European Commission

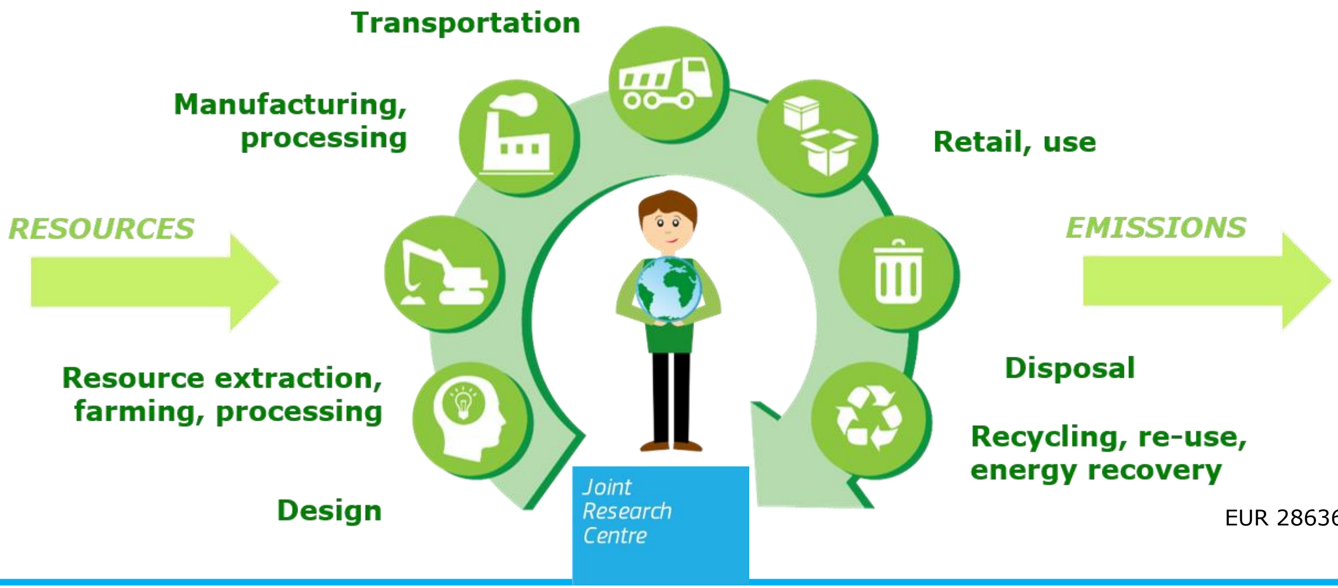
JRC TECHNICAL REPORTS

Suggestions for the update of the Environmental Footprint Life Cycle Impact Assessment

Impacts due to resource use, water use, land use, and particulate matter

Sala S, Benini L, Castellani V,
Vidal-Legaz B, De Laurentiis V, Pant R

2019



EUR 28636 EN

This publication is a Technical report by the Joint Research Centre (JRC), the European Commission's science and knowledge service. It aims to provide evidence-based scientific support to the European policymaking process. The scientific output expressed does not imply a policy position of the European Commission. Neither the European Commission nor any person acting on behalf of the Commission is responsible for the use that might be made of this publication.

Contact information
Name: Rana Pant
Address: Via E. Fermi, 2749
Email: rana.pant@ec.europa.eu

JRC Science Hub
https://ec.europa.eu/jrc

JRC106939

EUR 28636 EN

PDF	ISBN 978-92-79-69335-9	ISSN 1831-9424	doi:10.2760/78072
Print	ISBN 978-92-79-69336-6	ISSN 1018-5593	doi:10.2760/356756

Luxembourg: Publications Office of the European Union, 2019

How to cite this report: Sala S., Benini L., Castellani V., Vidal Legaz B., De Laurentiis V., Pant R. *Suggestions for the update of the Environmental Footprint Life Cycle Impact Assessment. Impacts due to resource use, water use, land use, and particulate matter*, EUR 28636 EN, Publications Office of the European Union, Luxembourg, 2019, ISBN 978-92-79-69335-9, doi:10.2760/78072, JRC106939.

Contents

Acknowledgements

We acknowledge the financial support of the Directorate General for the Environment (DG ENV) paying for this work under an Administrative Arrangement (AA JRC No 33446 – 2013-11 07.0307/ENV/2013/SI2.668694/A1). The EC-JRC team (Serenella Sala, Lorenzo Benini, Beatriz Vidal- Legaz, Valentina Castellani, Valeria De Laurentiis, Rana Pant) wants to thank Anne-Marie Boulay for the substantial contribution provided to the development of the water scarcity section of this report. The EC-JRC team wants to thank Jo Dewulf for the substantial contribution provided to the development of the resource use section of this report. The EC-JRC team wants to thank Assumpciò Anton, Danielle Maia De Souza, and Ricardo Teixeira for the contributions provided to the development of the evaluation of land use models as well as to thank Ulrike Bos and Rafael Horn for the support in the adaptation and update of the LANCA model The EC-JRC team thanks Luca Zampori and Erwin M Schau for reviewing sections of this document.

Executive summary

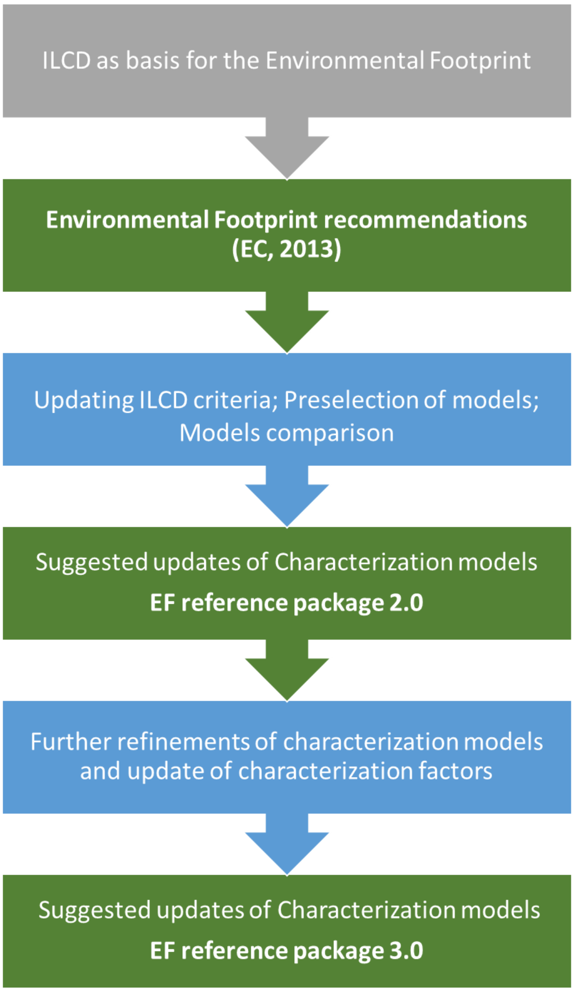
Companies and organisations that want to assess the environmental performance of their organisation or their products face numerous obstacles. They have to choose between several assessment methods promoted by public and private initiatives, and they are often forced to pay multiple costs for generating environmental information, and have to deal with the mistrust of consumers who are confused by the proliferation of too many communication tools with different information that makes products difficult to compare.

The Communication on Building the Single Market for Green Products (COM (2013) 196 final) and the related Recommendation 2013/179/EU on use of common methods to measure and communicate the environmental life-cycle performance of products and organisations, aim to ensure that environmental information in the EU market is comparable and reliable, and can be used confidently by consumers, business partners, investors, other company stakeholders, and policy makers. In this context, assessing the potential environmental impacts due to resource consumption and emissions into air, water and soil in a harmonised and robust way in the Life Cycle Impact Assessment (LCIA) phase is of high relevance to put stakeholders in a position to make better-informed decisions.

In 2011, the Joint Research Centre of the European Commission (EC-JRC) published the International Reference Life Cycle Data System (ILCD) Handbook recommendations on the use of Impact Assessment models for use in LCA (EC-JRC 2011). This created the basis for the Product and Organisation Environmental Footprint (PEF/OEF) recommendations for impact categories and characterisation models as per Recommendation 2013/179/EU on the use of common models to measure and communicate the life cycle environmental performance of products and organisations (EC 2013b).

The selection of LCIA characterisation models for the ILCD Handbook recommendations in 2011 was based on an assessment framework and related requirements and was limited to models available up to the year 2008. Over the years, a number of LCIA models have been developed for different impact categories, improving previous recommended models.

The present report illustrates the assessment of available characterisation models and factors in order to suggest an update of those recommended in the EF. The suggested updates where firstly reported in the EF reference package 2.0 and subsequently refined in the EF reference package 3.0) for the impact categories related to resource use, land use, water use, and particulate matter. It has to be noted that the EF reference package 2.0 includes as well updated characterisation factors for other impact categories (e.g. Climate change, ozone depletion) for which details are available in Fazio et al 2018a. The main steps followed to suggest updates of the recommendations for the Environmental Footprint Life Cycle Impact Assessment are reported in the workflow on the right.



The criteria for the evaluation of new models entailed: completeness of the scope; environmental relevance; scientific robustness and certainty; transparency, documentation and reproducibility; applicability of the model; robustness of characterization factors; stakeholders' acceptance. Details of the evaluation are reported as annexes to this document.

During the process of evaluating characterisation models, the name of the impact categories has been subject to changes compared to ILCD, and they are now referring to: impacts due to resource use, impacts due to land use, impacts due to water use and impacts due to emission of particulate matter. The suggested updates are reflected in the EF reference package 3.0. In 2017, a previous draft version of this report has provided updates for the EF reference package 2.0, which has been updated for what concern the land use characterisation model

The table below summarizes the models and the indicators suggested for the EF reference package 2.0 and 3.0, as well as the proposed level of recommendation.

Impact category	Suggested characterisation models and indicators
Impact due to resource use	Two mandatory indicators: - **"Abiotic resource Depletion"** (ADP ultimate reserves **-** for abiotic resources (metals and minerals) - **"Abiotic resource Depletion – fossil fuels"** (ADP fossil) for assessing depletion of energy carriers. based on the models of van Oers et al. 2002 and van Oers and Guinée 2016. Level of recommendation III
Impact due to land use	One mandatory indicator, applied at country scale: • **"Soil quality index"** resulted from the aggregation of selected indicators from LANCA model , namely LANCA Biotic Production; LANCA Erosion resistance; LANCA Mechanical filtration; LANCA Groundwater replenishment. For the EF reference package 2.0, adopting the model developed by Bos et al 2016. For the EF reference package 3.0, adopting the model of Horn and Maier 2018, as improved and implemented by De Laurentiis et al. 2019. Level of recommendation III
Impact due to water use	One mandatory indicator, applied at country scale, for consumptive uses and calculated with: • **"User deprivation potential"** resulted from the application of the AWARE model (Boulay et al. 2016 as recommended in UNEP, 2016) without: i) differentiating between agricultural and non-agricultural uses; and ii) monthly resolution Level of recommendation III
Impact due to emission of Particulate matter	One mandatory indicator: • **Disease incidences caused by 1 kg of PM emission**, calculated by the model developed by Fantke et al. (2016) as recommended in UNEP, 2016. Level of recommendation I

1 Introduction

1.1 Introduction for all impact categories

In 2011, the Joint Research Centre of the European Commission (EC-JRC) published the International Reference Life Cycle Data System (ILCD) Handbook recommendations on the use of Impact Assessment models for use in LCA (EC-JRC, 2011). This created the basis for the Product and Organisation Environmental Footprint (PEF/OEF) recommendations for impact categories and models as per Recommendation 2013/179/EU on the use of common models to measure and communicate the life cycle environmental performance of products and organisations (EC, 2013a). This Commission Recommendation is expected to contribute to the Building the Single Market for Green Products (EC, 2013b) by supporting a level playing field regarding the measurement of environmental performance of products and organisations.

The selection of LCIA models for the ILCD Handbook recommendations in 2011 was based on an initial analysis (EC-JRC, 2010a) and a description of an assessment framework and related requirements (EC-JRC, 2010a) and was limited to models available up to the year 2008. Between 2008 and 2016 a number of LCIA models had been published in scientific journals for several impact categories, with increased level of complexity, resolution and geographic coverage in comparison to those models available in 2008.

Impact assessment LCA is a fast developing area and any recommendation in that area requires periodically further development, maintenance and updates. Therefore, the existing default impact assessment categories and models for resource depletion, land use, water depletion, and respiratory inorganics for use in Environmental Footprint according to Recommendation 2013/179/EU are reviewed and suggestions for necessary updates are made. The resulting EF method has evolved over time. Annex 1.1 provides an overview of this process from the initial recommendation (EC 2013a), to EF reference package 2.0, up to EF reference package 3.0.

Moreover, the development of this assessment has proceeded in parallel with the activities of the United Nations Environment Programme and the Society for Environmental Toxicology and Chemistry Life Cycle Initiative (in the remainder, referred as UNEP-SETAC life cycle initiative) on life cycle impact assessment. Indeed, the UNEP-SETAC life cycle initiative undertook a global process aiming at global guidance and consensus building on a selected number of life cycle impact category indicators (Jolliet et al., 2014; UNEP, 2016). EC-JRC has been directly involved in the process, leading to the release of the first set of recommendations as results of 2 years of the work of working groups composed by international experts and practitioners. Those recommendations have been recently published in a report, built from a Pellston workshop held in January 2016 (UNEP, 2016). The present work of analysis capitalizes on the EC-JRC involvement in the process, namely for land use, water use and particulate matter.

This final version covers the impact categories related to resources use, land use, water use and the emission particulate matter. It has been developed following a number of steps, as in the following figure.

1.2 Update criteria for all impact categories

A review of the general criteria used for assessing and evaluating LCIA models was performed. Moreover, building on the criticism received on the some of the LCIA models recommended for use by ILCD, a section of the evaluation criteria dedicated to the aspects of the characterization factors was added to the groups of criteria. Those new criteria were added on top of the sections defined in EC-JRC (2011) (i.e. Completeness of Scope; Environmental Relevance; Scientific robustness and uncertainty, Transparency, Documentation, and Reproducibility; Applicability, and Stakeholders Acceptance). The list of modifications made to the previous set of criteria is reported below:

- **Introduction**: additional general aspects to be reported in the "Introduction" section were added so to provide readers with an increased description and understanding of the model. The following aspects have been added on top of those already included in the ILCD evaluation (EC-JRC, 2011): *'Model and its purpose'; 'Description of the impact pathway of the characterization model'; 'Midpoint indicator/s name and metric'; 'Range of values of the characterization factors'; 'Underlying model(s)';*
- **Completeness of the scope**: the criteria have been. The updated list includes: 'Impact pathway completeness'; 'Impact pathway consistency'; 'AoP coverage by the midpoint characterization model'; 'Midpoint indicator placement in the impact pathway regarding LCI flows'. A criterion was removed (i.e. use of empirical data) and some moved to the section Environmental Relevance or Applicability (i.e. geographical coverage and resolution);
- **Environmental Relevance**: although this section is specific for each impact category, a common structure composed by three groups of criteria was developed building on the previous list. The groups are: *'Coverage of the environmental mechanisms'; 'Spatial and temporal resolution'; 'Comprehensiveness - elementary flows';*
- **Scientific robustness and Certainty**: the title of the section was changed into 'Scientific robustness and Uncertainty'. Moreover, specific criteria were added:
- **Transparency, Documentation and Reproducibility:** no major modifications were made to this group of criteria; the following criterion was added for clarity: **'***Completeness of the characterization model documentation'*;
- **Applicability:** no major changes were introduced in this group of criteria, the additional criterion: *'Availability of normalization factors for LCA practitioners'* was added so to distinguish the level of readiness of the different models for use in LCA;
- **Characterization factors**: a new section dedicated to the analysis of the characterization factors was introduced with the aim of better assessing the relevance, usability and maturity of the models, including coverage of geographical and temporal scales. Ideally, indicators used in LCIA as characterization factors should allow for use at both high temporal and spatial resolution and large scales (year - country) in order to meet large background applications requirement and finer foreground assessment. The criteria added here were: *'Relevance of the characterization'; 'Usability of characterization factors for LCA practitioners'; 'Testing of the characterization factors'; 'Temporal resolution of characterization factors'; 'Spatial resolution of characterization factors'*;
- **Stakeholders' acceptance**: few modifications were made, as the criterion related to the understanding of the principles of the model was merged with the criterion: '*understandability and interpretability of the model'*.

1.3 References of the introduction

EC (2013a). Communication from the Commission to the European Parliament and the Council - Building the Single Market for Green Products - Facilitating better information on the environmental performance of products and organisations. COM(2013)196

EC (2013b). Recommendation 2013/179/EU on the use of common methods to measure and communicate the life cycle environmental performance of products and organisations, Annex III, OJ L 124, 4.5.2013, p. 1–210. Available at:https://eur-lex.europa.eu/legal-content/EN/TXT/?uri=CELEX%3A32013H0179

EC (2017). PEFCR Guidance document - Guidance for the development of Product Environmental Footprint Category Rules (PEFCRs), version 6.3, December 2017. Available at http://ec.europa.eu/environment/eussd/smgp/pdf/PEFCR_guidance_v6.3.pdf (Accessed July 2018)

EC-JRC (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

EC-JRC (2012). Characterisation factors of the ILCD Recommended Life Cycle Impact Assessment methods. Luxembourg: Publications Office of the European Union. EUR 25167EN

EC-JRC (2018a). Environmental Footprint reference package 2.0 (EF 2.0). Available at: https://eplca.jrc.ec.europa.eu/LCDN/developerEF.xhtml

EC-JRC (2018b). Environmental Footprint reference package 3.0 (EF 3.0). Available at: https://eplca.jrc.ec.europa.eu/LCDN/developerEF.xhtml

Fazio, S. Castellani, V. Sala, S., Schau, EM. Secchi, M. Zampori, L., (2018a) Supporting information to the characterisation factors of recommended EF Life Cycle Impact Assessment methods, New models and differences with ILCD. EUR 28888 EN, European Commission, Ispra.

Fazio, S., Biganzoli, F., De Laurentiis, V., Zampori, L., Sala, S. and Diaconu, E., (2018b) Supporting information to the characterisation factors of recommended EF Life Cycle Impact Assessment methods, version 2, from ILCD to EF 3.0, EUR 29600 EN, Publications Office of the European Union, Luxembourg

Zampori, L. and Pant, R., (2019a) Suggestions for updating the Product Environmental Footprint (PEF) method, EUR 29682 EN, Publications Office of the European Union, Luxembourg

Zampori, L. and Pant, R., (2019b) Suggestions for updating the Organisation Environmental Footprint (OEF) method, EUR 29681 EN, Publications Office of the European Union, Luxembourg

Zampori, L. and Pant, R., (2019a) Suggestions for updating the Product Environmental Footprint (PEF) method, EUR 29682 EN, Publications Office of the European Union, Luxembourg

Zampori, L. and Pant, R., (2019b) Suggestions for updating the Organisation Environmental Footprint (OEF) method, EUR 29681 EN, Publications Office of the European Union, Luxembourg

EC-JRC (2018b). Environmental Footprint reference package 3.0 (EF 3.0). Available at:

Fazio, S., Biganzoli, F., De Laurentiis, V., Zampori, L., Sala, S. and Diaconu, E., (2018b) Supporting information to the characterisation factors of recommended EF Life Cycle Impact Assessment methods, version 2, from ILCD to EF 3.0, EUR 29600 EN, Publications Office of the European Union, Luxembourg.

2 Introduction for the impact categories related to resources: metals and fossils, land, and water

2.1 Update impact framework

Current LCIA recommendations do not capture 'resource footprint' in a comprehensive way. As modern society, we fully depend not just on 'depletable' abiotic resources, but equally on land as a resource for urbanization, infrastructure and agro-industrial production (essential for the functions/end use shelter, mobility and food), natural biomass (fish stocks), air, water, abiotic renewable resources (solar, wind, hydro). By only considering 'depletable' abiotic resources, the full picture is not captured.

Through the historical development, LCA experts stay within the commonly agreed viewpoints with a huge emphasis on natural environment and human health as area of protection. Especially on natural assets like water bodies and land, they keep emphasis on their role in function of Natural Environment. This is highly justified as these assets host the natural environment, supporting biodiversity, ecosystem services beyond provisioning. Equally, the viewpoints on these assets need be to be broadened in light of the area of protection natural resources for their provisioning role: no land, no water as resources means no products.

Another consequence is that raw materials derived from biotic resources are not treated equally to abiotic resources in the assessment. Taking a simple example: a street bench made of tropical wood versus one made of metals or plastics is about 'for free' in terms of natural resource footprint. While there are certainly differences in terms of resource consumption between renewable biotic and non-renewable abiotic resources, the current situation cannot ensure that products using mainly abiotic 'depletable' resources are treated in an equal way.

To reflect the discussion around how "depletable" some of the abiotic resources are and to reflect the envisaged broadening of the scope of the impact category, the impact category is from now on titled "Resource Use" instead of the previous category title "Resource Depletion".

2.1.1 Framework

In function of the AoP Natural Resources in LCA, five types of Natural Resource Assets have been identified with their respective sub-assets (24 in total) that can be considered for the AoP Natural Resources and LCIA models (Table 2.1).

Table 2.1. Five assets and 24 sub-assets to be considered in function of the AoP Natural Resources and LCIA models (adapted from Swart et al., 2015, based on Lindeijer et al., 2002).

Asset	Sub-asset
1. Abiotic resources (stocks)	Minerals & metal ores
	Fossils
	Nuclear
	Elements from the hydrosphere
	Elements from the atmosphere
	Soil
2. Abiotic resources (flows)	Solar
	Wind
	Hydro

Asset	Sub-asset
	Tidal Geothermal
3. Air and water bodies[1]	Groundwater Rainwater Freshwater bodies Marine water bodies Air
4. Land and sea surface[1] (occupation/transformation) (man-made)	Agricultural land (→ manmade biomass) Forestry land (→ manmade biomass) Aquaculture surface (→ manmade biomass) Urban land Industrial/infrastructure land
5. Natural biomass	Terrestrial ecosystems (→ natural biomass) Marine ecosystems (→ natural biomass) Freshwater ecosystems (→ natural biomass)

2.1.2 Scope

Natural Resources are part of the natural environment but at the same time they are only 'resources' if they have a role in the manmade industrial system. Given this ambiguity, it turns out that many perspectives (viewpoints) can be adopted, with significant impacts on related LCIA models.

Preparatory work has been published for a better understanding of the area of protection Natural Resources (Dewulf et al., 2015), offering different viewpoints. Five perspectives on what should be safeguarded with respect to natural resources have been identified:

Perspective 1: Asset of Natural Resources as safeguard subject (S1). Natural Resources as such are seen as safeguard subject as such as we are conscious that in the end they have a function for humans directly or indirectly, irrespective of their further role, function or impact on humans and ecosystems.

Perspective 2: Provisioning Capacity of Natural Resources as safeguard subject (S2). The capacity of ecosystems to fulfil provisioning functions for humans, i.e. provide materials, energy, food, and space directly is to be safeguarded.

Perspective 3: Global functions of Natural Resources as a safeguard subject (S3). Next to provisioning, other non-provisioning functions for humans and the global (eco)system as a whole are recognized and should be safeguarded, e.g. role of tropical forests in climate regulation.

Perspective 4: Natural Resources as building block in the supply chain of Products and Services for human welfare as safeguard subject (S4). This safeguard subject includes the essential provisioning capacity of the natural resource base (perspective 2) but it is expanded in perspective 4, since a number of socio-economic mechanisms can hinder the human welfare benefits from natural resources.

Perspective 5: Natural Resources for human welfare as safeguard subject (S5). This is a more holistic point of view on the role natural resources play in human welfare

[1] Please refer to the section "Impact pathway" for a description on how ILCD deals with land and water as resources, including in relation to the impact categories "water depletion" and "land use".

through their direct and indirect functions they provide, encompassing perspectives 2, 3 and 4.

With respect to the perspective to be adopted, perspectives 4 and 5 have a socio-economic scope, hence, they can be seen as going beyond 'classical environmental LCA'. Perspective 3 - even if preferable against perspective 1 and 2 because it is more comprehensive in terms of the impacts that are covered -, looks to be unfeasible for the time being as there is currently insufficient modelling that can capture the complexity fully, as there is a lack of quantitative factors to characterize it.

Perspective 2 proves to be the perspective that matches currently best with what is to be protected with respect to Natural Resources in classical LCA: the Provisioning Capacity of Natural Resources as safeguard subject. Moreover, there are a significant number of LCIA models available that fit with this perspective (see Addendum 1). However, they typically cover only a particular (sub) asset of the Natural Resources.

Nevertheless, even if a set of LCIA models following perspective 2 (LCIA-P2) is proposed, the set may not be able to cover the full asset of Natural Resources. A couple of reasons can be mentioned: there is not a full set of LCIA models to cover the full asset of Natural Resources within perspective 2. Equally, there may not be a need to have the full asset covered as some are of higher priority compared to others. It is suggested that the LCA practitioner should at least be aware of the limited range of Natural Resources that is covered by the set of LCIA models he/she uses. Therefore, it is proposed to complement the set LCIA-P2 with an accounting of (sub) assets covered relative to the total asset of natural resources a production/consumption system relies on. This 'coverage' could simply come from models following perspective 1 making use of physical accounting.

Looking at the complexity and diversity and the existing different approaches of natural resources as an area of protection, the number of impact categories related to the area of protection natural resources may need to be increased, not only to achieve a more balanced picture related to the two other areas of protection but especially for the abovementioned reasons.

Secondly, there is a need to do prioritization for particular natural resource assets. LCIA for Natural Resources is approached, here, in a two-tier approach in function of the aforementioned perspectives:

- <u>Tier 1: Non-abundant Resource Accounting (NARA)</u> intends to protect the full asset of resources that are considered to have some supply constraint, either because they lack renewability or they have limited abundance or they are not widespread available.

In defining the range of natural resources in terms of this <u>N</u>on-<u>A</u>bundant <u>R</u>esource <u>A</u>ccounting (NARA) that are of concern, solar radiation/energy and wind are considered out of the scope. First of all, they are the only resources that are considered to be always available as they are renewed the fastest with renewal times below 0.05 years, see Figure 1 (Cummings and Seager,2008).

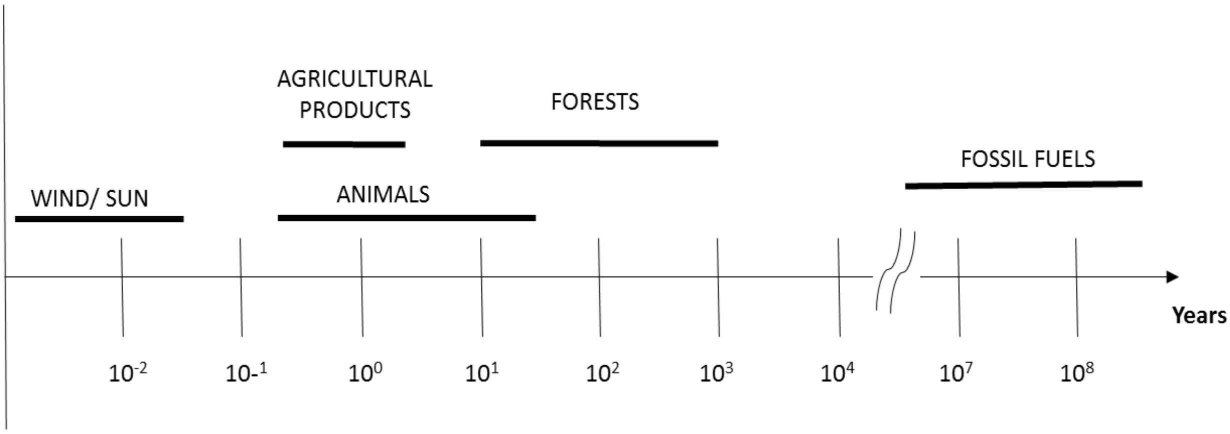


Figure 2.1. Renewal time for resources in logarithmic scale (modified from Cummings and Seager, 2008).

Secondly, their continuously renewed quantities are orders of magnitude higher than their currently used quantities. The exergy flow of solar radiation heating the land and oceans amounts to 43.200 TW: that is about three thousand times more than the present power needs of the whole world (17 TW in 2010). Knowing that geothermal, solar, wind and heat all together merely constituted 1.1% of the world production in 2012 (IEA, 2014): means that the available solar energy versus current production rates is at least in the order of 10^5. Similar reasoning is valid for wind. Global wind power continuously regenerated at locations with mean annual wind speeds ≥ 6.9 m/s at 80 m is found to be about 72 TW (Valero 2008). A technical potential of 72 TW installed global capacity at 20% average capacity factor would generate around 14.5 TW, which is about 250 times the 2005 exergy power capacity worldwide, 59 GW. Finally, the geographical spread of solar energy and wind is far better than the other renewable energy flows, i.e. geothermal and hydropower.

Based on these considerations the following natural resource assets are to be included in the impact category NARA at tier 1:

-Abiotic Resources – fossils (for energy or material purposes)

-Abiotic Resources – metal and minerals (for energy or material purposes)

-Abiotic Resources – hydropower and geothermal

-Land as a Resource (see also land use impact category)

-Water as a Resource (see also water impact category)

-Natural Biomass

At the LCI level, these resource assets can be accounted in just their basic physical units: mass (kg), energy content (MJ), or spatial units (m^2 or m^3). It makes sense to foresee accounting in all applicable units, this in function to keep the range of future applications as broad as possible (e.g. mass for MFA; e.g. exergy for technical resource efficiency).

However, in function of LC(I)A at perspective 1 level, more advanced models may be more advantageous because of their capability to cover a broader range of assets (e.g. ecological footprint, exergy based models).

Tier 2: PArticular Resource Assets: PARA

Among the non-abundant resources accounted for at tier 1, there are PArticular Resource Assets (PARA) to be assessed in function of their current and future provisioning capabilities with according impact assessment models. From this starting point, the capabilities of hydropower and geothermal are not considered under PARA because of their renewable character, this among other reasons. This makes that ideally impact assessment models are to be evaluated for the following categories:

PARA1: Abiotic Resources – fossils (for energy or material purposes)

PARA2: Abiotic Resources – metal and minerals (for energy or material purposes)

PARA3: Land as a Resource (see also land use impact category)

PARA4: Water as a Resource (see also water use impact category)

PARA5: Natural Biomass

Depending on the available impact assessment models and their respective scope, the number of models may be limited to just one model that is capable to cover all the 5 PARAs. However, given the different nature of the 5 PARAs, it is likely that several impact assessments need to be used, up to 5 different ones.

Since Land and Water are items that are also relevant for other areas of protection and the evaluation is hence dealt with separately from Natural Resources in the context of land use and water use impact categories, the work related to resources is limited here to evaluate and select models only for:

NARA: All non-abundant natural resources

PARA1: Abiotic Resources – fossils (for energy or material purposes)

PARA2: Abiotic Resources – metal and minerals (for energy or material purposes)

PARA5: Natural Biomass

3 Impact due to Resource use

3.1 Introduction

The model for resource impact assessment adopted in the ILCD recommendation (see EC-JRC 2011) was the CML model for abiotic depletion potential, based on the model developed by Guinée (2002). The characterization factors adopted for the impact category "Resource depletion – mineral and fossils" at midpoint, were the ones proposed by van Oers et al. (2002), building on Guinée (2002) but using reserve base as reference for resource stock (instead of "ultimate reserves" as proposed by Guinée). Characterisation factors (CFs) are given as Abiotic Depletion Potential (ADP), quantified in kg of antimony-equivalent per kg extraction, or kg of antimony-equivalent per MJ for energy carriers (see also EC-JRC, 2012). van Oers et al. (2002) suggest to define a separate impact category for fossil fuels, based on their similar function as energy carriers. However, this was not implemented in the CML model in 2002 (and not until 2009 version), nor in the current version of ILCD recommendation. Therefore, the separation of abiotic resources and energy carriers into two indicators is an issue that has been explored in the current evaluation and considered for recommendation.

Several critics have been raised to the abiotic depletion concept (applied in different forms by several LCIA models) and more specifically to its application in the context of life cycle assessment. They are reported and summarised in section 3.1.1. These critics have been taken into consideration within the process of updating the recommendation.

3.1.1 Critical issues related to the current recommendation and to resource assessment in LCA

The first critic moved by experts in the field of mining activities is related to the partial inconsistency between the terminology used in the LCA context and the terminology used by the mining industry. Therefore, it is important to clarify the terminology used, in order to be able to better understand and communicate across disciplines. Table 3.1 provides a glossary of the terminology used in the mining industry and in LCA.

As reported by Drielsma et al. (2016), the main difference in the terminology used in the mining industry and in LCIA is in the use of the term "reserve". "In the mining industry, anything that is referred to as a "reserve" has a high level of feasibility and is economic to extract in the current or short-term future. The Committee for Mineral Reserves International Reporting Standards (CRIRSCO) would class the reserve base of Guinée and Heijungs (1995) as mineral resources and economic reserves as mineral reserves." (Drielsma et al. 2016, p. 90). Figure 3.1 illustrates the relationship between crustal content, resources, reserves, and the technosphere.

Regarding the model recommended in ILCD ($ADP_{[Reserve\ Base]}$), some authors (e.g. Bringezu, 2015, Drielsma et al., 2016) question the environmental relevance of assessing the depletion of abiotic resources using economic data. They highlighted that this is a way to measure the availability for human use (also driven by economic and technological issues) but not to account for physical depletion of resources.

Drielsma et al. 2016 discussed also the inherent differences in the two possible perspectives available when accounting for resource availability and depletion. The first option is to apply the "fixed stock paradigm", i.e. to evaluate availability of resources considering their abundance in the Earth's crust and assuming that the whole content can be extracted and irreversibly depleted. The second option is to apply the "opportunity cost paradigm", i.e. to consider resource availability as an economic question driven by market demand. Drielsma et al., 2016 underline that in the first case, the crustal content is taken as planetary boundary upon which quantify the depletion potential, whereas when the second approach is chosen, it is not straightforward to quantify the stock, i.e. no fixed boundary can be identified. Reserve estimates are considered accurate, restricted and fluctuating. The fluctuations are due to: demand; policy and governance, technological improvements for discovery and extraction; access to energy; cost of capital; exchange rates. Instead, resource estimates are considered selective and uncertain (Drielsma et al., 2016).

Table 3.1: Glossary on abiotic resources definitions used in the geological and mining context and in LCIA (based on Drielsma et al., 2016).

<table>
<tr><th>Meaning</th><th>Name used by the mining industry</th><th>Name used in LCIA</th></tr>
<tr><td><u>Total amount of an element in a given layer of the Earth’s crust.</u> It is estimated by multiplying the average concentrations of chemical elements in the crustal layer by the mass of the same crustal layer. The crustal content of an element will never be extracted completely as some deposits/concentrations will remain unavailable under all foreseeable economic conditions.</td><td>Crustal content</td><td>Ultimate reserve</td></tr>
<tr><td><u>Amount of crustal content that will ultimately prove extractable by humans.</u></td><td>Extractable global resource</td><td>Ultimately extractable reserve</td></tr>
<tr><td><u>Concentration or occurrence of solid material of economic interest in or on the Earth’s crust in such form, grade or quality, and quantity that there are reasonable prospects for eventual economic extraction.</u> The location, quantity, grade or quality, continuity, and other geological characteristics of a mineral resource are known, estimated, or interpreted from specific geological evidence and knowledge, including sampling.</td><td>Mineral resource</td><td>Reserves base</td></tr>
<tr><td><u>Economically mineable part of a measured and/or indicated mineral resource.</u> It includes diluting materials and allowances for losses, which may occur when the material is mined or extracted and is defined by studies at pre-feasibility or feasibility level as appropriate that include application of modifying factors. Such studies demonstrate that, at the time of reporting, extraction could reasonably be justified.</td><td>Mineral reserve</td><td>Economic reserves</td></tr>
<tr><td><u>Process of physically reducing the global amount of a specific resource.</u> It refers to the reduction of geological/natural stocks over time—not of an individual mine or ore body.</td><td>Resource depletion</td><td rowspan="2">Resource depletion</td></tr>
<tr><td><u>Mining out of already identified mineral reserves.</u></td><td>Reserve depletion</td></tr>
</table>

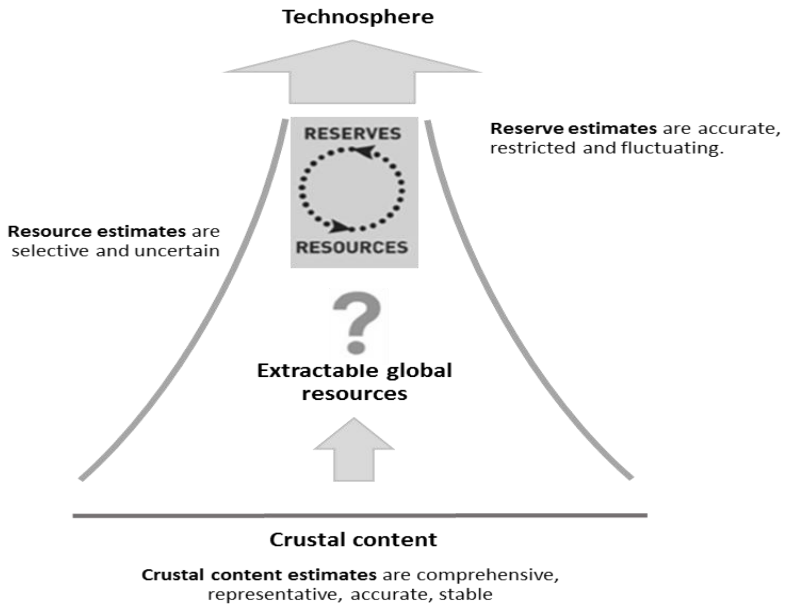


Figure 3.1 Relationship between crustal content, resources, reserves, and the technosphere (modified from Drielsma et al. 2016).

Within ADP model by Guinée (2002), the first approach is recommended (ultimate reserves), whereas ILCD recommends an option based on the second type of approach (reserves base). Van Oers and Guinée (2016) comment on this difference, stressing the need to go back to their original choice (crustal content, as "ultimate reserves" approach) because using the crustal content enables for a more environmental-oriented evaluation. Anyway, they also highlight that "data on the ultimately extractable reserve are unavailable and will never be exactly known because of their dependence on future technological developments. Nevertheless, one might assume that the "ultimate reserve" is a proxy for the "ultimately extractable reserve", implicitly assuming that the ratio between the ultimately extractable reserve and the ultimate reserve is equal for all resource types. In reality, this will not be the case, because the concentration-presence-distribution of different resources will most likely be different. Hence, there is insufficient information to decide which of these reserves gives the best indication of the ultimately extractable reserve" (van Oers and Guinée, 2016, p.16).

As pointed out also by Yellishetty et al. (2011), Klinglmair et al. (2014) and Rørbech et al. (2014), the exact quantification of the stocks of resources available following the opportunity cost paradigm (use-to-availability ratio) is a complex task, that can have a high level on uncertainty, due to several factors such as: i) the influence of price volatility on the mining activities; ii) the role of technological improvements in making the extraction economically viable or not, etc. Therefore, there can be high variability of results even if the same approach (e.g. ADP) is applied, depending on how the reference stock is measured (e.g. ultimate resources, reserve base, economic reserves, etc.). The most recent literature on this topic (e.g. Drielsma et al., 2016, van Oers and Guinée, 2016) suggests assessing availability of resources implementing methods outside the LCA context, to be used in parallel to and complementing LCIA. With reference to models for environmental LCA, there is more favour for models based on use-to-stock ratio paradigm rather than to use-to-availability ratio, because in the first case the denominator reflects the environmental stock and is more stable in time.

Moreover, the opportunity cost approach needs to take into account the temporal dimension, because economic data change annually in response to demand, exploration and supply cycles, politics and socio-economic trends (Drielsma et al. 2016). Therefore, input data for the calculation of resource availability should be periodically updated.

Finally, another critic posed to the framework of abiotic resource depletion refers to the assumption that once a resource is extracted from the Earth's crust, it is considered depleted. Several authors (Yellishetty et al. (2011), Klinglmair et al. (2014), Frischknecht (2014), van Oers and Guinée, 2016) discuss the possibility to consider also the amount of resources available in the technosphere (because they have been used but still available in the form of scraps or waste) as part of the stock potentially available and include them in the calculation. As for the other type of resources, also the quantification of the so-called anthropogenic stock of resources poses some challenges, such as the uncertainties related to quantification (e.g. the complexity of differentiating the recyclability potential of different metals Klinglmair et al. 2014) or the need to account for the time of residence in the products before the resources can be made available for reuse or recycling (Yellishetty et al. 2011).

Beyond the critics to the model underpinning the ILCD recommendation, also other issues deserve consideration when evaluating LCIA models and models for the Area of Protection Natural Resources. Klinglmair et al. (2014) identified some key issues that are poorly covered by the existing LCIA models and that need to be taken into consideration when looking for an improvement in the AoP natural resources:

- *Biotic resources and renewability issues*. Biotic resources are poorly covered at present in LCIA, and the current ILCD recommendation (EC-JRC, 2011) does not include CFs for biotic resources. With regards to this aspect, Brigenzu (2015) highlights the growing relevance of some categories of biotic resources, that need to be addressed when performing sustainability assessments. They are the *topsoil* (threatened by erosion), *forest biomass* and *fish stocks*. In addition, the authors point out that there is still a misleading perception that renewable resources do not pose any criticality problem and the stock of ecological capital is not subject to depletion. They also report the proposal by Lindeijer (2002) *to include biotic resources in the resource depletion assessment*. A review of methods accounting for biotic is reported in Crenna et al., 2018.
- *Recycling.* Recycling is currently considered only at the LCI phase, but usually not in LCIA phase (except for few models, e.g. Schneider et al., 2015 and Frischknecht, as mentioned in Vadenbo et al., 2014). To improve the coverage and the ability of abiotic resource depletion models, also the amount of resources already in the technosphere and potentially available (the so-called "*anthropogenic stock*") should be included in the evaluation of resource availability.
- *Criticality.* The ongoing work done, among others, by the European Commission (Chapman et al. 2013) on the assessment of criticality and *supply risk* of materials may be used to complement the current perspective (focused mainly on the provisioning function of resources, mentioned as "perspective 2") with a more supply-chain oriented approach (mentioned as "perspectives 4 and 5") (see also Dewulf et al. 2015, and Mancini et al., 2016) for a deeper discussion on this topic). It is worthy to note that other authors (such as van Oers and Guinée, 2016) are against the inclusion of criticality as a LCIA indicator, because it does not consider mainly environmental issues. Others (e.g. Drielsma et al. 2016), highlight that criticality switch the subject and the object of the assessment from assessing the impacts of the system on its surroundings to assessing the impacts of the surroundings on the system.
- *Dynamic approach to estimate future availability.* Due to the already discussed role of market demand of resources in driving the exploration and the mining activities, static indicators are –in general – not valid approaches to predict physical scarcity of resources in the future (Scholz et al. 2013). Dynamic models should therefore be preferred.

It is worthy to note also that the UNEP-SETAC Life Cycle Initiative has a task force on cross-cutting issues that include also the investigation on how to assess damages to the AoP 'Natural Resources' across all natural resources. The results of this work, still in progress, can contribute to the evaluation presented in the present document.

Additionally, from 2017, a group dedicated on resources, focusing on metals, has been established by UNEP-SETAC.

3.2 Framework and scope of the evaluation

The evaluation has been done taking into consideration the advancements in research regarding the topic of resource depletion, the weaknesses of the model currently recommended ($ADP_{reserve}$) highlighted by some researchers and by industries and the potential areas of improvements mentioned before.

Therefore, the scope of the evaluation was to assess models:

- Representing possible improvements within the area of resource depletion
- Able to cover a wider range of resources, especially biotic ones (enlarging the current scope, which include only abiotic resources)
- Able to take into account different approaches and key issues identified as priority for the AoP (e.g. renewability and criticality).

3.3 Environmental mechanism (cause-effect chain)

The environmental mechanism (impact pathway) taken as reference for the evaluation of LCIA models about resources in the present work is illustrated in Figure 3.2. The figure depicts the cause-effect chain from the human intervention (which define the border between the ecosphere and the technosphere) to the final effect on the Areas of Protection (AoPs). The intermediate steps between the cause and the effect are classified as:

- Pure accounting of resources extracted (e.g. in mass or energy).
- Intermediate accounting, starting from pure accounting and calculating the amount of resource extracted using some inherent properties more related to their final use, i.e. their value for the natural (e.g. emergy) or the human-made (e.g. exergy) systems.
- Midpoint impacts.
- Endpoint impacts.

According to the classification provided before, the models dealing with accounting and some of the ones for advance accounting adopt Perspective 1 (Asset of Natural Resources as safeguard subject) whereas some other models for advance accounting and all the models for midpoint and endpoint impact assessment adopt perspective 2 (Provisioning Capacity of Natural Resources as safeguard subject). Models adopting Perspective 3 (Global functions of Natural Resources as a safeguard subject) refer both to the impacts on the AoP Ecosystem health and to the AoP Natural resources. Models adopting Perspective 4 (Natural Resources as building block in the supply chain of Products and Services for human welfare as safeguard subject) regards the impacts on the AoP Human health.

As explained in the section "scope", the two latter perspectives (Perspectives 4 and 5) – even if relevant in a broader perspective - are considered out of the scope of the present exercise (i.e. to evaluate and recommend LCIA models to account for the effect on the AoP Natural resources: provisioning capacity). Hence, they are presented in grey in the figure and will not be discussed in the evaluation of the models.

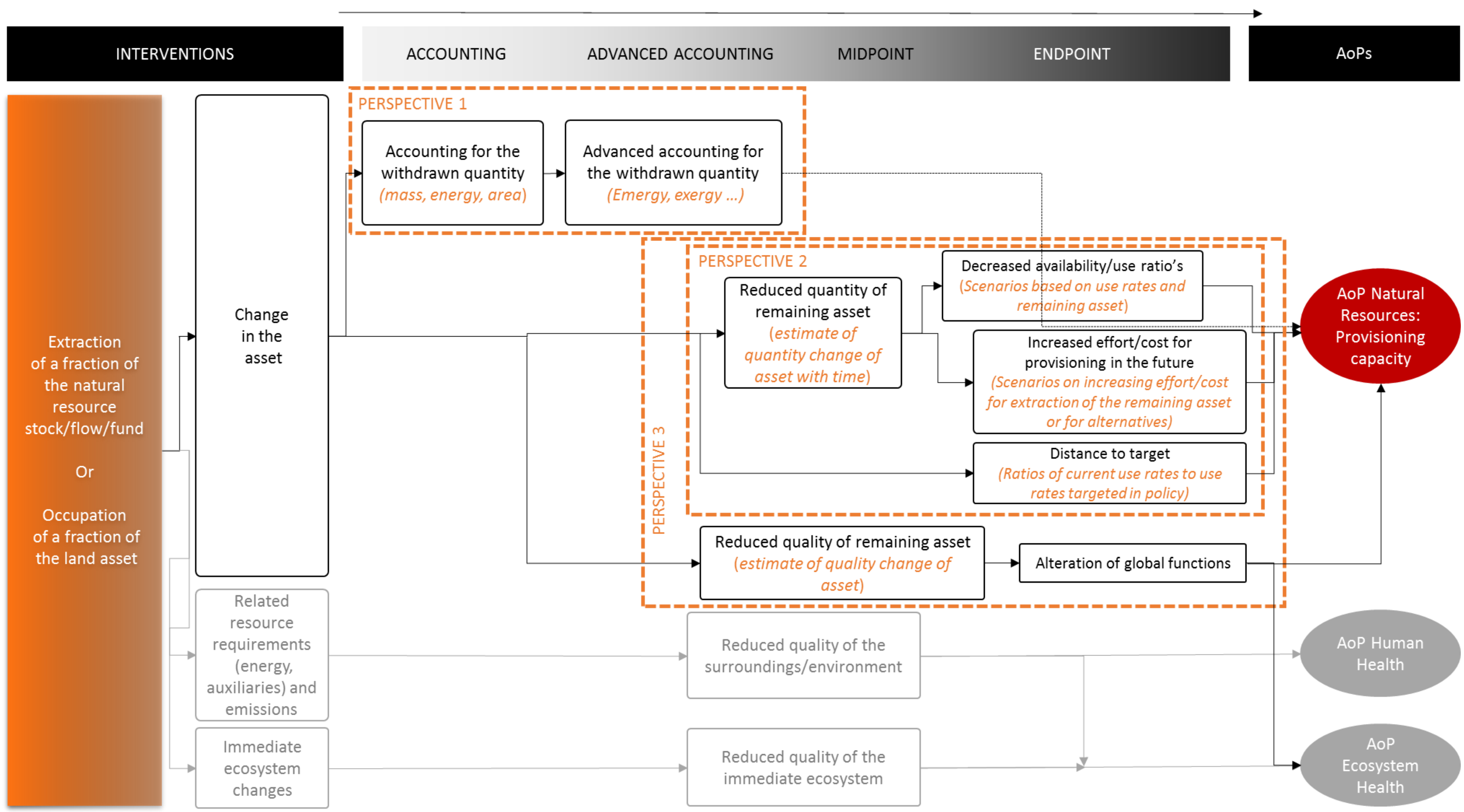


Figure 3.2 Impact pathway (cause-effect chain) for resources.

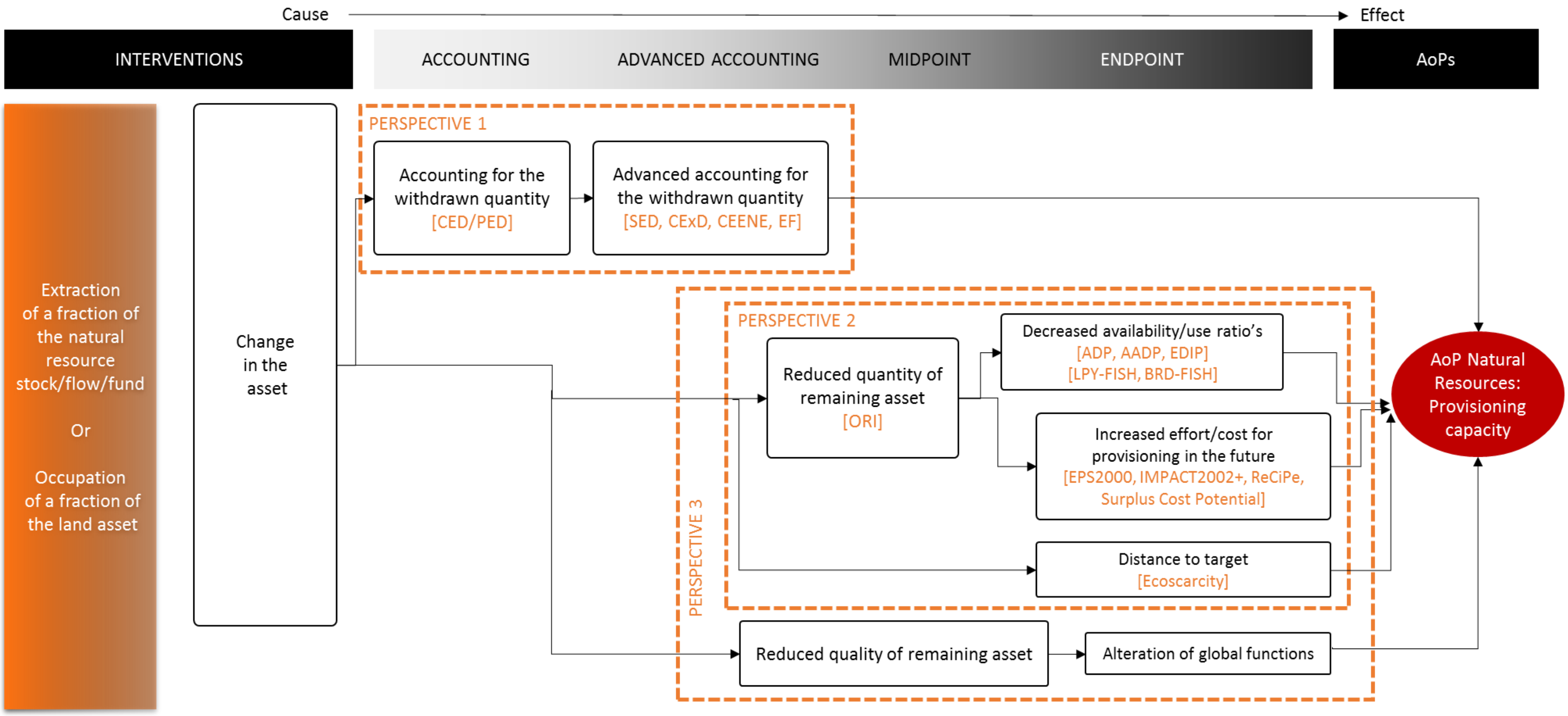


Figure 3.3 Positioning of LCIA models for natural resources, limited to perspective 1, perspective 2 and perspective 3, within the impact pathway described before. Within perspective 2, land as a resource and water as a resource are not covered, hence only abiotic resources (fossils, metals, minerals) and biotic resources (natural biomass) are included.

3.4 Criteria for the evaluation of this impact category

In addition to the general criteria defined for the evaluation of all the impact categories (reported in section 1.1), some other criteria have been selected to take into account specific features of the impact category resources. These additional criteria are described below.

Summary information.

Impact categories covered by the model: description of the coverage of the model in terms of the impact categories described before (NARA, PARA1, PARA2, PARA3, PARA4, PARA5), i.e. the type of resources (biotic, abiotic, abundant, non-abundant, etc.) considered by the model.

Relevance to the envisaged Area(s) of Protection.

Perspective adopted: perspective adopted by the model, as described in section 2 (Perspective 1: Asset of Natural Resources; Perspective 2: Provisioning Capacity of Natural Resources, Perspective 3: Global functions of Natural Resources, Perspective 4: Natural Resources as building block in the supply chain of Products and Services for human welfare, Perspective 5: Natural Resources for human welfare). The aim is to describe which is the main concern behind the rationale of the models in terms of safeguard of resources.

Within the general criterion "Comprehensiveness", that aims at evaluating if all critical parts of the environmental mechanism describing the cause-effect chain, as defined by JRC, are included with acceptable quality, some specific criteria are added, regarding the type of resources considered and the number of types covered by the model. The core of the evaluation regards the inclusion of "*Energy, metals and minerals, biotic resources*". Models that are able to consider all the three categories are considered as preferable.

The ability of the model to rank also *water as resource* and *land as resource* with the same metric is also considered in two specific criteria under "Comprehensiveness", while land and water use are treated also as separate impact categories in the context of the Environmental Footprint in their relation to other AoPs.

With reference to the general criteria on spatial and temporal resolution of the model, we wanted to assess if the model is time-dependent and/or spatial-dependent. The first aims at verifying if the model depends from inputs to be updated periodically or not, i.e. if it can be valid over time or needs to be periodically updated to maintain its validity. The second one aims at assessing if the model is valid globally or it is referred only to a specific region, or it has both global and regional/country values.

Applicability.

Regarding the compatibility with the most common LCI datasets, the specific criterion *"Are characterization factors provided for different ore grades?"* is added for resources impact category. The aim is to evaluate how the model is dealing with ore grades and to check if this is easily applicable with the structure of available datasets.

Annex 3.1 reports all the criteria used for the evaluation of models in the impact category Resource use.

3.5 Preselection of models for further evaluation

Given that the number of available resource related impact models is high, a two steps procedure has been adopted:

<u>In step 1</u>, a list of identified available models is collected. These are characterized in terms of three features and these three features are the criteria to select them for step 2:

- *Feature 1*: perspective adopted according to Dewulf et al., 2015. Only models that adopt perspective 1 or perspective 2 are candidates for step 2.

- *Feature 2*: in case they adopt perspective 1 or 2 they are analyzed in terms of covering NARA, PARA1, PARA2 and/or PARA5. In case they cover at least one of these, they are candidates for step 2.

- *Feature 3*: in case one model adopts perspective 1 or 2 and this model covers NARA, PARA1, PARA2 and/or PARA5, the models are evaluated in terms of its level of maturity, i.e. provided with applicable characterization factors.

In addition to this, also models assessing supply risk of resources are included, to be evaluated as potential additional environmental information.

In step 2, all models that fulfil the abovementioned criteria are going to a detailed evaluation following the general criteria adopted for all the impact categories and the specific ones defined for resources and described before.

3.5.1 Pre-selection of models (step 1)

Table 3.2 shows the list of models collected and considered for evaluation in the impact category resources and the related comments about pre-selection, following the approach described before as "step 1".

Table 3.2 List of models collected and considered for evaluation in the impact category resources with evaluation against the criteria for pre-selection

Model	Reference	Feature 1: Perspective	Feature 2: Coverage of resources impact categories	Feature 3: CFs available	Comment	Pre-selected
SED	Rugani et al. 2011	1	Y	Y		**Y**
CExD	Bösch et al. 2007	1	Y	Y		**Y**
CEENE	Dewulf et al. 2007, Alvarenga et al. 2013, Taelman et al. 2014	1	Y	Y		**Y**
CED/PED	VDI 1997	1	Y	Y		**Y**
EF	Global Footprint Network, 2009	1	Y	Y	The approach is more related to land use impact category	N
WF - Water Footprint	Hoekstra et al. 2011, Boulay et al 2015a and 2015b	1	N		Already included in the evaluation of Water impact category	N
AADP 2015	Schneider et al. 2011, Schneider et al. 2015	2	Y	Y		**Y**
ADP-CML Ultimate reserve Reserve base Economic reserve	van Oers 2002	2[2]	Y	Y	Updated version available (CML 2015), with differentiation between elements and fossils	**Y**
EDIP 2003 (rev of EDIP 97)	Hauschild and Wenzel 1998	2	Y	Y		**Y**
ORI	Swart and Dewulf, 2013	2	Y	Y		**Y**
Recipe midpoint - fossils	Goedkoop et al. 2009	2	Y	Y		**Y**
Recipe midpoint - elements	Goedkoop et al. 2009	2	Y	Y		**Y**
LPY-fish (lost potential yield - fish)	Emanuelsson et al. 2014	2	Y	Y		**Y**
BRD-fish (biotic resource depletion - fish)	Langlois et al. 2014	2	Y	Y		**Y**
EcoPoints/Ecoscarcity 2006	Frischknecht et al. 2009	2	Y	Y	Newer version available	**Y**
EcoPoints /Ecoscarcity 2013	Frischknecht and Büsser Knöpfel, 2013	2	Y	Y	Update and revision of Ecoscarcity 2006.	**Y**
Ecoindicator 99	Goedkoop and Spriensma 2001	2	Y	Y	Included even if a newer version available (Recipe), because still in use by some practitioners	**Y**
EPS 2000	Steen 1999	2	Y	Y		**Y**
IMPACT 2002+	Jolliet et al. 2003	2	Y	Y		**Y**
Surplus cost potential	Vieira et al. 2016	2	Y	Y		**Y**

[2] The $ADP_{ultimate\ reserve}$ model can be seen as a bridge between P1 and P2, because the final aim is to assess availability for human use, but the stock considered is the whole amount of natural resources.

Model	Reference	Feature 1: Perspective	Feature 2: Coverage of resources impact categories	Feature 3: CFs available	Comment	Pre-selected
ReCiPe endpoint	Goedkoop et al. 2009	2	Y	Y		**Y**
LIME2	Itsubo and Inaba, 2012	2	Y	Y	Interesting approach, but CFs are specific for Japan	N
BPP	Brandão and Mila I Canals, 2013	2	N		Already included in the evaluation of Land use impact category	N
Abiotic depletion potential water	Mila I Canals et al. 2009	2	N		Already included in the evaluation of Water impact category	N
Supply risk	Chapman et al. 2013	4	Y	Y		Y[3]
Supply risk_JRC	Mancini et al. 2016	4	Y	Y		Y[4]
ESP (Economic Scarcity Potential)	Schneider et al. 2014	N				N
ERP-Erosion Retention Potential	Saad et al. 2013	N			Already included in the evaluation of Land use impact category	N
WPP-Water Purification Potential	Saad et al. 2013	N			Already included in the evaluation of Land use impact category	N
FRP-Freshwater Regulation Potential	Saad et al. 2013	N			Already included in the evaluation of Land use impact category	N

[3] Even if the model reflects P4 instead of P2, it has been included in the pre-selection with the aim to explore the possibility to cover more than one perspective, in response to the need highlighted in section 2.1

Table 3.3 - List of models pre-selected for further evaluation

Model	Version	Indicator	Unit	Reference
CED	-	Cumulative Energy Demand	MJ/unit	VDI 1997, Frischknecht et al., 2007
SED – Solar Energy Demand	-	Solar Energy Factor (SEF)	MJse/unit	Rugani et al., 2011
CExD	-	Cumulative Exergy Demand	MJex/unit	Bösch et al., 2007
CEENE	2014	Cumulative Energy Extracted from the Natural Environment	MJex/unit	Dewulf et al., 2007, Alvarenga et al., 2013, Taelman et al., 2014
ADP-CML	2002	Abiotic Depletion Potential	kg Sb-eq/unit	Guinee et al., 2002; van Oers et al., 2002
ADP-CML - fossils	2012	Abiotic Depletion Potential	kg Sb-eq/MJ	
	2015	Abiotic Depletion Potential	MJ/MJ	
ADP-CML – elements: Ultimate reserve Reserve base Economic reserve	2012	Abiotic Depletion Potential	kg Sb-eq/kg	
	2015	Abiotic Depletion Potential		
AADP	2015	Anthropogenic stock extended Abiotic Depletion Potential (different reference values for resources' stock)	t Sb-eq/t	Schneider et al., 2011, Schneider et al. 2015
ORI	-	Ore Requirement Indicator	$kg_{additional\ ore\ required}/kg_{metal}*year^{-1}$	Swart and Dewulf, 2013
EDIP	2003 (rev of EDIP 97)	Resource consumption	PR/kg	Hauschild and Wenzel, 1998
Recipe midpoint – mineral[4]	version 1.08 Dec. 2012	Mineral depletion	kg Fe eq/kg	Goedkoop et al., 2009
Recipe midpoint – fossil[2]	version 1.08 Dec. 2012	Fossil depletion	kg oil eq/kg	Goedkoop et al., 2009
Supply risk	2013	Supply risk	dimensionless	Chapman et al., 2013
Supply risk_JRC		Supply risk	dimensionless	Mancini et al., 2016
LPY-fish (lost potential yield –fish)	-	Lost Potential Yields (LPY)	dimensionless	Emanuelsson et al., 2014
BRD-fish (biotic resource depletion – fish)	-	1 / maximum sustainable yield (MSY) or 1 / current fish catches (Ct) [to be applied in case of overexploitation]	yr/t[5]	Langlois et al., 2014
Ecoscarcity	2006	Ecopoints (calculated as distance to a target)	UBP/unit	Frischknecht et al., 2008
	2013			Frischknecht et al., 2013
Ecoindicator 99	-	Surplus energy	MJsurplus/kg or MJsurplus/MJ	Goedkoop and Spriensma, 2001
EPS	2000	Environmental Load Units (ELU)	ELU/kg	Steen, 1999
IMPACT 2002+	-	Surplus energy	MJ/kg or MJprimary/MJ	Jolliet, 2003
Surplus cost potential	2016	Surplus cost potential	USD2013/kg	Vieira et al., 2016
Recipe endpoint – mineral[2]	version 1.08 Dec. 2012	Damage to resource cost (RC)	$/kg	Goedkoop et al., 2009
Recipe endpoint – fossil[2]	version 1.08 Dec. 2012	Damage to resource cost (RC)	$/kg	Goedkoop et al. 2009

[4] The model uses increased costs as endpoint indicator and 'the slope (relation grade-yield) divided by availability' as midpoint indicator
[5] MSY is expressed in t/yr, i.e. mass produced every year

3.5.2 Description of pre-selected models

Advanced accounting models:

Cumulative Energy Demand (CED) (VDI 1997, Frischknecht et al., 2007)**.** The model aims to assess the energetic quality of resources, through energy. The Cumulative Energy Demand (CED) indicates the total energy withdrawn from nature to provide a product, summing up the energy of all the resources required. It measures the amount of energy required to provide a process or product. It is also mentioned as the accounting of Primary Energy Demand (PED). Several approaches for the calculation of CED are available. They are described and discussed in Frischknecht et al., 2015.

Solar Energy Demand (SED) (Rugani et al., 2011)**.** The model, based on the emergy concept with some modifications, is aimed at measuring the Solar Energy Demand (SED) of the extraction of atmospheric, biotic, fossil, land, metal, mineral nuclear and water resources. The purpose is to measure the amount of solar energy that would be needed to replace the resource that is extracted from the environment. SED does not account for energy available for human use after extraction. The model measures the flow of solar energy in the transformations occurred in the formation of the resource, before its extraction. It defines resources having a turnover time of less than year as renewable, whereas resources having a turnover time over one year as non-renewable. The main difference between SED and emergy is that emergy do not allow for allocation, whereas SED includes allocation between coproducts.

Cumulative Exergy Demand (CExD) (Bösch et al., 2007)**.** The model aims to assess the energetic quality of resources, through exergy. Exergy is a measure of the minimal work necessary to form the resource or the maximally obtainable amount of work when bringing the resource's components to their most common state in the natural environment. The Cumulative Exergy Demand (CExD) indicates the total exergy removal from nature to provide a product, summing up the exergy of all the resources required. The model aims to assess the consumption of exergy (through the production of entropy) due to the extraction of resources from nature to human systems. It measures the amount of exergy required to provide a process or product.

Cumulative Exergy Extracted from the Natural Environment (CEENE) (Dewulf et al., 2007, Alvarenga et al., 2013, Taelman et al., 2014)**.** The model aims to assess the energetic quality of resources, through exergy. Exergy is a measure of the minimal work necessary to form the resource or the maximally obtainable amount of work when bringing the resource's components to their most common state in the natural environment. The Cumulative Exergy Extracted from the Natural Environment (CEENE) indicates the total exergy deprived from nature to provide a product, summing up the exergy of all the resources required. The model is able to cover resources such as: fossils, metals and minerals, hydropower and geothermal energy sources, land, water and natural biomass.

Models based on the abiotic depletion model:

ADP-CML (Guinee et al. 2002; van Oers et al. 2002)**.**

The model is based on use-to-availability ratio. The remaining availabilities (economic reserves/reserve base/ultimate reserves) are squared in order to take into account that extracting 1 kg from a larger resource is not equivalent to extracting 1 kg from a small resource, even if the use-to-resource ratio is the same. The original model developed by Guinée. (1995) includes only the ultimate stock reserves as reference stock. In the Ultimate reserves version, the reference stock is the quantity of a resource (like a chemical element or compound) that is ultimately available. It is estimated by multiplying the average natural concentration of the resource in the primary extraction media (e.g., the earth’s crust) by the mass or volume of these media (e.g., the mass of the crust assuming a depth of e.g., 10 km). The model includes non-renewable resources (fossil fuels and minerals). This is the case also in Guinée et al. (2002). In Oers et al. (2002), additional characterisation factors have been listed on the basis of USGS economic reserve and reserve base figures beyond those of ultimate reserve.

The characterisation factors are named "abiotic depletion potentials" (ADP) and expressed in kg of antimony equivalent, which is the adopted reference element. The abiotic depletion potential is calculated for elements and, in the case of economic reserves and reserve base, several mineral compounds. Since 2009 version, ADP is distinguished in $ADP_{elements}$ and $ADP_{fossil\ fuels}$. CFs for fossil fuels are no more expressed in kg antimony equivalents (kg Sbeq) per unit (kg, m3, MJ) of resource but as MJ/MJ, i.e. the CF is equal to 1 for all fossil resources[6].

Several updates of the characterization factors has been released: a complete documentation on updates can be found at http://cml.leiden.edu/software/data-cmlia.html. The version considered for the evaluation is the one released April 2015.

The authors of Recipe model, who have been also involved in the development of CML 2000 and Ecoindicator 99, present Recipe as an improvement of CML 2000 and Ecoindicator 99.

EDIP 97 and 2003 (Hauschild and Wenzel 1998)**.** The EDIP 97 model defines the weighting factor for resource consumption as the reciprocal of the supply horizon for the resource, calculated based on consumption, computed reserves and any rate of regeneration in the reference year 1990. For renewable resources, the weighting factor is defined based on the total consumption where the process is occurring. If the resource is not used faster than it is regenerated, the supply horizon is infinite and the weighting factor is therefore zero. For non-renewable resources, this definition of the weighting factor means that consumption, R(j), of resource (j) in the product system is compared against reserves of the resource in question at the weighting. This model was updated in 2004. In the new version, non-renewable resources (fossil fuels and minerals) are included. The amount of the resource extracted is divided by the 2003 global production of the resource and weighted according to the quantity of the resources in economically-exploitable reserves. Effectively, this means that the global annual production drops out, so that the characterisation model is based on the economic reserves only. The characterisation factors are expressed in person-reserve, meaning the quantity of the resource available to an average world citizen.

The anthropogenic stock extended abiotic depletion potential (AADP) (Schneider et al. 2011, Schneider et al. 2015)**.** It is a modification of the original ADP model in two ways: (1) it takes as reference stock an estimation of ultimately extractable reserves as a percentage of ultimate reserves instead of (ultimate) reserves as such (crustal content); (2) it adds the stock available in the anthroposphere. The model follows the same calculation principle of ADP, i.e. it makes the ratio of extraction rates versus stocks (in this case, equal to anthropogenic + natural resources; squared), relative to a reference compound (Sb).

Models taking into account the variation of ore grade over time:

Recipe 2008 (fossils and elements) – midpoint and endpoint (Goedkoop et al. 2009)**.** The model propose two different sets of characterization factors (at the midpoint and at the endpoint) for resources: one for mineral resources and the other for fossil resources. In the description of the Area of protection natural resources, the damage is defined as the additional net present costs that society has to pay as a result of an extraction. These are the costs incurred due to the fact that, after the extraction of the "best" (highest grade) resources, future mining becomes more expensive. This cost can be calculated by multiplying the marginal cost increase of a resource by an amount that is extracted during a certain period.

[6] In case the elementary flow is expressed in mass, an additional factor need to be calculated, related to the energy content of the mass considered (e.g. ADP for "oil crude (41.87 MJ/kg)" is 41.87 [MJ/kg]).

In this cost calculation, a depreciation rate of 3% is chosen. Current technology is assumed to determine the costs. For minerals, the model focuses on the depletion of deposits, instead of individual commodities. It uses increased costs as endpoint indicator and 'the slope (relation grade-yield) divided by availability' as midpoint indicator. For fossil fuels, the marginal increase of oil production costs (due to the need to mine non-conventional oils) is used. Characterization factors at the endpoint and midpoint are calculated as for mineral resources.

The authors of Recipe model, who have been also involved in the development of CML 2002 and Ecoindicator 99, present Recipe as an improvement on CML 2002 and Ecoindicator 99.

Ecoindicator 99 (Goedkoop and Spriensma 2001). The endpoint characterisation factor is expressed as Surplus Energy. This expresses the additional energy requirements due to mining resources with a decreased grade at some point in the future. This point is arbitrarily chosen as the time mankind has mined 5 times the historical extraction up to 1990. Current technology (for the time the model was developed) is assumed. This model includes non-renewable resources (fossil fuels and minerals). The model calculates the depletion of elements, not of ores.

The authors of Recipe model, who have been also involved in the development of CML 2002 and Ecoindicator 99, present Recipe as an improvement on CML 2002 and Ecoindicator 99.

IMPACT 2002+ (Jolliet 2003). It is based on the surplus energy concept (future scenario), using Ecoindicator 99, egalitarian as source model and factors. An infinite time horizon for fossil energy is assumed. This implies that the total energy content of the fossil energy are lost due to their consumption; hence, damage is quantified simply by the energy content. For resources, the model adopt the same model as Ecoindicator 99 (endpoint indicator as surplus energy required for extraction of an additional unit of resource), but using the egalitarian scenario instead of the hierarchist one, which is suggested as default by Ecoindicator developers. This model includes non-renewable resources (fossil fuels and minerals).

Ore requirement indicator (ORI) (Swart and Dewulf, 2013). The Ore Requirement Indicator is a model that relies on the annual increase in ore requirements as a function of declining ore grades due to present mining activities. The model characterizes the average annual increase of ore required per kg metal. It relies on a database with a substantial worldwide coverage of mining over the period 1998-2010.

Surplus cost potential (Vieira et al. 2016). The model calculates the surplus cost potential (SCP) of mining and milling activities. Main differences from similar models (e.g. ORI) are: 1) all future metal extractions are considered, via cumulative cost-tonnage relationships 2) the operating mining costs account for co-production and are allocated across all mine products in proportion to the revenue that they provide. As ORI, the model is based on the assumption that mines with lower operating costs are explored first. Therefore, increased primary metal extraction results in a subsequent increase in mining and milling costs. These costs are assumed as measure of depletion.

Models based on the Distance-to-target concept:

Ecoscarcity 2006 and 2013 (Frischknecht et al., 2008, Frischknecht et al., 2013)**.** Distance-to-target methodology developed based on the Swiss context. The model measures the current resource use versus 2030 policy targets. The Ecofactors are derived on the basis of policy targets (2030) versus current resource flows, expressed in units of eco-points. Resources impact category is modelled based on targets for 2030, with characterization done in 2006 and updated in 2013. In case of minerals and metals, the 2013 version uses the characterisation model of $ADP_{reserves}$ with updated data on reserves and production. The Ecofactors are applied to dissipative use of resources, which is derived as the difference between the amounts of resources extracted and recycled, i.e. the aggregated amount lost during manufacture, use and end-of-life treatment (Vadenbo et al., 2014).

Model based on Willingness to Pay:

Environmental Priority Strategies in product design (EPS) (Steen, 1999)**.** The model consists of weighting factors obtained by applying monetisation to environmental impacts of production. It is based on willingness to pay (WTP) for restoring damage done to the safe-guard subject**.** This model includes non-renewable resources (fossil fuels and minerals) and renewable resources (water, fish, meat and wood). The amount of resource depleted by the system under study is directly normalized and weighted using monetization. Weighting factors are expressed in Environmental Load Units (ELU) per unit of resource and are the sum of direct and indirect (i.e. environmental) costs for obtaining one unit of the resource in the reference system. The reference system is defined considering the optimized sustainable alternative to the current production route for each resource. Each weighting factor is reported with the related uncertainty.

Models only for biotic resources:

Lost Potential Yield (LPY) for fish (Emanuelsson et al., 2014)**.** The model aims at the quantification of overfishing by comparing the current with target fisheries management by the Lost Potential Yield (LPY). It relies on simplified biomass projections to assess the lost catches due to ongoing overfishing.

The model starts from the impact of the current (over)fishing practice on the future overall fish landings (e.g. 30 years period) and compares it to the potential optimal yield based on the maximum sustainable yield concept. In this way, it characterizes the current (over)exploitation of the natural resource versus the optimal exploitation and it comes up with a midpoint indicator that characterizes the impact on (reduction in) future provisioning.

Biotic Resource Depletion (BRD) for fish (Langlois et al., 2014). The model aims to characterise the impacts on biotic natural resources at (fish) species level. It characterizes the current mass caught with the maximum sustainable yield for sustainably fished stocks and with the actual (last 5 years) catches for depleted or overexploited stocks.

The model relates the mass caught in relation to either the maximum sustainable yield (MSY, based on fisheries science) or the current fish catches (Ct) in case of overexploitation.

Models accounting for criticality of resources:

Supply risk (Chapman et al., 2013)**.** The model defines the level of criticality of resources, considering the environmental dimension (e.g. aspects like depletion of reserves, recyclability, overuse of ecosystems), the economic dimension (e.g. concentration of supply, import dependency, etc) and the socio-political dimension (e.g. human rights violations, resource conflicts, illicit trade, precarious working conditions). The background framework is based on the concept of availability of resources for human use. A threshold is set for each of the two variables which characterize any raw material (EI and SRWGI), and the combination of the two leads to the definition of the "criticality area". If a raw material is characterized by values of EI and SRWGI higher than the thresholds, it is then to be considered as critical.

Supply risk_JRC (Mancini et al., 2016)**.** It is an elaboration of the supply risk model, aimed at adapting it for the use in LCIA. The model applies an exponent to the criticality factors (supply risk value, SRWGI) identified by Chapman et al. (2013), with the aim to magnify their effect in LCIA and, then, to highlight the use of critical raw materials in the supply chain, even in small quantities.

3.5.3 Characterization factors at midpoint

For the pre-selected models, all the characterization factors available were collected and evaluated. Background documentation for each model as they are published (e.g. with their own elementary flows) was also collected and, when needed, a mapping of the characterization factor to ILCD elementary flow list was performed. Name correspondence was the first criterion followed in the process of mapping the original CFs to the ILCD elementary flows. For models providing just one value for the aggregated flow "heavy and light rare earths" or "rare earths" or "Platinum Group Materials", the value was attributed to the different minerals in the group, when needed differentiated according to their characteristics (heavy or light). Uranium is considered as an energy carrier in the ILCD and its flows is expressed in MJ. When in the original model the CF for Uranium was referred to an elementary flow in mass (kg), a conversion factor of 544284 MJ/kg was applied.

There is a quite high variability of CFs' characteristics among the models considered. Some models are able to cover a wide range of resources, e.g. the one by Langlois et al (2014), that provides CFs for 127 fish resources, SED (92 abiotic resources), CExD (82 abiotic resources). On average, most of the models are able to cover between 20 and 50 different resources (mostly abiotic). The models with the lowest coverage are ORI (9 mineral resources), and the Surplus Cost Potential (13 mineral resources). CED is able to cover 13 energy carriers out of 14, but of course, its ability to cover resource flows in ILCD is limited to energy resources.

On the other hand, some of the models that cover the highest number of resources show a quite limited range of values assigned to them, i.e. a limited capacity to discriminate and rank resources when characterizing them in the LCIA phase. E.g., the supply risk set of values by Chapman et al. 2013 covers a good number of resources, but has a quite limited span (1 order of magnitude between the minimum and maximum values of CFs). However, it has to be remembered that this model was not developed to be applied in LCIA. On the contrary, the set proposed by Mancini et al (2016) and named as "supply risk_JRC" was specifically developed for use in LCIA, starting from the original model of supply risk. Hence, it ensure a good coverage of resources (60) and a wide range of values for CFs (10 orders of magnitude). Again, the CFs coming from the SED model show one of the widest range of values for CFs (10 orders of magnitude).

Regarding the applicability in the ILCD framework, almost all the models can be easily adapted to the ILCD set of elementary flows and related nomenclature, except from the two models specifically referring to fish resources, because currently there is no elementary flow related to fish in the ILCD set. This is also a more general problem, because most of the LCIs available for free or by purchase do not account for fish resources' use.

Apart from these two models, the coverage of the ILCD set of elementary flows for resources varies among the models considered. On average, the models considered are able to cover about 25% of the ILCD flows. As expected, the ones with the lowest coverage are the ones with the smallest set of CFs (ORI, Surplus Cost Potential and CED). On the other hand, due to some differences between the ILCD list of flows and the list of resources considered in some models, there are some models such as ADP and AADP, with an average number of resources covered, that have a quite good score in terms of ILCD flows coverage (respectively, 34% and 35%). These numbers need to be interpreted in light of the coverage of the current ILCD recommendation (around 46%) and the fact that the list of ILCD flows for resources is very large (it includes 157 elementary flows).

In order to compare the CFs values of models using different approaches and different units, the CFs of each model were normalized over copper, to show the relative ranking of resources given by each model (i.e. the higher impact potential assigned to resources with higher CFs). The results show that resources ranked amongst the first 20 positions are to some extent common in many of the models (e.g. Germanium, Rhenium, Platinum are considered relevant in ADP-based models and Recipe), whereas other models adopting a totally different approach like emergy and exergy rank first resources that for most of the other models are at the end of the list (e.g. Cinnabar and Rhodium). Finally, as expected, Supply risk models give highest importance to resources of the Rare Earth Elements group.

A correlation analysis was performed to verify to what extent models applying the same approach lead to similar results in terms of characterization. Table 3.4 illustrates the correlation among CFs of most of the models evaluated. CEENE and Supply risk models have very low correlation with other models (of course excluding the natural correlation between the two models relying on the same supply risk assessment). In general, models for advanced accounting show a low level of correlation among themselves. A quite high level of correlation is shown by models applying the surplus energy or cost approach (Ecoindicator 99, IMPACT 2002+, Recipe and Surplus Cost Potential). Surprisingly, the Surplus Cost Potential model shows a correlation of 1 with ADP-CML based on ultimate resources. On the other hand, the level of correlation among the ADP-based models is quite low, probably because of the influence of the different assumptions used in the calculation of stock availability.

A linear correlation analysis based on Pearson coefficient was performed with the aim of assessing similarities and differences amongst models and verifying to what extent models applying the same approach lead to similar results in terms of characterization. The results are shown in Table 3.5. For the models presenting different versions the most recent one has been evaluated in this analysis.

The highest positive correlation emerges among the models reflecting the second perspective. ILCD, AADP, EcoPoints and ADP-CML 2015 economic/reserve base show significant correlation scores among them (correlation coefficients ranges from 0.6 to 1).

On the other hand, models for advanced accounting show a different pattern, despite they all refer to perspective 1: CExD and SED are negatively correlated while almost no correlation is registered between CEENE and CExD/SED. In addition, SED appears to be well correlated to Perspective 2 models, like ILCD (0.61), AADP (0.98), CML economic (0.97) and Ecopoints (0.98). In a similar way CExD presents a very high correlation scores, i.e. 0.7-0.9, with EDIP 97, SRwgi/P, ReCiPe and EPS whereas CEENE is poorly or negatively correlated with all the other indicators, presenting part of the most significant negative values. Similar trends are evident also for SRwgi^6, EI99 and IMPACT 2002.

A different pattern is shown by models applying the surplus energy or cost approach: the correlation is quite high between Ecoindicator 99 and IMPACT 2002+ and between Recipe and Surplus Cost Potential; in spite of that, any different combination of these models shows negative correlation.

Finally, concerning models based on criticality approach, Supply Risk presents a positive correlation with EDIP 97, EPS and Surplus cost potential, while its correlation with ADP-CML ultimate shows the highest negative score in the table (-0.35).

Table 3.4 Correlation among the characterisation factors of a selection of the models evaluated

	ILCD	SED	CExD original	CEENE 2014	AADP 2015	CML 2015 ultimate	CML 2015 reserve	CML 2015 economic	EDIP 97	Supply Risk (SR)	SR $_{WGI}$^6	ReCiPe 2008 Midpoint	EcoPoints 2013	EI99	EPS 2000	IMPACT 2002+	Surplus cost potential	ReCiPe 2008 Endpoint
ILCD	1.00	0.61	0.24	-0.16	0.61	0.71	1.00	-0.17	0.67	-0.09	-0.13	0.52	0.70	0.86	0.55	0.86	0.84	0.52
SED	0.61	1.00	-0.14	0.05	0.98	-0.05	0.61	0.97	-0.03	0.13	-0.11	0.07	0.98	0.93	0.51	0.93	0.17	0.07
CExD original	0.24	-0.14	1.00	-0.15	-0.13	0.89	0.24	-0.11	0.97	0.47	0.04	0.82	-0.10	0.84	0.76	0.84	0.94	0.82
CEENE 2014	-0.16	0.05	-0.15	1.00	0.07	-0.24	-0.16	0.03	-0.18	-0.28	-0.20	-0.18	0.04	-0.25	-0.04	-0.25	-0.33	-0.18
AADP 2015	0.61	0.98	-0.13	0.07	1.00	-0.07	0.61	0.99	0.94	0.12	-0.08	0.67	0.99	0.16	0.53	0.16	0.85	0.67
CML 2015 ultimate	0.71	-0.05	0.89	-0.24	-0.07	1.00	0.71	0.08	0.42	-0.35	-0.11	0.37	0.06	0.66	-0.01	0.66	1.00	0.37
CML 2015 reserve	1.00	0.61	0.24	-0.16	0.61	0.71	1.00	0.71	0.67	-0.09	-0.13	0.52	0.70	0.86	0.55	0.86	0.84	0.52
CML 2015 economic	0.71	0.97	-0.11	0.03	0.99	0.08	0.71	1.00	0.53	0.07	-0.10	0.43	1.00	0.42	0.54	0.42	0.84	0.43
EDIP 97	0.67	-0.03	0.97	-0.18	0.94	0.42	0.67	0.53	1.00	0.26	0.00	0.70	0.59	0.82	0.92	0.82	0.82	0.70
Supply Risk (SR)	-0.09	0.13	0.47	-0.28	0.12	-0.35	-0.09	0.07	0.26	1.00	0.71	0.16	0.08	0.35	0.42	0.36	0.74	0.16
SR $_{WGI}$^6	-0.13	-0.11	0.04	-0.20	-0.08	-0.11	-0.13	-0.10	0.00	0.71	1.00	-0.01	-0.10	0.05	-0.01	0.05	0.84	-0.01
ReCiPe 2008	0.52	0.07	0.82	-0.18	0.67	0.37	0.52	0.43	0.70	0.16	-0.01	1.00	0.47	0.99	0.65	0.99	0.96	1.00
EcoPoints 2013	0.70	0.98	-0.10	0.04	0.99	0.06	0.70	1.00	0.59	0.08	-0.10	0.47	1.00	0.98	0.55	0.98	0.83	0.47
EI99	0.86	0.93	0.84	-0.25	0.16	0.66	0.86	0.42	0.82	0.35	0.05	0.99	0.98	1.00	0.44	1.00	-0.23	0.99
EPS 2000	0.55	0.51	0.76	-0.04	0.53	-0.01	0.55	0.54	0.92	0.42	-0.01	0.65	0.55	0.44	1.00	0.44	0.85	0.65
IMPACT 2002+	0.86	0.93	0.84	-0.25	0.16	0.66	0.86	0.42	0.82	0.36	0.05	0.99	0.98	1.00	0.44	1.00	-0.28	0.99
Surplus cost potential	0.84	0.17	0.94	-0.33	0.85	1.00	0.84	0.84	0.82	0.74	0.84	0.96	0.83	-0.23	0.85	-0.28	1.00	0.96
ReCiPe 2008 Endpoint	0.52	0.07	0.82	-0.18	0.67	0.37	0.52	0.43	0.70	0.16	-0.01	1.00	0.47	0.99	0.65	0.99	0.96	1.00

Table 3.5 Correlation analysis between models' characterization factors based on Pearson coefficient.

	ILCD	SED	CExD	CEENE 2014	AADP 2015	CML 2015 (ultimate)	CML 2015 (reserve base)	CML 2015 (economic)	EDIP 97	Supply Risk (SR)	SR WGI/P	SR WGI^6	ReCiPe 2008	EcoPoints 2013	EI99	EPS 2000	IMPACT 2002+	Surplus cost potential	ReCiPe 2008
ILCD	1.00	0.61	0.10	-0.16	0.61	0.71	1.00	0.71	0.44	-0.09	0.19	-0.13	0.35	0.70	-0.17	0.54	-0.17	0.09	0.35
SED	0.61	1.00	-0.12	0.05	0.98	-0.05	0.61	0.97	-0.15	0.13	-0.14	-0.11	-0.09	0.98	0.08	0.51	0.08	-0.10	-0.09
CExD	0.10	-0.12	1.00	-0.09	-0.11	-0.07	0.10	-0.10	0.84	0.44	0.97	0.06	0.73	-0.10	-0.12	0.76	-0.12	0.95	0.73
CEENE 2014	-0.16	0.05	-0.09	1.00	0.07	-0.24	-0.16	0.03	-0.19	-0.28	-0.11	-0.20	-0.19	0.04	-0.16	-0.05	-0.17	-0.09	-0.19
AADP 2015	0.61	0.98	-0.11	0.07	1.00	-0.07	0.61	0.99	-0.14	0.12	-0.12	-0.08	-0.10	0.99	-0.09	0.53	-0.09	-0.10	-0.10
CML 2015 (ultimate)	0.71	-0.05	-0.07	-0.24	-0.07	1.00	0.71	0.08	0.42	-0.35	0.04	-0.11	0.37	0.06	-0.10	-0.01	-0.10	-0.06	0.37
CML 2015 (reserve base)	1.00	0.61	0.10	-0.16	0.61	0.71	1.00	0.71	0.44	-0.09	0.19	-0.13	0.35	0.70	-0.17	0.54	-0.17	0.09	0.35
CML 2015 (economic)	0.71	0.97	-0.10	0.03	0.99	0.08	0.71	1.00	-0.06	0.07	-0.10	-0.10	-0.04	1.00	-0.11	0.54	-0.11	-0.09	-0.04
EDIP 97	0.44	-0.15	0.84	-0.19	-0.14	0.42	0.44	-0.06	1.00	0.24	0.92	0.01	0.70	-0.07	-0.17	0.70	-0.17	0.73	0.70
Supply Risk (SR)	-0.09	0.13	0.44	-0.28	0.12	-0.35	-0.09	0.07	0.24	1.00	0.42	0.71	0.15	0.08	0.04	0.42	0.05	0.39	0.15
SR WGI/P	0.19	-0.14	0.97	-0.11	-0.12	0.04	0.19	-0.10	0.92	0.42	1.00	0.06	0.66	-0.09	-0.14	0.78	-0.14	0.86	0.66
SR WGI^6	-0.13	-0.11	0.06	-0.20	-0.08	-0.11	-0.13	-0.10	0.01	0.71	0.06	1.00	0.00	-0.10	-0.11	-0.01	-0.11	0.06	0.00
ReCiPe 2008	0.35	-0.09	0.73	-0.19	-0.10	0.37	0.35	-0.04	0.70	0.15	0.66	0.00	1.00	-0.05	-0.12	0.49	-0.12	0.87	1.00
EcoPoints 2013	0.70	0.98	-0.10	0.04	0.99	0.06	0.70	1.00	-0.07	0.08	-0.09	-0.10	-0.05	1.00	-0.11	0.55	-0.11	-0.08	-0.05
EI99	-0.17	0.08	-0.12	-0.16	-0.09	-0.10	-0.17	-0.11	-0.17	0.04	-0.14	-0.11	-0.12	-0.11	1.00	-0.18	1.00	-0.12	-0.12
EPS 2000	0.54	0.51	0.76	-0.05	0.53	-0.01	0.54	0.54	0.70	0.42	0.78	-0.01	0.49	0.55	-0.18	1.00	-0.18	0.67	0.49
IMPACT 2002+	-0.17	0.08	-0.12	-0.17	-0.09	-0.10	-0.17	-0.11	-0.17	0.05	-0.14	-0.11	-0.12	-0.11	1.00	-0.18	1.00	-0.12	-0.12
Surplus cost potential	0.09	-0.10	0.95	-0.09	-0.10	-0.06	0.09	-0.09	0.73	0.39	0.86	0.06	0.87	-0.08	-0.12	0.67	-0.12	1.00	0.87
ReCiPe 2008	0.35	-0.09	0.73	-0.19	-0.10	0.37	0.35	-0.04	0.70	0.15	0.66	0.00	1.00	-0.05	-0.12	0.49	-0.12	0.87	1.00

3.6 Models evaluation

The table below shows the summary results of the land use models evaluation. The complete evaluation can be found in Annex 3.1

		CED		SED		CExD	
Science-based criteria	Summary information (descriptive)	The model aims to assess the use of energy due to the extraction of resources from nature to human systems. The **Cumulative Energy Demand (CED) indicates the total energy withdrawn from nature** to provide a product, summing up the energy of all the resources required.		The model, based on the emergy concept with some modifications, is aimed at measuring the **Solar Energy Demand (SED) of the extraction of atmospheric, biotic, fossil, land, metal, mineral nuclear and water resources**. The purpose is to measure the amount of solar energy that would be needed to replace the resource that is extracted from the environment. SED does not account for energy available for human use after extraction.		The model aims to assess the **energetic quality of resources**, through exergy. Exergy is a measure of the minimal work necessary to form the resource or the maximally obtainable amount of work when bringing the resource's components to their most common state in the natural environment. The **Cumulative Exergy Demand (CExD) indicates the total exergy removal from nature** to provide a product, summing up the exergy of all the resources required.	
	Completeness of the scope	C	Within its limited scope (accounting of resources for energy provision), it can be useful for depicting the use of fossil and renewable energy carriers in light of energy-related emissions reduction. No geographical resolution.	C	As an advanced accounting method (early stage in cause and effect chain), SED is quite complete; no geographical resolution.	C-B	As an advanced accounting method (early stage in cause and effect chain), CExD is quite complete but misses land as a resource and geographical resolution.
	Relevance to the envisaged Area(s) of Protection	C-B	Limited coverage of the types of resources. Good coverage if the scope of the assessment is accounting of (fossil and renewable) energy use.	A-B	Highly relevant to the envisaged advanced accounting; very good coverage of elementary flows.	C-B	Relevant to the envisaged advanced accounting; relative good coverage of elementary flows.
	Scientific robustness & Uncertainty	B-C	Peer reviewed method; quite well recognized by the scientific community. Uncertainties quite well known but not quantified.	C	Peer reviewed method; but not recognized as the most solid scientific method in function of elementary flows. Uncertainties not quantified.	B-C	Peer reviewed method; quite well recognized by the scientific community. Uncertainties described; not quantified.
	Documentation, Transparency & Reproducibility	A-B	Quite well documented. Data and model assumptions are accessible.	A	Well documented. Data and model assumptions are accessible.	B	Well documented; accessible except Suppl. Info.
	Applicability	B	Already in use in LCIA. Easily applicable in ILCD. Normalisation factors not available	B	Already in use in LCIA. Easily applicable in ILCD. Normalisation factors not available	B	Already in use in LCIA. Easily applicable in ILCD. Normalisation factors not available
	Characterization factors	B	Characterization factors relevant and usable; not specifically elaborated for natural biomass.	B	Characterization factors relevant and usable; not specifically elaborated for natural biomass.	B	Characterization factors relevant and usable; not specifically elaborated for natural biomass.
Overall evaluation of science-based criteria		B	Accounting method covering only energy carriers. Good model if used to account for energy use from non-renewable and renewable sources.	B	Model is quite complete; good coverage of elementary flows, robustness of accounting method in function of elementary flows may be questioned.	B	Proper advanced accounting method but misses land as a resource.
Overall evaluation of stakeholders acceptance criteria		C	In function of accounting for natural resources with limitation to energy carrying resources, quite well accepted.	C	In function of accounting for natural resources, limited acceptance of the solar energy demand.	C	In function of accounting for natural resources, quite well accepted. Model not easy to understand; uncertainties not clear
Final evaluation		B-C	Simple accounting method, suitable for (renewable and non-rnewable) energy accounting.	B	Reasonably well accounting method, but with some scientific aspects under discussion.	B	Reasonably well accounting method, but with incomplete natural resource asset coverage.

		CEENE		ADP elements - ultimate reserve/crustal content		ADP elements reserve base/mineral reserve	
Science-based criteria	**Summary information (descriptive)**	The model aims to assess the **energetic quality of resources**, through exergy. Exergy is a measure of the minimal work necessary to form the resource or the maximally obtainable amount of work when bringing the resource's components to their most common state in the natural environment. The **Cumulative Exergy Extracted from the Natural Environment (CEENE) indicates the total exergy deprived from nature** to provide a product, summing up the exergy of all the resources required.		Based on use-to-availability ratio; the remaining availabilities (economic reserves/reserve base/ultimate reserves) are squared in order to take into account that extracting 1 kg from a larger resource is not equivalent to extracting 1 kg from a small resource, even if the use-to-resource ratio is the same. In the ULTIMATE RESERVES version, the reference stock is the quantity of a resource (like a chemical element or compound) that is ultimately available. It is estimated by multiplying the average natural concentration of the resource in the primary extraction media (e.g., the earth's crust) by the mass or volume of these media (e.g., the mass of the crust assuming a depth of e.g., 10 km)		Based on use-to-availability ratio; the remaining availabilities (economic reserves/reserve base/ultimate reserves) are squared in order to take into account that extracting 1 kg from a larger resource is not equivalent to extracting 1 kg from a small resource, even if the use-to-resource ratio is the same. In the RESERVE BASE version, the reference stock includes that part of an identified resource that meets specified minimum physical and chemical criteria relating to current mining practice.	
	Completeness of the scope	B	CEENE is very complete, and foresees geographical resolution for land as a resource. However, as an advanced accounting method refers to the early stage of the cause and effect chain, not assessing impacts at the midpoint)	C	Quite good consistency with the scope of the AoP. However, it ignores the provisioning capacity of the anthropogenic stock in the technosphere	C	Quite good consistency with the scope of the AoP. However, it ignores the provisioning capacity of the anthropogenic stock in the technosphere
	Relevance to the envisaged Area(s) of Protection	A-B	Highly relevant to the envisaged advanced accounting; good coverage of elementary flows.	C	Is a relevant way to express reduction on a total stock.	D	Limited relevance to the AoP due to its use of the Reserve Base - an economic measure of no relevance to resources in nature.
	Scientific robustness & Uncertainty	B-C	Peer reviewed method; quite well recognized by the scientific community. Uncertainties described; not quantified.	B-C	Only original model published in peer-reviewed papers (not CFs). Ultimate Reserve data published in leading scientific literature for several decades.	D	Only original model published in peer-reviewed papers (not CFs). Apart from the uncertainties on the use rate, the weakness of the method is the availability estimate.
	Documentation, Transparency & Reproducibility	A	Well documented. Data and model assumptions are accessible.	B	Well documented; but diffidulties in accessing input data may lead to weaker reproducibliity.	B	Well documented; but diffidulties in accessing input data may lead to weaker reproducibliity.
	Applicability	B	Already in use in LCIA. Easily applicable in ILCD. Normalisation factors not available	A	Already in use in LCIA. Normalisation factors available	A	Already in use in LCIA. Already recommended in ILCD. Normalisation factors available
	Characterization factors	B	Characterization factors relevant and usable.	B	Characterization factors almost relevant and usable.	C	Quite high relevance of CFs. The Reserve Base data no longer being generated by the USGS is a significant issue for updates and therefore relevance.
Overall evaluation of science-based criteria		A-B	Proper advanced accounting method among those termodynamic-based with coverage of all non-abundant natural resource assets.	B-C	Suffers from weaknesses inherent to the concept (estimates of availabilities) and overestimates impact by failing to reflect actual depletion. However, its reproducability, robustness of data and long period of use in LCIA make it advantageous.	D	Suffers from weaknesses inherently to the concept (estimates of availabilities), requires updates from a no longer available dataset and loses connection with the AoP by using economic data.
Overall evaluation of stakeholders acceptance criteria		C	In function of accounting for natural resources, quite ~~well~~ accepted. Model not easy to understand; uncertainties not clear	B	Most agreed upon method currently available.	C	Reserves as a base is controversial.
Final evaluation		A-B	Most complete advanced accounting method.	B	Method has high stakeholder acceptance and the lowest uncertainty of those available, but does not reflect actual depletion and needs improvement.	C	Method has an inherent controversial base, only for reasons of continuity to some extent recommendable. Use of Reserve Base presents significant data gap.

		ADP elements - Economic reserve		ADP fossils		AADP 2015	
Science-based criteria	**Summary information (descriptive)**	Based on use-to-availability ratio; the remaining availabilities (economic reserves/reserve base/ultimate reserves) are squared in order to take into account that extracting 1 kg from a larger resource is not equivalent to extracting 1 kg from a small resource, even if the use-to-resource ratio is the same. In the RESERVE BASE version, the reference stock includes the part of the natural reserve base which can be economically extracted at the time of determination.		Based on use-to-availability ratio; the remaining available quantity (economic reserves/reserve base/ultimate reserves) is squared in order to take into account that extracting 1 MJ from a larger resource is not equivalent to extracting 1 MJ from a small resource, even if the availability-to-use ratio is the same.		The anthropogenic stock extended abiotic depletion potential (AADP). It is a modification of the original ADP method in two ways: (1) it takes the resources in nature (economically extractable today) instead of (ultimate) reserves; (2) it adds the stock we have in the anthroposphere.	
	Completeness of the scope	C	Quite good consistency with the scope of the AoP. However, it ignores the provisioning capacity of the anthropogenic stock in the technosphere	B	Reflects the impact on decreasing provisioning capacity quite well.	B	Quite good consistency with the scope of the AoP; a specific fature is that anthropogenic stocks are considered.
	Relevance to the envisaged Area(s) of Protection	D	Limited relevance to the AoP due to its use of the Economic reserve - an economic measure of no relevance to resources in nature.	C	Is a relevant way to express the impact on the AoP by use of fossils.	D	Is a relevant way to express reduction on a total stock, but uses a stock estimate that is highly uncertain.
	Scientific robustness & Uncertainty	D	Only original model published in peer-reviewed papers (not CFs). Apart from the uncertainties on the use rate, the weakness of the method is the availability estimate.	C	Only original model published in peer-reviewed papers (not CFs). Apart from the changing use rate, the weakness of the method is in the availability estimate.	C-D	Published in peer-reviewed paper. Estimates of Ultimately Extractable Reserve are highly uncertain and not accepted in the geological community.
	Documentation, Transparency & Reproducibility	B	Well documented; but diffidulties in accessing input data may lead to weaker reproduciblity.	B	Well documented; but diffidulties in accessing input data may lead to weaker reproduciblity.	C	Quite well documented; but diffidulties in accessing input data may lead to weaker reproduciblity.
	Applicability	A	Already in use in LCIA. Normalisation factors available	A	Already in use in LCIA. Already recommended in ILCD. Normalisation factors available	B	Applicable in LCIA. Easily applicable in ILCD. Normalisation factors not available.
	Characterization factors	C	Quite high relevance of CFs. The Reserve Base data no longer being generated by the USGS is a significant issue for updates and therefore relevance.	B-C	Quite high relevance of CFs. The need to be periodically updated is a weakness	C	Rather high relevance and usability. Quite limited set of CFs. The need to be periodically updated is a weakness
Overall evaluation of science-based criteria		D	Suffers from weaknesses inherently to the concept (estimates of availabilities), requires updates from a no longer available dataset and loses connection with the AoP by using economic data.	B	Well reflecting the decreasing resource availability.	C-D	Suffers from weaknesses inherently to the concept (estimates of availabilities). Although this is an Interesting effort to account for resources available in the technosphere, its estimation of Ultimately Extractable Reserves is not robust and increases the uncertainty of the result and therefore CF's.
Overall evaluation of stakeholders acceptance criteria		C	Reserves as a base is controversial.	C	Exhaustion of available stocks is controversial.	C	Rather recent method with limited exposure/feedback; resources/reserves as a base stays controversial.
Final evaluation		C	Method has an inherent controversial base. Use of Economic reserves presents significant data gap.	C-B	Despite the controversy on available quantities, this method reflects reasonably the impact.	D	Although this method takes into account stocks in the technosphere, it uses a controversial denominator which increases uncertainty.

		EDIP 97		ORI		Supply risk (SR)	
Science-based criteria	**Summary information (descriptive)**	EDIP is a distance-to-target weighting method. Based on availability-to-use ratio (economic reserves). The global production of a substance or a specific year is divided by the world population from that year. In the second step, the economic reserve of the substance is divided by the global production from the same substance in a particular year, providing the supply horizon of the substance, in years.		The model characterizes the average annual increase of ore required per kg metal. It relies on a database with a substantial worldwide coverage of mining over the period 1998-2010.		The model defines the level of criticality of resources, considering the environmental dimension (e.g. aspects like depletion of reserves, recyclability, overuse of ecosystems), the economic dimension (e.g. concentration of supply, import dependency, etc) and the socio-political dimension (e.g. human rights violations, resource conflicts, illicit trade, precarious working conditions).	
	Completeness of the scope	C	Quite good consistency with the scope of the AoP. However, it ignores the provisioning capacity of the anthropogenic stock in the technosphere	C	Quite good consistency with the scope of the AoP. However, it ignores the provisioning capacity of the anthropogenic stock in the technosphere	B	Good level of consistency with EC-JRC impact pathway and perspective 2 in tems of criticality
	Relevance to the envisaged Area(s) of Protection	C	Is a relevant way to express the impact on the AoP by use of fossils.	D	Is a relevant way to express the impact of resource extraction on the future provisioning capacity in physical terms. Flow coverage is very low	B-C	Relevant method to account for scarcity of resources. Good coverage of flows.
	Scientific robustness & Uncertainty	C	Only normalisation factors published in peer-reviewed paper. Apart from the changing use rate, the weakness of the method is the availability estimate.	B	Published in peer-reviewed paper. Scientific robustness is high; uncertainties very well documented.	B	Not peer-reviewed, but quite well substiantiated and described. Uncertainty not assessed.
	Documentation, Transparency & Reproducibility	B	Well documented; but diffidulties in accessing input data may lead to weaker reproduciblity.	B	Well documented; but diffidulties in accessing input data may lead to weaker reproduciblity.	A	Well documented, transparent and reproducible.
	Applicability	A-B	Already in use in LCIA. Easily applicable in ILCD. Normalisation factors available	B	Applicable in LCIA. Easily applicable in ILCD. Normalisation factors not available.	B	Applicable in LCIA. Easily applicable in ILCD. Normalisation factors not available.
	Characterization factors	C	Rather high relevance and usability. Quite limited set of CFs. The need to be periodically updated is a weakness	C	Limited relevance of characterisation factors, but stability over time	C	Quite relevant number of CFs, but limited range of values. Need to be updated, but relying on frequently updated sources.
Overall evaluation of science-based criteria		B-C	Suffers from weaknesses inherently to the concept (estimates of resources), well reflecting the decreasing resource availability.	B-C	Well reflecting the decreasing resource availablity but limited coverage of elementary flows.	B-C	Interesting method for the criticality approach
Overall evaluation of stakeholders acceptance criteria		C	Exhaustion of available stocks is controversial.	D	Rather recent method with limited exposure/feedback from stakeholders.	B	Reasonably well-accepted method
Final evaluation		C-B	Despite the controversy on available quantities, this method reflects reasonably the impact.	C	Given the limited coverage of elementary flows only recommendable after further development.	B-C	The method could be recommended for the assessment of criticality. The only limit is the relevance of CFs in terms of range of values.

		Supply risk - JRC		Recipe - fossils		Recipe - elements	
Science-based criteria	Summary information (descriptive)	The model applies an exponenet to the criticality factors (supply risk value, SR_{WGI}) identified by Chapman et al., in order to magnify their effect in LCIA.		At the endpoint, the damage is defined as the additional net present costs that society has to pay as a result of an extraction. These are the costs incurred due to the fact that, after the extraction of the "best" (highest grade) resources, future mining becomes more expensive.		The method considers a cost increase over time (future scenario) due to decreasing ore quality and grade. It quantifies the marginal cost increase per year and per kg and expresses it relatively to a reference.	
	Completeness of the scope	B	Good level of consistency with EC-JRC impact pathway and perspective 2 in tems of criticality	A-B	Reflecting the impact on decreasing provisioning capacity quite well. High level of flow coverage.	B	Reflects the increasing efforts due to the declining quality of lithospheric stocks, in relative terms.
	Relevance to the envisaged Area(s) of Protection	B-C	Relevant method to account for scarcity of resources. Good coverage of flows.	C-D	Relativevly limited coverage of AoP and flows	C-D	Is a way to express the impact of resource extraction on the future provisioning capacity in relative economic terms. Quite low coverage of flows
	Scientific robustness & Uncertainty	B	Peer-reviewed and based on a quite well-substantiated model (supply risk). Uncertainty not assessed.	C	Model well-known and accepted, but not peer-reviewed	C	Model well-known and accepted, but not peer-reviewed. Assumption on a systematic cost increase has been questioned.
	Documentation, Transparency & Reproducibility	A	Well documented, transparent and reproducible.	A	Well documented, transparent and reproducible.	B	Reasonably well documented, transparant. Reproducibility over time may be affected by changing marginal cost.
	Applicability	B	Applicable in LCIA. Easily applicable in ILCD. Normalisation factors not available.	A-B	Already in use in LCIA. Easily applicable in ILCD. Normalisation factors available	A-B	Already in use in LCIA. Easily applicable in ILCD. Normalisation factors available
	Characterization factors	B	Quite relevant number of CFs, and wider range of values (compared to supply risk as such). Need to be updated, but relying on frequently updated sources.	B	The limited set of characterisation factors are ok, with some stability over time.	B	The limited set of characterisation factors are ok, with some stability over time.
Overall evaluation of science-based criteria		B	Interesting method for the criticality approach. Wider range of values than in the original version of Supply risk	C	Robust method but not really reflecting the decreasing resource availabity.	C	Well reflecting the relative decreasing availability, but based on economic terms (midpoint) and suffering from uncertainties on cost estimates.
Overall evaluation of stakeholders acceptance criteria		B	Reasonably well-accepted method	B	Reasonably well-accepted method	C	Relative cost increase may be questioned
Final evaluation		B	The method can be recommended for the assessment of criticality. The limit of Supply Risk method about the relevance of CFs is overcome in this version of CFs.	C	The method makes use of cost estimates, which introduces inherent uncertainties; equally economics as a base at midpoint is questionable.	C	The method makes use of cost estimates, which introduces inherent uncertainties; equally economics as a base at midpoint is questionable.

		LPY-FISH		BRD-FISH		EcoPoints/Scarcity 2014	
Science-based criteria	**Summary information (descriptive)**	The model aims at the quantification of overfishing by comparing the current with target fisheries management by the Lost Potential Yield (LPY). It relies on simplified biomass projections to assess the lost catches due to ongoing overfishing.		The model aims to characterise the impacts on biotic natural resources at (fish) species level. It characterizes the current mass caught with the maximum sustainable yield for sustainably fished stocks and with the actual (last 5 years) catches for depleted or overexploited stocks.		The model is ecoscarcity: Swiss distance-to-(policy) target method. This impact category is modeled as many others, based on targets for 2030, with characterization done in 2006 and updated in 2013.	
	Completeness of the scope	C	As such a proper impact pathway and evaluation at midpoint, but limited coverage of natural biomass and geographic area.	C-D	As such a proper but simple impact pathway and evaluation at midpoint, but limited coverage of natural biomass and geographic area.	D	Not much linked to impact or decreasing provisioning capacities; rather to policy objectives.
	Relevance to the envisaged Area(s) of Protection	D	Relevant; a proper way to express the impact on the provisioning capacity at midpoint; however limited geographical and flows coverage	D	Relevant; a simple way to express the impact on the provisioning capacity at midpoint. However limited geographical and flows coverage	C	Intermediate relevance to the AoP. Intermediate coverage of flows and limited geographical scope.
	Scientific robustness & Uncertainty	B-C	Published in peer-reviewed paper. Quite well elaborated scientifically, but with projection over 30 years loaded with a significant level of uncertainty.	D	Published in peer-reviewed paper. Not yet fully elaborated.	C	Original model published in scientific literature. Following updates not. Moderately robust method, including some normative assumptions.
	Documentation, Transparency & Reproducibility	A	Well documented, transparent and reproducible.	A	Well documented, transparent and reproducible.	A	Well documented, transparent and reproducible.
	Applicability	D	Potentially applicable in LCIA, but very limited coverage of elementary flows that are not common in LCI datasets. Normalisation factors not available.	D	Potentially applicable in LCIA, but very limited coverage of elementary flows that are not common in LCI datasets. Normalisation factors not available.	B-C	Already in use in LCIA. Easily applicable in ILCD. Normalisation factors available but method referred to Swiss targets.
	Characterization factors	D	Recent method where characterization factors are relevant but poorly usable by LCA practitioners today.	D	Recent method where characterization factors are relevant but poorly usable by LCA practitioners today.	D	CFs are based on distance to Swiss policy targets. Relevance in other contexts may be questioned.
Overall evaluation of science-based criteria		C-D	Well reflecting the relative decreasing resource availability, but very limited coverage of natural biomass and weak compatibility with LCA practice.	D	Intends to mimic abiotic resource depletion (ADP), but method is not yet fully mature and is being further developed.	C	Robust method but not really reflecting the decreasing resource availabity. CFs specifically referred to Swiss policy targets.
Overall evaluation of stakeholders acceptance criteria		D	Very recent method with limited exposure to and feedback from the stakeholders.	D	Very recent method with limited exposure to and feedback from the stakeholders.	C	Distance to policy target apporach may be questioned
Final evaluation		C-D	One of the first scientifically sound methods but given the limited coverage and poor compatibility with current LCA practice today only recommendable after further	D	One of the first methods but given the limited coverage, poor compatibility and state of development: to be re-evaluated in a later stage.	C	Quite robust method, but distance to target approach may be questionable and relevance outside Switzerland is questionable as well.

		Ecoindicator 99 - minerals		Ecoindicator 99 - fossil fuels		EPS 2000	
Science-based criteria	Summary information (descriptive)	The model assess depletion of resources using the surplus energy as a proxy for the additional effort needed to obtain resources from a lower quality deposit. Surplus energy is dedined as the difference between the energy needed to extract a resource now and in the future.		The model assess depletion of resources using the surplus energy as a proxy for the additional effort needed to obtain resources from a lower quality deposit. Surplus energy is dedined as the difference between the energy needed to extract a resource now and in the future.		The method consists of weighting factors obtained by applying monetisation to environmental impacts of production. It is based on willingness to pay (WTP) for restoring damage done to the safe-guard subject.	
	Completeness of the scope	C	Quite good consistency with the scope of the AoP. However, it ignores the provisioning capacity of the anthropogenic stock in the technosphere	B	The model is relatively complete for fossil resources.	C-D	Reflecting the damage in economic terms on WTP approach. Lack of consistency with EC-JRC impact pathway.
	Relevance to the envisaged Area(s) of Protection	C	The model is relatively complete for minerals and fossil, but does not include biotic resources. It is also based on old input data.			B	Consequences of fossil resource use are quantified by the economic consequences, estimated by the WTP principle.
	Scientific robustness & Uncertainty	D	No review process documented. Quite robust method but not up-to-date.	D	No review process documented. Quite robust method but not up-to-date.	D	No review process documented. Robustness of the model (based on WTP) has been questioned. Assumptions on technology can be out-of-date.
	Documentation, Transparency & Reproducibility	B	The method is well documented and fully accessible, but not easily replicable	B	The method is well documented and fully accessible, but not easily replicable	C	Well documented and transparant. Stability over time unsure with WTP concept.
	Applicability	B	Applicable (already in use), even if quite outdated and superseded by Recipe. Easily applicable in ILCD. Normalisation factors available.	B	Applicable (already in use), even if quite outdated and superseded by Recipe. Easily applicable in ILCD. Normalisation factors available.	B	Applicable (already in use), even if quite outdated. Easily applicable in ILCD. Normalisation factors not available.
	Characterization factors	D	Rather high relevance of CFs, but they are not up-to-date and no update is foreseen (because it has been replaced by Recipe)	D	Rather high relevance of CFs, but they are not up-to-date and no update is foreseen (because it has been replaced by Recipe)	C	Characterization in monetary terms is ok; but temporal resolution is an issue.
Overall evaluation of science-based criteria		D	Quite well-known and used method, but not up-to-date and no update is foreseen (because it has been replaced by Recipe)	D	Quite well-known and used method, but not up-to-date and no update is foreseen (because it has been replaced by Recipe)	C	Reflects the consequences of decreasing resource availability, but the WTP as quantification weakens the method.
Overall evaluation of stakeholders acceptance criteria		B	Reasonably well accepted.	B	Reasonably well accepted.	B	Reasonably well accepted.
Final evaluation		D	The method is out-of-date and superseded by Recipe	D	The method is out-of-date and superseded by Recipe	C	Method reflects the consequences in economic terms but is limited by the WTP quantification.

		IMPACT 2002+		Surplus cost potential		ReCiPe 2008 endpoint - fossils		ReCiPe 2008 endpoint - elements	
Science-based criteria	Summary information (descriptive)	Based on the surplus energy concept (future scenario), using Eco-indicator 99, egaliatarian as source model and factors. An infinite time horizon for fossil energy is assumed. This implies that the total energy content of the fossil energy are lost due to their consumption; hence damage is quantified simply by the energy content.		The model calculates the surplus cost potential (SCP) of mining and milling activities. Main differences from similar models (e.g. ORI) are: 1) all future metal extractions are considered, via cumulative cost-tonnage relationships 2) the operating mining costs account for co-production and are allocated across all mine products in proportion to the revenue that they provide		The approach for evaluating damage is based on the marginal cost increase (future scenario). The marginal increase is to the shift from conventional to unconventional sources.		The approach for evaluating damage is based on the marginal cost increase (future scenario). The marginal increase is to the shift from conventional to unconventional sources.	
	Completeness of the scope	C	Quite good consistency with the scope of the AoP. However, it ignores the provisioning capacity of the anthropogenic stock in the technosphere	C	Quite good consistency with the scope of the AoP. However, it ignores the provisioning capacity of the anthropogenic stock in the technosphere	B	Quite high level of coverage and consistency with EC-JRC impact pathway.	C	Quite good consistency with the scope of the AoP. However, it ignores the provisioning capacity of the anthropogenic stock in the technosphere
	Relevance to the envisaged Area(s) of Protection	D	Quite limited coverage of resources and of flows	D	Limited coverage of AoP and flows	C-D	Relativevly limited coverage of AoP and flows	C-D	Relativevly limited coverage of AoP and flows
	Scientific robustness & Uncertainty	D	Indicator for resources based on Eco-indicator 99, for which no review process is documented. Method based on EI99, which is not up-to-date.	B-C	Published in peer-reviewed paper. The model relies on up-to-date input data, but CFs are provided only for some minerals and metals. Assumption on a systematic cost increase has been questioned.	C	No review process documented. Assumption on a systematic cost increase has been questioned.	C	No review process documented. Assumption on a systematic cost increase has been questioned.
	Documentation, Transparency & Reproducibility	B	The method is well documented and fully accessible, but not easily replicable	B	The method is well documented and fully accessible, but not easily replicable	B	Reasonably well documented, transparant. Reproducibility over time may be affected by changing marginal cost.	B	Reasonably well documented, transparant. Reproducibility over time may be affected by changing marginal cost.
	Applicability	B	Applicable (already in use), even if quite outdated (to be noted that EI 99 was superseded by Recipe, but Impact 2002+ was not updated accordingly). Normalisation factors available.	B	Applicable in LCIA. Easily applicable in ILCD. Normalisation factors not available.	A-B	Already in use in LCIA. Easily applicable in ILCD. Normalisation factors available	A-B	Already in use in LCIA. Easily applicable in ILCD. Normalisation factors available
	Characterization factors	D	Rather high relevance of CFs, but they are not up-to-date	C	Limited number of CFs available at the moment	C	Characterization in monetary terms is ok; but temporal resolution is an issue.	C	Characterization in monetary terms is ok; but temporal resolution is an issue.
Overall evaluation of science-based criteria		C	Robust method but not well reflecting the decreasing resource availabity.	C	Interesting new method, but at the moment with limited number of CFs and still not tested.	C	Reflects the consequences of decreasing resource availability, with inherent uncertainties related to cost change shifts.	C	Reflects the consequences of decreasing resource availability, with inherent uncertainties related to cost change shifts.
Overall evaluation of stakeholders acceptance criteria		C	Method is accepted to a limited extent.	B	Reasonably well accepted.	B	Reasonably well accepted.	B	Reasonably well accepted.
Final evaluation		D	The method does not reflect the consequences of decreasing availability.	C	The approach might be interesting in case the costs are taken as measure of potential depletion, but the method is still immature	B	The method reflects the consequences of decreasing availability reasonably.	B	The method reflects the consequences of decreasing availability reasonably.

3.7 Discussion on models evaluation

As previously mentioned in relation to their description and in the presentation of the impact pathway, the models evaluated can be grouped in some clusters, according to their perspective and their approach to the AoP resources.

The evaluation of available models highlighted that there is no model able to assess midpoint or endpoint impacts for all the types of natural resources (minerals and metals, biotic resources, energy carriers, etc.). The models for advance accounting are generally able to cover a good range of resources (and of ILCD flows), even if many resources, and especially biotic ones, are still not covered by the existing LCIA models. Moreover, advance accounting models are not aimed at linking resource use to changes in the provisioning capacity of a resource; therefore, they are not suitable for recommendation as LCIA models.

In addition, the tests done on the models (not only advance accounting ones) have underlined that distinguishing biotic and abiotic resources from energy carriers helps to better highlight the intensity of the use of materials and energy in the system under evaluation.. This is in line with what was proposed by the developers of the ADP model previously recommended (see van Oers and Guinée 2016, as discussed in section 3.1 of this document), and recognizes that the main target for assessing the use of energy carriers (i.e. acknowledging the use of non-renewable resources or of renewable ones) is different from the target of the other types of resources (i.e. assessing depletion and provisioning capacity).

Among the models for pure accounting of energy carriers use, CED is considered the most suitable one, also because it is already widely used in LCA and has been included in existing standards (e.g. EN 15978 on sustainability assessment of construction works) and labelling systems (e.g. Environmental Product Declarations, EPDs). Among the models for impact assessment of depletion of energy carriers, the ADPfossil is considered the most suitable to be recommended.

Models based on the abiotic depletion model have received several critics related mainly to the uncertainty of the calculation. However, at the moment there is no robust alternative to substitute this approach for the assessment of the reduced availability of resources due to human use. Among this type of models, the approach adopted for the AADP model by Schneider et al. (2015) is considered the most advanced one because it is the first attempt to take into consideration the need to consider recycling not only in the LCI but also in the LCIA phase. In fact, the model includes in the calculation of available stock also the amount of resources already extracted from nature but potentially still available for use after the end of life of the products in which they have been used (called "anthropogenic stock"). However, a relevant drawback of the AADP model is the lack of robustness of the assumptions underlying the calculation of the available stock (both the "ultimately extractable reserves" and the "anthropogenic stock") and the consequent lack of acceptance of the model by the geological community.

Moreover, as suggested by van Oers and Guinée (2016), the inclusion of the anthropogenic stock in the calculation of resource availability would require a further change also in how the extraction rate is calculated, i.e. moving from a depletion problem to a dissipation approach, as proposed and framed by Frischknecht in the Ecoscarcity 2013 model (where only dissipative use of resources is accounted for). However, at the moment, there is no set of CFs available and applicable at the global (or at least European) scale. Therefore, this should be taken as a further need for future improvements.

Most of the models taking into account the variation of ore grade over time rely on quite outdated data (e.g. Ecoindicator 99, IMPACT 2002+) and are not suitable for being recommended as an improvement of the existing ILCD recommendation. Recipe model, even if largely used and already provided in the most common commercial software for LCIA, if its

CFs are mapped to ILCD flows, they are able to cover only 15% of them. Therefore, it cannot be recommended as an improvement of the current ILCD recommended model as used as default model for the Environmental Footprint. Similar applies ORI (at the midpoint) and Surplus Cost Potential (at the endpoint) that are able to cover, respectively, 5% and 10% of the flows. In addition, the use of an economic evaluation is considered not robust enough at midpoint level.

Ecoscarcity 2013 adopts an interesting approach. However, it is highly country-specific, because the CFs measure the distance of Swiss environmental conditions from the ones foreseen in the Swiss policies. Therefore, the CFs cannot be recommended for use in other context.

EPS 2000 model could be potentially interesting because it provides CFs for some biotic resources (wood, fish and meat). However, the suitability of the WTP approach has been questioned, especially because it implies a lot of assumptions about the alternative scenarios. In addition, all these assumptions are made on the technology that was in place in late 90s or foreseen for the near future, and this makes the model out-of-date for use in present times.

The model by Emanuelsson et al. (2014) related to fish resources is promising as one of the first attempts to include this sector of biotic resources into LCIA, but its applicability in the context of ILCD is at present very limited. The main gap to be filled to ensure applicability is the lack of elementary flows for fish resource, both in the ILCD list and in the existing background dataset.

In summary, the analysis of the critics posed to the model and calculation at the base of the current ILCD recommendation, of the needs emerged from the most recent research in the area of resources, of the results coming from the evaluation of the models pre-selected and of the comments received by stakeholders, led to the decision to recommend again two models based on the resource depletion concept.

Two main changes occur with respect to the previous recommendation:

i) ADP is split into two indicators, one for abiotic resource depletion and the other for energy carriers,

ii) the recommended indicator for abiotic resource depletion is ADPultimate reserves, because it is the one with the highest stakeholder acceptance and the lowest uncertainty in the estimation of the reference stock.

It is recognized that biotic resources remain not covered by the current recommendation.

3.8 Recommended default model for midpoint

The former impact category “resource depletion”, now “resource use”, consists of two mandatory indicators for impact, reflecting the conclusions illustrated before.

The two mandatory indicators recommended for impact assessment are:

1) “ADPultimate reserves”. The ADPultimate reserve is considered the more suitable for this impact category. The model still does not consider the anthropogenic stock and does not include biotic resources. This should be taken into account for future improvements.
2) “ADPfossil” is recommended for assessing depletion of energy carriers.

3.9 Additional environmental information

In order to include naturally occurring resource in the evaluation, an additional environmental information for biotic resource may be added, with the following indicator:

- "Biotic resource intensity". The indicator consists of a mass accounting of biotic resources (in kg) as for the LCI of the system under evaluation. A list of elementary flows of naturally occurring biotic resources is available at the European Platform on Life cycle Assessment (EPLCA) website at http://eplca.jrc.ec.europa.eu/LCDN/developerEF.xhtml. The list is based on the study performed by Crenna et al .2018.

3.10 Models for endpoint

At the endpoint level, all models evaluated are considered too immature to be recommended. However, Surplus Cost Potential may be used as interim solution.

3.11 Consistency between midpoint and endpoint models

As the recommended model at the midpoint level and the interim model at the endpoint level rely on different approaches and rationales, there is poor consistency between them.

3.12 Classification of the recommended default midpoint models

The recommendation of the two indicators listed before - as mandatory for resource use impact assessment- is level III, because still some improvements are needed (which are discussed in sections 3.8 and 3.15).

3.13 Recommended characterization factors

Characterisation factors are available to be downloaded at the European Platform on Life cycle Assessment (EPLCA) website at http://eplca.jrc.ec.europa.eu/LCDN/developerEF.xhtml

3.14 Normalisation factors

Source and data used to calculate the normalisation factors are available in Crenna et al. 2019. The EF normalisation factors to be used are available at http://eplca.jrc.ec.europa.eu/LCDN/developerEF.xhtml

3.15 Research needs

Some of the needs highlighted in the introduction and discussed throughout the evaluation cannot be fully satisfied by the models currently available and suitable for application in LCIA. Therefore, some needs for improvements in the future remain still open and should be addressed by future research in the field of resource conservation within LCIA. They are listed and discussed below.

- **Biotic resources**. The choice to have an indicator accounting for resource intensity, i.e. for the mass of resources used within the system under evaluation, helps to keep track, at least partially, of the use of biotic resources. However, this is still far from a proper impact assessment of the environmental impact of the use of biotic resources, to be added and/or compared with the assessment of abiotic resources and energy carriers. A preliminary review of methods used so far in LCA and a proposal for an approach based on renewability of biotic resources is presented in Crenna et al., 2018.
- **Recycling**. The AADP model by Schneider et al. 2015 is a first attempt to improve the ability of abiotic resource depletion models to take into account also the amount of resources already in the technosphere and potentially available (the so-called "*anthropogenic stock*"). However, this model still suffers from some of the weaknesses identified for the overall ADP concept and can be further improved in the future.

- **Dissipation**. As proposed by Frischknecht (2014) the amount of resources extracted from the natural environment and the amount of resources used in a dissipative way should be considered separately. This means to move from looking only at the interface between the ecosphere and the technosphere (by measuring the amount of resources extracted), to look at what happens within the technosphere, once the resources are available for (multiple) human uses, and to reflect this at the inventory stage. A proposal and a way forward has been described in the OEFSR pilot on copper production (Technical secretariat of the OEFSR pilot on copper production, 2016) as well as by Zampori and Sala, 2017.
- **Dynamic approach to estimate future availability**. Dynamic models to predict future availability of resources were not available for recommendation. Therefore, the recommended model for the indicator "resource depletion" still relies on static models. Future research should be oriented to develop more dynamic models for resource availability evaluation.

3.16 References of the chapter on the general introduction on resources and on resource use impacts

- Alvarenga, R. A., Dewulf, J., Van Langenhove, H., & Huijbregts, M. A. (2013). Exergy-based accounting for land as a natural resource in life cycle assessment. The International Journal of Life Cycle Assessment, 18(5), 939-947.
- Bösch, M. E., Hellweg, S., Huijbregts, M. A., & Frischknecht, R. (2007). Applying cumulative exergy demand (CExD) indicators to the ecoinvent database. The International Journal of Life Cycle Assessment, 12(3), 181-190.
- Boulay, A. M., Motoshita, M., Pfister, S., Bulle, C., Muñoz, I., Franceschini, H., & Margni, M. (2015a). Analysis of water use impact assessment models (part A): evaluation of modeling choices based on a quantitative comparison of scarcity and human health indicators. The International Journal of Life Cycle Assessment, 20(1), 139-160.
- Boulay, A. M., Bayart, J. B., Bulle, C., Franceschini, H., Motoshita, M., Muñoz, I., ... & Margni, M. (2015b). Analysis of water use impact assessment models (part B): applicability for water footprinting and decision making with a laundry case study. The International Journal of Life Cycle Assessment, 20(6), 865-879.
- Brandão, M., Milà i Canals, L., (2013). Global characterisation factors to assess land use impacts on biotic production. Int. J. Life Cycle Assess. 18(6), 1243−1252.
- Bringezu, S. (2015). Possible target corridor for sustainable use of global material resources. Resources, 4(1), 25-54.
- Chapman A, Arendorf J, Castella T, Thompson P, Willis P, Espinoza LT, Klug S and Wichmann E. (2013). Study on Critical Raw Materials at EU Level. http://ec.europa.eu/DocsRoom/documents/5605/attachments/1/translations/en/renditions/native (Accessed March 2016).
- Crenna E., Sozzo S., Sala S., (2018) Natural biotic resources: towards an impact assessment framework for sustainable supply chain management. Journal of Cleaner Production 172:3669-3684, https://doi.org/10.1016/j.jclepro.2017.07.208
- Crenna E, Secchi M, Benini L, Sala S (2019) Global environmental impacts: data sources and methodological choices for calculating normalisation factors. International Journal of LCA, doi.org/10.1007/s11367-018-1534-9

- Cummings, C. D. and Seager, T.P. (2008). Estimating exergy renewability for sustainability assessment of corn ethanol. Conference paper IEEE, doi: 10.1109/ISEE.2008.4562893
- Dewulf, J., Benini, L., Mancini, L., Sala, S., Blengini, G. A., Ardente, F., ... & Pennington, D. (2015). Rethinking the area of protection "Natural Resources" in life cycle assessment. Environmental science & technology, 49(9), 5310-5317.
- Dewulf, J., Bösch, M. E., Meester, B. D., Vorst, G. V. D., Langenhove, H. V., Hellweg, S., & Huijbregts, M. A. (2007). Cumulative exergy extraction from the natural environment (CEENE): a comprehensive life cycle impact assessment model for resource accounting. Environmental science & technology, 41(24), 8477-8483.
- Drielsma, J. A., Russell-Vaccari, A. J., Drnek, T., Brady, T., Weihed, P., Mistry, M., & Simbor, L. P. (2016). Mineral resources in life cycle impact assessment—defining the path forward. The International Journal of Life Cycle Assessment, 21(1), 85-105.
- Emanuelsson, A., Ziegler, F., Pihl, L., Sköld, M., & Sonesson, U. (2014). Accounting for overfishing in life cycle assessment: new impact categories for biotic resource use. The International Journal of Life Cycle Assessment, 19(5), 1156-1168.
- European Commission (EC) (2013a) Commission Recommendation of 9 April 2013 on the use of common models to measure and communicate the life cycle environmental performance of products and organisations (2013/179/EU)
- EC (2013b) Communication: Building the Single Market for Green Products Facilitating better information on the environmental performance of products and organisations (COM (2013) 196 final)
- European Commission, Joint Research Centre (EC-JRC) (2010a). International Reference Life Cycle Data System (ILCD) Handbook - Analysis of existing Environmental Impact Assessment methodologies for use in LCA - Background Document to the International Reference Life Cycle Data System (ILCD). Luxemburg. Publications Office of the European Union.
- EC-JRC (2010b). International Reference Life Cycle Data System (ILCD) Handbook - Framework and Requirements for Life Cycle Impact Assessment Models and Indicators. EUR 24586 EN. Luxembourg. Publications Office of the European Union.
- EC-JRC (2011). International Reference Life Cycle Data System (ILCD) Handbook - Recommendations for Life Cycle Impact Assessment in the European context. EUR 24571 EN. Luxemburg. Publications Office of the European Union.
- EC-JRC (2012). Characterisation factors of the ILCD Recommended Life Cycle Impact Assessment models. Database and Supporting Information. First edition. February 2012. EUR 25167. Luxembourg. Publications Office of the European Union
- Frischknecht R., Jungbluth N., Althaus H.-J., Bauer C., Doka G., Dones R., Hellweg S., Hischier R., Humbert S., Margni M. and Nemecek T. (2007) Implementation of Life Cycle Impact Assessment Methods. ecoinvent report No. 3, v2.0. Swiss Centre for Life Cycle Inventories, Dübendorf, CH, retrieved from: www.ecoinvent.org.
- Frischknecht R, Steiner R, Jungbluth N (2008). Ökobilanzen: Modele der ökologischen Knappheit – Ökofaktoren 2006. Modele für die Wirkungsabschätzung in Ökobilanzen. Öbu SR 28/2008. Öbu, Zürich, Switserland http://www.oebu.ch/oebu/downloads/oekofaktoren_sr28.pdf
- Frischknecht R., Büsser Knöpfel S., (2013). Swiss Eco-Factors 2013 according to the Ecological Scarcity Model. Methodological fundamentals and their application in

Switzerland. Environmental studies no. 1330. Federal Office for the Environment, Bern: 254 pp.

- Frischknecht R. (2014). Impact assessment of abiotic resources: the role of borrowing and dissipative resource use. LCA Forum 55, April 11, 2014, ETH Zürich. Available at: http://www.lcaforum.ch/portals/0/df55/DF55-05%20Frischknecht.pdf (Accessed March 2016).
- Frischknecht R., Wyss F., Buesser S., Lützkendorf T. and Balouktsi M. (2015) Cumulative energy demand in LCA: the energy harvested approach. Int J LCA, 20(7), pp. 957-969.
- Goedkoop, M., Spriensma, R. (2001). The Ecoindicator99: A damage oriented model for life cycle impact assessment: Methodology report.
- Goedkoop, M., Heijungs, R., Huijbregts, M., De Schryver, A., Struijs, J., & van Zelm, R. (2009). ReCiPe 2008. A life cycle impact assessment model which comprises harmonised category indicators at the midpoint and the endpoint level, 1.
- Guinée, J.B. (Ed.), Gorrée, M., Heijungs, R., Huppes, G., Kleijn, R., de Koning, A., Van Oers, L., Wegener Sleeswijk, A., Suh, S.,. Udo de Haes, H.A, De Bruijn, J.A., Van Duin R., Huijbregts, M.A.J. (2002). Handbook on Life Cycle Assessment: Operational Guide to the ISO Standards. Series: Eco-efficiency in industry and science. Kluwer Academic Publishers. Dordrecht (Hardbound, ISBN 1-4020-0228-9; Paperback, ISBN 1-4020-0557-1).
- Guinée, J. B., & Heijungs, R. (1995). A proposal for the definition of resource equivalency factors for use in product life-cycle assessment. Environmental toxicology and chemistry, 14(5), 917-925.
- Hauschild, M. Z., & Wenzel. (1998). H. Environmental Assessment of Products. Vol. 2. Scientific Background, 1998.
- Hischier R., Weidema B., Althaus H.-J., Bauer C., Doka G., Dones R., Frischknecht R., Hellweg S., Humbert S., Jungbluth N., Köllner T., Loerincik Y., Margni M., Nemecek T. (2010) Implementation of Life Cycle Impact Assessment Models. Ecoinvent report No. 3, v2.2. Swiss Centre for Life Cycle Inventories, Dübendorf.
- Itsubo N, Inaba A (2012) LIME 2 Life-cycle impact assessment method based on endpoint modeling chapter 1—outline of LIME 2. JLCA News Letter English Edition, No. 14. http://lca-forum.org/english/pdf/No14_C1_Outline.pdf
- Jolliet, O., Margni, M., Charles, R., Humbert, S., Payet, J., Rebitzer, G., & Rosenbaum, R. (2003). IMPACT 2002+: a new life cycle impact assessment methodology. The International Journal of Life Cycle Assessment, 8(6), 324-330.
- Jolliet, O., Frischknecht, R., Bare, J., Boulay, A. M., Bulle, C., Fantke, P., & McKone, T. E. 2014. Global guidance on environmental life cycle impact assessment indicators: findings of the scoping phase. *The International Journal of Life Cycle Assessment*, *19*(4), 962-967.
- IEA 2014. International Energy Agency. Key World Energy Statistics 2014. Accessed at: http://www.iea.org/publications/freepublications/publication/keyworld2014.pdf
- Klinglmair, M., Sala, S., Brandão, M. (2014). Assessing resource depletion in LCA: a review of models and methodological issues. The International Journal of Life Cycle Assessment, 19(3), 580-592.

- Langlois, J., Fréon, P., Delgenes, J. P., Steyer, J. P., & Hélias, A. (2014). New models for impact assessment of biotic-resource depletion in life cycle assessment of fisheries: theory and application. Journal of Cleaner Production, 73, 63-71.
- Lindeijer E, Mueller-Wenk R, Steen B (2002) Impact assessment of resources and land use. In: Udo de Haes HA et al (eds) Life-cycle impact assessment: striving towards best practice. SETAC, Pensacola
- Mancini L, Benini L, Sala S., (2016). Characterization of raw materials based on supply risk indicators for Europe. International Journal of LCA, p 1-13, doi: 10.1007/s11367-016-1137-2
- Milà i Canals, L., Chenoweth, J., Chapagain, A., Orr, S., Antón, A., Cliftm R. (2009). Assessing freshwater use impacts in LCA: part I—inventory modelling and characterisation factors for the main impact pathways. Int J Life Cycle Assess 14(1):28–42
- Rørbech, J. T., Vadenbo, C., Hellweg, S., & Astrup, T. F. (2014). Impact assessment of abiotic resources in LCA: quantitative comparison of selected characterization models. Environmental science & technology, 48(19), 11072-11081.
- Rugani, B., Huijbregts, M. A., Mutel, C., Bastianoni, S., & Hellweg, S. (2011). Solar energy demand (SED) of commodity life cycles. Environmental science & technology, 45(12), 5426-5433.
- Schneider, L., Berger, M., & Finkbeiner, M. (2011). The anthropogenic stock extended abiotic depletion potential (AADP) as a new parameterisation to model the depletion of abiotic resources. The International Journal of Life Cycle Assessment, 16(9), 929-936.
- Schneider, L., Berger, M., Schüler-Hainsch, E., Knöfel, S., Ruhland, K., Mosig, J., Bach, V., Finkbeiner, M. (2014). The economic resource scarcity potential (ESP) for evaluating resource use based on life cycle assessment. The International Journal of Life Cycle Assessment, 19(3), 601-610.
- Schneider, L., Berger, M., & Finkbeiner, M. (2015). Abiotic resource depletion in LCA—background and update of the anthropogenic stock extended abiotic depletion potential (AADP) model. The International Journal of Life Cycle Assessment, 20(5), 709-721.
- Scholz, R. W., & Wellmer, F. W. (2013). Approaching a dynamic view on the availability of mineral resources: what we may learn from the case of phosphorus? Global Environmental Change, 23(1), 11-27.
- Steen, B. (1999). A Systematic Approach to Environmental Priority Strategies in Product Development (EPS): Version 2000-General System Characteristics. Gothenburg: Centre for Environmental Assessment of Products and Material Systems.
- Swart, P., & Dewulf, J. (2013). Quantifying the impacts of primary metal resource use in life cycle assessment based on recent mining data. Resources, Conservation and Recycling, 73, 180-187.
- Swart, P., Alvarenga, R. A. F., Dewulf, J. (2015). Abiotic resource use. In Encyclopedia of LCA, Vol. IV: Life Cycle Impact Assessment; Hauschild, M. Z., Huijbregts, M. A. J., Ed.; Springer Press , 2015.
- Taelman, S. E., De Meester, S., Schaubroeck, T., Sakshaug, E., Alvarenga, R. A., & Dewulf, J. (2014). Accounting for the occupation of the marine environment as a natural resource in life cycle assessment: an exergy based approach. Resources, Conservation and Recycling, 91, 1-10.

- Technical secretariat of the OEFSR pilot on copper production (2016). OEFSR v. 2.2, available at: https://webgate.ec.europa.eu/fpfis/wikis/display/EUENVFP/Stakeholder+workspace%3A+OEFSR+pilot+Copper+production
- UNEP (2016) Global guidance for life cycle impact assessment indicators. Volume 1. ISBN: 978-92-807-3630-4. Available at: http://www.lifecycleinitiative.org/life-cycle-impact-assessment-indicators-and-characterization-factors/
- Vadenbo, C., Rørbech, J., Haupt, M., Frischknecht, R. (2014). Abiotic resources: new impact assessment approaches in view of resource efficiency and resource criticality--55th Discussion Forum on Life Cycle Assessment, Zurich, Switzerland, April 11, 2014. The International Journal of Life Cycle Assessment, 19(10), 1686.
- Valero A. (2008). Exergy evolution of the mineral capital on earth. PhD thesis, University of Zaragoza, 2008, 461p
- van Oers L, de Koning A, Guinee JB, Huppes G. (2002): Abiotic Resource Depletion in LCA. Road and Hydraulic Engineering Institute, Ministry of Transport and Water, Amsterdam.
- van Oers, L., Guinée, J. (2016). The Abiotic Depletion Potential: Background, Updates, and Future. Resources, 5(1), 16.
- VDI, Verein Deutscher Ingenieure. (1997). VDI-richtlinie 4600: Cumulative Energy Demand, Terms, Definitions, Models of Calculation; Verein Deutscher Ingenieure: Düsseldorf, Germany, 1997.
- Vieira, M., Ponsioen, T. C., Goedkoop, M. J., & Huijbregts, M. A. 2016. Surplus Cost Potential as a Life Cycle Impact Indicator for Metal Extraction. Resources, 5(1), 2.
- Yellishetty, M., Mudd, G. M., & Ranjith, P. G. (2011). The steel industry, abiotic resource depletion and life cycle assessment: a real or perceived issue?. Journal of Cleaner Production, 19(1), 78-90.
- Zampori L., Sala S., (2017) Feasibility study to implement resource dissipation in LCA, EUR 28994 EN, Publications Office of the European Union, Luxembourg, ISBN 978-92-79-77238-2, doi:10.2760/869503

4 Impact due to Land use

4.1 Introduction

The intensification and expansion of human activities, with the increase use of land and associated land use change, have been leading over the years to increased pressure on land resources, resulting in soil quality degradation (MEA, 2005). However, due to the challenge of quantifying impacts on soils (Li, 2007), soil qualities, properties and functions have been incorporated only in a very limited way in LCA studies. Clearly, land use impact assessment needs to be more inclusive (Koellner et al., 2013a) and, specifically, following the recent recommendations of the United Nations Environmental Programme/- Society of Environmental Toxicology and Chemistry Life Cycle Initiative (UNEP/SETAC LC Initiative), LCA land use models should incorporate the impact of the supply chain on soil quality (Curran et al., 2016).

Soil quality is defined in Doran (2002) 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". Soils deliver essential ecosystem services, such as freshwater purification and regulation (Garrigues et al., 2013), food and fibres production and maintain the global ecosystem functions as well. Ensuring the maintenance of high quality standards for the state of soils is therefore a fundamental requirement for global sustainability (Doran, 2002). Indeed, a lot of attention has been given to the development of indicators for monitoring pressure on soil due to human activities (van Oudenhoven et al., 2012; Niemi et al., 2015). Yet, there is a clear need to assess to which extent soil quality is affected by current human interventions as well as to detect hotspots along supply chains and possible "sustainable land management" options. However, the quantification of impacts on soils functions is rather challenging given the complexity of soil processes, as well as the spatial and temporal variability of soil properties. This variability determines, for instance, the adequacy of the soil quality indicator to represent local conditions (Doran and Parkin, 1996). Therefore, defining a robust single soil indicator −or a minimum data set of indicators− remains a difficult task. This holds especially true in a life cycle assessment context, in which detailed information and data on location and local conditions is often scarce.

In the literature, three main quantitative approaches to the so-called "land footprinting" could be identified: i) mere land accounting, which reports the area of land use associated with certain activities/crops (e.g. m^2); ii) weighted accounting, which estimates the amount of land standardized to factors as the productivity of the land (e.g. Ecological Footprint, Wackernagel, 2014); iii) quantification of the change of a specific soil quality or property, resulting from a land interventions (e.g., soil organic matter, Milà i Canals et al., 2007a).

4.2 Framework and scope

Within an LCA context, midpoint indicators so far usually consist of the mere sum of the area of land occupied and/or transformed for the production of a certain amount of product. Occupation-related data are generally available in LCA software and inventories. Endpoint indicators have generally focused on the damage caused by land use and land use change to biodiversity (e.g. species richness loss: De Baan et al., 2013; Souza et al., 2015).

The International Reference Life Cycle Data System (ILCD) Handbook (EC-JRC, 2011) has recommended the model developed by Milà i Canals et al. (2007a, 2007b) for the assessment of the impact of the supply chain on land use at midpoint level. The model adopts soil organic matter (SOM) as a stand-alone indicator for the assessment of land use impacts. Although

SOM is considered one of the most important indicators for the sustainability of cropping systems (Fageria et al., 2012) and it has a crucial role in provisioning (e.g. biotic production) and supporting services (e.g. climate regulation), important soil functions are disregarded. Examples of these ignored functions are soil resistance to erosion, compaction and salinization (Mattila et al., 2011). Therefore, the model was considered not fully satisfactory (EC-JRC, 2011).

Due to the limitations of the currently recommended model and the need to more comprehensively assess the impacts of land use, there is a pressing need to improve currently available models. Here the focus is put on assessing land use impact models at midpoint level, building on the extended analysis is reported in Vidal Legaz et al. 2017. At endpoint, a similar process has been followed in a parallel review conducted by the UNEP-SETAC Life Cycle Initiative task force on land use impact on biodiversity (Curran et al., 2016).

4.3 Environmental mechanism (cause-effect chain)

The environmental cause effect chain of ILCD (EC-JRC, 2011) has been updated as there was the need of a clearer and more consistent impact pathway depicting the causal relationships from the inventory data (amount and typology of land use and soil conditions) to the mid- and endpoint indicators and further to the areas of protection (AoPs). The updated impact pathway serves also to identify which parts of the cause-effect chain are covered by the currently available land use models and which are still lacking. Furthermore, it serves to unravel unclear links between the LCI data, midpoints and endpoints.

This new impact pathway, presented in Vidal-Legaz et al. 2017, was built considering the last developments regarding land use impact assessment in LCA (Milà i Canals et al., 2007c; Koellner et al., 2013b; Souza et al., 2015; Curran et al., 2016) as well as the impact pathways of pre-selected models (Garrigues et al., 2013; Núñez et al., 2013). in particular, Curran et al. (2016) proposed a land use impact pathway with a focus on biodiversity, where the link between the impacts on soil quality and habitats and their ultimate impact on biodiversity is made explicit.

The impact pathway proposed here (Figure 4.1) starts from the different soil properties and functions of the soil related to geomorphological and pedological features of soils before any land interventions. Soil functions refer, among others, to the soil capacity to supply nutrients to plants (soil fertility), to regulate water flow and erosion etc. 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, or 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, soil organic carbon 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 will 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, e.g. exergy, emergy. Overall, it is clear that the different soil qualities and properties are intimately related to the capability of soil for providing ecosystem services of different typologies.

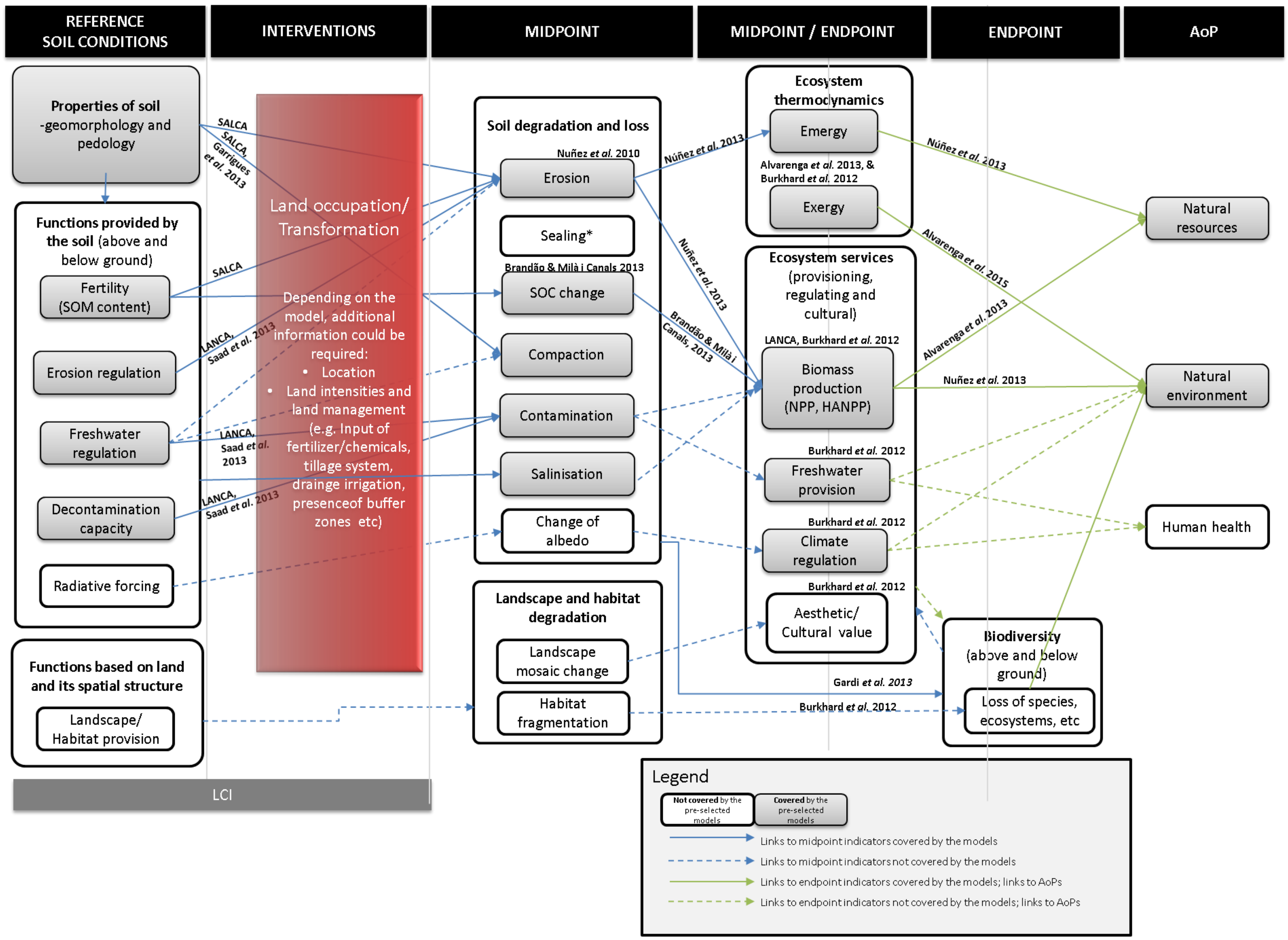


Figure 4.1: Reviewed impact pathway for land use. The positioning of the pre-selected models is displayed as: Brandão and Milà i Canals, 2013; LANCA (Beck et al., 2010); Saad et al., 2013; SALCA (Oberholzer et al., 2012); Núñez et al., 2010; Garrigues et al., 2013; Núñez et al., 2013; Alvarenga et al., 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 'natural resources' AoP, but a midpoint indicator when referring to the 'natural environment' AoP, on which endpoints (e.g. biodiversity) will rely on.

4.4 Criteria for the evaluation of this impact category

Specific criteria for the evaluation of land use models were developed. Compared with those used for evaluating the models for EC-JRC, 2011, additional ones were added under the *environmental relevance* set of criteria, more specifically under the *comprehensiveness* criteria. Moreover, they are partially based on the evaluation criteria set developed by Curran et al*.* (2016) for the assessment of land use models with a focus on biodiversity.

The land use-specific criteria developed here assess the coverage of land use inventory flows, following the International Reference Life Cycle Data System (ILCD). This reference is based on the land use classification proposed by Koellner et al*.* (2013b), a harmonized classification of land use/cover types derived from scientific efforts of the UNEP-SETAC Life Cycle Initiative to guarantee a better coverage of land use typologies and improve the comparability of modeling results. Building on this, the adopted land use classification includes a rather complete coverage of land use types, and aggregates them in four hierarchical levels.

Next, the land use-specific criteria assess whether the models address the following aspects:

- impacts of both extensive and intensive land uses (e.g. high/low input agriculture, clear-cut/selective forestry);
- permanent impacts, i.e. whether the model allows for quantifying irreversible impacts on the soil;
- direct and indirect land use change, i.e. whether the land intervention causes additional land interventions in other areas (e.g. the expansion of a specific type of agriculture might change the market conditions leading to additional land interventions);
- impacts of both land occupation and transformation;

Then, the criteria specify the typology of indicators that the models incorporate:

- soil properties: e.g. soil fertility, Soil Organic Carbon (SOC), Soil Organic Matter (SOM), soil density, soil texture, etc.;
- soil functions: biotic production, erosion regulation, water regulation, biodiversity support, climate regulation and aesthetic/cultural value;
- soil threats, i.e. relevant degradation processes of soil quality: e.g. erosion, compaction, sealing, salinization, or contamination.
- land availability, i.e. whether land competition with other uses or land scarcity are addressed.

The full set of criteria used for the evaluation of land use models, which includes the criteria that are common to all impact categories plus the land use-specific criteria detailed here above, are reported in Annex 4.1 (as a separate excel file). This annex provides a description of the aspects to be considered for the assessment of each criterion as well as the guidance used for models scoring during the evaluation process.

4.5 Pre-selection of models for further evaluation

We investigated whether relevant new developments have been introduced for the land use models already evaluated in the ILCD handbook (EC-JRC, 2011) that would allow for the modeling of land use impact at midpoint level with a focus on soil-related indicators. Moreover, we carried out a literature review to incorporate models assessing soil properties/functions/threats that had not been previously considered in the ILCD handbook, *i.e.* models developed after those assessed in the ILCD (*i.e.* until year 2009). The collection of studies covers those available in June 2016.

Among the models identified, eleven models were pre-selected for further evaluation according to the following minimum requirements:

- models had to compute indicators for assessing soil properties/functions/threats;
- models had to be compatible with LCA (e.g. they could be used to calculate impact indicators starting from elementary flows presented in Life Cycle Inventories) − but they did not necessarily have to come from LCA-specific studies;
- models had to produce characterization factors (CFs) or an output that could be easily converted into characterization factors.

4.5.1 Pre-selection of midpoint models

Table 4.1 shows the list of models pre-selected for evaluation within the impact category land use, with a focus on midpoint. All these models fulfil the three minimum requirements for pre-selection specified in the section above.

Table 4.1 Land use midpoint models pre-selected for evaluation

Model	Indicator(s)	Unit **	Reference	Relevant soil indicators	Compatibility LCA***
Brandão and Milà i Canals (2013)	-Soil Organic Carbon (SOC) –as indicator of Biotic Production Potential (BPP)	t C-year/ (ha-year)	Brandão and Milà i Canals (2013)	Yes	Yes
LANCA Baitz (2002) and Bos et al. (2016)	-Erosion resistance -Mechanical filtration -Physicochemical filtration -Groundwater replenishment -Biotic production	kg/m² year m³/m² year mol/m² m³/m² year kg/m² year	Beck et al. (2010), and Bos et al. (2016) for the characterization factors	Yes	Yes
Saad et al. (2013)	-Erosion resistance -Mechanical filtration -Physicochemical filtration -Groundwater recharge	t/(ha year) cm/day $cmol_c/kg_{soil}$ mm/year	Saad et al. (2013)	Yes	Yes
SALCA-SQ Oberholzer et al., (2012)	-Soil properties indicators: rooting depth, macro-pore volume, aggregate stability, organic carbon, heavy metals -organic pollutants, earthworm biomass, microbial biomass, microbial activity -Impact indicators: risk of soil erosion, risk of soil compaction	many different	Oberholzer et al. (2012)	Yes	Yes
Núñez et al. (2010)	-Desertification index	dimensionless	Núñez et al. (2010)	Yes	Yes
Garrigues et al. (2013)	-Total soil area compacted -Loss of pore volume	m^2/ha, m^2/t m^3/ha, m^3/t	Garrigues et al. (2013)	Yes	Yes
Núñez et al. (2013)	-Emergy -Net Primary Production (NPP) depletion	MJse g^{-1} soil loss m^2 year	Núñez et al. (2013)	Yes	Yes
Alvarenga et al. (2013)	-Exergy of natural land (biomass extraction-based) -Exergy of human-made land (potential NPP-based)	MJ ex/m^2 year	Alvarenga et al. (2013)	Yes	Yes
Alvarenga et al. (2015)	-Human Appropriation of NPP (HANPP)	kg dry matter/m^2 year	Alvarenga et al. (2015)	Yes	Yes
Gardi et al. (2013)	-Soil pressure (on biodiversity)		Gardi et al. (2013)	Yes	Yes
Burkhard et al. (2012)	-Ecosystem integrity indicators (7) -Ecosystem services indicators (22) -Demand of ecosystem services (22)	dimensionless (ranking)	Burkhard et al. (2012)	Yes	Yes

*The new release of IMPACT WORLD + will include soil functioning indicators based on Cao *et al.*, 2015, which is a further development of Saad *et al.*, 2013.
**for land occupation impact
***including availability of CF's

4.5.2 Description of pre-selected models

The first model evaluated, Brandão and Milà i Canals, 2013, is an updated version of the model currently recommended in the ILCD handbook (Milà i Canals et al., 2007a, 2007b):

- Brandão and Milà i Canals (2013) includes Soil Organic Carbon (SOC) as stand-alone soil quality indicator. SOC is used as a way to approach the productive capacity of the soil, which in turn affects the AoP 'natural resources' and 'natural environment'. Unlike the previous version of the land use framework (Milà i Canals et al., 2007a, 2007b), which characterized impacts only in the United Kingdom, Brandão and Milà i Canals (2013) provide CFs for a global application of the model.

Second, three models were pre-selected that consider soil properties and functions: LANCA (Beck et al., 2010); LANCA application by Saad et al., (2013); and SALCA-SQ (Oberholzer et al., 2012).

- LANCA is an updated version of the model developed by Baitz (2002), which was already evaluated in the ILCD Handbook −although not recommended, partly because of the lack of CFs and the large amount of input data requirement. LANCA calculates indicators for soil functions (e.g. erosion and water regulation, filtration capacity) originally based on site-specific data. LANCA developers have also recently developed CFs directly associated to land use flows (Bos et al., 2016).
- Saad et al. (2013) developed a global application of the LANCA model including some minor methodological modifications and deriving the indicators directly and solely from land use inventory flows.
- SALCA-SQ, also based on site-specific data, focuses on soil properties (e.g. macropore volume, microbial activity), and threats to soil (e.g. erosion, compaction). LANCA and SALCA-SQ do not establish explicit links to endpoint indicators or AoPs.

Next, we included three threat-specific models (Núñez et al., 2010, Garrigues et al., 2013 and Núñez et al., 2013):

- Núñez et al. (2010) calculate a desertification index based on aridity, erosion, aquifer over-exploitation and fire risk.
- Garrigues et al. (2013) focus on soil compaction, as a result of the use of agricultural machinery, calculating auxiliary indicators, such as water erosion and soil organic matter (SOM) change. The model is meant to be part of a broader framework, which should include other processes (e.g. erosion, change in SOM and salinization).
- Núñez et al. (2013) compute the loss of Net Primary Production (NPP) and emergy, as indicators of damage to the 'natural environment' (ecosystems) and resources, respectively. Both indicators are based on the soil loss calculated through the application of the Universal Soil Loss Equation (USLE, Wischmeier and Smith, 1978); and NPP depletion is calculated as a function of SOC loss.

Both Núñez et al. (2010) and Garrigues et al. (2013) use indicators that assess the capacity of the soil to provide ecosystem services and support biodiversity, although these links are not explicitly addressed by the authors. In Núñez et al. (2013), the AoP 'natural resources' is related to soil loss by means of an emergy indicator, which expresses all the energy embodied in the system.

All three models show limitations regarding the availability of CFs: while CFs for Núñez et al. (2013) and Núñez et al. (2010) are not related to land use inventory flows, CFs for Garrigues et al. (2013) are not detailed in the study.

Further, two models based on thermodynamics accounting were pre-selected:

- Alvarenga et al*.*, (2013) compute exergy distinctly for natural and human-made land: exergy of biomass extracted is calculated for natural land covers, while the exergy associated to potential NPP is used for human-made land.
- Alvarenga et al*.*, (2015) focus on the Human Appropriation of Primary Production (HANPP), i.e. the amount of NPP that is not available for nature due to human use of land.

Both exergy and HANPP, as stated by the authors, pose impacts on the 'natural resources' and 'natural environment' AoPs, and the resulting CFs are directly associated to land use flows.

Two models not specifically developed for LCA were selected:

- Gardi et al*.* (2013) developed a composite indicator on pressures to soil biodiversity, which is a weighted index of variables related to land use (agriculture intensity, land use change), threats to soil (compaction, erosion, contamination, SOC loss), and threats to biodiversity (invasive species). This index may serve to approach impacts on the 'natural environment' AoP.
- Burkhard et al*.* (2012) provide a model to score land use types according to a set of ecosystem integrity and ecosystem services indicators. These indicators include, among others, soil functions, water provision, biodiversity loss, and exergy capture. The scores are based on expert judgment and several case studies. The model does not establish any link between indicators but rather calculate them directly and solely associated to each land use type. The model includes also endpoint indicators among ecosystem services indicators (water provision).

4.5.3 Characterization factors at midpoint

All the characterization factors (CFs) available from the different characterisation models have been collected[7]. When needed, we proceeded with the adaptation (mapping) of the CFs to the ILCD elementary flow list.

The availability, geographic coverage, and level of usability of the compiled CFs differ among the pre-selected models (as it is summarized in Table 4.2). Almost models (except SALCA-SQ) provided CFs or an output that could be considered similar to CFs – e.g. the non-LCA model Burkhard et al*.* (2012) which anyway has a scoring system easily adapted to ILCD elementary flows. Yet, usability was not always guaranteed. Only five out of the eleven pre-selected models (Brandão and Milà i Canals (2013), LANCA, Saad et al*.* (2013), Alvarenga et al*.* (2013) and Alvarenga et al*.* (2015)) provide characterization factors that meet the following applicability requirements:

- being associated to land use inventory flows (i.e. land occupation and/or land transformation), which are the flows more easily available for the practitioner
- being associated to a practically usable spatial unit (country, world) – models providing CFs only by ecological/climate region would require an adaptation to be easily incorporated in the LCA software.

[7] In order to have a CFs compilation more complete, CFs from three additional models beyond those pre-selected have been also compiled. This includes the model currently recommended by the ILCD Handbook (Milà i Canals et al., 2007a, 2007b), Cao et al. (2015), which makes a further refinement of the CFs developed by Saad et al. (2013), and De Baan et al. (2013) (as applied in Impact World +), added as representative of endpoint models, and which accounts for the impact of land interventions on biodiversity.

Further, only the CFs for LANCA and Saad et al. (2013) followed a land use classification fully compatible with the ILCD, while the remaining models has required an adaptation to the ILCD nomenclature. As for the geographic coverage, models generally allow for the global application of the characterization, except for SALCA-SQ −site-specific−, Gardi et al., (2013) −only for Europe−, Garrigues et al. (2013) and Burkhard et al., (2012) –both being based on local case studies.

Considering these aspects, LANCA stands out in terms of applicability, with CFs available for the global use of the model as well as for calculating country-specific impacts. Similarly, Saad et al. (2013) has a good applicability although allowing only for the characterization of few generic land use inventory flows.

The coverage of ILCD inventory flows by the models' CFs was low overall with the exception of LANCA and Burkhard et al. (2012). Most models cover only the impacts resulting from land occupation, while both transformation and occupation impacts are estimated only by Brandão and Milà i Canals (2013), the recent further development of LANCA (Bos et al. 2016), and Saad et al. (2013).

Table 4.2 Applicability aspects of CFs of the pre-selected models, which determine their ability to be globally applicable. The models that provide CFs associated to land use flows have been highlighted in bold and with grey background color. Level 4 of land use flows partially incorporates land management practices.

Model	Characterisation factors (CFs) applicability			
	CFs associated to land use flows	Land use flows coverage by the CFs (hierarchical level and compatibility with ILCD flows)*	CFs geographic coverage	CFs spatial resolution
Brandão and Milà i Canals (2013)	Yes, adaptation to ILCD nomenclature required	-Level 2-3 -Adaptation to ILCD nomenclature required	Global	Regional (climatic) and world default
LANCA as in Bos et al. (2016)	Yes	-Level 4 -Compatible ILCD	Global	Country, world default and local (site-specific)
Saad et al. (2013)	Yes	-Level 1 -Compatible ILCD	Global	Regional (biogeographical regions) and world default
SALCA-SQ Oberholzer et al., (2012)	No	-n.a.**	Local (specific for Europe)	Local (site-specific)
Núñez et al. (2010)	No	-n.a. **	Global	Regional (ecoregions)
Garrigues et al. (2013)	No	-n.a. **	Some crops in some countries	Country
Núñez et al. (2013)	No	-n.a. **	Global	Local and country
Alvarenga et al. (2013)	Yes, adaptation to ILCD nomenclature required	-Level 2-4 -Adaptation to ILCD nomenclature partially required	Global	Higher than country (grid size of 5' or 10×10 km at the Equator), and world default
Alvarenga et al., (2015)	Yes, adaptation to ILCD nomenclature required	-Level 2 - Adaptation to ILCD nomenclature partially required	Global	Country and world default
Gardi et al. (2013)	Partly	-Level 1 - Adaptation to ILCD nomenclature partially required	Europe (but easily replicable globally	Local (grid size 1x1 km)
Burkhard et al. (2012)	Yes	-Level 3 -Adaptation to ILCD nomenclature partially required	Local	Local

*Following the recommendations given by Koellner et al. (2013) and consistently with the current ILCD elementary flow list, the classification of land use consists of four levels of detail: Level 1 uses very general land use and land cover classes; Level 2 refines the categories of level 1 (using mainly the classification of ecoinvent v2.0 and

GLOBIO3); Level 3 gives more information on the land management (e.g., irrigated versus non-irrigated arable land), and Level 4 mostly specifies the intensity of the land uses (extensive versus intensive land use).

**n.a. the models propose indicators that make use of specific flows, which differ from those usually adopted at the inventory (e.g. m^2 of a certain land use type)

Significant differences were also observed regarding the ability of CFs to grasp soils impacts associated to each land use intervention, as derived from our cross comparison of CFs values (see Figure 4.2, where it is to be noted that not only original but also mapped values are displayed). Thus, the models proposed by Brandão and Milà i Canals (2013) and Gardi et al. (2013) had the most relevant impact characterization, providing different CFs for many different typologies of land use/cover. The characterization proposed by Alvarenga et al. (2013) was the one with the lowest relevance since it bundles all human-made land use systems into one. The models by Brandão and Milà I Canals (2013) and Gardi et al. (2013) are also able to distinguish between extensive and intensive land uses, allowing also for discriminating between the impacts of production systems based on different land management practices. 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 on erosion resistance by LANCA, for which bare areas pose the strongest impact. The model by Alvarenga et al. (2013) is another exception since, as mentioned above, does not differentiate the impact of the variety of land use flows. Interestingly, CFs values reflecting the impact of agricultural and forest land uses on biotic production differ between the models by Milà i Canals et al. (2007a, 2007b) – based on SOM−, Brandão and Milà i Canals (2013) –based on SOC− and LANCA –based on biotic production. However, this might be partially due to the adaptation of land use flows to the ILCD nomenclature, since CFs provided by the models followed different land use classifications. Finally, it is important to note the strong correlation between indicators of multi-indicators models (LANCA and Saad et al., 2013). This means that, the information they provide might be redundant.

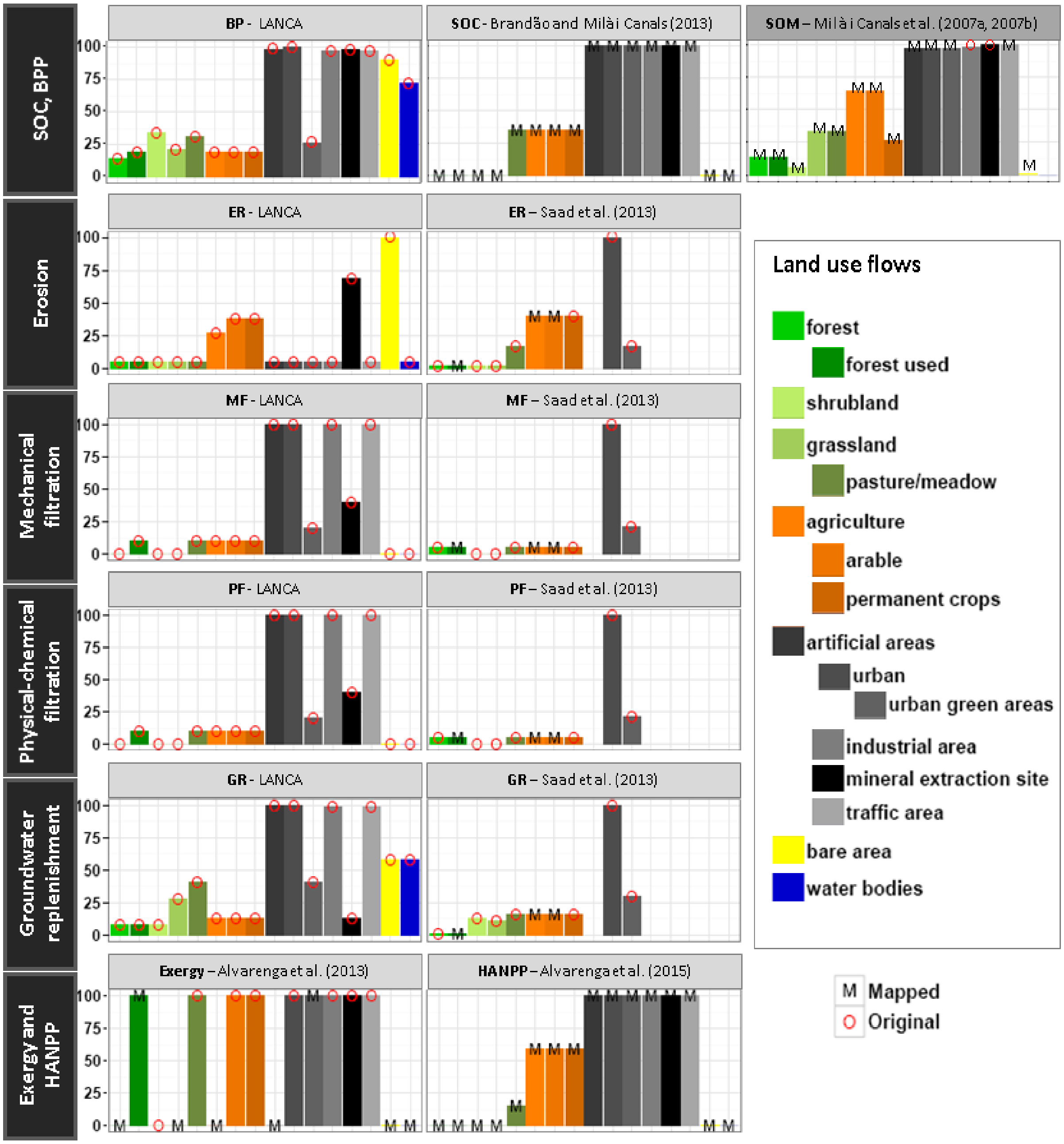


Figure 4.2 Comparison of the land occupation CFs from the pre-selected models and the model by Mila I Canals et al. (2007a, 2007b) – the ILCD recommendation (EC-JRC, 2011). CF's were selected, when available, for a set of representative land use types (according to the ILCD nomenclature, up to a hierarchical level 3). For comparability reasons, rather that the CF values as such, values displayed correspond to the percentage CF value relative to the maximum CF value for each typology of indicators. BP: biotic production; SOC: soil organic carbon; SOM: soil organic matter; ER: erosion resistance; MF: mechanical filtration; PF: physico-chemical filtration; GR: groundwater recharge/replenishment. CFs not requiring the adaptation to ILCD land use flows are marked as "O" (original). Conversely, flows requiring this adaptation appear as "M" (mapped, as this process is denominated in an LCA context). CFs values for Mila i Canals et al. (2007a, 2007b) are the ones reported in the ILCD.

Additionally, a correlation between characterisation factors of those models that had more coverage in terms of elementary flows has been calculated (Tables 4.3 and 4.4). This could help in assessing, for instance, how the common source of some models, e.g. Saad et al. (2013) indicator developed from LANCA (Beck et al., 2010), could influence the magnitude and type of information it provides to land use impact assessment. The Pearson correlation coefficients are calculated by taking into consideration only the elementary flows the models have in common in their original (not mapped) version. This allows for highlighting redundancies or discrepancies inter- and intra-model. Due to the limitation in De Baan et al. (2013), i.e. it provides CFs for Occupation only, this model is not included in the correlation analysis focused on transformation flows.

Erosion resistance (ER) as calculated by LANCA tool turned to be the indicator with the lowest correlation with the others. That is why we assumed its information as quite specific and impossible to extrapolate from the other indicators. On the other hand, correlation coefficients were always very high between mechanical (MF) and physiochemical filtration (PF), both intra- and inter-models. Those two indicators in a comparison would then provide a similar kind of information; hence, in a recommended model just one of them should be maintained, in order to avoid redundancy. Groundwater recharge indicators had a quite similar behaviour but their correlation coefficients were lower with regard to filtration indicators. Therefore, this indicator is expected to provide additional information that is not captured by the other indicators.

Table 4.3 - Pearson correlation between "Occupation CFs" of: SOM, SOC, LANCA, Saad et al. (2013) and de Baan et al. (2013) indicators[8]. Red cells present high positive correlation, light blue cells present lower positive correlation and dark blue cells present negative correlation.

	SOM	**SOC**	**ER**	**MF**	**PF**	**GR**	**BP**	**ER**	**MF**	**PF**	**GR**	**Biodiversity damage potential (e)**
SOM (a)	1.00											
SOC (b)	0.59	1.00										
ER (c)	-0.02	-0.07	1.00									
MF (c)	0.71	0.90	-0.17	1.00								
PF (c)	0.71	0.90	-0.17	1.00	1.00							
GR (c)	0.76	0.85	-0.28	0.92	0.92	1.00						
BP (c)	0.55	0.88	-0.32	0.96	0.96	0.88	1.00					
ER (d)	0.66	0.88	0.23	0.92	0.92	0.81	0.82	1.00				
MF (d)	0.69	0.87	-0.23	0.99	0.99	0.91	0.95	0.88	1.00			
PF (d)	0.69	0.87	-0.23	0.99	0.99	0.91	0.95	0.88	1.00	1.00		
GR (d)	0.73	0.88	-0.19	0.99	0.99	0.93	0.96	0.90	0.99	0.99	1.00	
Biodiversity damage potential (e)	-0.04	0.55	0.52	0.37	0.37	0.16	0.38	0.62	0.31	0.31	0.33	1.00

(a) Milà i Canals, 2007; (b) Brandão and Milà i Canals, 2013; (c) Bos et al., 2016; (d) Saad et al., 2013; (e) de Baan et al., 2013 as applied in Impact World

BP: biotic production; SOC: soil organic carbon; SOM: soil organic matter; ER: erosion resistance; MF: mechanical filtration; PF: physico-chemical filtration; GR: groundwater recharge/replenishment

[8] This correlation is carried out on the flows that are common to all the models taken into consideration. This is why some factors are slightly different from the values reported in correlation focused on SOM/SOC/LANCA

Table 4.4 - Pearson correlation between "Transformation CFs" of: SOM, SOC, LANCA, Saad et al. (2013) and de Baan et al. (2013) indicators[9]. Red cells present high positive correlation, light blue cells present lower positive correlation and dark blue cells present negative correlation.

	SOM	**SOC**	**ER**	**MF**	**PF**	**GR**	**BP**	**ER**	**MF**	**PF**	**GR**
SOM (a)	1.00										
SOC (b)	0.62	1.00									
ER (c)	-0.31	-0.14	1.00								
MF (c)	0.76	0.98	-0.17	1.00							
PF (c)	0.76	0.98	-0.17	1.00	1.00						
GR (c)	0.77	0.88	-0.28	0.92	0.92	1.00					
BP (c)	0.67	0.96	-0.32	0.96	0.96	0.88	1.00				
ER (d)	0.63	0.99	-0.04	0.98	0.98	0.86	0.93	1.00			
MF (d)	0.69	0.99	-0.23	0.98	0.98	0.89	0.97	0.98	1.00		
PF (d)	0.69	0.99	-0.23	0.98	0.98	0.89	0.97	0.98	1.00	1.00	
GR (d)	0.71	0.99	-0.23	0.99	0.99	0.90	0.97	0.98	1.00	1.00	1.00

(a) Milà i Canals, 2007; (b) Brandão and Milà i Canals, 2013; (c) Bos et al., 2016; (d) Saad et al., 2013.

BP: biotic production; SOC: soil organic carbon; SOM: soil organic matter; ER: erosion resistance; MF: mechanical filtration; PF: physico-chemical filtration; GR: groundwater recharge/replenishment

4.5.4 Pre-selection of endpoint models

As there are ongoing activities by UNEP- SETAC life cycle initiative[10] regarding the identification of an endpoint indicators of impact due to land use on biodiversity, JRC was taking part to working group analyzing the different models, as reported in the review by Curran et al. (2016).

In January 2016, a Pellston Workshop™ on "Global Guidance for Life Cycle Impact Assessment Indicators and Models" was held in Valencia, Spain. The goal of the workshop was to reach consensus on recommended environmental indicators and characterisation factors for Life Cycle Impact Assessment (LCIA), including biodiversity. As result of the workshop, model and related characterization factors representing global potential species loss from land use are provisionally recommended (Chaudhary et al. 2016) as suitable to assess impacts on biodiversity due to land use and land use change as hotspot analysis in LCA only (not for comparative assertions). According to UNEP-SETAC recommendations, further testing of the CFs as well as the development of CFs for further land use types are required to provide full recommendation (UNEP, 2016)[11].

[9] This correlation is carried out on the flows that are common to all the models taken into consideration. This is why some factors are slightly different from the values reported in correlation focused on SOM/SOC/LANCA

[10] (Flagship Project 1b) Environmental Life Cycle Impact Assessment Indicators

[11] More details could be found at http://www.lifecycleinitiative.org/reaching-consensus-on-recommended-environmental-indicators-and-characterisation-factors-for-life-cycle-impact-assessment-lcia/

4.6 Models evaluation

The table below shows the summary results of the land use models evaluation. The complete evaluation can be found in Annex 4.1 (separate file).

Table 4.5 Summary of the land use models evaluation results.

	Criteria	Brandão and Milà i Canals, 2013		LANCA		Saad et al., 2013		SALCA-SQ		Núñez et al., 2010		Garrigues et al., 2013	
Science-based criteria	**Completeness of the scope**	B	Model complete in scope; limited link to AoP	A-B	Model complete in scope	B	Model complete in scope; AoP coverage more reduced than LANCA	C	Limited link to AoP due to the very specific nature of indicators; and limited geographic coverage since it is site-specific	C	Limited link to AoP and endpoint	D	Limited link to AoP and endpoint; and limited geographic coverage
	Environmental relevance	C	Moderate LCI flows coverage; good performance in addressing land use-related aspects (intensive uses, occupation and transformation). Focus on one single soil quality indicator and moderate resolution	B-C	Full LCI flows coverage; good performance in addressing land use-related aspects (intensive uses, occupation and transformation). Delivery of 5 soils quality-related indicators. Both high and lower resolution	C	Very limited LCI flows coverage; good performance in addressing land use-related aspects (intensive uses, occupation and transformation). Delivery of 4 soils quality-related indicators and low resolution	C-D	High relevance, very detailed soil information and high spatial resolution (plot level). However, no land use LCI flows stated and not addressing land use-related aspects (e.g. occupation and transformation)	D	Very limited LCI flows coverage; not addressing land use-related aspects (intensive uses, occupation and transformation). Focus on desertification	D	Very limited LCI flows coverage; not addressing land use-related aspects (intensive uses, occupation and transformation). Focus on compaction
	Scientific robustness & Uncertainty	C	Peer-reviewed model; uncertainty of estimates not assessed but many based on validated data sources; model partially up-to-date	C	Peer reviewed model; uncertainty of estimates not assessed; not all underlying models up-to-date	B-C	Peer-reviewed model; uncertainty partially assessed; not all underlying models up-to-date	C	Peer-reviewed model; uncertainty of estimates not assessed; not all underlying models up-to-date	C-D	Peer-reviewed model; uncertainty of estimates not assessed	B-C	Peer-reviewed model; uncertainty partially assessed
	Documentation, Transparency & Reproducibility	A	Well documented model; documentation, CFs and model accessible	B-C	Documentation and CFs accessible; limited access to some input data and no access to the model in an operational manner	B-C	Documentation and CFs accessible; limited access to some input data and no access to the model in an operational manner	C	Documentation accessible; CFs not available; limited access to input data and no access to the model in an operational manner	B	Documentation accessible; CFS available; some limitations in the access to input data and no access to the model in an operational manner	C	Documentation accessible; CFs not available; some limitations in the access to input data; no access to the model in an operational manner
	Applicability	C	LCA compatible; LCI flows available and relatively compatible nomenclature; no normalization factors; LCI flows by climatic regions	B	LCA compatible; LCI flows available and compatible nomenclature; no normalization factors; LCI flows by country and global	C	LCA compatible; LCI flows available and compatible nomenclature; no normalization factors; LCI flows by biogeographical regions	D	LCA compatible; LCI flows required not available since site-specific; no normalization factors	C	LCA compatible; LCI flows required partially available; no normalization factors	C	LCA compatible; LCI flows required partially available; no normalization factors; high spatial resolution of LCI flows

	Criteria	Brandão and Milà i Canals, 2013		LANCA		Saad et al., 2013		SALCA-SQ		Núñez et al., 2010		Garrigues et al., 2013	
	Characterization factors	C	Relevant and usable CFs- although mapping required; values by climatic region and global; partially tested	B	Moderately relevant, usable CFs; values by country and global; partially tested	C	Moderately relevant, usable CFs; values by biogeographical and global; no relevant test	D	Moderately relevant but not usable CFs; partially tested	C-D	Moderately relevant but not usable CFs; high resolution; partially tested	C-D	Moderately relevant but not usable CFs; high resolution; tested
	Overall evaluation of science-based criteria	B-C	Complete in scope with moderate coverage of LCI flows and relevant and usable CFs	B	Complete in scope with full coverage of LCI flows and relevant and usable CFs	C	Complete in scope with limited coverage of LCI flows and usable but moderately relevant CFs	C-D	Limited scope;, comprehensive set of of very specific indicators, no coverage of land use flows and no usable CFs	C-D	Limited scope and very limited LCI flows coverage, with no usable CFs	C-D	Limited scope and very limited LCI flows coverage, with no usable CFs
Stakeholders acceptance criteria	**Overall evaluation of stakeholders acceptance criteria**	C	Results relatively easy to understand; lack of authority endorsement; focus on agriculture	C-D	Results difficult to understand; lack of authority endorsement	D	Results interpretation relatively complex; focused on agriculture only; lack of authority endorsement and limited academic endorsement	C-D	Relatively complex results; focus on agriculture	C-D	Relatively complex results; lack of authoritative body	C-D	Relatively complex results; limited authoritative body endorsement; focus on agriculture
	Final evaluation	B-C	Adequate in terms of scope and relevance, although it still shows some applicability limitations, its use will give continuity to the currently recommended model	B-C	One of the most complete models in terms of scope and applicability, although number of indicators could be reduced; limited approach to organic matter (addressed as NPP); model transparency needs to improve	C	Similar to LANCA but with a more reduced scope and LCI flows coverage	D	Comprehensive set of indicators. Suitable for a site-specific, focused analysis of foreground. Needs further development in terms of applicability	D	The main model limitation is the scope, focused on desertification, which would be more suitable for a complementary analysis. It needs further development in terms of CFs usability and LCI flows coverage	D	Limited scope, focused on soil compaction, that would be more suitable for a complementary analysis, needs further development in terms of coverage of CFs usability and LCI flows coverage

		Núñez et al., 2013		Alvarenga et al., 2013		Alvarenga et al., 2015		Gardi et al., 2013		Burkhard et al., 2012
Completeness of the scope	B	Limited geographic coverage since it is site-specific	B	Model complete in scope; limited AoP coverage	B	Model complete in scope; limited AoP coverage	B-C	Model complete in scope; limited AoP coverage and limited geographic coverage	B-C	Model complete in scope; limited AoP coverage and limited geographic coverage
Environmental relevance	D	No distinction of land use LCI flows coverage; not addressing land use-related aspects (intensive uses, occupation and transformation). Focus on erosion	D	Very limited LCI flows coverage; not addressing land use-related aspects (intensive uses, occupation and transformation). Focus on the NPP component	c-D	Limited LCI flows coverage; not addressing land use-related aspects (intensive uses, occupation and transformation). Focus on the NPP component	C	Limited LCI flows coverage; mostly not addressing land use-related aspects (intensive uses, occupation and transformation). Limited coverage of soil impacts	B-C	Good LCI flows coverage; not addressing land use-related aspects (intensive uses, occupation and transformation). Delivery of a complete set of impact indicators.
Scientific robustness & Uncertainty	C	Peer-reviewed model; uncertainty of estimates not explicitly assessed; underlying models partially up-to-date	C	Peer-reviewed model; limited assessment of uncertainty of estimates; underlying model partially up-to-date	B-C	Peer-reviewed model; limited assessment of uncertainty of estimates	A-B	Peer-reviewed model; comprehensive assessment of uncertainty of estimates	C	Peer reviewed model; uncertainty of estimates not assessed; model partially up-to-date
Documentation, Transparency & Reproducibility	B	Documentation accessible; CFs and input data accessible no access to the model in an operational manner	A-B	Well documented model; documentation, input data and CFs accessible	A-B	Well documented model; documentation, input data and CFs accessible	C	Documentation and some input data accessible; CFs not available	C	Documentation and CFs (model output assimilable to CFs) accessible; no access to the model to the model in an operational manner since it is expert judgement-based
Applicability	C	LCA compatible; LCI flows required partially available; no normalization factors; high spatial resolution of LCI flows	B	LCA compatible; LCI flows available and relatively compatible nomenclature; no normalization factors; high spatial resolution of LCI flows	B-C	LCA compatible; LCI flows available and relatively compatible nomenclature; no normalization factors; LCI flows by country	C	Non LCA model but compatible; LCI flows required partially available; no normalization factors; LCI flows at country and lower resolution level	C	Non LCA model but compatible; LCI flows required available and relatively compatible; no normalization factors; LCI flows for case studies
Characterization factors	C-D	Moderately relevant but not usable CFs; high resolution; partially tested	C-D	Usable CFs but of very low relevance; values by country and at higher resolution; partially tested	C	Moderately relevant and usable CFs; country and at higher resolution; partially tested	C-D	Relevant CFs but limited usability; values by country and at higher resolution; not tested	C-D	Moderately relevant and relatively usable (in the future) CFs; values for specific case studies; not tested
Overall evaluation of science-based criteria	C-D	Relatively complete scope and no coverage of LCI flows, with no usable CFs	C-D	Complete scope but limited coverage of LCI flows, with usable but not relevant CFs	C	Complete scope but very limited coverage of LCI flows with usable and moderately relevant CFs	C-D	Complete scope and limited coverage of LCI flows, with moderately relevant CFs with limited usability; comprehensive uncertainty assessment	C-D	Complete scope and good coverage of LCI flows, with moderately relevant and CFs potentially usable in the future
Overall evaluation of stakeholders acceptance criteria	C-D	Relatively complex results; lack of authoritative body	C-D	Relatively complex results; lack of authoritative body	C	Relatively complex results; lack of authoritative body	B	Relatively complex results	C	Relatively complex results; lack of authoritative body
Final evaluation	C-D	Promising combination of midpoint indicator with a link to damage in the AoP, yet needs further development in terms of environmental relevance	C-D	Although robust and presenting a promising approach, for the time being the model proposes a complex output without straightforward association to land management and no relevant CFs	C-D	The model proposes a complex output and shows limitations regarding environmental relevance	C-D	Promising model in terms of building a potential link between land use midpoint and endpoint indicators, which needs further research in terms of suitability in an LCA context	C-D	A promising, rather complete model in terms of scope, which needs further research in terms of suitability in an LCA context

4.7 Discussion on models evaluation

Current models that could be applicable in LCA are unable to comprehensively depict the multiple impacts derived from land use and land use change. The current evaluation found that none of the models here meets all the features required by the defined criteria. In fact, no model entirely combines a relevant characterization of the multiple impacts on soil with a sufficient applicability in an LCA context. Nevertheless, compared to the evaluation conducted in 2011 in the 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 as well more data-intensive, but their input data is more accessible, as are characterizations factors and the models themselves. In the following, the main finding and conclusions are summarized.

Derived from the results summarized in Table 3, we found that the models SALCA-SQ, as well as the models by Núñez et al. (2010), Alvarenga et al. (2013), and Garrigues et al. (2013) do not appear as suitable for its recommendation since they show important applicability limitations, especially considering their application for foreground processes. The scope of these models shows also limitations: while indicators provided by SALCA-SQ correspond to a very highly disaggregated level of detail, the model by Núñez et al. (2010) focusses only on desertification, and Garrigues et al. (2013) on soil compaction in agriculture. The models by Núñez et al. (2013) and Alvarenga et al. (2015) show less limitations as compare to the previous three models, yet none of them fits the current needs. Conversely, LANCA and the model by Brandão and Milà i Canals, 2013 obtained the best evaluation results since they are rather complete in terms of scope while at the same time overcome the applicability limitations shown by the other models. Yet, the model by Brandão and Milà i Canals (2013), while providing continuity to the currently recommended model, would require an additional effort to be adapted to ILCD flows. Moreover, both LANCA and the model by Brandão and Milà i Canals (2013) show room for improvement in terms of their capacity to grasp differential impacts on the soil derived from different land interventions. In addition, although LANCA model incorporates more indicators on soil functions than Brandão and Milà i Canals (2013), the set of indicators should be reduced since in some cases they provide redundant information that would add more complexity to the impact assessment of land use interventions. The complexity itself of LANCA, consisting on several coupled models, may challenge the acceptance of the model by some stakeholders. However, this is currently the model which has the higher coverage of elementary flows and the best attempt of modelling impact on different soil properties and it is seen as applicable in an EF context.

Apart from that, the model developed by Burkhard et al. (2012) appears as promising approach with a very complete coverage. However, it builds mainly on expert judgment leading to the necessity of improving the way in which scores are backed by evidences. As for Gardi et al. (2013), the model could be an interesting approach in the future, in terms of its capacity to build a link between the mid- and endpoint analysis.

4.8 Recommended default model for midpoint (pilot phase, EF reference package 2.0)

The recommended model for midpoint LCIA for land use in the EF pilot phase is a soil quality index (SQI). The soil quality index builds upon the aggregation of selected indicators from LANCA model (Beck et al. 2010) - as further developed by Bos et al. 2016:

- LANCA Erosion resistance
- LANCA Mechanical filtration
- LANCA Groundwater replenishment

- LANCA Biotic Production

The soil quality index should be applied at country scale, if country specific elementary flows are available. In case country-specific information at inventory are not available, global average CFs are to be used. Details of the aggregation towards the calculation of the soil quality index are reported in the next section.

4.8.1 LANCA model aggregation for calculating the soil quality index

According to the results of the models evaluation and the correlations between the CF's, LANCA model (Bos et al. 2016) was considered the model best suited for assessing impact on soil quality. However, given the high correlation between the different indicators, only some of the indicators proposed by LANCA model were selected (namely the indicators with the lowest correlation coefficients) in order to build an index capable of covering distinct soil properties. Hence, in order to reduce the complexity of the multi-indicator model and to simplify the interpretation of the results, an aggregation system for LANCA to a single score is proposed. The development of the aggregation for calculating the soil quality index included the following steps:

1- Identification of the most representative indicators avoiding redundancy in the type of information they provide. In the case of LANCA model, physicochemical filtration and mechanical filtration showed a very high correlation (i.e. 1). Therefore, in this aggregation the physicochemical filtration was not taken into account.

2- Identification, for each indicator separately, of the minimum and maximum value amongst the global characterization factors for "occupation" elementary flows. Then, these values were respectively replaced by the values 1 and 100.

3- Re-scaling of the remaining occupation CFs to the 1-100 range.

4- As the CFs for "transformation to" flows in LANCA correspond to the "occupation" CFs, and the CFs for "transformation from" flows correspond to the opposite of the "occupation" CFs, by applying the same logic to these flows the rescaled "transformation to" values ranged between 1 and 100, while the rescaled "transformation from" values ranged between -100 and -1.

5- The rescaled values thus obtained for each indicator were aggregated by adding them together in order to obtain just one number for each elementary flow to be used as soil quality index. In the aggregation scheme proposed here, each indicator has the same weight regarding the contribution to the final index (1-1-1-1).

6- Steps 2 to 6 were repeated using the country-specific characterization factors from LANCA® to calculate the soil quality index characterization factors at country level.

The result is a dimensionless single characterisation factors (the soil quality index) attributing to each elementary flow a score (namely, for occupation, ranging from 55.4 to 301 for the global CFs). The soil quality index is expressed in Points (Pt).

It is noteworthy to highlight that, when the four indicators are re-scaled to 1 - 100 range, their new values maintain the same meaning compare to the original indicator, i.e. higher values are associated to higher impacts. This means that, for instance, a high CF value in erosion resistance potential indicates a potentially higher soil loss.

This approach is a flexible way of aggregating even if it does not address modelling uncertainties that may be associated with each impact indicator.

4.9 Recommended default model for midpoint (transition phase, EF reference package 3.0)

During the EF pilot phase, a number of shortfalls in the original LANCA® model and characterisation factors (as used in the calculation of the soil quality index presented in Section 4.8) were identified. This led to:

- the refinement of the original model and characterisation factors, which underpins LANCA® v2.5 (Horn and Maier 2018)
- the update of the aggregation approach used to derive the soil quality index, calculated using the characterisation factors from LANCA® v2.5.

Such activities were the result of a cooperation between EC-JRC and the Frahunofer Institute. The shortcomings identified in the previous version of the LANCA® model (LANCA® v2.3) were mainly related to the modelling of the reference situation. This is the reference state against which the additional damaging effects on nature caused by the studied land uses are measured. In the LANCA® model, the reference situation is considered to be the potential natural vegetation. In the model refinement of LANCA® (leading to version 2.5), the source previously used to derive the global distribution of potential natural vegetation was replaced by a more updated one (FAO, 2012). Furthermore, the calculation of the reference situation in each country was modified to be more representative of the country considered, including considerations on where certain land use activities can and cannot take place (e.g. agricultural activities in desertic biomes). A comprehensive description of the shortcomings identified and the model refinement is presented in detail in De Laurentiis et al. (2019).

The characterization factors from LANCA® v2.5 (Horn and Maier 2018) were then used to build an updated version of the soil quality index, as presented in De Laurentiis et al. (2019). This is the recommended model for midpoint LCIA for land use in the EF transition phase. The soil quality index builds upon the aggregation of selected indicators from the LANCA® model (Beck et al. 2010, Bos et al. 2016) using the characterisation factors presented in Horn and Maier (2018):

- Erosion resistance
- Mechanical filtration
- Groundwater replenishment
- Biotic Production

The soil quality index should be applied at country scale, if country specific elementary flows are available. In case country-specific information at inventory are not available, global average CFs are to be used. Details of the aggregation towards the calculation of the soil quality index are reported in the section 4.9.1.

4.9.1 LANCA® model aggregation for calculating the soil quality index

According to the results of the models evaluation and the correlations between the CFs, the LANCA® model (Bos et al. 2016), in its latest update LANCA® v2.5 (Horn and Maier, 2018), was considered the best suited model for assessing the impact on soil quality. However, given the high correlation between the different indicators, only some of the indicators proposed by LANCA® model were selected (namely the indicators with the lowest correlation coefficients) in order to build an index capable of covering distinct soil properties. In order to reduce the complexity of the multi-indicator model and to simplify the interpretation of the results, an aggregation system for LANCA® to a single score is performed. The development of the aggregation for calculating the soil quality index included the following steps:

1- Identification of the most representative indicators avoiding redundancy in the type of information they provide. In the case of LANCA® model, physicochemical filtration and mechanical filtration showed a very high correlation (i.e. 1). Therefore, in this aggregation the physicochemical filtration was not taken into account.

2- Identification, for each indicator separately, of the value corresponding to the 5th and 95th percentile of the distribution of characterization factors for "occupation" elementary flows (CF^5 and CF^{95}) and application of a cut-off to all the characterization factors smaller than CF^5 and larger than CF^{95} (Table 4.6).

3- Linear re-scaling of the remaining occupation CFs, obtained by calculating the ratio between each value and the CF^{95} and multiplying by 100 (Figure 4.3).

4- The rescaled values thus obtained for each indicator were aggregated by adding them together in order to obtain just one number for each elementary flow. This number represents the characterization factor.

The result is an index attributing to each elementary flow a score (namely, for occupation, ranging from -17 to 165 Pt/m^{2}a for the global set of CFs and from -47 and 318 Pt/m^{2}a for the country-specific set). This approach is a flexible way of aggregating even if it does not address modelling uncertainties that may be associated with each single impact indicator. It is noteworthy to highlight that, when the four indicators are re-scaled, their new values maintain the same meaning compared to the original indicator, i.e. higher values are associated with higher impacts. This means that, for instance, a high CF value in erosion resistance potential indicates a potentially higher soil loss.

Figure 4.3 provides a visualisation of the rescaling process: the estimated probability density function of the global and country specific CFs is represented for each indicator. The original values of the 5th and 95th percentile of the distribution of CFs are provided underneath the plot (in red) and the re-scaled values are provided below (in green). Table 4.6 reports the minimum and maximum of the distribution of original CFs, and of the re-scaled sets of CFs, together with the values of the applied cut-offs. The full list of cases (combinations of country and land use type) excluded by the cut-off criteria for each indicator is provided in the Annex 4.2.

Table 4.6 Overview of the re-scaling technique adopted. BP: biotic production; ER: erosion resistance; GR: groundwater regeneration; MF: mechanical filtration.

Indicator	Original values		Cutoff values		Re-scaled values	
	CF_{MIN}	CF_{MAX}	CF^{5th}	CF^{95th}	CF_{MIN}	CF_{MAX}
BP	-1.93	1.75	-0.54	1.49	-36	100
ER	-8.15	624.9	-0.46	68.57	-1	100
GR	-1.17	1.74	-0.05	0.46	-11	100
MF	0	1149.75	0	255.5	0	100

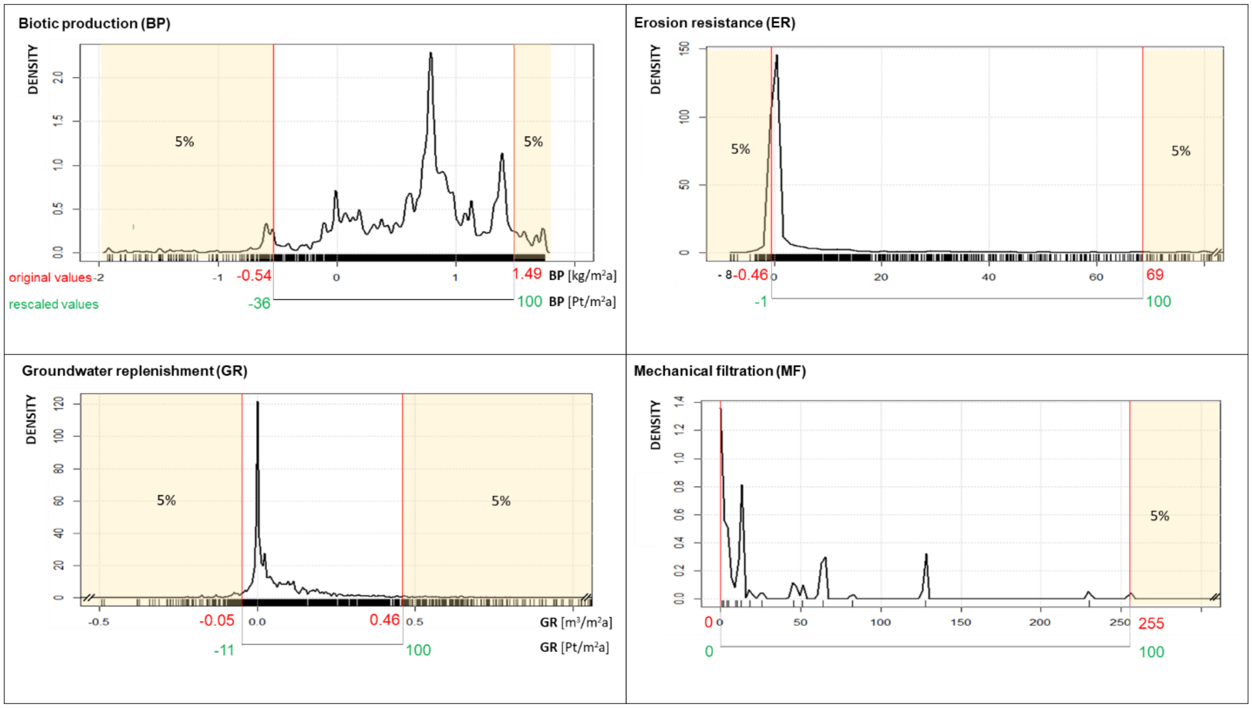


Figure 4.3 Visualisation of the re-scaling technique for biotic production, erosion resistance, groundwater replenishment and mechanical filtration indicators. Black line: kernel density estimation of the country-specific occupation CFs for all land use types. Numbers in red: values of the 5th and 95th percentile of the distribution. Numbers in green: corresponding re-scaled values. Yellow shaded areas: portion of CFs excluded by the applied cut-off.

4.9.1.1 Global and country-specific soil quality index characterization factors

A comparison between the global CFs provided by Horn and Maier (2018) for a selection of six land use types and the soil quality index CFs obtained for each land use type is presented in Figure 4.4. It is possible to see that artificial areas are assigned the highest value of soil quality index (equal to 139 Pt/m^{2}a), having the highest CFs across all impact indicators other than erosion resistance. This is due to the fact that artificial areas have a high sealing factor (a parameter describing the degree of surface sealing caused by different land uses). In contrast, wetlands present the lowest CFs for all impact indicators other than groundwater regeneration, and consequently present the lowest soil quality index (-17 Pt/m^{2}a). In this case, the negative value indicates a potential improvement against the reference situation.

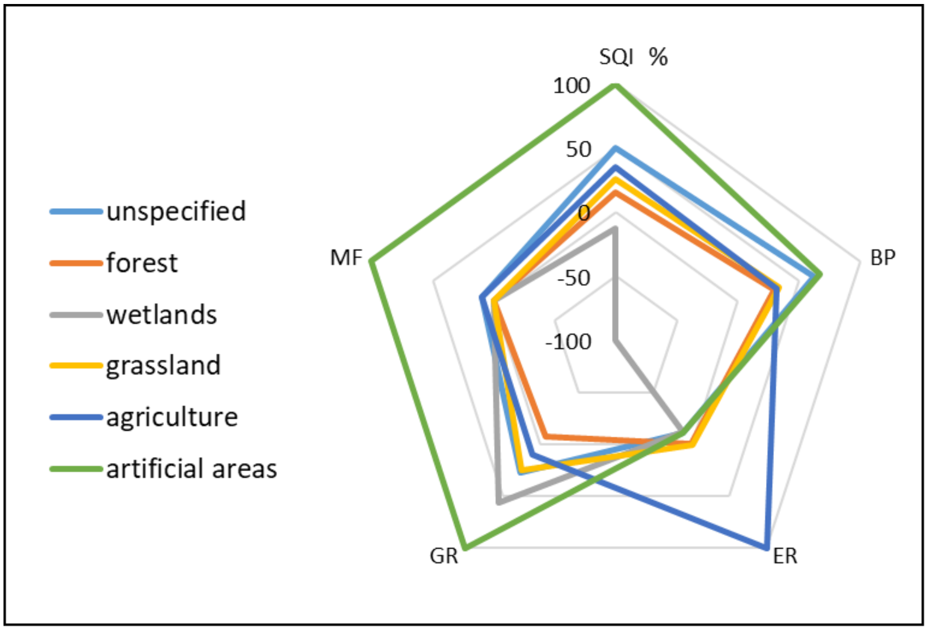


Figure 4.4 Comparison between the global SQI CFs and the global CFs calculated for the single LANCA® impact indicators for land occupation. In each case, the highest CF (absolute value) is taken as a reference (i.e. 100% or -100%) and the others are expressed as percentage with reference to it.

The contribution of the four indicators to the soil quality index varies according to the elementary flow and to the country. In Figure 4.5 a comparison between the soil quality index global default CFs and the CFs calculated for Sweden and Greece is provided for a selection of 12 occupation elementary flows. It is possible to see that the biotic production indicator tends to be predominant over the remaining indicators, the only exception being the case of the global CF for artificial areas, where instead the indicator mechanical filtration is predominant. The ranking of land use types (presented in Figure 4.6) is similar across the two countries presented and overall aligned with the ranking at global level, nevertheless there are some variations across the three sets of CFs. The most obvious difference between the results obtained for Greece and for Sweden is the contribution of the erosion resistance indicator to the soil quality index CFs obtained for the occupation of bare area, construction and mineral extraction sites, agricultural and arable land. In the case of Sweden the erosion resistance indicator does not contribute to the soil quality index, as the original LANCA® model provided extremely low CFs for this indicator. This demonstrates that the soil quality index is able to reflect country specific differences in the relative share of a driver of soil quality impact compared to another.

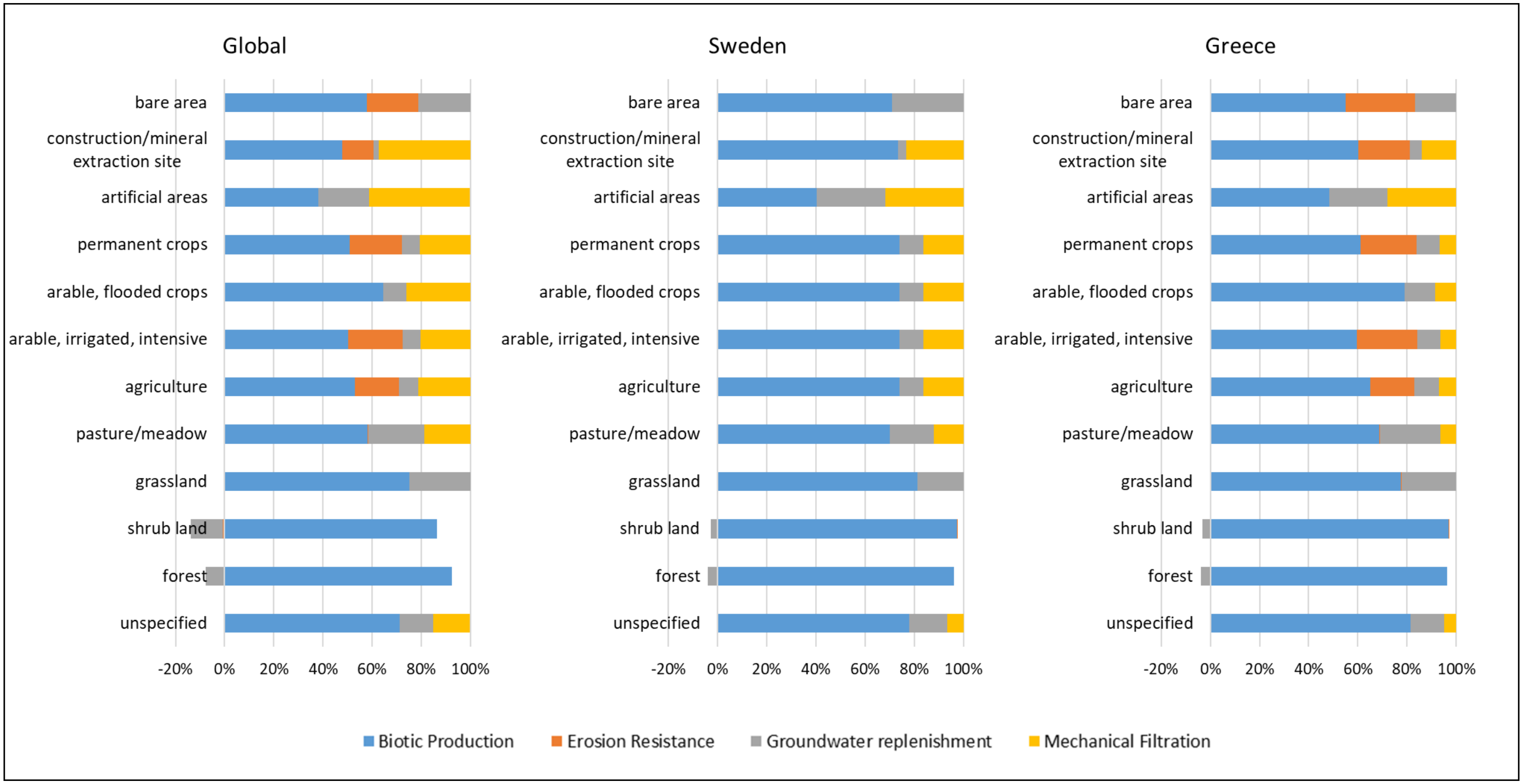


Figure 4.5 Contribution of the four indicators to the soil quality index for a selection of land use types, calculated using the global set of CFs and the country-specific sets for Sweden and Greece; the results are presented as percentages of the total soil quality index CFs.

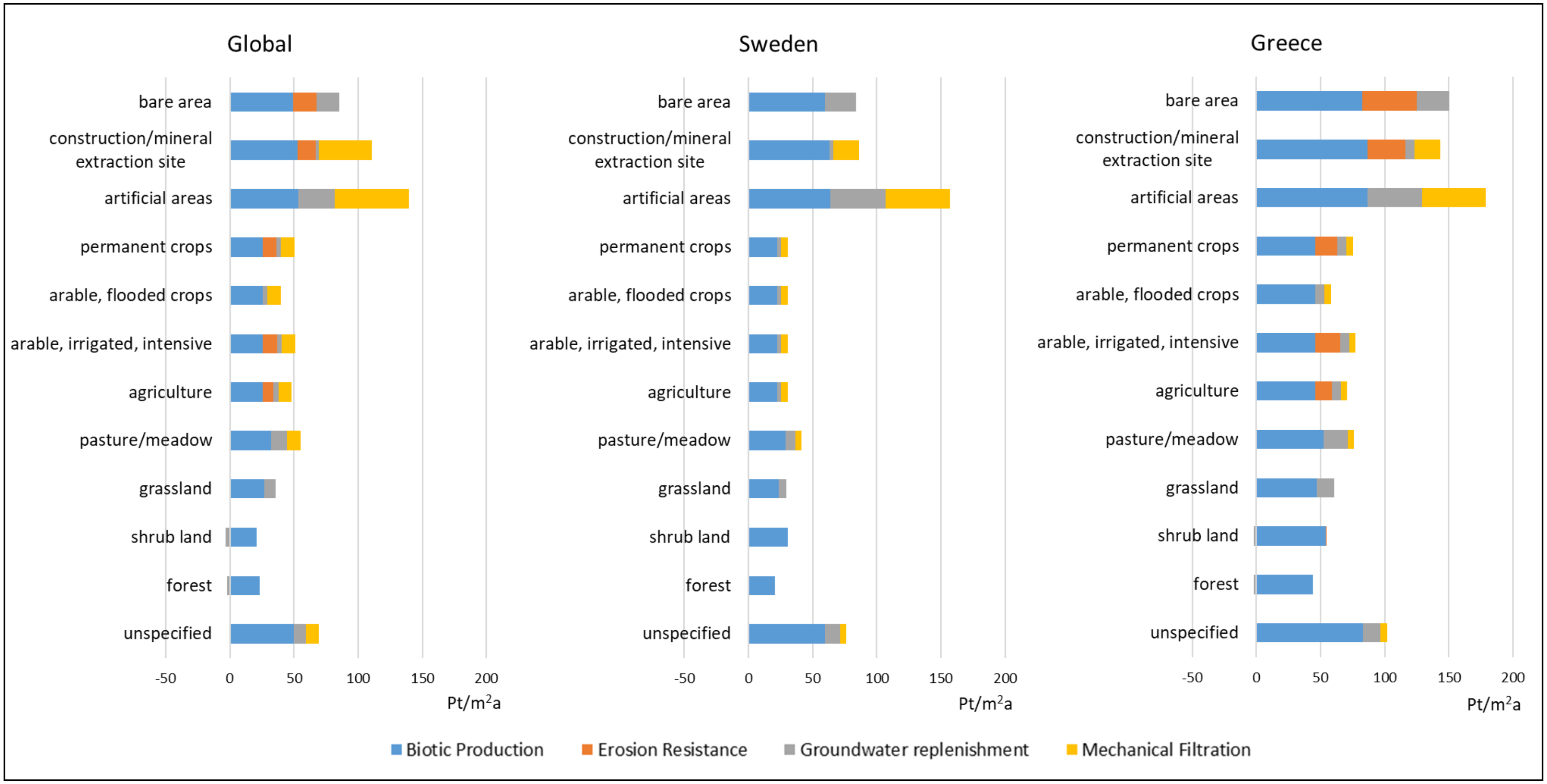


Figure 4.6 Ranking of soil quality index CFs for a selection of land use types, calculated using the global set of CFs and the country-specific sets for Sweden and Greece, and contribution of the different indicators to the soil quality index CFs.

4.9.1.2 Aggregation of land occupation and land transformation impacts

The CF for occupation of a specific land use type (j) in LANCA® is calculated for each indicator (i) (e.g. biotic production) as the ecosystem quality (Q) difference between the reference situation and the respective chosen land use, as illustrated in Equation 1 (Bos et al. 2016). Therefore, a land use activity associated with a low CF is expected to cause a small difference in the ecosystem quality compared to a situation in which it would not take place.

$CF_{occ,i,j} = Q_{i,ref} - Q_{i,j}$ Eq 1

The CFs provided by the LANCA® model can be used to calculate the impacts due to land occupation and land transformation. As can be seen from Table 4.7, the LANCA® CFs have the same unit regardless the type of land use intervention (i.e. occupation, permanent transformation). As the inventory flow for land occupation records the area occupied (A) and the occupation time (T_{occ}), while the inventory flow for land transformation only records the area occupied, the life cycle impact assessment (LCIA) results of land occupation (Eq. 2) and transformation (Eq.3) are not directly additional in case of permanent transformation (Koellner et al. 2013).

Occupation impact = $\Delta Q \times T_{occ} \times A$ Eq 2

Permanent transformation impact = $\Delta Q \times A$ Eq 3

In both equations ΔQ represents the difference in the ecosystem quality between the reference situation and the current (occupation impacts) or prospective (transformation impacts) land use. In both cases, the CF is equal to ΔQ (as illustrated in Equation 1).

In case of reversible transformation, according to Koellner et al. (2013), the impact is calculated by taking into account the regeneration time (T_{reg}) as illustrated by Equation 4. In this case, occupation and transformation impacts have the same unit of measure and, therefore, can be added together.

Reversible transformation impact = $\Delta Q \times T_{reg} \times 0.5 \times A$ Eq 4

The CF for reversible transformation ($CF_{transf,r}$) is, therefore, calculated following Equation 5:

$CF_{transf,r} = \Delta Q \times T_{reg} \times 0.5$ Eq 5

Currently, the LANCA® model only provides CFs for permanent transformations (Table 4.7). Hence, in order to obtain soil quality index CFs for reversible transformations, new CFs were calculated by assuming a regeneration time and following Equation 5. The regeneration time depends on the intensity of the land use type during the transformation phase, on the impact pathway and on the ecosystem type (i.e. warm humid climates favor a faster regeneration) (Koellner et al. 2013). Although there is limited knowledge on ecosystems regeneration times, a number of publications have listed estimations of regeneration times (e.g. Koellner and Scholz 2007, van Dobben et al. 1998).

Therefore, reversible transformation CFs were calculated following Equation 6 and assuming a regeneration time of 20 years for biotic land uses and of 85 years for artificial land uses (sealed land), following Brandão and Mila i Canals (2013).

$CF_{transf,r} = CF_{occ} \times T_{reg} \times 0.5$ Eq 6

Table 4.7 LANCA® impact indicators and soil quality index with related units (De Laurentiis et al., 2019)

Indicator	Land use activity	LCI unit	CF unit	LCIA result unit
Erosion resistance	Occupation	m^2*a/fu	kg soil/(m^2*a)	kg soil loss
	Permanent transformation	m^2/fu		(kg soil loss)/a
Mechanical filtration	Occupation	m^2*a/fu	m^3 water/(m^2*a)	m^3 reduced water infiltration
	Permanent transformation	m^2/fu		(m^3 reduced water infiltration)/a
Groundwater regeneration	Occupation	m^2*a/fu	m^3 groundwater/(m^2*a)	m^3 reduced groundwater regeneration
	Permanent transformation	m^2/fu		(m^3 reduced groundwater regeneration)/a
Biotic production	Occupation	m^2*a/fu	kg biotic production/(m^2*a)	kg reduced biotic production
	Permanent transformation	m^2/fu		(kg reduced biotic production)/a
Soil quality index	Occupation	m^2*a/fu	Pt/(m^2*a)	Pt
	Reversible transformation	m^2/fu	Pt/(m^2)	

4.10 Additional environmental information

Considering the high relevance of biodiversity for many product groups, biodiversity should be addressed separately (in addition to the EF impact categories). Each EF study shall explain whether biodiversity is relevant for the product in scope. If that is the case, the user of the PEF method shall include biodiversity indicators under additional environmental information.

The following suggestions may be taken into account to cover biodiversity:

- To express the (avoided) impact on biodiversity as the percentage of material that comes from ecosystems that have been managed to maintain or enhance conditions for biodiversity, as demonstrated by regular monitoring and reporting of biodiversity levels and gains or losses (e.g. less than 15% loss of species richness due to disturbance, but the PEF studies may set their own level provided this is well justified and not in contradiction to a relevant existing PEFCR). The assessment should refer to materials that end up in the final products and to materials that have been used during the production process. For example, charcoal that is used in steel production processes, or soy that is used to feed cows that produce dairy etc.
- To report additionally the percentage of such materials for which no chain of custody or traceability information can be found.
- To use a certification system as a proxy. The user of the PEF method should determine which certification schemes provide sufficient evidence for ensuring biodiversity maintenance and describe the criteria used. A useful overview of standards is available on http://www.standardsmap.org/.

This applies to both recommended default model for midpoint reported above (i.e. the one adopted during the pilot phase – as reflected in EF reference package 2.0 - and the one for the transition phase - as reflected in EF reference package 3.0).

4.11 Models for endpoint

As explained in section 4.5.4, in line with the results of the UNEP-SETAC flagship initiative on LCIA Pellston workshop (UNEP, 2016), at the endpoint, Chaudhary et al. 2016 may be adopted for hotspots analysis only, in order to assess impact to biodiversity due to land use. Being meant for hotspot analysis only, and not for product comparison, it is considered sufficiently robust to be placed as additional environmental information.

4.12 Consistency between midpoint and endpoint models

As the recommended model at the midpoint level and the more promising models at the endpoint level operate with different environmental impact pathways, there is poor consistency between them. This is identified as a research need for this impact category.

4.13 Classification of the recommended default models

At midpoint, the soil quality index (SQI) -developed aggregating the indicators of LANCA model (Bos et al 2016) as explained in section 4.8.1 (EF pilot phase) and in section 4.9.1 (EF transition phase) - is classified as recommended, but to be applied with caution (Level III).

4.14 Recommended characterization factors

The recommended characterisation factors for the EF pilot phase and for the EF transition phase refer to the soil quality index calculated as reported in section 4.8.1 and section 4.9.1, respectively. These are based on four out of five indicators proposed by Bos et al 2016. Both country- specific and global default CFs are provided via the EPLCA website at at http://eplca.jrc.ec.europa.eu/LCDN/developerEF.xhtml.

4.15 Normalisation factors

Source and data used to calculate the normalisation factors are available in Crenna et al. 2019. The EF normalisation factors to be used are available at http://eplca.jrc.ec.europa.eu/LCDN/developerEF.xhtml.

4.16 Research needs

Based on the review and the assessment of current models, there are a number of research needs that have emerged. An extended analysis is reported also in Vidal Legaz et al. 2017 and in De Laurentiis et al. 2019.

4.16.1 Single or multiple indicators

The need of multiple indicators to assess – thoroughly- soil quality was expressed both by modellers that account for various drivers of impact (LANCA; Saad et al., 2013; SALCA-SQ) as well as by those using only one indicator (e.g. Garrigues et al., 2013). However, our comparison of the CFs shows that the information given by some of these indicators could be redundant, which points out the need of further i) statistical analyses of the redundancy of CF values in multi-indicator models; and, ii) analysis of the sensitivity of results to using multiple instead of a single indicator. Basically, this means answering the question "is the role of each different indicator relevant in the overall contribution of a land use flow to the total impact- which will determine the ranking of two production options-?". Moreover, none of the multi-indicators models provides guidance on how to integrate the different indicators, which introduces an additional complexity in the analysis. Thus, model developers should either provide guidance on how to assess the relevance of each of the multiple indicators, or propose an aggregation algorithm.

4.16.2 Environmental relevance

Current LCA models are hardly able to rank interventions considering if “good” agricultural practices are put in place. 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. This highlights the need for more comprehensive land use flows – although this may encounter data availability limitations both by LCA practitioners and by model developers. To this regard, the CFs of the globally applicable models (Brandão and Milà I Canals, 2013; LANCA, Saad et al*.*, 2013; and Alvarenga et al*.*, 2013 and 2015) 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 characterizations values for many land use types simultaneously. An example of the latter is LANCA, which, although having a full coverage of the ILCD land use flows, assigns the same CF value to all arable land use types.

Moreover, CFs for the calculation of transformation impacts were missing for most models, which will require further development of the models. In addition, 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 following the ILCD handbook criteria. Salinization is also an important threat to soil: although it takes 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). 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 modes, once applicability limitations have been overcome. However, it should be note that a coarse scale, the one available for the applicable models, might be not be adequate for indicators that require a very detailed spatial analysis (Koellner et al*.*, 2013a), e.g. erosion.

Finally, guidance for the calculation of normalization factors should be provided, which was absent from all evaluated models.

4.16.3 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, and none of them model the link to the 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 endpoint indicator, based on Saad et al*.* (2013). Continuing in the line of linking mid- and endpoint, the model proposed by Gardi et al. (2012) could be further explored. Also, NPP and HANPP, indicators used by the Alvarenga et al. (2013, 2015) models – currently with important applicability limitations –, may be used for supporting endpoint modelling covering two AoPs (‘natural environment’ but also ‘natural resources’).

4.16.4 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 required given the needs of the models. To this regard, building the inventory on land use area-related flows is questioned by Helin et al. (2014), who state 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. In addition, given the site-dependent character of soils, the models with site-specific, more accurate, calculations of soil properties and functions required inventory data such as soil data, climate, location, etc.

4.16.5 Land use, climate change and resource use: sharing elements of the impact pathways

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

4.17 References of the chapter on land use impacts

- 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. The International Journal of Life Cycle Assessment 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 model. The International Journal of Life Cycle Assessment 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.
- Baulcombe, D., Crute, I., Davies, B., Dunwell, J., Gale, M., Jones, J., Pretty, J., Sutherland, W. Toulmin, C., (2009). Reaping the benefits: science and the sustainable intensification of global agriculture. London, UK: The Royal Society. ISBN: 978-0-85403-784-1.
- Beck, T., Bos, U., Wittstock, B., Baitz, M., Fischer, M., Sedlbauer, K., (2010). 'LANCA Land Use Indicator Value Calculation in Life Cycle Assessment – Model 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., Milà i Canals, L., (2013). Global characterisation factors to assess land use impacts on biotic production. The International Journal of Life Cycle Assessment 18(6), 1243−1252.
- Burkhard, B., Kroll, F., Nedkov, S., Müller, F., (2012). Mapping ecosystem service supply, demand and budgets. Ecological Indicators 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. Journal of Cleaner Production 94, 56–66.
- Chaudhary, A., Pfister, S., Hellweg, S. (2016). Spatially Explicit Analysis of Biodiversity Loss due to Global Agriculture, Pasture and Forest Land Use from a Producer and Consumer Perspective. Environmental science & technology 50 (7), 3928–3936
- Crenna E, Secchi M, Benini L, Sala S (2019) Global environmental impacts: data sources and methodological choices for calculating normalisation factors. International Journal of LCA, doi.org/10.1007/s11367-018-1534-9
- Curran, M.P., Souza, D.M., Antòn, A., Teixeira, R.F.M., Michelsen, O., Vidal-Legaz, B., Sala, S., Milà i Canals, L., (2016). How well does LCA model land use impacts on biodiversity? —A comparison with approaches from ecology and conservation. Environmental science & technology 50 (6), pp 2782–2795
- De Baan, L., Alkemade, R., Koellner, T. (2013). Land use impacts on biodiversity in LCA: a global approach. The International Journal of Life Cycle Assessment 18, 1216-1230.
- De Laurentiis V, Secchi M, Bos U, Horn R, Laurent A, Sala S (2019). Soil quality index: exploring options for a comprehensive assessment of land use impacts in LCA. Journal of Cleaner Production 215, 63-74
- 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). Models for Assessing Soil Quality. Madison, Wisconsin, USA.
- Doran, J.W., (2002). Soil health and global sustainability: translating science into practice. Agriculture Ecosystem and Environment 88, 119–127.
- European Commission, Joint Research Centre (EC-JRC), (2010). International Reference Life Cycle Data System (ILCD) Handbook - Framework and Requirements for Life Cycle Impact Assessment Models and Indicators. EUR 24586 EN. Luxembourg. Publications Office of the European Union.
- EC-JRC (2010a). ILCD Handbook - Analysis of existing Environmental Impact Assessment methodologies for use in LCA - Background Document to the ILCD. Luxemburg. Publications Office of the European Union.
- EC-JRC (2010b) - Institute for Environment and Sustainability: International Reference Life Cycle Data System (ILCD) Handbook - Nomenclature and other conventions. EUR 24384 EN. Luxembourg. Publications Office of the European Union
- EC-JRC, (2011). Reference Life Cycle Data System (ILCD) Handbook- Recommendations for Life Cycle Impact Assessment in the European context. EUR 24571 EN. Luxemburg. Publications Office of the European Union.
- Fageria, N. K., (2012). Role of soil organic matter in maintaining sustainability of cropping systems. Commun. Soil Sci. Plant Anal. 43(16), 2063−2113.
- Gardi, C., Jeffery, S., Saltelli, A., (2013). An estimate of potential threats levels to soil biodiversity in EU. Global Change Biology 19, 1538–1548.
- Garrigues, E., Corson, M., Angers, D., van der Werf, H.G., Walter, C., (2013). Development of a soil compaction indicator in life cycle assessment. The International Journal of Life Cycle Assessment 18, 1316–1324.
- 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. The International Journal of Life Cycle Assessment 19, 770–785.

- Horn, R., Maier, S., 2018. LANCA®- Characterization Factors for Life Cycle Impact Assessment, Version 2.5. November 2018. Available at: http://publica.fraunhofer.de/documents/N-379310.html. Accessed October 2018.
- Koellner, T., de Baan, L., Beck, T., Brandão, M., Civit, B., Margni, M., i Canals, L., Saad, R., de Souza, D., Müller-Wenk, R., (2013a). UNEP-SETAC guideline on global land use impact assessment on biodiversity and ecosystem services in LCA. The International Journal of Life Cycle Assessment 18, 1188–1202.
- 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., (2013b). Principles for life cycle inventories of land use on a global scale. The International Journal of Life Cycle Assessment 18, 1203–1215.
- 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.
- Milà i Canals, L., Romanyà, J., Cowell, S.J., (2007a). Model for assessing impacts on life support functions (LSF) related to the use of 'fertile land' in Life Cycle Assessment (LCA). Journal of Cleaner Production 15, 1426–1440.
- Mila i Canals, L., (2007b). LCA methodology and modeling considerations for vegetable production and consumption. Centre for Environmental Strategy, University of Surrey.
- Milà i Canals, L., Bauer, C., Depestele, J., Dubreuil, A., Knuchel, R.F., Gaillard, G., Michelsen, O., Müller-Wenk, R., Rydgren, B., (2007c). Key elements in a framework for land use impact assessment within LCA. The International Journal of Life Cycle Assessment 12(1), 5e15.
- Niemi, G. J., Johnson, L. B., Howe, R. W., (2015). Environmental Indicators of Land Cover, Land Use, and Landscape Change. In Environmental Indicators (pp. 265-276). Springer Netherlands.
- Núñez, M., Civit, B., Muñoz, P., Arena, A., Rieradevall, J., Antón, A., (2010). Assessing potential desertification environmental impact in life cycle assessment. The International Journal of Life Cycle Assessment 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. The International Journal of Life Cycle Assessment 18, 755–767.
- Oberholzer, H.-R., Freiermuth Knuchel, R., Weisskopf, P., Gaillard, G., (2012). A novel model for soil quality in life cycle assessment using several soil indicators. Agronomy. Sustainable Development 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.,& Huizenga, D. (eds). Proceedings of the 9th International Conference on LCA in the Agri-Food sector. American Center for Life Cycle Assessment.
- 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. The International Journal of Life Cycle Assessment 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? Global Change Biology 21, 32-47.
- UNEP, (2016). Global guidance for life cycle impact assessment indicators. Volume 1. ISBN: 978-92-807-3630-4. Available at: http://www.lifecycleinitiative.org/life-cycle-impact-assessment-indicators-and-characterization-factors/
- Van Oudenhoven, A. P., Petz, K., Alkemade, R., Hein, L., de Groot, R. S., (2012). Framework for systematic indicator selection to assess effects of land management on ecosystem services. Ecological Indicators 21, 110-122.
- Vidal Legaz B., De Souza D. M., Teixeira R. F. M., Antón A., Putman B., Sala S. (2017). Soil quality, properties, and functions in Life Cycle Assessment: an evaluation of models. Journal of Cleaner Production 140 (2), 502-515.
- Wackernagel, M., (2014). Comment on "ecological footprint policy? land use as an environmental indicator". Journal of Industrial Ecology 18 (1), 20-23.
- Wischmeier, W.H., Smith, D.D., (1978). Predicting rainfall erosion losses—a guide to conservation planning. Agricultural Handbook no. 537. Department of Agriculture, USA.

5 Impact due to Water use

5.1 Introduction

Water is a fundamental resource unevenly distributed across the globe. According to UNEP (2012) the world is entering a period of growing water scarcity and estimates show that by 2030, global demand for water could outstrip supply by over 40% if no changes are made. Wada et al. (2013) report that over the past 50 years human water use has more than doubled and affected streamflow over various regions of the world, increasing frequency and intensity of low flows in rivers and streams over Europe, North America and Asia. Moreover, as reported by the European Environment Agency (EEA, 2015) '*population growth, demand for food and climate change are expected to create significant threats to freshwater availability (Murray et al., 2012). Scenarios on global food demand for 2050 point to severe water stress in many regions, even if strong efficiency gains in its use are made (Pfister et al., 2011), implying threats to both human water security and to the functioning of ecosystems. Already today, around half of the world's major river basins, home to 2.7 billion people, face water scarcity in at least one month a year (Hoekstra et al., 2012) and water restrictions are projected to be further amplified by climate change'*.

Rockström et al. (2009) had recently proposed a set of planetary boundaries, including a maximum amount of freshwater (or 'blue' water) that can be appropriated by humans without '*significantly increase the risk of approaching green and blue water-induced thresholds (collapse of terrestrial and aquatic ecosystems, major shifts in moisture feedback, and freshwater/ocean mixing)'*. Gerten et al. (2013) had recently improved the calculation of the planetary boundary for water, finding that the threshold for water is being approached rapidly. As a consequence, '*cascading macro-impacts due to shifts in the hydrological cycle may result in yield declines or even collapses of rainfed or irrigated agricultural systems'* (Rockström et al. 2009) or in '*collapses of riverine, estuary, limnic and coastal ecosystems as a consequence of excessive blue water consumption or other forms of streamflow and lake level reduction'* (Gerten et al., 2013).

Addressing water scarcity and increasing water-use efficiency is also included within the United Nations' Sustainable Development Goals (UN, 2015) – Goal 6: *Ensure availability and sustainable management of water and sanitation for all,* and one of the specific targets to be reached by 2030 is to '*substantially increase in water-use efficiency across all sectors and ensure sustainable withdrawals and supply of freshwater to address water scarcity and substantially reduce the number of people suffering from water scarcity.* According to UNEP (2012) several complementary tools to the quantification of water uses and their environmental impacts are needed at several levels of water management. They have been identified by UNEP (2012) as: i) statistical water accounting on a macroeconomic level and as input-output analysis; ii) Water Footprint Assessment (WFA); and, iii) Water-use assessment and impact assessment in the context of Life Cycle Assessment (LCA). The latter two had been indicated as mutually complementary by scholars in the field (Boulay et al., 2013; Hoekstra, 2015) although several criticisms had been raised on specific aspects of both WFA and LCA methodologies (e.g. Pfister and Hellweg 2009; Hoekstra et al., 2009; Ridoutt and Huang, 2012; Hoekstra and Mekonnen, 2012; Hoekstra, 2016).

The recently published ISO standard on water footprint (ISO 14046:2014) aims at developing a basis for improved water management, providing guidance towards the application of water footprint based on life-cycle assessment to products and services. In this document, only life cycle impact assessment (LCIA) models addressing the impacts of water consumption were considered for evaluation in the context of the Product and Organization Environmental Footprint (PEF/OEF) (EC 2013a; 2013b).

5.2 Framework and scope

LCA and water-related assessments are two fields of their own with a large variety of indicators. Water stress, water scarcity, social water stress, aridity and other water-related indexes abound and have emerged coming from several branches of science, from water management, to ecology, social sciences and LCA. Many of these indices and models have been reviewed (Kounina et al. 2013; Brown et al. 2011; Boulay et al. 2015a) and others keep on emerging (Gleeson et al., 2012; Gassert et al. 2013; Loubet et al., 2013, Berger et al., 2014; Wada & Bierkens 2014).

A clear need of updating this impact category in the context of the Product and Organization Environmental Footprint (PEF/OEF) (EC 2013a; 2013b), including the update of its definition, has emerged because of the following:

- a significant number of LCIA models assessing this impact category were published after 2009;
- two important international initiatives took place: the launch of the UNEP/SETAC Life Cycle Initiative on Water Use in LCA (WULCA) (ongoing) and the publication of the ISO standard on water footprint ISO 14046;
- the fact that the previous ILCD recommendations had identified this impact category as one amongst those classified as Level III – i.e. 'recommended but to be applied with caution';
- the fact that well-founded criticism was brought forward against the current recommendation (Finkbeiner 2014).

The ILCD recommendations (EC-JRC, 2011) for models assessing 'water depletion' were developed by evaluating life cycle impact assessment models against criteria designed for assessing abiotic resources. This was justified by given the limited number of LCIA models available on the subject of water use at the time of the evaluation. No specific framework and criteria were developed for this impact category. In order to overcome this limitation a framework for water use was developed in this document, together with specific criteria aimed at evaluating recently published LCIA models assessing midpoint potential impacts associated with water scarcity, building on the ISO 14046 definition of water scarcity. This was performed building on the outcomes of the UNEP/SETAC LC Initiative – WULCA, as well as in collaboration with members of the WULCA working group - consensus-based indicator. In order to limit the scope of this analysis for recommendation of models, a selection of model was performed based on the following criteria: LCA relevance and perspective adopted, both described in section 5.5. Other indicators which had been developed in literature for non-LCA applications are valuable models which should be considered in future assessments in case these indicators will be made more relevant, and applicable to LCA as well as robust.

Water resources types and uses in LCA

Kounina et al. (2013) and other authors (Milà i Canals et al. 2009, Bayart et al. 2010) identified four types of water resources which are currently used in LCA to model water flows: surface water (river, lake, and sea), groundwater (renewable, shallow, and deep), precipitation (or water stored as soil moisture - also called green water), and fossil groundwater, referring to groundwater coming from fossil aquifers. Another way of categorizing water resources in green, blue, and grey types was proposed by Hoekstra et al., (2011), where green water represents the water stored as soil moisture and available for evaporation through crops and terrestrial vegetation, blue water being surface or groundwater available for abstraction and grey water being a virtual amount of water which should be used to dilute pollutants in water released to water bodies so for the concentration of major pollutants being below specified thresholds.

Different types of water uses were identified by Bayart et al. (2010), on the basis of the work of Owens (2002) and Milà i Canals et al. (2009), as *in-stream* and *off-stream*, and as *consumptive* (evaporative) or *degradative* (non-evaporative) uses. The first differentiation refers to either the use of water in situ (e.g. navigation, turbine use) or off the site (e.g. pumping or diversion of water for agriculture, industry or households needs). The latter distinction specifies whether water resources are withdrawn and discharged into the same watershed, with alteration of the water quality, or the release into the original watershed doesn't occur because of evaporation, product integration, or discharge into different watersheds or the sea. According to Kounina et al. (2013) the impact of degradative use can be defined as withdrawal of surface of groundwater at a given quality followed by release at another quality. Instead, 'borrowing' of water resources refers to the process for which water is withdrawn and released into the same watershed without changes in water quality (e.g. turbine water) (Flury et al., 2012).

As suggested by Boulay et al. (2011a) and Kounina et al. (2013), in addition to the type of source (e.g. surface or groundwater) water resources could also be classified by quality parameters including organic and inorganic contaminants and users for which a particular type of water can be of use.

5.3 Environmental mechanism (cause-effect chain)

The use and consumption of water might lead to impacts at the level of all of three areas of protection (AoPs) defined by Jolliet et al., (2004): human health, ecosystem quality and natural resources. The underlying impact pathways and frameworks have been described by several authors (Milà i Canals et al. 2009, Bayart et al. 2010, Kounina et al. 2013, Loubet et al., 2013, Boulay et al. 2015a, 2015c). According to Milà i Canals et al. (2009) the direct use of freshwater, groundwater and changes in land use may lead to reduced availability of water for other users (i.e. deprivation), locally lower levels of rivers and lakes with effects on aquatic ecosystems, and ultimately impacts on human health due to insufficient water availability and poor water quality. Along similar lines, Bayart et al. (2010) identified impact pathways based on three elements of concern, namely: sufficiency of freshwater resources for contemporary human users, sufficiency of freshwater resources for existing ecosystems, sustainable freshwater resources for future generations and the future use of present-day generations.

Quality aspects were recognized as relevant by Milà i Canals et al. (2009) and discussed in detail by Bayart et al (2010), Boulay et al. (2011a; 2011b) and Kounina et al. (2013). According to Kounina et al. (2013), both the degradative use and the consumption of water can lead to water deprivation for other users because of: changes in availability (scarcity), modifications of functionality (i.e. degradation), reduction of the renewability rate as well as because of the fact that water resources have an ecological value, where the water ecological value is defined as the physical relation to, and dependency of, ecosystems on freshwater (Bayart et al. 2010). The cause-effect chain diagram identified by Kounina et al. (2013) is further elaborated in this work (Figure 5.1) based on the latest findings from the WULCA working group of the UNEP-SETAC Life Cycle Initiative[12] (Boulay et al., 2015a; 2015b; 2015c). A description of the impact pathway associated with the human intervention: 'water consumption' and associated LCIA models is provided in the sections below.

According to the scheme (Figure 5.1), an additional consumption of water in a watershed leads to a reduction in availability in comparison to a given reference state and potentially affects downstream users, therefore generating impacts at the level of the AoPs (arrows in green, purple and orange in Figure 5.1). In general, impacts due to consumption of water resources are generated by the appropriation of water resources by one or more user(s), which leads to the reduction of availability for others, most typically being down-stream users,

[12] www.wulca-waterlca.org

ecosystems or humans (Kounina et al. 2013). Such pathway is described by arrows linking water consumption as (inventory section of the diagram) to a set of state variables such as flow quantity, groundwater table level, flow regimes, etc. (cyan box in Figure 5.1), which water consumption would have an impact to (midpoint impact) and further connected to AoPs.

The extent to which such reduction in availability leads to deprivation depends on the specific needs of the users (Bayart et al., 2010; Boulay et al., 2011b), as water of a given quality is no longer available anymore for specific user (Kounina et al., 2013). Impacts on human health are generated when deprivation occurs for specific human uses, which might need water resources with specific qualities for different uses (e.g. drinking, sanitation, irrigation, production of goods, etc.), as modelled by Pfister et al., (2009), Boualy et al. (2011), Motoshita et al. (2010a; 2010b; 2014). The severity of such impacts varies according to the level of water scarcity and competition within a specific region, as well as on other socio-economic parameters characterizing the society and its ability to avoid, compensate or buffer deprivation.

Similarly, the deprivation of water resources of water flows and funds, might severely affect terrestrial and freshwater ecosystems and reducing their quality. Reduced flow quantities, modified flow regimes, groundwater levels and changes in the availability of water quality on which ecosystems rely upon are amongst the most frequent causes of freshwater ecosystems degradation, with variable intensity depending on their vulnerability to such stress.

When over-use occurs (i.e. the use of a renewable resource beyond its renewability rate), then impacts affects future generations as the resource becomes, under a set of assumptions, unavailable for use in the future. The impacts accounted within the area of protection “natural resources” are associated to the concept of depletion, as future generations might be deprived because of today’s excessive use of water resources; this is typically the case for fossil (non-renewable) groundwater and for groundwater resources characterized by a low natural recharge rate.

Generic midpoint models and indicators

By adopting a mechanistic perspective, impact assessment models should aim at quantifying the extent to which an increase in water consumption leads to users’ deprivation. However, as a matter of facts, the majority of LCIA models assessing water scarcity (or stress, deprivation, depletion) make use of scarcity/stress/deprivation/depletion indices (i.e. the generic midpoints – black dotted box in Figure 5.1) as proxies of severity of the phenomenon they are referring to (e.g. water scarcity, stress, deprivation or depletion) so to characterize water consumption. Therefore, the assumption underlying generic midpoint models is that the impact generated by the consumption of water in a region is proportional to the level of water scarcity, stress, deprivation or depletion in that given region. Few exceptions to this are represented by ecosystem endpoint models, in which mechanistic impact pathways are modelled instead. Modelling according to a mechanistic approach would require instead a thorough modelling of the watersheds and water users at a very detailed scale, a level of detail, which is not currently compatible with the need for global coverage of LCIA models.

Water scarcity is defined as a situation in which water use is approaching or exceeding the natural regeneration of water in a given area, and it is considered by several LCIA models a parameter leading to freshwater deprivation by limiting freshwater availability (Kounina et al., 2013). Different terminologies can be found in LCIA literature i.e. stress, deprivation or depletion, all of them sharing a meaning similar to the one of scarcity, with specific nuances. For instance, a watershed is highly stressed when scarcity is high and deprivation or (long-term) depletion are likely to be high as well.

Several types of generic midpoint models have been described by Kounina et al. (2013) on the basis of their specificity (i.e. positioning towards an area of protection or generic towards all AoPs) and approaches to the determination of scarcity i.e. withdrawal-to-availability (WTA) ratio (such as Pfister et al. 2009; Ridoutt and Pfister 2010b; Frischknecht et al., 2009; Milà i Canals et al. 2009; Frischknecht and Büsser Knöpfel 2013; Pfister and Bayer 2013; Motoshita et al. 2014) or consumption-to-availability (CTA) ratio (Boulay et al., 2011b; Hoekstra et al., 2012; Loubet et al., 2013; Berger et al., 2014). The two most recent models, Yano et al., (2015) and Boulay et al. (2018) build on different rationales than WTA and CTA. The model proposed by Yano et al. (2015) express scarcity in terms of land or time equivalents needed to obtain a reference volume of water, by distinguishing between rainfall, surface water and groundwater.

According to the WULCA working group of the UNEP-SETAC Life Cycle Initiative, if a generic midpoint indicator has to be used for water footprinting in LCA for assessing water scarcity, it should better allow for generic quantification of potential water deprivation on water users and independently of which user is affected. Therefore, as outcome of the WULCA initiative, Boulay et al. (2016) proposed a model based on the residual available water after demand by humans and ecosystem is met, representing a proxy of potential deprivation occurring to any of the two users as a consequence of increased consumption in water.

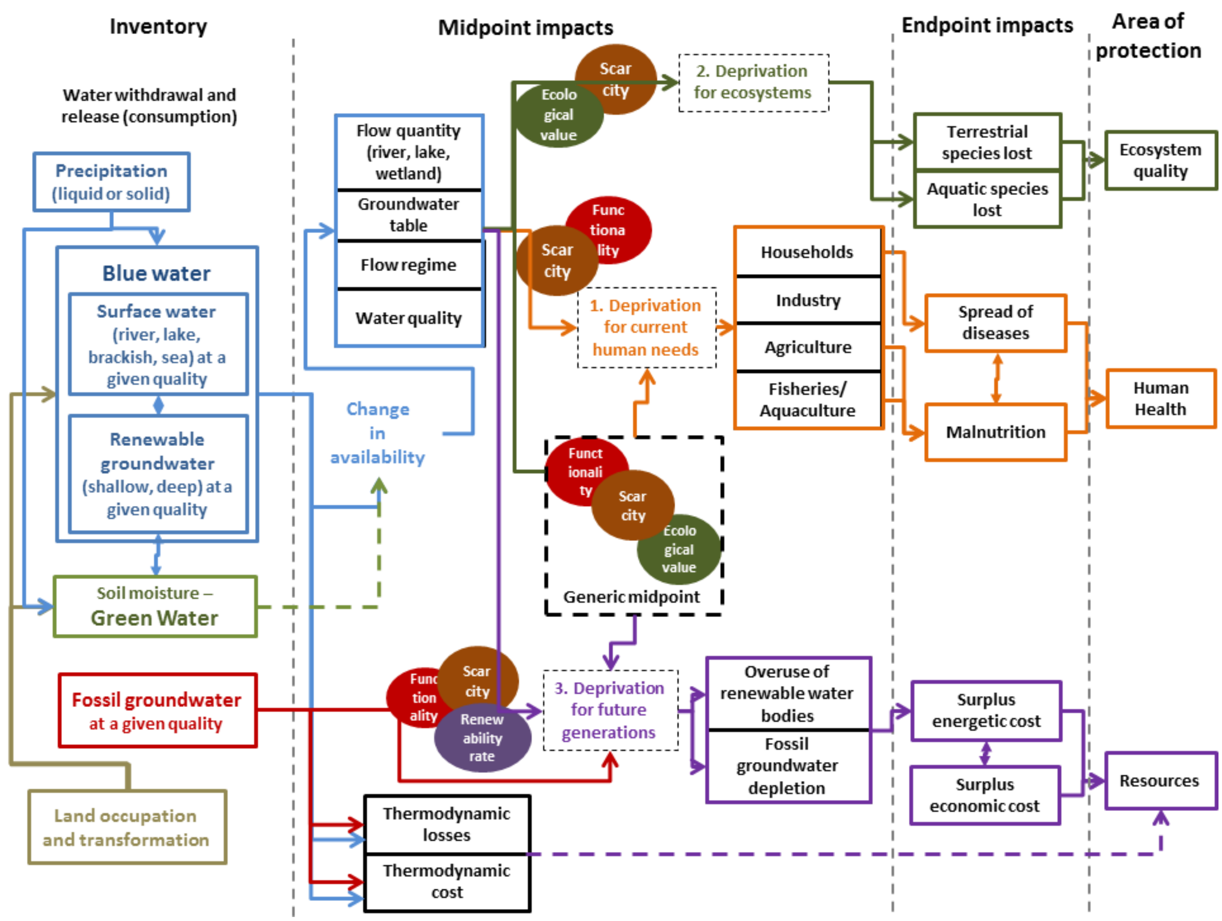


Figure 5.1: Cause-effect chains leading from the inventory to the midpoint and endpoint impacts (modified from Kounina et al., 2013). Continuous arrows identify the impact pathways associated with blue waters (blue), with fossil groundwater (red arrows) and with green-water (green arrows). Dashed arrows represent hypothesized links based on literature, but not modelled yet by any LCIA model. Boxes represent indicators linked to the areas of protection natural resources (in violet), ecosystems quality (in green) and human health (orange) or to inventory flows freshwaters (blue), green-water (blue), fossil groundwater (red) or land

occupation and transformation (yellow). Circles define variables, which are included in one or more LCIA model to perform the characterization of the impacts.

Impacts on Human Health

As reported by Kounina et al. (2013), current endpoint models (Motoshita et al., 2010; 2011; Boulay et al., 2011b; Pfister et al. 2009; Motoshita et al., 2014) agree that the way human health is affected by water use depends on the level of economic development and welfare (Boulay et al., 2011b; Bayart 2008; Motoshita et al., 2014) and their capability to develop backup technologies. According to Kounina et al. (2013) if the level of economic development is not sufficient to introduce compensation mechanisms (e.g. desalination), freshwater use will lead to water deprivation for a set of societal functions. The functions identified in literature are: domestic use (hygiene and ingestion), agriculture, and aquaculture/fisheries, whereas industrial functions are assumed to be more likely to consider compensation strategies. Water quality degradation leads to water deprivation when it creates a loss of functionality for users who need water at a higher quality level than the released one. The withdrawal of freshwater represents an adverse impact depriving users from a given amount of water at ambient quality; the released freshwater results in a burden reduction by making available water for users capable to use water at that quality. Current endpoint models express aggregated impacts on human health through disability-adjusted life years.

Impacts on Ecosystem Quality

As depicted in Figure 5.1, water use can affect ecosystems by changes in the river, lake, or wetland flow quantity; changes in the level of groundwater table; changes in flow regimes; and loss of freshwater quality. Similarly, to human health, degradation corresponds to the consumption of freshwater of a given quality and release of freshwater with lower quality. The midpoint impacts related to freshwater deprivation eventually lead to species diversity change in aquatic and terrestrial ecosystems. Endpoint models such as Van Zelm et al. (2011), Hanafiah et al., (2011) express such impacts as potentially disappeared fraction of species (PDF) in an area (or volume) in a year, whereas Verones et al. (2013) propose a different approach able to account for absolute loss of species due to wetland area loss, including the species vulnerability. Milà i Canals et al. (2009) suggest that changes caused by production systems on the amount of rainwater available to other users through changes in the fractions of rainwater that follow infiltration, evapotranspiration and runoff should be included as well.

However, the environmental mechanisms leading to environmental impacts on ecosystem quality has been deemed not yet sufficiently developed, consistent and complete to be harmonized (Kounina et al., 2013; Boulay et al., 2015b; 2015c). Existing models address different impact pathways and are not sufficiently consistent in the type of modelling and underlying hypothesis to be harmonized yet (Nunez et al. 2016). Work is ongoing within the WULCA initiative to propose a harmonized framework solely for ecosystem impacts from water use supporting the development of a harmonized model.

Impacts on Natural resources

According to Milà i Canals et al., (2009) water can be a flow, fund or stock resource. Flow resources cannot be depleted but there can be competition over its use, whereas depletion may be an issue for funds and stocks. For instance, the use of groundwater may reduce its availability for future generations, when aquifers are over-abstracted or fossil water is used. According to Kounina et al. (2013), the overuse of renewable water bodies can occur depending on the water renewability rate. For calculating midpoint indicators Milà i Canals et al. (2009) had proposed to use a modified version of the abiotic depletion potential model (Van Oers et al., 2002). Pfister et al. (2009) translated changes in water availability into surplus energy needed, whereas exergy associated to the resource water was accounted by Bösch et al. (2007) through the CExD methodology and by DeWulf et al. (2007) through the

CEENE methodology. The model proposed by Rugani et al. (2011) accounts for solar energy demand on the basis of emergy accounting principles (Odum, 1996), with substantial differences in the way allocation is performed.

In Figure 5.1, land occupation and transformation as well as rainwater harvesting are identified as drivers for change in surface water availability run-off and in precipitation water stored as soil moisture as the modification of the hydrological balance following land transformation or occupation corresponds to a modification of the amount of water that reaches the groundwater and surface water (Kounina et al., 2013). Nevertheless, these impact pathways haven't been clearly described and assessed in literature.

5.4 Criteria for the evaluation of this impact category

A set of criteria was specifically defined to assess in detail the completeness of scope, relevance, transparency and robustness, as well as applicability of scarcity-type indicators for LCA impact assessment models, coherently with the general structure provided within the ILCD (EC-JRC, 2011). The set of specific criteria which are selected for evaluating models assessing water depletion is explicitly designed for the evaluation of midpoint models and not for endpoint models, because of the relatively low level of development and maturity which characterizes the endpoint models.

- The criteria described below complement the general ILCD structure by incorporating the outputs of previous works dedicated to the qualitative and quantitative comparison and review of LCIA models assessing water scarcity (Kounina et al., 2013; Frischknecht et al., 2013; Boulay et al., 2015a), as well as the outcomes of expert workshops held in the context of the WULCA initiative (Boulay et al., 2015c). The full list of criteria, their description and the evaluation of models, is provided in Annex 5.1, whereas a brief explanation on the newly introduced criteria is provided below according to the main ILCD sections.
-
- Completeness of the scope:
- Two specific criteria were added to this impact category: comprehensiveness and geographic coverage. The first criteria aims at answering the questions: "Does the model assess water scarcity, deprivation, stress, depletion and/or potential effects on water users? Does it include water quality aspects?" where the capability of the model in dealing with both scarcity and quality aspects is considered advantageous, whereas the second criterion aims at assessing the comprehensiveness in terms of geographic coverage.
-
- Environmental relevance:
- The outcomes of the WULCA experts' workshops pointed out towards the set of recommendations for a generic midpoint indicator (Boulay et al. 2015c). They cover the following aspects:
    - inclusion of both human and ecosystems water demand with respect to availability;
    - inclusion of arid areas with special attention in the model, as data quality is generally low in those regions and because of the fact that WTA, CTA or demand to availability (DTA) indices may be unable to properly reflect aridity when withdrawal, consumption or demand is low in arid areas;
    - inclusion of the Environmental Water Requirement (EWR), although no complete agreement was found on this and some members/experts believe exclusion may

be a better choice as CTA or WTA could be considered a "proxy" (albeit an unspecific one) for ecosystem impacts. Recommendation was made to use EWR median and maximum of different models or a range of 30 to 80 % of pristine (natural) conditions to account for the uncertainty and account for the temporal variability

- exclusion of green water and hence terrestrial ecosystem water requirements from the generic midpoint indicator because these impact pathways are not well known (Gerten et al., 2013), the link between blue water consumption and change in green water availability (i.e. terrestrial ecosystems deprivation) is not well known, although impacts from blue water consumption on terrestrial ecosystems are described in literature (Van Zelm et al., 2011);
- to use WaterGAP as recognized to be the only water availability model of this suite that is post-calibrated to actual river discharge measurements and hence represents the current reality better, nevertheless some of its modules are better resolved in other models (after comparison with WaterGAP - Müller Schmied et al. 2014; LPJmL - Rost et al. 2008; Aqueduct 2.0 - Gassert et al. 2013; H08 - Hanasaki et al. 2010);
- suggestion to include infrastructure in water availability assessment (reservoirs, water transfers, etc.) as well as to use datasets that will facilitate regular updates of the indicator;
- not distinguishing surface and groundwater and providing only one generic values, as this indicator aims to assess the overall pressure/impacts;
- to model monthly indicators to be used for those LCA practitioners who have access to temporal data related to water use and average the monthly values to obtain an annual one using a weighted average, to account for less-informed studies;
- to allow for differentiation at the sub-basin level;
- to perform the aggregation of the indicator to the country level using consumption-based weighted averages, in order to represent the geographic probability distribution of the water use within the selected country.

Therefore, on the basis of the recommendations above and the work performed by Kounina et al. (2013), the following criteria were introduced under the sub-section 'Coverage of the environmental mechanisms': Environmental water requirement by ecosystems, Water demand by humans, Downstream impacts of water consumption, Seasonal variability, Arid areas, Consumptive use of water, Groundwater renewability rate. All of them aim to assess the extent to which the LCIA model takes into account relevant aspects of the environmental mechanisms underlying impacts associated to water scarcity, in line with the specifications above.

Under the sub-section 'Comprehensiveness - elementary flows' the following criteria were included: Coverage of water types and coverage of water uses, reflecting the ability of the model to cover different typologies of water types (i.e. surface, groundwater, rainwater, precipitation stored as soil moisture, fossil groundwater, sea/ brakishwater) or different types of uses (i.e. withdrawal, release and time-lapse (borrowing)). As pointed out by Boulay et al., (2011b) and Berger et al. (2014) amongst others, it is preferable that a scarcity indicator considers human consumption since water that is abstracted and returned, like cooling water, does not contribute to water scarcity.

The spatial and temporal resolution of the bio-physical model(s) have been evaluated for each of the relevant sub-models, namely: Environmental water requirement by ecosystems, Water availability, Human water demand – agriculture and Human water demand - households and industry.

Scientific robustness and uncertainty:

On top of the general criteria described in the beginning of the section, the scientific robustness of water scarcity LCIA models is evaluated by assessing the representativeness of the underlying biophysical models, including their temporal representativeness, with specific reference to water availability and human water demand. An evaluation of the techniques used to downscale or upscale and aggregating data at different resolutions was performed as well so to take into account the soundness of such choices. Similarly, uncertainty was evaluated by considering how uncertain the underlying hydrological models and/or data are, including human demand and environmental water requirements.

Importance of the criteria:

The importance of the selected criteria was defined on the basis of the goal of the current recommendations as well as on the results provided by Boulay et al. (2015a; 2015b) on quantitative comparison amongst water scarcity models. A key aspect of these recommendations is the applicability and the level of readiness of the LCIA for implementation in LCA software, together with the completeness and coverage of the characterization factors. This is because of the fact that LCIA models have to be directly implementable and usable by practitioners as the context of application of these recommendations is the application of updated LCIA models within PEF/OEF.

In order to understand how methodological choices may affect the results and therefore how much important should be these aspects in defining an overall score of LCIA models, Boulay et al. (2015a; 2015b) was considered as starting point. Boulay and colleagues had recalculated four midpoint indicators (Boulay et al., 2011; Frischknecht and Büsser Knöpfel, 2013, Pfister et al., 2009; Hoekstra et al., 2012) on 250 watersheds on the basis of the same data source (WaterGap2), with the aim of quantitatively compare generic midpoint indicators assessing water scarcity. According to the results the most sensitive choices underpinning water scarcity indicators are: the definition of the spatial scale at which the modeling data are used to calculate the index as important differences are observed between sub-watershed and country scales; the function defining scarcity as the choice of the curve (direct, exponential or logistic) as well as the use of threshold values describing scarcity as function of CTA (or WTA) plays an important role and it is not generally based on scientific data. Instead, the definition of the temporal scale although showing large variation throughout the year, shows high correlation between regions, meaning that comparative results would not be excessively affected provided that the same temporal inventory information is used. The source of data i.e. the choice between WaterGap2, Aquaduct (Fekete et al., 2002) and Mekonnen and Hoekstra (2011) is not important for most of the world regions, with some exceptions. The type of model and data reference year might be possible sources of discrepancy. The differentiation between withdrawn surface water versus groundwater and the use of WTA-based or CTA-based indicator made less of a difference at a global level, with, however a few important exceptions. According to Boulay et al. (2015a; 2015b), it is important to notice that the relevance of the methodological choices might change depending on the region of the globe.

5.5 Pre-selection of models for further evaluation

As introduced in the 'Framework and scope' section, the selection of models for further evaluation was performed by following two criteria: relevance of the model in the LCA context and the perspective adopted by the model. The rationale of this choice is discussed below.

Relevance for LCA

In the context of this report, a pre-selection of models was done in order to consider only models which: i) have already been used in LCA, or ii) have been developed for use in LCA, or iii) have been recommended (by their authors or others) for use in LCA.

Reasons for indicators to not be used or recommended in LCA can vary, but it is important to remember that LCA aims to assess potential environmental impacts on humans, ecosystems and, when relevant, natural resources. Moreover, this should be done by multiplying a characterization factor with an inventory, providing a meaningful indicator result. The characterization factor should describe as much as possible an actual impact pathway, minimizing value choices and describing the potential impacts associated with a marginal human intervention such as one assessed in LCA. Following this reasoning, some indicators are valuable but not necessarily adapted for LCA, including indices reflecting water per capita (Falkenmark et al., 1989; Gleick, 1996; Ohlsson, 2000; Asheesh, 2007), further socio-economic political assessment or water security related index (Vörösmarty et al. 2010; Chaves & Alipaz 2007; Sullivan 2002), or water criticality (Sonderegger et al., 2015). As mentioned above, other indicators are useful to assess current surface and groundwater scarcity (Gassert et al. 2013, Wada and Bierkens, 2014) or scenarios of future freshwater availability (Döll, 2009; Hejazi et al. 2014; Veldkamp et al. 2016; Wada and Bierkens, 2014) however they have not been currently implemented within the LCA framework.

Perspective adopted – water scarcity

In the context of water use impact assessment, it is important that the model selected be consistent with the international standard on water footprint ISO 14046 (2014): an insight on understanding what is LCA and ISO compliant water Footprinting was provided by Pfister et al. (2017). In this document, several types of footprints are described, either addressing water degradation (through other existing impact assessment methodologies, such as ecotoxicity, eutrophication, acidification, etc.), or addressing water availability. It is specified that water availability can be affected by consumption or degradation, which may render water unusable (as described above with the concept of functionality). If water availability is assessed only based on the quantity and not the quality, then it is called water scarcity. In the context of this recommendation, it is desired to recommend a model that describes water scarcity, and hence describes "*the extent to which demand for water compares to the replenishment of water in an area*" (ISO 14026, 2014). The scope of the present models comparison was thus limited to scarcity models for generic midpoint indicators, with recommended use for LCA (Table 5.1). Nevertheless, as clear guidance on the use of impact assessment models in the context of ISO 14046 is not yet available, therefore all LCA-consistent midpoint models addressing water from a scarcity/stress/deprivation or depletion (even long-term and based on thermodynamics), were included in the analysis.

Other LCA-relevant models for water use impact assessment in LCA include wider scope availability models (Boulay et al. 2011b; Ridoutt and Pfister, 2010; Bayart et al., 2014) or area-of-protection specific impact models at the endpoint (Verones et al. 2013; van Zelm et al. 2010; Hanafiah et al. 2011; Boulay et al. 2011b; Pfister et al. 2009; Motoshita et al. 2011; Motoshita et al. 2014). Availability models describe a loss of availability, or functionality of water due to consumption or degradation of the resource that renders it unavailable/non-functional for other users. Although they are relevant and adding important additional information, they were not considered for two main reasons: 1. more research is needed on the potential for double counting when used in parallel with specific water degradation impact assessment models (i.e. human toxicity, aquatic ecotoxicity, etc.), and 2. data availability on input water quality and additional calculations required to obtain output water quality from available data both add a level of uncertainty and operationalisation burden that were not desired at this point. Future research and implementation efforts are welcome on this topic.

Impact assessment on specific areas of protection is currently going through a harmonization and consensus building effort by the WULCA group (Boulay et al., 2015c) which suggests that it would be too early to provide recommendations for a comprehensive assessment of impacts from water use via individual areas of protections at the endpoint level, for the areas of protection human health, ecosystem quality and even more so, resources.

Similarly, green water scarcity indicators (Hoekstra et al., 2011; Schyns et al., 2015) were excluded from the analysis because the impact pathway identifying impacts associated with green-water depletion is still under debate, and it is not clear whether it should be tackled under the impact category land use or within water depletion, although an example of application of green water scarcity index in LCIA was provided by Nuñez et al. (2013). Also Quintero et al. (2015) proposed a contribution to the environmental impact assessment of green water flows. However, the model provides factors only for Portugal. According to the outcomes of the expert workshops held within the WULCA initiative (Boulay et al., 2015c) the mechanisms underlying consumption from terrestrial ecosystems from groundwater are not yet well understood. Moreover, as pointed out by Schyns et al. (2015), the operational implementation of the green water scarcity indices is seen problematic. This is due to the following reasons: i) the determination of which areas and periods of the year the green water flow can be used productively is not straightforward; ii) the estimation of green water consumption of forestry is difficult because it entails separation of production forest evaporation into green and blue parts; iii) research is required to determine the environmental green water requirements, i.e. the green water flow that should be preserved for nature, similar to the environmental flow requirements for blue water.

5.5.1 Selection of midpoint models

The list of selected midpoint models is provided in table 5.1.

Description of the midpoint models

The midpoint models selected for further evaluation have been grouped according to the three categories: i) generic midpoint models; ii) human health-specific midpoint models and iii) resource-specific midpoint models, coherently with the impact pathways defined in Figure 5.1.

Generic midpoint indicators for human health and ecosystem quality

1. Category 1: generic midpoint indicators
    - Swiss Eco-scarcity (Frischknecht and Büsser Knöpfel, 2013)

The following text heavily relies on Boulay et al. (2015a), Frischknecht and Büsser Knöpfel (2013)

The Swiss ecological scarcity method 2013 (Frischknecht and Büsser Knöpfel, 2013) is the update of the previous versions (Frischknecht et al., 2006; Frischknecht et al., 2008; 2009) already included in the review of models the ILCD (EC-JRC, 2011). The method converts environmental pressures into points (eco-factors) which are determined from the current environmental situation (current flow, i.e. water withdrawal) and the target situation aimed at by the environmental policy i.e. the critical flow, set equal to 20% of renewable water supply accordingly to OECD[13] (2003).

[13] "*the ratio in the range of 10 to 20% indicates that water availability is becoming a constraint on development and that significant investments are needed to provide adequate supplies. When the ratio is over 20%, both supply and demand will need to be managed and conflicts among competing uses will need to be resolved*".

Table 5.1: LCIA models chosen for further evaluation, fulfilling both scarcity and LCA relevance criteria

Model	Indicator (CF x inventory)	Units of the indicator	Characterization factors (CFs)	References
GENERIC MIDPOINT				
Milà i Canals et al. 2009 – Smathkin et al. 2004	Water stress (stress-weighted water consumption)	m^3 eq.	Water stress index (WSI) [m^3 eq./m^3]	Milà i Canals et al., 2009; Smathkin et al., 2004
Swiss Ecological scarcity	Ecological scarcity (scarcity-weighted water consumption)	Eco-points (UBP)	Eco-factors [UBP /m3]	Frischknecht and Büsser Knöpfel, 2013 (update of Frischknecht et al., 2009)
Boulay et al. 2011 – simplified (a)	Water scarcity (scarcity-weighted water consumption)	m^3 eq.	Water scarcity index (α) [m^3 eq./m^3]	Boulay et al., 2011
Pfister et al. 2009 - WSI	Water stress (stress-weighted water consumption)	m^3 eq.	Water stress index (WSI) [m^3 eq./m^3]	Pfister et al., 2009; Pfister & Bayer 2013
Hoekstra et al. 2012	Blue water scarcity (scarcity-weighted water consumption)	m^3 eq.	Blue water scarcity index [m3 eq./m3]	Hoekstra et al., 2012
Berger et al. 2014	Water depletion (depletion-weighted water consumption)	m^3 eq depleted	Water depletion index (WDI) [m^3 eq. depleted/m^3]	Berger et al., 2014
Loubet et al., 2013	Water deprivation (deprivation-weighted water consumption)	m^3 eq deprived	Water deprivation index [m^3 eq. /m^3]	Loubet et al., 2013
AWARE	User deprivation potential (deprivation-weighted water consumption)	m^3 world eq. deprived	m^3 world eq. deprived/m^3	UNEP, 2016
Yano et al. 2015	Water scarcity footprint (unavailability-weighted water consumption)	m^3H_2O eq.	Water unavailability factors [m^3H_2O eq./m^3]	Yano et al. 2015
HUMAN HEALTH – MIDPOINT				
Motoshita et al. 2014	Agricultural water scarcity	m^3 eq.	Agricultural water scarcity factors [m^3 eq./m^3]	Motoshita et al. 2014
NAURAL RESOURCES – MIDPOINT				
Pfister et al., 2009 - resources	Surplus energy	Joules	Additional energy cost [MJ/m3]	Pfister et al., 2009
Milà i Canals et al., 2009	Abiotic Depletion Potential (ADP)	Sb eq.	ADP factors [Sb eq./m^3]	Milà i Canals et al., 2009
Dewulf et al., 2007	Cumulative Energy Extracted from the Natural Environment (CEENE)	Joules of exergy	Exergy factors [MJex/m^3]	DeWulf et al., 2007
Rugani et al. 2011	Solar energy demand (SED)	solar energy Joules	Solar Energy Factors [MJse /m^3]	Rugani et al., 2011

$$Eco - factor = K \cdot \frac{1 \cdot UBP}{F_n} \cdot \left(\frac{F}{F_k}\right)^2 \cdot c$$

$$Weighting = \left(\frac{current\ flow}{critical\ flow}\right)^2 = \left(\frac{water\ withdrawal}{water\ supply_{renewable} \cdot 20\%}\right)^2$$

Where: K is the characterization factor of a pollutant or a resource; Flow is the load of a pollutant, quantity of a resource consumed or level of a characterized environmental pressure; Fn represents the normalization flow (Switzerland as reference); F = Current flow: Current annual flow in the reference area; Fk = Critical annual flow in the reference area; c = Constant (10^{12}/a); UBP = Ecopoint: the unit of the assessed result. Results, calculated on the basis of AQUASTAT statistics at country level (FAO 1998-2010), are given in eco-points at the country level for OECD and non-OECD countries (Frischknecht and Büsser Knöpfel, 2013). The calculation of the indicator on a more refined spatial scale is available based on data of WaterGap2 model (grid-level 0.5° x 0.5°) (Frischknecht et al., 2013) and regionalized water stress index (WSI) values[14]. Differences with the previous versions consist of: the application to consumptive water use instead of overall withdrawals, and the average factor for OECD countries is now higher than the previous one because of using the consumption weighted average .

- Water Stress Indicator (Milà i Canals et al., 2009 – Raskin et al., 1997 and Milà i Canals et al., 2009 – Smakhtin et al., 2004)

Milà i Canals et al. (2009) propose the water stress index (WSI) from Smakhtin et al. (2004) or Raskin et al. (1997) as characterization factor for assessing freshwater ecosystem impact. As reported by Frishcknecht et al. (2013) both models focus on impacts from surface and groundwater evaporative use and land use transformation, accounting for all evaporative uses. As proposed by Frishcknecht et al. (2013) the models are hereafter referred as 'Milà i Canals & Smakhtin' and 'Milà i Canals & Raskin'.

The water stress index values calculated by Smakhtin et al. (2004) compares water withdrawals to renewable water resources minus environmental water requirements (see equations below). Instead, Raskin et al. (1997) developed a water-to-availability index comparing withdrawals to renewable water resources available in a given country. According to Frischknecht et al. (2013) both indicators might lead to underestimation of local effects when non-evaporative uses are considered to have no impact on freshwater ecosystems. Both indicators can be used to indicate generic scarcity; therefore they are classified in this document as generic midpoint indicators.

'Milà i Canals & Raskin': $WSI = \frac{WU}{WR}$; 'Milà i Canals & Smakhtin': $WSI = \frac{WU}{WR - EWR}$

Where: WU = water use – withdrawals; WR = Water Resources; EWR = environmental water requirement.

Within this analysis only 'Milà i Canals et al. & Smakhtin' was considered as the use of simple withdrawal to availability (WTA) indicators such as Milà i Canals et al.& Raskin', is considered

[14] see http://treeze.ch/fileadmin/user_upload/downloads/UBP/WS_class_UBP06.zip, June 2016

superseded for use in LCA in several publications (Kounina et al., 2013; Boulay et al., 2015c). Characterization factors at watershed scale were made available by Milà i Canals and colleagues, for a limited set of world watersheds. The geographic and temporal coverage of the characterization factors is country-year or watershed-year according to the selected model (as provided by Milà i Canals et al., 2009), the main data sources were WaterGapv2 (Alcamo et al., 2003).

- Water scarcity α (simplified) (Boulay et al. 2011b)

This scarcity model is based on a consumption-to-availability (CTA) ratio, calculated using statistical low-flow to account for seasonal variations, and modeled using a logistic function (S-curve) in order to obtain resulting indicator values between 0 and 1 m^3deprived/m^3consumed. The curve is tuned using the same water scarcity thresholds as the OECD thresholds in Pfister (Pfister et al., 2009; Alcamo et al., 2000; OECD, 2003) but converted with an empirical correlation between WTA and CTA. In the original version of the model (Boulay et al., 2011b) different values of water scarcity were calculated for different types of water qualities/functionalities, in this document only the simplified version (α) is considered, as described by CIRAIG (2016). Water consumption and availability data for surface and ground water are taken from the WaterGap v2.2 model (Alcamo et al., 2003a; 2003b). Results are available at a scale that originates from the intersection of the watershed and country scales, resulting in 808 cells worldwide.

The simplified version of the model does not consider changes in water quality, unlike the original one, which aims to assess the equivalent amount of water of which other competing users are deprived as a consequence of water use. The model is based on consumed water volume (i.e. assesses consumptive water use only). The characterization factors are available at the country scale, per year, covering the majority of the world.

$$CF_i = \alpha_i = f\left(\frac{CU}{Q90}\right) = \frac{1}{1+\left(0.305\cdot \mathrm{e}^{(-0.567\cdot(\mathrm{CTA}-2.1899))}\right)^{1/0.0053}}$$

Where: f(x) = logistic function matching CU/Q90 with scarcity thresholds (see Boulay et al. 2011b – SI); CU = human consumption; Q90 = statistical low flow

- Water stress index (WSI) (Pfister et al. 2009; Pfister & Bayer 2014)

The following text heavily relies on Boulay et al. (2015a) and Pfister et al. (2009)

This scarcity indicator is based on the withdrawal-to-availability (WTA) ratio, modified to account for seasonal variations, and modeled using a logistic function (S-curve) in order to obtain resulting indicator values between 0.01 and 1 m^3deprived/m^3consumed. The curve is tuned using OECD water scarcity (stress) thresholds, which define moderate and severe water stress as 20% and 40% of withdrawals, respectively (Alcamo et al., 2003; OECD, 2003). The model is available at the grid-cell level (0.5° x 0.5°), and data for water withdrawals and availability were obtained from the WaterGap v2 model (Alcamo et al. 2003a; 2003b). The indicator is applied to the consumed water volume (i.e. assesses consumptive water use only). In other publications the authors suggest dividing the indicator by the world average and to characterize also grey water inventories together with blue water consumption (Ridoutt and Pfister 2010). Characterization factors are available at both watershed and country scales, per year, covering the majority of the globe. Pfister and Bayer (2014) calculated monthly values at the level of the watersheds.

$$\mathrm{WTA}_i = \frac{\sum_j \mathrm{WU}_{ij}}{\mathrm{WA}_i} \qquad \mathrm{WTA}^* = \begin{cases} \sqrt{\mathrm{VF}} \times \mathrm{WTA} & \text{for SRF} \\ \mathrm{VF} \times \mathrm{WTA} & \text{for non} - \text{SRF} \end{cases}$$

$$\mathrm{VF} = e^{\sqrt{\ln(s^*_{\mathrm{month}})^2 + \ln(s^*_{\mathrm{year}})^2}} \qquad \mathrm{WSI} = \frac{1}{1 + e^{-6.4 \cdot \mathrm{WTA}^*}\left(\frac{1}{0.01} - 1\right)}$$

Where: WSI: is the Water Stress Index of a given catchment; WTAi : is WTA in watershed i and user groups j are industry, agriculture, and households; VF: is the variation factor estimated as combination of the standard deviations of the precipitations (monthly S*month and yearly S*year); SRF are watersheds with strongly regulated flows as reported by Pfister et al. (2009) on the basis of Nilsson et al. (2005); VFws: is the variation factor calculated for a given watershed 'ws'; P = rainfalls.

- Blue water scarcity index (Hoekstra et al. 2012)

The following text heavily relies on Boulay et al. (2015a) and Hoekstra et al., (2012)

This scarcity indicator is based on a consumption-to-availability ratio (CTA) calculated as the fraction between consumed (referred to as blue water footprint) and available water. The latter considers all runoff water, of which 80% is subtracted to account for environmental water needs, assuming that depletion over 20% of a river's natural flow increases risks to ecological health and ecosystem services (Richter et al., 2011). The methodological difference between the approach developed by Hoekstra et al. (2012) and the one proposed Milà i Canals et al. (2009) (so-called 'Milà i Canals & Smathkin') consists in the fact that the former model is based on a consumption-to-availability ratio, whereas, the latter is based on withdrawal-to-availability. Hoekstra et al. (2011) is the methodology followed for the calculation of blue water consumption, building on Mekonnen and Hoekstra (2011) as well as FAO statistics on total withdrawal (FAO, 2010). Water availability was calculated as 'natural runoff' minus environmental water requirements. The data for water runoff is from Fekete et al. (2002) and re-adjusted by Hoekstra et al. (2012) so to approximate the natural undepleted run-off. Storage of water is not considered as available water resource. The indicator is applied to the consumed water volume (i.e. assesses consumptive water use only). The time scale of the calculation is monthly and the spatial resolution is five arc minutes for the world. Results are available for the main watersheds worldwide but some regions are not covered. The characterization factors are available at both watershed and country scales, however the coverage of the world area is lower in comparison to other models as it only covers major world catchments.

$$WSI = \frac{CU}{WR - EWR}$$

Where: CU = consumptive use of water – withdrawals; WR = Water Resources; EWR = environmental water requirement; WSI = blue water scarcity index

- Water depletion index (Berger et al. 2014)

The Water Accounting and Vulnerability Evaluation (WAVE) model analyzes the vulnerability of basins to freshwater depletion. Based on local blue water scarcity, the water depletion index (WDI) denotes the risk in a basin that water consumption can lead to depletion of freshwater resources. The index is based on a modified consumption-to-availability (CTA) ratio which relates annual water consumption (C) to annual availability (A). It can be understood as an equivalent volume of depleted water resulting from a volume of water consumption.

Water scarcity is determined by relating annual water consumption to availability in more than 11 000 basins based on WaterGap v2.2 (Alcamo et al., 2003a; 2003b; Florke et al., 2013). The freshwater availability of a drainage basin (A) expresses the annually renewable freshwater volumes within the basin which can be quantified by means of runoff (plus upstream inflows if the basin is divided into sub-catchments). Annually usable surface water stocks (SWS) are added to A in order to consider lakes, wetlands, and dams as available water resources. As volumes of groundwater stocks (GWS) are not available on a global level an adjustment factor (AF_{GWS}) defined on geological structure and annual recharge (WHYMAP - Richts et al., 2011) was introduced by the authors so to account for availability of groundwater.

Berger et al. (2014) use a logistic function which is fitted to turn 1 above a CTA of 0.25 which is regarded as extreme water stress, this way the indicator values range between 0.01 and 1.00. In order to avoid mathematical artifacts of previous indicators which turn zero in deserts if consumption is zero WDI was set to the highest value in (semi)arid basins, therefore absolute freshwater shortage is taken into account in addition to relative scarcity. The resulting characterization factors are available at both watershed and country scale, with yearly resolution.

$$WDI = \frac{1}{1 + e^{-40 \cdot CTA^*}\left(\frac{1}{0.01} - 1\right)} \quad ; \quad CTA^* = \frac{C}{A + SWS} \cdot AF_{GWS}$$

Where: CTA* = modified consumption-to-availability (CTA) ratio; C = annual water consumption; A = annual availability; SWS = annually usable surface water stocks; AF_{GWS} = adjustment factor accounting for the availability of groundwater.

- Water deprivation index (Loubet et al., 2013)

The model proposed by Loubet et al. (2013) is based on a two-step approach. First, water scarcity is defined at the sub-river basin scale with the consumption-to-availability (CTA) ratio, and second, characterization factors for water deprivation (CF_{WD}) are calculated, integrating the effects on downstream sub-river basins. This effect is considered at a finer scale because a river basin is split into different subunits. This approach reflects the fact that the water consumed at a specific location only affects Sub-River Basins (SRB) downstream from the location of consumption: specific water consumption in sub-river basin–*i* (SRBi) will affect SRBi to SRBn. This causes a cascade effect on potential downstream usages and ecosystems, something that is not captured by water scarcity indicators. This effect can be measured by the sum of downstream CTA ratios. The characterization factor for water deprivation in SRBi is the weighted sum of all downstream CTA ratios. The available water

(WA) is estimated as regulated discharge (Fekete et al., 2002) to which the share of environmental water requirements (EWR) estimated by Smathkin et al. (2004) are subtracted. The set of characterization factors was calculated only for two case studies in France and Spain, therefore factors are not available.

$CTA = \frac{tWC}{WA}$; $WA_i = (1 - \%EWR) \cdot D_i^{reg}$

$CF_{WD,i} = \frac{1}{\bar{p} \cdot \bar{N}_{down}} \sum_{j=i}^{n} (CTA_j \cdot p_j)$; $I_{WD} = WW \cdot CF_{WD,A} - WR \cdot CF_{WD,B}$

Where: WA= available water (m^3) in the river basin; D^{reg} = regulated discharge is that in which natural discharge is altered by reservoir operations; tWC = total water consumption; CF_{WD} = characterization factor for downstream water deprivation; p☐ = is the average value of the weighting parameters among all the SRBs within the river basin and N☐down is the average number of SRBs downstream from each SRB within the river basin; pj is the chosen weighting parameter of downstream SRBj; I_{WD} is the midpoint impact of water deprivation (m^3equivalent or m^3 equiv), WW is the water withdrawal volume of the studied system that occurs at location A (m^3), WR is the water release volume of the studied system that occurs at location B (m^3), and CFWD,A and CFWD,B characterize locations A and B, respectively.

- Water unavailability factor Yano et al. (2015)

Yano et al. (2015) developed a model for assessing water scarcity footprints as indicators of the potential impacts of water use. The model assumes that the potential impact of a unit amount of water used is proportional to the land area or time required to obtain a unit of water from each water source. This approach is based on renewability only, not resulting from ratio of water use to availability.

The potential impacts of a unit amount of water used can be expressed using the land area or collection time required to obtain a unit of water from each source. The characterization factor for each source is defined as water unavailability factor (*fwua*) and calculated using a global hydrological model (H08 - Hanasaki et al., 2008) with a global resolution of 0.5 × 0.5 degrees. It can be calculated as ratio between the required land area per unit of time to obtain the reference volume of water from the water source *x* at location *l* (A *x,l*) and the required land area per unit of time to obtain the reference volume of water from the reference condition (Aref) or, similarly, by using required collection time per unit area instead (see equations below).

Precipitation, surface water, and groundwater are characterized separately to reflect the location and source variability of renewable water resources of each source. The characterization factors are provided with yearly resolution at the spatial scale of world countries, covering the majority of the globe.

$fwua_{x,l} = \frac{A_{x,l}}{A_{ref}} = \frac{T_{x,l}}{T_{ref}}$; $A_{x,l} = \frac{Q_{A,ref}}{P_{x,l}}$; $T_{x,l} = \frac{Q_{T,ref}}{P_{x,l}}$

Where: $fwua_{x,l}$ is the characterization factor for water source x at location l; QA,ref is the reference volume of water per unit of time (m3/year); QT,ref is the reference volume of water over unit land area (m^3/m^2); and Px,l is the annual renewability rate of the water cycle of water source x at location l (m/year).

- Available WAter REmaining (AWARE) (UNEP, 2016)

The AWARE model is built on a water use midpoint indicator representing the relative "Available Water Remaining" per area in a watershed, after the demand of humans and aquatic ecosystems has been met. It assesses the potential of water deprivation, to either humans or ecosystems, building on the assumption that the less water remaining available per area, the more likely another user will be deprived (UNEP, 2016). It is first calculated as the water Availability Minus the Demand (AMD) of humans and freshwater ecosystems and is relative to the area (m^3 m^{-2} $month^{-1}$). In a second step, the value is normalized with the world average result and inverted, and hence represents the relative value in comparison with the average m^3 consumed in the world. The world average is calculated as a consumption-weighted average. Once inverted, 1/AMD can be interpreted as a surface-time equivalent to generate unused water in this region. Minimum and maximum thresholds have been set for this indicator. When demand is higher than availability the maximum threshold value for ADMi (equal to 100) is used instead of the equation (which otherwise would lead to a negative result).

The indicator is calculated at the sub-watershed level and monthly time-step, the underlying hydrological model from which water availability and human consumption of water is WaterGap v2.2/v3 (Müller Schmied et al., 2014), whereas water demand model EWR relies on values estimated by Pastor et al. (2013). Characterization factors are available at watershed-month scale as well as country and/or annual scales, for agricultural and non-agricultural water use as well as unknown use.

AWARE is the recommended model from WULCA to assess water consumption impact assessment in LCA. The Life Cycle initiative Flagship project on LCIA indicators also chose AWARE as a consensus impact model, following the Pellston workshop held in Valencia (Spain) in January 2016. They specify that this recommendation has to be considered interim until 10 case studies have been performed and made available publicly (and that no unexplainable/unjustifiable issues have been found).

$$AMD_i = \frac{(Availability - HWC - EWR)}{Area}$$

$$CF = \frac{STe_i}{STe_{world\ avg}} = \frac{\overline{AMD_i}}{AMD_i} = \frac{AMD_{world\ avg}}{AMD_i}, \text{ for Demand<Availability}$$

Where: AMD = Availability-Minus-Demand per area; Demand = HWC + EWR; HWC = human water consumption; STe = Surface-Time equivalent required to generate one cubic meter of unused water i.e. 1/AMDi

Human health-specific midpoint indicators

1. Category 2: Human Health-oriented midpoint indicators

    •

        - Agricultural water scarcity (Motoshita et al., 2014)

Motoshita et al. (2014) developed a midpoint characterization model, which focuses specifically on shortages in food production resulting from agricultural water scarcity. The model takes into account country-specific compensation factors for physical availability of water resources and socio-economic capacity in relation to the irrigation water demand for agriculture. The underlying equation is as follows:

$$CF_{AgrMidpoint,i} = R_{Agr,i} \cdot IDR_i \cdot (1 - PCF_i) \cdot (1 - SCF_i)$$

Where $R_{Agr,i}$ is the ratio of agricultural water use to total water withdrawal in country i, IDR_i is the irrigation dependency ratio for crop production, PCF_i expresses the physical compensation capacity of country i and SCF expresses the social compensation capacity of country i. For physical vulnerability (i.e. 1-PCF), the midpoint indicators calculated in other studies can be applied without any modifications. The compensation capacity (SCF) is calculated by comparing the average annual production of commodities to their average annual stock. In case the former is higher than the latter, the ratio of average surplus stocks and total production is computed for a list of agricultural commodities. The higher is the ratio, the higher is the social vulnerability factor (i.e. 1-SCF).

The underlying hydrological model for agricultural water and total withdrawal per country is Aquastat (FAO, 2010), whereas the WSI values were taken from (Pfister et al., 2009), which in turns builds on WaterGap v2.2 (Alcamo et al., 2003a; 2003b). Characterization factors are available at country scale as some of the input variables are available only at that level of spatial resolution. The proposed midpoint model is connected to the endpoint model developed within the same publication.

Resource-specific midpoint models

2. Category 3: resource depletion-oriented midpoint indicators
   -
     - Freshwater depletion (Pfister et al., 2009)

According to Pfister et al. (2009), water stock exhaustion can be caused by the extraction of fossil groundwater or the overuse of other water bodies. Pfister and colleagues adopt the concept of back-up technology introduced by Stewart and Weidema (2005) for assessing abiotic resource depletion in Ecoindicator99 (Goedkoop and Spriensma, 2001) for assessing damage to freshwater resources, as endpoint indicator. The indicator, expressed in "surplus energy" (MJ) assumes that desalination of seawater is applied as a backup technology to compensate for water resource depletion, although Pfister et al. (2009) recognize that "*it merely serves as a theoretical indicator to make water use comparable to other types of resource use*" as not necessarily all water depleted will be desalinated.

$$\Delta R = E_{des} \cdot F_{dep} \cdot WU_{cons} \quad ; \qquad \mathrm{F_{dep,i}} = \begin{Bmatrix} \frac{\mathrm{WTA}-1}{\mathrm{WTA}} \text{ for WTA } > 1 \\ 0 \quad \text{for WTA } \leq 1 \end{Bmatrix}$$

The damage to freshwater resources ($\Delta \mathrm{R}$) is therefore assessed by multiplying the energy required for seawater desalination ($\mathrm{E_{des}}$) times the fraction of freshwater consumption that contributes to depletion ($\mathrm{F_{dep}}$), times the consumptive use of water. According to Pfister and colleagues, $\mathrm{F_{dep}}$ serves also as characterization factor for the midpoint indicator "freshwater depletion", and is calculated by aggregating the values for $\mathrm{F_{depn,i}}$ of all watersheds in the country, using total annual withdrawal within the watershed i as a weighting factor. The underlying hydrological model providing values of water withdrawal and availability is WaterGap v2.2 (Alcamo et al., 2003a; 2003b). Cross-boundary watersheds located in several countries are assigned to countries according to the area share of watershed i within the specific country. Characterization factors for $F_{depletion}$ are not publicly available, whereas CF values for ΔR are available by catchment and the vast majority of world countries.

- Freshwater depletion (Milà i Canals et al., 2009)

Milà i Canals et al. (2009) propose a modified formula of the abiotic depletion potential (ADP) (Guinée et al., 2002) (Guinée and Heijungs, 1995) to be applied for assessing freshwater depletion. R_{Sb} is the reserve of the reference flow (Antimony) and DR_{Sb} is its extraction rate. R is the ultimate reserve of resource stored in the aquifer. The regeneration rate (RR) of the resource is added to the equation. According to the developers in case its value is higher than the extraction rate (ER), the ADP value for the resource should be zero, otherwise it would lead to negative values. Considering the limited knowledge of groundwater resources in relation to their use, the authors suggest that if there is knowledge that the relevant aquifer is being over-abstracted, or that fossil water is being used, then the LCA practitioner should find the necessary values to develop ADP factors for the specific water bodies in question.

$$\mathrm{ADP_i} = \frac{\mathrm{ER_i} - \mathrm{RR_i}}{(\mathrm{R_i})^2} \times \frac{(\mathrm{R_{Sb}})^2}{\mathrm{DR_{Sb}}}$$

On the basis of the data published by Custodio (2002) on groundwater consumption and availability, Milà i Canals et al. (2009) estimated ADP values for Califorina and Almeria over-exploited aquifers. The resulting factors are of several orders of magnitude higher than those of scarce resources such as fossil fuels or metals. Characterization factors have not been calculated for other aquifers.

- Cumulative Exergy Extraction from the Natural Environment CEENE (Dewulf et al. 2007)

According to Bösch et al., (2007) "*the exergy of a resource accounts for the minimal work necessary to form the resource or for the maximally obtainable amount of work when bringing the resource's components to their most common state in the natural environment. Exergy measures are traditionally applied to assess energy efficiency, regarding the exergy losses in a process system. However, the measure can be utilised as an indicator of resource quality demand when considering the specific resources that contain the exergy".*

Cumulative Exergy Extraction from the Natural Environment (CEENE) (DeWulf et al., 2007) depicts total exergy removal from nature to provide a product, summing up the exergy of all resources required. The chemical exergy of any species can be calculated from the exergy values of the reference compounds, considering its reference reaction (De Meester et al., 2006). Water is therefore characterized because of its chemical and potential exergy, on the basis of the reference state for water defined by Szaegut et al. (1988). CEENE consists in an update and refinement of the Cumulative Exergy Demand (CExD) (Bösch et al., 2007) and, according to DeWulf et al. (2007), is the most comprehensive resource indicator which evaluates energy carriers, non-energetic resources (including water) and land occupation. Water is only one of the resources covered by the CExD and CEENE methodologies. The methodology does not look at the scarcity aspect and no factors for spatial differentiation are provided.

- Solar Energy Demand (Rugani et al., 2011)

The model, based on the emergy concept with some modifications, amis at measuring the Solar Energy Demand (SED) of the extraction of atmospheric, biotic, fossil, land, metal, mineral nuclear and water resources. The purpose is to measure the amount of solar energy that would be needed to replace the resource that is extracted from the environment. SED does not account for energy available for human use after extraction.

The model measures the flow of solar energy in the transformations occurred in the formation of the resource, before its extraction. It defines resources having a turnover time of less than year as renewable, whereas resources having a turnover time over one year as non-renewable. The main difference between SED and emergy is that emergy do not allow for allocation, whereas SED include allocation between co-products.

$SEF_i = S/F_i$

Where: SEF: is Solar Energy Factor (MJse/unit); S is the annual baseline of energy that flows in the geobiosphere, i.e., sum of emergy in sun, tide, and crustal heat; Fi is annual flow of the resource i (e.g., kg/year), estimated by the ratio of the stored quantity and its turnover time.

5.5.2 Characterization factors at midpoint

Not all of the selected models could be considered for the comparison as characterization factors (CFs) were not made available. Loubet et al. (2013) made available only CFs for two case studies; 'Milà i Canals et al. (2009) – Smathkin' only provide a limited set of watershed-specific factors but no country values and 'Milà i Canals et al. (2009) – ADP' only provide site-specific characterization factors for two watersheds. Instead, all the other pre-selected models were considered as they provided at least country-scale CFs.

The ILCD version used (2016) covered a low number of elementary flows, which are relevant for the impact category water depletion. These were:

- for resources: 'ground water', 'lake water', 'river water', 'freshwater', 'sea water', 'water'. Moreover, 7 water stress-levels are specified for each of the flows reported above, with exclusion of 'sea water' and 'water'. Similarly, the elementary flows 'ground water', 'lake water', 'river water' and 'freshwater' are further specified for 29 OECD countries and OECD average values.
- for emissions: 'water vapor' (to all environmental media: air, water, soil). No further regionalization is available for this flow.

Common life cycle inventories datasets and LCA software currently cover country-specific elementary flows, including both water withdrawals and releases with regional resolution. Moreover, the majority of the recently published LCIA models for water scarcity are highly spatially-resolved, providing characterization factors for countries as well as for watersheds, as the spatial dimension is extremely relevant for assessing water scarcity. All the models recommend consumption to be characterized instead of withdrawals. Therefore, they are currently implemented in LCA software by including negative characterization factors for water releases (emissions).

The majority of the models provide factors that are generic for the following flows: "ground water, lake water, river water, freshwater" (i.e. blue water) and specific factors for geographic locations (countries and watersheds). A limited number of models provide factors for other elementary flows i.e. precipitation (Yano et al., 2015), water use-specific factors (Boulay et al., 2016), water stress levels (Frischknecht and Büsser Knöpfel, 2013). Models based on thermodynamics (Dewulf et al., 2007; Rugani et al. 2011) do not provide regionalized characterization factors.

A number of elementary flows have been preliminarily added to the ILCD list, including geographic locations currently missing (countries), by making use of the ISO country code - level 2, as well as two additional types of water based on the typology of water use (irrigation and non-irrigation). LCIA models' coverage of elementary flows was assessed by comparing the list of available CFs for characterizing blue water (i.e. freshwater, groundwater) to the list of ISO country codes. This was done for all pre-selected LCIA models, by c. When factors characterizing flows others than freshwater resources (including groundwater, rivers and lakes) were available within models, they had been accounted for, based on the same rationale. In table 5.2 the list of flows covered by characterization factors by each of the models is reported. The most recent models are, in general, those covering the majority of the countries. All the models are suitable for the ILCD current flows.

A correlation analyses between the sets of characterization factors provided at the country level was performed with the aim of assessing similarities and differences amongst models. The results are presented in Table 5.3.

Several patterns amongst models emerge from the results. Average factors for AWARE (YR_AVG) are highly correlated with water use-specific factors (correlation coefficient – r =0.91 and 0.88 respectively for YR_AGRI and YR_NON-AGRI); average correlation is observed, in general, between generic CTA, WTA models and AWARE, with correlation coefficients ranging from 0.47 to 0.67.

The CFs developed by Yano et al. (2015) are poorly correlated with all of the others, with highest correlation coefficient being 0.62 with Pfister et al. (2009) – resources. The model developed by Berger et al. (2014) is positively correlated with AWARE, Pfister et al. (2009) – WSI, Boulay et al. (2011b) and Hoekstra et al. (2012). The model developed by Frischknecht and Büsser Knöpfel (2013) is not correlated with other models, the highest factor observed is with ILCD (2011) (r = 0.5) as the latter is based on a former version of Frischknecht and Büsser Knöpfel (2013). Pfister et al. (2009) – WSI and Boulay et al. (2011b) show relatively high correlation (r = 0.76); this is due to the fact that both LCIA models rely on the same version of the hydrogeological model WaterGap v2, and due to the fact that both approaches model scarcity as an indicator ranging from 0.01 to 1 and attempt to capture scarcity as defined for humans, based on threshold values from Alcamo et al. (2000) and OECD (2003). The Blue water scarcity model is weakly correlated with other models based on WTA or CTA ratios (e.g. Pfister et al., 2009 – WSI; Boulay et al., 2011b; Berger et al., 2014) or absolute scarcity (AWARE).

The model 'Motoshita et al., 2014' is poorly correlated with other models due to its specificity in assessing potential impacts occurring to humans due to lack of agricultural production in reaction to water shortage. Similarly, 'Pfister et al., 2009 – resources' is poorly correlated with other models, a part from Pfister et al., 2009 – WSI, due to the its specific focus on additional effort required to extract water depleted. The models 'CEENE' (Dewulf et al., 2007) and 'SED' (Rugani et al. 2011) could not be compared as they provide only generic factors but no country-specific ones.

Table 5.2: Coverage of ILCD flows, assuming ILCD flows list with extended country coverage

ILCD flows covered (as from ISO country codes)	AWARE-UNEP (2016)	Yano et al. 2015	Berger et al. (2014) (WDI)	Frischknecht and Büsser Knöpfel (2013)	Pfister et al. 2009 - WSI	Blue water scarcity (Hoekstra et al. 2012)	Boulay et al. (2011b)	Motoshita et al. 2014 (default WSI from Pfister)	Pfister et al. (2009) - resources	CEENE (DeWulf et al., 2007)	SED (Rugani et al., 2011)
freshwater	210	153	235	157	173	131	188	138	173	generic factor	NA
ground water	210	153	235	157	173	131	188	138	173	generic factor	1 CF specific for this water type
lake water	210	153	235	157	173	131	188	138	173	generic factor	
river water	210	153	235	157	173	131	188	138	173	generic factor	1 CF specific for this water type
sea water	NA	NA	NA	NA	NA	NA	NA	NA	NA	generic factor	NA
surface water	210	153	235	157	173	131	188	138	173	generic factor	NA
water	210	153	235	157	173	131	188	138	173	generic factor	NA
stress levels	NA	NA	NA	7 for the flows: freshwater, groundwater, lake, river (21 CFs in total)	NA	NA	NA	NA	NA	NA	NA
World generic factors (unspecified type, unspecified country)	yes	Not provided in the publication	Not provided, calculated in LCA software	yes	Not provided, calculated in LCA software	Not provided, calculated in LCA software	Not provided, calculated in LCA software	Not provided in the publication	Not provided, calculated in LCA software	yes	yes

Table 5.3: Correlation analysis between models' characterization factors, based on country values as implemented in LCA software

		ILCD (2011)	Boulay et al. (2016) AWARE100 AVG			Yano et al. (2015)			Berger et al. (2014) (WDI)	Frischknecht and Büsser Knöpfel (2013)	Pfister et al. 2009 -WSI	Blue water scarcity (Hoekstra et al. 2012)	Boulay et al. (2011b) - simplified	Motoshita et al. 2014 (assuming WSI from Pfister)	Pfister et al. (2009) - resources
			YR_AVG	YR_AGRI	YR_NON_AGRI	Precipitation - Country avg	Surface water - Country avg	Ground water - Country avg							
ILCD (2011)		1.00	0.02	0.02	0.03	0.04	-0.06	-0.13	0.22	0.50	0.21	-0.12	0.11	0.05	0.00
UNEP (2016) AWARE100 AVG	YR_AVG	0.02	1.00	0.91	0.88	0.38	0.16	0.17	0.66	0.08	0.67	0.57	0.57	0.29	0.45
	YR_AGRI	0.02	0.91	1.00	0.71	0.35	0.12	0.13	0.59	-0.06	0.67	0.57	0.59	0.29	0.40
	YR_NON_AGRI	0.03	0.88	0.71	1.00	0.52	0.35	0.34	0.66	0.23	0.61	0.49	0.47	0.30	0.55
Yano et al. (2015)	Precipitation - Country avg	0.04	0.38	0.35	0.52	1.00	0.61	0.63	0.44	0.13	0.42	0.16	0.35	0.17	0.53
	Surface water - Country avg	-0.06	0.16	0.12	0.35	0.61	1.00	0.98	0.44	0.31	0.37	0.20	0.34	0.33	0.62
	Groundwater - Country avg	-0.13	0.17	0.13	0.34	0.63	0.98	1.00	0.43	0.23	0.34	0.22	0.33	0.30	0.55
Berger et al. (2014) (WDI)		0.22	0.66	0.59	0.66	0.44	0.44	0.43	1.00	0.15	0.76	0.59	0.66	0.36	0.56
Frischknecht and Büsser Knöpfel (2013)		0.50	0.08	-0.06	0.23	0.13	0.31	0.23	0.15	1.00	0.20	0.12	0.13	0.13	0.48
Pfister et al. 2009 -WSI		0.21	0.67	0.67	0.61	0.42	0.37	0.34	0.76	0.20	1.00	0.62	0.76	0.47	0.72
Blue water scarcity (Hoekstra et al. 2012)		-0.12	0.57	0.57	0.49	0.16	0.20	0.22	0.59	0.12	0.62	1.00	0.60	0.35	0.47
Boulay et al. (2011b) - simplified		0.11	0.57	0.59	0.47	0.35	0.34	0.33	0.66	0.13	0.76	0.60	1.00	0.33	0.52
Motoshita et al. 2014 (assuming WSI from Pfister)		0.05	0.29	0.29	0.30	0.17	0.33	0.30	0.36	0.13	0.47	0.35	0.33	1.00	0.47
Pfister et al. (2009) -resources		0.00	0.45	0.40	0.55	0.53	0.62	0.55	0.56	0.48	0.72	0.47	0.52	0.47	1.00

5.6 Models evaluation

Table 5.4 – part 1: Summary table of the evaluation of the models are reported below. The extended table with the detailed scores for each model is reported in Annex 5.1.

	Boulay et al. 2016 (AWARE)		Yano et al. (2015)		Berger et al. (2014) (WDI)		Ecological scarcity (Frischknecht and Büsser Knöpfel, 2013)		Pfister et al. (2009) - (WSI)	
Completeness of the scope	B/C	Generic model in terms of scarcity, more oriented towards human health and ecosystem quality, global in scope.	B/C	Generic model in terms of scarcity, more oriented towards resource depletion, global in scope.	B/C	Generic model in terms of scarcity, more oriented towards resource depletion, global in scope.	B/C	Generic distance-to-target model in terms of scarcity, more oriented towards human health, global in scope.	B/C	Generic model in terms of scarcity/stress, more oriented towards human health, global in scope.
Environmental relevance	B	High environmental relevance to assess surface and groundwater use, considering monthly environmental water requirements and detailed human water consumption at a sub-watershed scale; underlying models are amongst the most resolved. Arid areas are well reflected and different uses are reflected in the aggregation at lower resolution levels. Does not consider downstream impacts, rain water, green or fossil groundwater use, nor water quality. ILCD flows are covered with exception for stress-levels	B/C	Medium environmental relevance to assess generic surface, ground and rain water use. It considers unavailability/ run-off on a monthly and detailed resolution at a sub-watershed scale. Arid areas are well reflected. Does not consider: human water demand, environmental water requirements, downstream impacts, green or fossil groundwater use, nor water quality. ILCD flows are covered, with exception for stress-levels	B/C	Average environmental relevance to assess surface and groundwater use and depletion for human use. Low environmental relevance to assess ecosystems. It considers detailed human water consumption at a sub-watershed scale, the underlying models are highly resolved. Arid areas are modelled separately. Does not consider: environmental water requirements, downstream impacts, rain water, green or fossil groundwater use, nor water quality. ILCD flows are covered with exception for stress-levels	C	Average environmental relevance, it assesses stress levels due to surface and groundwater use (incl. fossil), low in assessing ecosystems. It considers human water withdrawals at country scale instead of consumption; the underlying models have a coarse resolution although they can be substituted with more resolved ones. Arid areas are not reflected. Does not consider: environmental water requirement, downstream impacts, rain water, green or fossil groundwater use, nor water quality. ILCD flows are fully covered.	B/C	Average environmental relevance, it assesses surface and groundwater use, considering human water withdrawals at sub-watershed scale instead of consumption. The underlying models have high resolution. Arid areas are not reflected. Does not consider: environmental water requirements, downstream impacts, rain water, green or fossil groundwater use, nor water quality. ILCD flows are covered with exception for stress-levels
Scientific robustness & Uncertainty	B	Modelling choices related to the characterization model show average scientific robustness, as some sensitive assumptions were made, although they well performed against reported cases of watersheds affected by high scarcity levels (closed basins). Moreover, the choices were legitimated by experts through consensus building process. Uncertainty and sensitivity are partially provided. Underlying models	B	Modelling choices related to the characterization model show high scientific robustness. Aggregation to country-scale CFs is performed through relevant proxies and their uncertainty distribution is reported. Underlying models are partially analysed for goodness of fit for water availability, and	B/C	Modelling choices related to the characterization model show average scientific robustness; the curve is set consistently with OECD recommendations on stress values for human uses of water. Uncertainty and sensitivity are illustrated in a comprehensive manner; no uncertainty ranges are provided. Underlying models are recent, post-calibrated but their uncertainty assessments are limited; temporal representativeness is good.	C/D	Modelling choices related to the characterization model show average scientific robustness. No uncertainty is described. The underlying bio-physical models are not validated, although their temporal representativeness is good to average. The model is not published in a peer-reviewed article.	B/C	Modelling choices related to the characterization model show average scientific robustness; as some speculative assumptions and described qualitative uncertainty. Underlying models are recent, post-calibrated but their uncertainty assessments are limited; temporal representativeness is good.

	Boulay et al. 2016 (AWARE)		Yano et al. (2015)		Berger et al. (2014) (WDI)		Ecological scarcity (Frischknecht and Büsser Knöpfel, 2013)		Pfister et al. (2009) - (WSI)	
		are state-of-the-art, post-calibrated but their uncertainty assessments are limited; their representativeness is good to high.		qualitative discussion of uncertainty sources was provided; their representativeness is good to high.						
Documentation, Transparency & Reproducibility	B	High transparency, documentation and reproducibility. Documentation and factors are readily accessible (report, web link and scientific journal) and value choices are transparent, only the model is not operationalized for re-calculation. Background data needs to be requested to data provider.	B/C	High transparency, documentation and reproducibility. Documentation and factors are readily accessible (scientific article in open-access journal) and value choices are transparent, the model is not operationalized for re-calculation; background data needs to be requested to data provider.	B/C	High transparency, documentation and reproducibility. Documentation and factors are accessible in form of scientific article upon fee-payment. Value choices are transparent; the model is not operationalized for re-calculation. Background data needs to be requested to data provider.	A/B	High transparency, documentation and reproducibility. Documentation and factors are readily accessible (reports) and value choices are transparent, however not discussed. The model is not operationalized for re-calculation; background data is accessible.	B/C	High transparency, documentation and reproducibility. Documentation and factors are accessible in form of scientific article upon fee-payment. Value choices are implicit in the equations. The model is not operationalized for re-calculation, background data needs to be requested to data provider.
Applicability	B	The model is compatible with LCA, readily available for LCA software, normalization factors can be calculated, and the flows can be conformed to ILCD nomenclature and units	B/C	The model is compatible with LCA, normalization factors can be calculated, and the flows can be conformed to ILCD nomenclature and units. Only rain water flows would be missing to apply all aspects of the model.	B	The model is compatible with LCA, readily available for LCA software, normalization factors can be calculated, and the flows can be conformed to ILCD nomenclature and units.	A/B	The model is compatible with LCA, readily available for LCA software, normalization factors can be calculated, and the flows can be conformed to ILCD nomenclature and units	A/B	The model is compatible with LCA, readily available for LCA software, normalization factors can be calculated by use of average factors provided by the authors, and the flows can be conformed to ILCD nomenclature and units
Characterization factors	B	Factors are readily usable, at high spatial-temporal resolution as well as lower one, including user-specific resolution. Maturity is relatively low.	B/C	Factors are available at low spatial-temporal resolution. Maturity is relatively low as factors have been tested in a simplified case study only.	B	Factors are readily usable, at high spatial-temporal resolution as well as lower one. Maturity is relatively low.	C	Factors are readily usable, at low spatial /temporal resolution. Some issues with characterization factors have been reported in literature	A/B	Factors are readily usable, at high spatial-temporal resolution as well as lower one. Maturity is relatively high.
Overall evaluation of science based criteria	B	The model has positive features for what concerns environmental relevance as it includes environmental water requirements as well as it captures aridity. Applicability is good but lower than other models as this models is newly developed and not yet extensively tested, although studies are ongoing. Resolution is high, factors characterize also different types of water uses	B/C	This model is scientifically robust but less relevant than others as it doesn't include important elements to the definition of scarcity. It's implementation is low in LCA softwares, although it is able to characterize elementary flows such as groundwater and	B/C	The model well performs in terms of applicability and robustness of the characterization factors, as well as resolution of the underlying models. the environmental relevance is limited as it does not include environmental water requirements and treat aridity inconsistently as special case. The impacts are based on a CTA ratio and further modelled as logistic function matching OECD/Alcamo et al. 2000 thresholds for which scarcity/stress is defined as affecting humans rather than freshwater ecosystems	B/C	The model well performs in terms of applicability and robustness of the characterization factors. the environmental relevance is limited as it does not include environmental water requirements; underlying hydrogeological models have low resolution but others can be used instead. The impacts are	B/C	The model well performs in terms of applicability and robustness of the characterization factors, as well as resolution of the underlying models, although a bit outdated. the environmental relevance is limited as the model does not include environmental water requirements, is based on WTA and does not treat aridity issues. The impacts

	Boulay et al. 2016 (AWARE)		Yano et al. (2015)		Berger et al. (2014) (WDI)		Ecological scarcity (Frischknecht and Büsser Knöpfel, 2013)		Pfister et al. (2009) - (WSI)	
				precipitation separately from surface water				based on a squared WTA ratio where availability is defined according to OECD for humans rather than for freshwater ecosystems.		are based on a WTA ratio and further modelled as logistic function matching OECD/Alcamo et al. 2000 thresholds for which scarcity/stress is defined as affecting humans rather than freshwater ecosystems
Stakeholders acceptance	B	High acceptance, as endorsed by an international group of experts, average understandability, could be integrated in policies and is neutral across industries, product or processes	C	Average acceptance, as not endorsed, but easily understandable, could be integrated in policies and is neutral across industries, product or processes	C	Average acceptance, as not endorsed, average understandability, could be integrated in policies and is neutral across industries, product or processes	C	Good acceptance, although eco-points are not so easily understandable. It was integrated in policies in the past and it is neutral across industries, product or processes; however it has been criticized by stakeholders	C	Average acceptance, as not endorsed, average understandability, could be integrated in policies and is neutral across industries, product or processes
Final Evaluation	B	It results in the most complete and robust model, also accepted by experts, it overcomes many of the limitations of the other models. Still, assumptions play an important role in the modelling and it represents a proxy of potential deprivation, with some degrees of evidence	B/C	Relatively robust model, however it lacks of environmental relevance due to the fact that current water demand is not taken into account	B/C	Relatively relevant and robust model, limited by some arbitrary assumptions and by the lack of important environmental aspects	C	Simple model based on distance to target assumptions and little resolution of input data. It has been contested by some stakeholders within the pef/oef activities	B/C	Relatively relevant and robust model, limited by some arbitrary assumptions and by the lack of important environmental aspects

Table 5.4 – part 2: Summary table of the evaluation of the models are reported below. The extended table with the detailed scores for each model is reported in Annex 5.1.

	Blue water scarcity (Hoekstra et al. 2012)		Boulay et al. (2011b)		Loubet et al. (2013)		Milà i Canals et al., 2009 - Smathkin		Motoshita et al. 2014 (assuming WSI from Pfister)	
Completeness of the scope	B/C	Generic model in terms of scarcity, more oriented towards human health and ecosystem quality, global in scope.	B	Generic model in terms of scarcity, more oriented towards human health, it also includes losses of water quality and functionality, global in scope.	B/C	Generic model in terms of scarcity, more oriented towards human health and ecosystem quality.	B/C	Generic in terms of scarcity, more oriented towards human health and ecosystem quality, global in scope.	B/C	Model covering a specific impact pathway and AoP (HH), global in scope.
Environmental relevance	B/C	High environmental relevance, it assesses surface and groundwater use considering annual environmental water requirements and detailed human water consumption at a sub-watershed scale. The underlying models have high resolution a part from the EWR which is based on presumptive assumptions. Arid areas are not reflected. Does not consider downstream impacts, rain water, green or fossil groundwater use, nor water quality. ILCD flows are covered with exception for stress-levels	B/C	Average environmental relevance in assessing human-oriented water stress, low in assessing potential ecosystems impacts. It covers surface and groundwater use (incl. fossil) and considers human water withdrawals at country scale. Arid areas are not reflected. Underlying models have relatively high resolution. Does not consider: environmental water requirement, downstream impacts, green water use or fossil groundwater use. ILCD flows are covered with exception for stress-levels.	B/C	High environmental relevance, it assesses surface and groundwater use, considering annual environmental water requirements and detailed human water consumption at a sub-watershed scale. Arid areas are not reflected. Considers downstream impacts. Does not consider rain water, green or fossil groundwater use, nor water quality. ILCD flows are covered with exception for stress-levels.	C	High relevance to assess surface and groundwater use, considering environmental water requirements and detailed human water withdrawals at a country scale. Arid areas are not reflected. Underlying models have relatively high resolution. Does not consider: downstream impacts, rain water, green or fossil groundwater use, nor water quality. ILCD flows are covered with exception for stress-levels.	C	High relevance to assess agricultural water deprivation from surface and groundwater use. Low environmental relevance for assessing ecosystems as environmental water requirements are not included as the model is designed for addressing scarcity of water in agriculture specifically. It considers detailed human water withdrawal at country scale. Arid areas are not reflected. Underlying models have relatively high resolution. Does not consider: downstream impacts, rain water, green or fossil groundwater use, nor water quality. ILCD flows are covered with exception for stress-levels.

	Blue water scarcity (Hoekstra et al. 2012)		Boulay et al. (2011b)		Loubet et al. (2013)		Milà i Canals et al., 2009 - Smathkin		Motoshita et al. 2014 (assuming WSI from Pfister)	
Scientific robustness & Uncertainty	B/C	Modelling choices related to the characterization model show average scientific robustness. Some speculative assumptions were made and uncertainty is partially described qualitatively. Underlying models are recent but their uncertainty assessments are limited; temporal representativeness is high.	B/C	Modelling choices related to the characterization model show average scientific robustness, although some speculative assumptions and described uncertainty. Underlying models are recent, post-calibrated but their uncertainty assessments are limited; temporal representativeness is good.	B/C	Modelling choices related to the characterization model show average scientific robustness. Speculative assumptions are made in the definition of the scarcity index, uncertainty sources and limitations are well described although no quantitative estimate is reported. Underlying models are recent, post-calibrated but their uncertainty assessments are limited, for EWR specifically. Temporal-representativeness is high.	C	Modelling choices related to the characterization model show average scientific robustness, as the modelling is based on speculative assumptions. Uncertainty is not described. Underlying bio-physical models are not validated, only; temporal representativeness is good.	B	Modelling choices related to the characterization model show average scientific robustness; the modelling curve is set consistently with OECD recommendations on stress values for human uses of water, additional modelling is based on speculative assumptions. A number of sources of uncertainty is discussed, some of them are reported numerically, not all of them are discussed in detail. Underlying models are recent, post-calibrated but their uncertainty assessments are limited. Temporal representativeness is good.
Documentation, Transparency & Reproducibility	B/C	Average transparency, documentation and reproducibility. . Characterization factors at country level are available only in LCA software and no documentation is provided on their calculation. Value choices are transparent, only the model is not operationalized for re-calculation. Background data is readily available.	B/C	High transparency, documentation and reproducibility. Documentation and factors are accessible in form of scientific article upon fee-payment. Value choices are transparent, only the model is not operationalized for re-calculation. Background data needs to be requested to data provider.	C	Relatively transparent and reproducible, documentation of the characterization model is accessible in form of a scientific paper, but limited accessibility as the factors were not calculated.	C	High transparency, documentation and reproducibility. Documentation and factors are accessible in scientific journals; value choices are transparent and qualitative discussed. The model is not operationalized for re-calculation. Background data needs to be requested to data provider.	C	High transparency, documentation and reproducibility. Documentation and factors are accessible in scientific journals; value choices are transparent and qualitative discussed. The model is not operationalized for re-calculation. Background data needs to be requested to data provider.
Applicability	B	The model is compatible with LCA, readily available for LCA software, normalization factors can be calculated, and the flows can be conformed to ILCD nomenclature and units	B	The model is compatible with LCA, readily available for LCA software, normalization factors can be calculated, and the flows can be conformed to ILCD nomenclature and units	D	The model is compatible with LCA, in principle available for LCA software, however the characterization factors, elementary flows and normalization factors are not available and cannot be easily calculated as detailed data is needed	C	The model is compatible with LCA, in principle available for LCA software, however the characterization factors, elementary flows and normalization factors are not available although they can be easily calculated	B	The model is compatible with LCA, readily available for LCA software, normalization factors can be calculated, and the flows can be conformed to ILCD nomenclature and units

	Blue water scarcity (Hoekstra et al. 2012)		Boulay et al. (2011b)		Loubet et al. (2013)		Milà i Canals et al., 2009 - Smathkin		Motoshita et al. 2014 (assuming WSI from Pfister)	
Characterization factors	B/C	Factors are readily usable, at high spatial /temporal resolution as well as lower one. Maturity is relatively low, coverage is partial.	B	Factors are readily usable, at high spatial resolution. Temporal resolution is limited to year. Maturity is relatively high.	D	Factors are not available, maturity is low.	D	Factors are not available, maturity is low.	C	Factors are available at low spatial-temporal resolution. Maturity is relatively low as they have not been implemented in software
Overall evaluation of science based criteria	B/C	The model considered environmental water requirement and human consumption. However arid areas are not addressed. The model was not originally developed for LCIA applications and its coverage is limited; factors are available but for a limited part of the globe.	B/C	The model well performs in terms of applicability and robustness of the characterization factors, as well as resolution of the underlying models, although a bit outdated. The environmental relevance is limited as the model does not include environmental water requirements, it is based on CTA and does not treat aridity issues. The impacts are based on a CTA ratio and further modelled as logistic function matching OECD/Alcamo et al. 2000 thresholds for which scarcity/stress is defined as affecting humans rather than freshwater ecosystems	C/D	The model is relevant and robust in the way it deals with impacts of downstream users, however characterization factors were not calculated due to the lack of detailed information available at the sub-watershed scale, at the level of the globe. Therefore the applicability of this model is low	C/D	The model has high environmental relevance, as it includes both human consumption and environmental water requirements; however it has been calculated at the watershed level only and its applicability is low	B/C	The model is specific for human health - lack of water resources for agricultural production. Therefore it is specific in scope but lacks other impact pathways. The values of the midpoint characterization factors are made available at the country scale only.
Stakeholders acceptance	C	Average acceptance, as not endorsed, average understandability, could be integrated in policies and is neutral across industries, product or processes	C	Average acceptance, as not endorsed, average understandability, could be integrated in policies and is neutral across industries, product or processes	C	Average acceptance, as not endorsed, average understandability, could be integrated in policies and is neutral across industries, product or processes	C	Average acceptance, as not endorsed, average understandability, could be integrated in policies and is neutral across industries, product or processes	C	Average acceptance, as not endorsed, average understandability, could be integrated in policies and is neutral across industries, product or processes
Final Evaluation	B/C	Relatively relevant model, limited by some arbitrary assumptions, coverage is partial	B/C	Relatively relevant and robust model, limited by some arbitrary assumptions and by the lack of important environmental aspects	C/D	Highly relevant methodological development, however far from being fully operational at the resolution needed	C/D	Relatively relevant model, limited by some arbitrary assumptions, coverage is partial; factors are not provided at the needed scale	C	Relatively robust modelling focussing on human health impacts; limited evidence of the impact pathway; not relevant for freshwater ecosystems

Table 5.4 - part 3: Summary table of the evaluation of the models are reported below. The extended table with the detailed scores for each model is reported in Annex 5.1. -

	Pfister et al. 2009 – resources (F_depletion)		Milà i Canals et al., 2009 (ADP)		CEENE (Dewulf et al. 2007)		SED (Rugani et al. 2011)	
Completeness of the scope	B/C	Model covering a specific impact pathway and AoP (Resources).	B/C	Model covering a specific impact pathway and AoP (Resources).	C	Coverage of a specific impact pathway and AoP (Resources), scarcity aspects aren't taken into account, whereas thermodynamic aspects are considered.	C	Coverage of a specific impact pathway and AoP (Resources), scarcity aspects aren't taken into account, whereas thermodynamic aspects are considered.
Environmental relevance	C	High relevance to assess resource depletion from surface and groundwater use (efforts required for desalinating), considering detailed human water withdrawals at a sub-watershed scale. Low environmental relevance to assess short-term scarcity for humans and ecosystems. Arid areas are not reflected. Does not consider: environmental water requirements, downstream impacts, rain water, green or fossil groundwater use, nor water quality. ILCD flows are covered with exception for stress-levels.	D	High environmental relevance for assessing groundwater (long-term) resource depletion from groundwater use, considering detailed human water withdrawal at a sub-watershed scale. however, low environmental relevance in capturing generic scarcity for humans and ecosystems. Arid areas are not reflected. Does not consider: downstream impacts, rain water, surface water green or fossil groundwater use, nor water quality. ILCD flows are covered with exception for stress-levels.	D	Low environmental relevance. The majority of the environmental aspects which are relevant in defining scarcity, stress and depletion on humans or ecosystems are missing from the model, as the model aims to quantify something different i.e. the eXergetic cost of extraction and use of a resource. ILCD flows are covered with exception for stress-levels.	D	Low environmental relevance. The majority of the environmental aspects which are relevant in defining scarcity, stress and depletion on humans or ecosystems are missing from the model, as the model aims to quantify something different i.e. the eMergetic cost of extraction and use of a resource. ILCD flows are covered with exception for stress-levels.
Scientific robustness & Uncertainty	B/C	Modelling choices related to the characterization model show average scientific robustness; speculative assumptions are made in the definition of the equation. Limitations are not clearly discussed; variability associated with aggregation at country scale is not discussed. Underlying models are recent, post-calibrated but their uncertainty assessments are limited. temporal representativeness is good.	C/D	Modelling choices related to the characterization model show average scientific robustness; speculative assumptions are made in the definition of the equation. Limitations are not discussed. Underlying models are recent, post-calibrated but their uncertainty assessments are limited. Temporal representativeness is good.	B/C	Modelling choices related to the characterization model show high scientific robustness based on solid thermodynamic theory. Little discussion on quality of the input data and uncertainty is provided. No sensitivity analysis was performed on the results.	B/C	Modelling choices related to the characterization model show high scientific robustness based on thermodynamic theory, however the model through which calculations are performed is highly uncertain, being all estimations dependent on a specific baseline. In spite of this, the quality of the input data is discussed and the uncertainty of the outcomes is provided together with sensitivity analysis.
Documentation, Transparency & Reproducibility	C/D	High transparency, documentation and reproducibility. Documentation is accessible upon fee payment, value choices are implicitly defined in the equations. CFs are not available for the midpoint indicator. The model is not operationalized for re-calculation. Background data needs to be requested to data provider.	C/D	Limited transparency and reproducibility as input data is not specified and limited accessibility as the factors were not calculated	B	the model is well documented, transparent and reproducible, however it is published in form of scientific article, not freely accessible	B	the model is well documented, transparent and reproducible, however it is published in form of scientific article, not freely accessible

	Pfister et al. 2009 – resources (F_depletion)		Milà i Canals et al., 2009 (ADP)		CEENE (Dewulf et al. 2007)		SED (Rugani et al. 2011)	
Applicability	C	the model is compatible with LCA, readily available for LCA software, normalization factors can be calculated, and the flows can be conformed to ILCD nomenclature and units; midpoint factors have been calculated by the authors but not made available	C	the model is compatible with LCA, in principle available for LCA software, however the characterization factors, elementary flows and normalization factors are not available	B	the model is compatible with LCA, readily available for LCA software, normalization factors can be calculated, and the flows can be conformed to ILCD nomenclature and units	B	the model is compatible with LCA, readily available for LCA software, normalization factors can be calculated, and the flows can be conformed to ILCD nomenclature and units
Characterization factors	C/D	Factors are readily usable for endpoint, at high spatial-temporal resolution as well as lower one. Maturity is relatively high. No factors were made available for the midpoint indicator, can be calculated	D	Factors are not available, can be easily calculated, however maturity is low.	C	the characterization factors have been tested over a number of case studies and journal papers, however their ability to distinguish between water resources types and space is low	C	the characterization factors have been tested over a number of processes, however their ability to distinguish between water resources types and space is low
Overall evaluation of science based criteria	B/C	the model is specific for resource depletion, it is specific in scope but lacks other impact pathways. The values of the midpoint characterization factors underlying the endpoint are not made publicly available. The underlying model has good resolution although it is a slightly outdated.	D	the model is specific for resource depletion, it is specific in scope but lacks other impact pathways. The values of the midpoint characterization factors were not calculate d by the authors due to the difficulty in getting estimates for the availability of groundwater resources	C	the model is developed to account for aspects others than water scarcity, as it focuses on thermodynamics. Therefore, in this context, the model is not environmentally relevant; moreover factors do not allow for spatially and temporarily explicit evaluations	C	the model is developed to account for aspects others than water scarcity, as it focuses on thermodynamics. Therefore, in this context, the model is not environmentally relevant; moreover factors do not allow for spatially and temporarily explicit evaluations
Stakeholders acceptance	C	Average acceptance, as not endorsed, average understandability, could be integrated in policies and is neutral across industries, product or processes	D	Low acceptance, as not endorsed, not easily understandable, could be integrated in policies and is neutral across industries, product or processes	D	Low acceptance, as not endorsed, not easily understandable, could be integrated in policies and is neutral across industries, product or processes	D	Low acceptance, as not endorsed, not easily understandable, could be integrated in policies and is neutral across industries, product or processes
Final Evaluation	C	relatively robust modelling based on WTA ratio, however midpoint factors are not made available by the authors, only at the endpoint	D	weak modelling based on available, high relevance for long term scarcity. Issues in the communications of the unit and of the meaning of the indicators can be expected	C	robust modelling based on thermodynamics, however with little environmental relevance for water scarcity. Issues in the communications of the unit and of the meaning of the indicators can be expected	C	relatively robust modelling based on thermodynamics, however with little environmental relevance for water scarcity. Issues in the communications of the unit and of the meaning of the indicators can be expected

5.7 Discussion on models evaluation

The results presented in table 5.4 are summarized below.

Completeness of the scope: none of the models considered is complete in scope as each of the models has a specific focus. Therefore, only part of the impact pathways is covered. AoP-specific models such as CEENE and SED show low completeness as the way water consumption is addressed builds on a different rational than hydrologic scarcity. The majority of the models score B/C, showing that none of them is significantly better than the others, in general terms. In relative terms, the better performing are those models, which include human water consumption, water availability and/or ecosystems water requirement.

Environmental Relevance: this group of criteria is the one which provides more information, allowing for significant distinctions to be made across models. Overall, the model AWARE scores better than the others as it does consider both human and ecosystem demands and it accounts consistently for aridity. Moreover, it is based on a hydrologic model that is more recent, detailed and complete than the others. Some of the other models are characterized by some interesting features, such as the coverage of downstream users (Loubet et al., 2013), the inclusion of climate variability (Pfister et al., 2009 – WSI), the coverage of a relatively high number of elementary flows (Yano et al., 2015), on top of relatively good underlying models which make them suitable for LCIA assessments, although not the best performing in general.

Scientific robustness and uncertainty: models have been evaluated on the basis of the underlying methodological choices as well as on the robustness of the theory and underlying data used to calculate characterization factors. None of the models can be defined as 'robust' as all of them heavily rely on modelling assumptions, which cannot be empirically tested against observations. Few attempts have been recently made (e.g. Boulay et al., 2018) to compare results with metrics of scarcity others than those rooted in LCA. The AWARE model performed reasonably well against a set of world watersheds known to be severely affected by water scarcity (i.e. closed basins), providing partial validation to the model. The model developed by Yano et al. (2015) minimizes value choices and it is based on physical properties only. Other models make use of a set of thresholds of stress, which are somehow set arbitrarily.

Documentation, Transparency and Reproducibility: all models are relatively well documented, with some differences in accessibility of the input data, underlying models and in the availability of the characterization model for practitioners. Many of the models are published in scientific journals accessible upon fee payment; whereas others are made accessible to the practitioners through technical reports or web-pages. Other differences observed in the scores can be attributed to whether value choices were transparently reported and discussed in the underlying documentation.

Applicability: different levels of applicability can be found across the models selected for analysis. Some of them in fact are not yet made fully operational in LCA software and relatively high effort would be required for that. For some models (Mila i Canals et al. 2009 – Smathkin; Mila i Canals et al. 2009 – ADP, Loubet et al., 2013) factors were not made available.

Characterization factors: no particular issues were identified while testing the available characterization factors, for those models reporting values. Ongoing studies within WULCA are assessing whether AWARE factors would well perform in a number of case studies, whereas other models had been already tested by practitioners due their availability in LCA software.

Stakeholder's acceptance: the AWARE model scores higher than the others as it is the outcome of a consensus-building process led by UNEP/SETAC. All the other models scored similarly with exception of the models based on thermodynamics and for Mila i Canals et al. 2009 – ADP for which communicability remains a challenge (being the latter expressed in kg of Antimony eq.).

5.8 Recommended default model for midpoint

Based on the evaluation reported in table 5.4, the recommended model for midpoint LCIA is AWARE (Boulay et al. 2016 as presented in UNEP 2016), applied at country scale, without: i) differentiating between agricultural and non-agricultural uses; and ii) monthly resolution.

5.9 Additional environmental information

In order to include an overall assessment of the water consumed, an additional environmental information for water may be added, with the following indicator:

- "net blue water consumption" (i.e. net freshwater balance)

5.10 Models for endpoint

In this assessment, endpoint models were not considered for evaluation and recommendations. This is because the level of development of endpoint models is less mature than midpoint ones and research activities on human health, ecosystem quality and resources AoPs are still ongoing within the WULCA working group. UNEP/SETAC recommendations for a specific part of the human health impact pathway have been published for human health (Boulay et al. 2016), whereas recommendations of models for a mechanistic model structure for assessing impacts to ecosystems quality and resources are expected to be made in the timeframe 2017-2018.

5.11 Consistency between midpoint and endpoint models

The model recommended at the midpoint level is not consistent with endpoint models as it aims at assessing potential water deprivation for a generic user of water resources regardless of the fact it is humans or freshwater ecosystems. Instead, endpoint models are user-specific by definition.

5.12 Classification of the recommended default models

Although being developed for overcoming major limitations of other models, the AWARE model (Boulay et al. 2016) is characterized by a series of modelling choices, which are based upon expert judgment rather than on pure scientific evidence. This stems from the fact that it attempts to provide a generic value of scarcity at the midpoint, which applies regardless of the fact that scarcity is potentially affecting a specific user amongst humans and freshwater ecosystems. Nevertheless, the model is being tested by a significant number of LCA case studies and it well performed already against other measures of scarcity such as closed basins (see Boulay et al., 2017), showing its ability to identify highly stressed situations, at least. Besides, being characterized by epistemic uncertainty, the model is expected to show a proper behaviour in identifying areas in which at least a water user potentially suffers water deprivation in reaction to the consumption of an additional volume of water. Therefore, the model AWARE (Boulay et al. 2016) is classified as 'recommended but to be applied with caution'' i.e. Level III.

5.13 Recommended characterization factors including calculation principles for midpoint

The requirement for the PEF/OEF is that all assessments are as default to be conducted at country level. The country-scale characterization factors recommended for use within the PEF/OEF context are available at the EPLCA website at http://eplca.jrc.ec.europa.eu/LCDN/developerEF.xhtml. Characterisation factors are recommended for blue water only. The original flows developed for AWARE, available at http://www.wulca-waterlca.org/project.html, were mapped to updated ILCD compliant elementary flows for use in Environmental Footprint.

Notwithstanding the characterization factors of AWARE are available at different temporal and spatial scales (month/year, watershed/country) as well as water use types (agriculture/non-agriculture), due to applicability reasons, they are not part of the recommendation.

5.14 Normalisation factors

Source and data used to calculate the normalisation factors are available in Crenna et al. 2019. The EF normalisation factors to be used are available at http://eplca.jrc.ec.europa.eu/LCDN/developerEF.xhtml.

5.15 Research needs

As pointed out in the previous sections, the following research needs can be identified for this impact category:

- better understanding of the relationships between land/use green water and blue water;
- better definition of water functionalities for assessing water availability;
- better understanding and modelling of endpoint impacts on human health, ecosystem quality and resource depletion, with specific reference to mechanistic approach linking water consumption to problems and damages on ecosystems;
- consideration and characterization of non-consumptive water uses such as thermal emissions.
- watersheds seem to be a logical choice for assessing the impacts of water consumption; nevertheless 'adjusting the geographic resolution of data to a scale that warrants an appropriate assessment, without making the application too complex, is one of the challenges that remains to be confronted' (Hellweg and Milà i Canals (2014)).

Ongoing research activities within WULCA members are being focused on these aspects and are expected to provide results in the timeframe 2017-2018. As a preliminary result a model covering a part of the impact pathways leading to damages to human health (i.e. on agricultural water deprivation only) was recommended within the UNEP/SETAC Pellston workshop, whereas another component of the same model dealing with impacts associated to lack of water for sanitation was considered to be not yet robust enough for recommendations.

Additional developments in the field of life cycle inventory datasets. As pointed out by Frischknecht and Büsser Knöpfel (2013), as well as Pfister et al. (2015), an advisable feature of life cycle inventory datasets is that unit processes are modelled so that they allow for the quantification of consumptive water by ensuring the entire water mass balance, including water embodied in products and wastes, as well as the use of the most appropriate characterization factors.

Moreover, as well-known by the LCA community, the combination of LCA software with geographic information systems (GIS), together with the systematic regionalization of background processes at geographical scales such as countries or lower, would allow for an effective use of available LCIA models currently available in literature. In fact, the majority of LCIA models assessing water scarcity already provide both country- and watershed-specific characterization factors which, in order to be properly applied, would need geographical specification for both background and foreground inventories. This would significantly improve the robustness as well as reduce the uncertainty associated with the assessment of impacts associated to water scarcity.

5.16 References of the chapter on water impacts

- Alcamo, J., Henrichs, T. & Rosch, T. (2000). World water in 2025 - Global modeling and scenario analysis for the World Commission on Water for the 21st century. Kassel World Water series.
- Alcamo, J., Döll, P., Henrichs, T., Kaspar, F., Lehner, B., Rösch, T., & Siebert, S. (2003a). Development and testing of the WaterGAP 2 global model of water use and availability. Hydrological Sciences Journal, 48(3), 317-337.
- Alcamo, J., Döll, P., Henrichs T., Kaspar, F., Lehner, B., Rösch, T., Siebert, S. (2003b). Global estimates of water withdrawals and availability under current and future "business-as-usual" conditions. Hydrological Sciences Journal, 48(3), pp.339–348. Available at: http://dx.doi.org/10.1623/hysj.48.3.339.45278.
- Asheesh, M. (2007). Allocating the gaps of shared water resources (the scarcity index) case study Palestine Israel. In Water Resources in the Middle East Eds Shuval H. and Dweik H. pp 241-248 ISBN-13 978-3-540-69508-0 Springer
- Bayart, J.B. (2008). Quantification des impacts reliés à l'utilisation de la ressource eau en analyse de cycle de vie: définition d'un cadre d'étude et développement de facteurs de caractérisation (Maîtrise és Sciences Appliquées). Ecole Polytechnique de Montréal, Montréal
- Bayart, J.B., Bulle C., Deschênes L., Margni M., Pfister S., Vince F., Koehler A. (2010). A framework for assessing off-stream freshwater use in LCA. Int J Life Cycle Assess 15(5):439–453
- Bayart, J.B., Worbe, S., Grimaud, J., Aoustin, E. (2014). The Water Impact Index: a simplified single-indicator approach for water footprinting. The International Journal of Life Cycle Assessment, 19(6), pp.1336–1344.
- Berger, M., van der Ent, R., Eisner, S., Bach, V., Finkbeiner, M. (2014). Water accounting and vulnerability evaluation (WAVE): considering atmospheric evaporation recycling and the risk of freshwater depletion in water footprinting. Environmental science & technology, 48(8), 4521-4528.
- Bjørn, A., Hauschild, M. Z. (2015). Introducing carrying capacity based normalisation in LCA: Framework and development of midpoint level references. Int. J. Life Cycle Assess. DOI: 10.1007/s11367-015-0899-2.

- Bösch, M., Hellweg, S., Huijbregts, M., Frischknecht, R. (2007). Applying cumulative exergy demand (CExD) indicators to the ecoinvent database. Int J Life Cycle Assess 12(3):181–190
- Boulay, A.M., Bouchard, C., Bulle, C., Deschênes, L., Margni, M. (2011a). Categorizing water for LCA inventory. Int J Life Cycle Assess 16 (7):639–651
- Boulay, A.M., Bulle, C., Bayart J-B, Deschênes, L., Margni, M. (2011b). Regional characterization of freshwater use in LCA: modeling direct impacts on human health. Environ Sci Technol 45(20): 8948–8957
- Boulay, A. M., Hoekstra, A. Y., & Vionnet, S. (2013). Complementarities of water-focused life cycle assessment and water footprint assessment. Environmental science & technology, 47(21), 11926-11927.
- Boulay, A. M., Motoshita, M., Pfister, S., Bulle, C., Muñoz, I., Franceschini, H., & Margni, M. (2015a). Analysis of water use impact assessment methods (part A): evaluation of modeling choices based on a quantitative comparison of scarcity and human health indicators. The International Journal of Life Cycle Assessment, 20(1), 139-160.
- Boulay, A. M., Bayart, J. B., Bulle, C., Franceschini, H., Motoshita, M., Muñoz, I., ... & Margni, M. (2015b). Analysis of water use impact assessment methods (part B): applicability for water footprinting and decision making with a laundry case study. The International Journal of Life Cycle Assessment, 20(6), 865-879.
- Boulay A.M., Bare J., De Camillis C., Döll P., Gassert F., Gerten D., Humbert S., Inaba A., Itsubo N., Lemoine Y., Margni M., Motoshita M., Núñez M., Pastor A.V. , Ridoutt B., Schencker U., Shirakawa N., Vionnet S., Worbe A., Yoshikawa S., Pfister S. (2015c). Consensus building on the development of a stress-based indicator for LCA-based impact assessment of water consumption: outcome of the expert workshops. Int J Life Cycle Assess 20:577-583
- Boulay A.M., Pfister S., Motoshita M., Schneker U., Benini L., Shabbir H. Gweewala, Brito de Figueredo M.C., Harding K., (2016). Water use related impacts: water scarcity and human health. In: Frischknecht, R., Jolliet, O. (Eds.), Global Guidance for Life Cycle Impact Assessment Indicators: Volume 1. UNEP/SETAC Life Cycle Initiative, Paris, pp. 100-124 (available at: www.lifecycleinitiative.org/applying-lca/lcia-cf/, accessed January 2017).
- Boulay A.M., Bare J., Benini L., Berger M., Lathuillière M.J., Manzardo A., Margni M., Motoshita M., Núñez M., Pastor A.V., Ridoutt B., Oki T., Worbe S., Pfister S. (2018). The WULCA consensus characterization model for water scarcity footprints: Assessing impacts of water consumption based on available water remaining (AWARE). Int J Life Cycle Assess 23 (2): 368–378 doi 10.1007/s11367-017-1333-8
- Chaves, H.M.L. & Alipaz, S., (2007). An Integrated Indicator Based on Basin Hydrology, Environment, Life, and Policy: The Watershed Sustainability Index. Water Resources Management, 21(5), 883–895. Available at: http://link.springer.com/10.1007/s11269-006-9107-2 [Accessed March 20, 2015].
- CIRAIG - Interuniversity Research Centre for the Life Cycle of Products – Polytechnique Montreal (2016). http://www.ciraig.org/fr/wateruseimpacts.php [accessed on March 2016]
- Crenna E, Secchi M, Benini L, Sala S (2019) Global environmental impacts: data sources and methodological choices for calculating normalisation factors. International Journal of LCA, doi.org/10.1007/s11367-018-1534-9

- Custodio E. (2002). Aquifer overexploitation: what does it mean. Hydrogeol J 10:254–277
- De Meester, B., Dewulf, J., Janssens, A., Van Langenhove, H. (2006). An improved calculation of the exergy of natural resources for exergetic life cycle assessment (ELCA). Environ. Sci. Technol. 40:6844–6851.
- Dewulf, J., Bösch, M.E., Demeester, B., Vandervorst, G., Vanlangenhove, H., 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
- Döll, P. (2009). Vulnerability to the impact of climate change on renewable groundwater resources: a global-scale assessment. Environmental Research Letters, 4(035006), p.12.
- EC – European Commission (2017). PEFCR Guidance 6.1, draft status.
- EC - European Commission (2013a). Communication COM/2013/0196 final Building the Single Market for Green Products Facilitating better information on the environmental performance of products and organisations. available at http://eur-lex.europa.eu/legal-content/EN/ALL/?uri=CELEX:52013DC0196
- EC - European Commission (2013b). Commission Recommendation of 9 April 2013 (2013/179/EU) on the use of common methods to measure and communicate the life cycle environmental performance of products and organisations. Official Journal of the European Union. Available at: http://eur-lex.europa.eu/legal-content/EN/ALL/?uri=OJ:L:2013:124:TOC
- 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. First edition EUR24571EN. ISBN 978-92-79- 17451-3. Available at http://eplca.jrc.ec.europa.eu/
- EEA. (2015). European environment — state and outlook 2015: Assessment of global megatrends, European Environment Agency, Copenhagen.
- Falkenmark, M., Lundqvist, J. & Widstrand, C., (1989). Macro-scale water scarcity requires micro-scale approaches. Natural Resources Forum, 13(4), pp.258–267. Available at: http://dx.doi.org/10.1111/j.1477-8947.1989.tb00348.x.
- FAO (2010). Aquastat: FAO's Information System on Water and Agriculture, data from 1998-2010 retrieved from: http://www.fao.org/ag/agl/aglw/aquastat/dbase/index.stm.
- Fekete, B., Vörösmarty, C. & Grabs, W. (2002). High-resolution fields of global runoff combining observed river discharge and simulated water balances. Global Biogeochem, 16(3): 15-1 15-6
- Finkbeiner M (2014) Product environmental footprint—breakthrough or breakdown for policy implementation of life cycle assessment? Int J Life Cycle Assess 19(2):266–271
- Flörke, M., Kynast, E., Bärlund, I., Eisner, S., Wimmer, F., & Alcamo, J. (2013). Domestic and industrial water uses of the past 60 years as a mirror of socio-economic development: A global simulation study. Global Environmental Change, 23(1), 144-156.
- Flury K., Frischknecht R., Muñoz I., Jungbluth N. (2012). Recommendation for Life cycle inventory analysis for water use and consumption. ESU-services Ltd., Treeze Ltd., Unilever, Zürich, Uster, London.

- Frischknecht, R., Steiner, R., Jungbluth, N. (2008). Methode der ökologischen Knappheit - Ökofaktoren 2006. Umwelt-Wissen Nr. 0906. Bundesamt für Umwelt (BAFU), Bern, retrieved from: www.bafu.admin.ch/publikationen/publikation/01031/index.html?l ang=de.
- Frischknecht, R., Steiner, R., Jungbluth, N. (2009). The Ecological Scarcity Method – Eco-Factors 2006. A method for impact assessment in LCA. Environmental studies no. 0906. Federal Office for the Environment (FOEN), Bern: 188 pp. retrieved from: www.bafu.admin.ch/publikationen/publikation/01031/index.html?lang=en.
- Frischknecht, R., Büsser Knöpfel, S. (2013). Swiss Eco-Factors 2013 according to the Ecological Scarcity Method, - methodological fundamentals and their application in Switzerland. Environmental studies no. 1330. Federal Office for the Environment, Bern: 254 pp.
- Frischknecht, R., Itten, R., Büsser Knöpfel, S. (2013). Tracking important Environmental Impacts Related to Domestic Consumption - A Feasibility Study on Environmental Life Cycle Indicators for Land Use/Biodiversity, Air Pollution, Nitrogen, Water Use, and the Use of Materials. commissioned by Federal Office for the Environment (FOEN). Treeze Ltd., fair life cycle thinking.
- Gassert, F., M. Luck, M. Landis, P. Reig, and T. Shiao. (2013). Aqueduct Global Maps 2.0. Working Paper. Washington, DC: World Resources Institute. Available online at http://wri.org/publication/aqueduct-global-maps-20.
- Gerten, D., Heinke, J., Hoff, H., Biemans, H., Fader, M., and Waha, K. (2011). Global water availability and requirements for future food production, J. Hydrometeorol., 12, 885–899, doi:10.1175/2011jhm1328.1
- Gerten, D., Hoff, H., Rockström, J., Jägermeyr, J., Kummu, M., & Pastor, A. V. (2013). Towards a revised planetary boundary for consumptive freshwater use: role of environmental flow requirements. Current Opinion in Environmental Sustainability, 5(6), 551-558.
- Gleeson, T., Wada, Y., Bierkens, M. F. P., van Beek, L. P. H. (2012). Water balance of global aquifers revealed by groundwater footprint. Nature 488:197-200
- Gleick, P.H., (1996). Basic Water Requirements for Human Activities: Meeting Basic Needs. Water International (IWRA), 21, pp.83–92.
- Goedkoop, M.; Spriensma, R. (2001). The Ecoindicator 99: a Damage Oriented Method for Life Cycle Impact Assessment: Methodology Report, Publikatiereeks produktenbeleid; nr. 36A; Ministerie van Volkshiusvesting, Ruimtelijke Ordening en Milieubeheer: Den Haag.
- Graedel T.E., Barr, R., Chandler C., Chase, T., Choi J., Chirstoffersen L., Friedlander E., Henly C., Jun C., Nassar N.T., et al. (2012). Methodology of metal criticality determination. Environ. Sci. Technol. 46 (2) 1063-1070
- Guinée, J. B., & Heijungs, R. (1995). A proposal for the definition of resource equivalency factors for use in product life-cycle assessment. Environmental toxicology and chemistry, 14(5), 917-925.
- Guinée, J. B., Gorée., M., Heijungs, R., Huppes, G., Kleijn, R., de Koning, A., van Oers, L., Wegener Sleeswijk, A., Suh, S., Udo de Haes, H. A., de Bruijn, J. A., van Duin, R., Huijbregts, M. A. J. (2002). Handbook on life cycle assessment. Operational guide to the ISO standards. Series: eco-efficiency in industry and science. Kluwer, Dordrecht, The Netherlands

- Hanafiah, M. M., Xenopoulos, M. A., Pfister, S., Leuven, R. S., & Huijbregts, M. A. (2011). Characterization factors for water consumption and greenhouse gas emissions based on freshwater fish species extinction. Environmental science & technology, 45(12), 5272-5278.
- Hanasaki, N., Inuzuka, T., Kanae, S., & Oki, T. (2010). An estimation of global virtual water flow and sources of water withdrawal for major crops and livestock products using a global hydrological model. Journal of Hydrology, 384(3), 232-244.
- Hejazi, M. I., Edmonds, J., Clarke, L., Kyle, P., Davies, E., Chaturvedi, V., ... & Calvin, K. (2014). Integrated assessment of global water scarcity over the 21st century under multiple climate change mitigation policies. Hydrology and Earth System Sciences, 18(8), 2859-2883.
- Hellweg, S., & i Canals, L. M. (2014). Emerging approaches, challenges and opportunities in life cycle assessment. Science, 344(6188), 1109-1113.
- Hoekstra, A. Y., Gerbens-Leenes, W., & van der Meer, T. H. (2009). Reply to Pfister and Hellweg: Water footprint accounting, impact assessment, and life-cycle assessment. Proceedings of the National Academy of Sciences, 106(40), E114-E114.
- Hoekstra, A.Y., Chapagain, A.K., Aldaya, M.M., Mekonnen, M.M. (2011). The water footprint assessment manual: setting the global standard. Water Footprint Network. Enschede, The Netherlands
- Hoekstra, A. Y., & Mekonnen, M. M. (2012). Reply to Ridoutt and Huang: From water footprint assessment to policy. Proceedings of the National Academy of Sciences, 109(22), E1425-E1425.
- Hoekstra, A. Y., Mekonnen, M. M., Chapagain, A. K., Mathews, R. E., & Richter, B. D. (2012). Global monthly water scarcity: blue water footprints versus blue water availability. PLoS One, 7(2), e32688.
- Hoekstra, A. Y. (2015). The sustainability of a single activity, production process or product. Ecological Indicators, 57, 82-84.
- Hoekstra, A. Y. (2016). A critique on the water-scarcity weighted water footprint in LCA. Ecological Indicators, 66, 564-573.
- ISO (2014). ISO 14046 water footprint – principles, requirements and guidelines. International Organization for Standardization, Geneva
- Jolliet, O., Müller-Wenk, R., Bare, J., Brent, A., Goedkoop, M., Heijungs, R., ... & Rebitzer, G. (2004). The LCIA midpoint-damage framework of the UNEP/SETAC life cycle initiative. The International Journal of Life Cycle Assessment, 9(6), 394-404.
- Kounina, A., Margni, M., Bayart, J. B., Boulay, A. M., Berger, M., Bulle, C., ... & Núñez, M. (2013). Review of methods addressing freshwater use in life cycle inventory and impact assessment. The International Journal of Life Cycle Assessment, 18(3), 707-721.
- Loubet, P., Roux, P., Nunez, M., Belaud, G., Bellon-Maurel, V. (2013). Assessing Water Deprivation at the Sub-river Basin Scale in LCA Integrating Downstream Cascade Effects. Environ. Sci. Technol. 47:14242−14249
- Mekonnen, M.M., Hoekstra, A.Y. (2011). National water footprint accounts: the green, blue and grey water footprint of production and consumption. Value of Water Research Report Series No. 50, UNESCO-IHE, Delft, The Netherlands. Available: www.waterfootprint.org/Reports/Report50-NationalWaterFootprints-Vol1.pdf

- Milà i Canals, L., Chenoweth, J., Chapagain, A., Orr, S., Antón, A., Cliftm R. (2009). Assessing freshwater use impacts in LCA: part I—inventory modelling and characterisation factors for the main impact pathways. Int J Life Cycle Assess 14(1):28–42
- Motoshita, M., Itsubo, N., & Inaba, A. (2010, November). Damage assessment of water scarcity for agricultural use. In Proceedings of 9th international conference on EcoBalance. D1-1410. National Institute of Advanced Industrial Science and Technology (AIST).
- Motoshita, M., Itsubo, N., & Inaba, A. (2011). Development of impact factors on damage to health by infectious diseases caused by domestic water scarcity. The International Journal of Life Cycle Assessment, 16(1), 65-73.
- Motoshita, M., Ono, Y., Pfister, S., Boulay, A. M., Berger, M., Nansai, K., ... & Inaba, A. (2014). Consistent characterisation factors at midpoint and endpoint relevant to agricultural water scarcity arising from freshwater consumption. The International Journal of Life Cycle Assessment, 1-12.
- Murray, S. J., Foster, P. N., & Prentice, I. C. (2012). Future global water resources with respect to climate change and water withdrawals as estimated by a dynamic global vegetation model. Journal of Hydrology, 448, 14-29.
- Müller Schmied, H., Eisner, S., Franz, D., Wattenbach, M., Portmann, F. T., Flörke, M., & Döll, P. (2014). Sensitivity of simulated global-scale freshwater fluxes and storages to input data, hydrological model structure, human water use and calibration. Hydrology and Earth System Sciences, 18(9), 3511-3538.
- Nilsson, C., Reidy, C. A., Dynesius, M., & Revenga, C. (2005). Fragmentation and flow regulation of the world's large river systems. Science, 308(5720), 405-408.
- Núñez, M., Pfister, S., Antón, A., Muñoz, P., Hellweg, S., Koehler, A., Rieradevall, J. (2013). Assessing the Environmental Impact of Water Consumption by Energy Crops Grown in Spain. Journal of Industrial Ecology 17(1):90-102.
- Núñez, M., Bouchard, C.R., Bulle, C., Boulay, A.M., Margni, M. (2016). Critical analysis of life cycle impact assessment methods addressing consequences of freshwater use on ecosystems and recommendations for future method development. The International Journal of Life Cycle Assessment, in press. DOI 10.1007/s11367-016-1127-4.
- Odum, H. T. (1996). Environmental accounting: EMERGY and environmental decision-making.
- OECD. (2003). Environmental Performance Reviews: Water; Performance and Challenges in OECD Countries. Organisation for Economic Cooperation and Development, OECD, Paris.
- Ohlsson, L., (2000). Water conflicts and social resource scarcity. Physics and Chemistry of the Earth, Part B: Hydrology, Oceans and Atmosphere, 25(3), pp.213–220.
- Owens, J. W. (2002). Water resources in life-cycle impact assessment. Considerations in choosing category indicators. J Ind Ecol 5:37–54
- Pfister, S., Koehler, A., & Hellweg, S. (2009). Assessing the environmental impacts of freshwater consumption in LCA. Environmental science & technology, 43(11), 4098-4104.

- Pfister, S., Bayer, P., Koehler, A., & Hellweg, S. (2011). Projected water consumption in future global agriculture: scenarios and related impacts. Science of the Total Environment, 409(20), 4206-4216.
- Pfister, S., & Bayer, P. (2014). Monthly water stress: spatially and temporally explicit consumptive water footprint of global crop production. Journal of Cleaner Production, 73, 52-62.
- Pfister, S., Vionnet, S., Levova, T., & Humbert, S. (2015). Ecoinvent 3: assessing water use in LCA and facilitating water footprinting. The International Journal of Life Cycle Assessment, 1-12.
- Pfister, S., Boulay A.M., Berger, M., Hadjikakou, M., Motoshita, M., Hess, T., Ridoutt, B., Weinzettel, J., Scherer, L., Doll, P., Manzardo, A., Núñez, M., Verones, F., Humbert, S., Buxmann, K., Harding, K., Benini, L., Oki, T., Finkbeiner, M., Henderson, A. (2017). Understanding the LCA and ISO water footprint: A response to Hoekstra (2016) A critique on the water-scarcity weighted water footprint in LCA. Ecological Indicators, 72, 352-359.
- Pfister, S., & Hellweg, S. (2009). The water "shoesize" vs. footprint of bioenergy. Proceedings of the National Academy of Sciences, 106(35), E93-E94.
- Quintero, P., Dias, A.C., Silva, M., Ridoutt, B.G., Arroja, L. (2015). A contribution to the environmental impact assessment of green water flows. Journal of Cleaner Production 93, 318-329.
- Raskin P. Gleick, P., Kirshen, P., Pontius, G., Strzepek, K. (1997). Water Futures: Assessment of Long-Range Patterns and Problems. Stockholm Environment Institute. ISBN: 91 88714454
- Reig, P., Shiao, T., Gassert, F. (2013). Aqueduct Water Risk Framework. Working Paper. Washington, DC: World Resources Institute. Available online at http://www.wri.org/publication/aqueduct-waterrisk-framework.
- Richter BD, Davis MM, Apse C, Konrad C (2011). A presumptive standard for environmental flow protection. River Res Appl;doi:10.1002/rra.1511.
- Richts, A., Struckmeier, W., Zaepke, M. (2011). WHYMAP and the groundwater resources of the world 1:25,000,000. In Sustaining Groundwater Resources; Jones, J. A. A., Ed.; Springer: Heidelberg; pp 159−173.
- Ridoutt, B.G. & Pfister, S. (2010). A revised approach to water footprinting to make transparent the impacts of consumption and production on global freshwater scarcity. Global Environmental Change, 20(1), pp.113–120.
- Ridoutt, B. G., & Huang, J. (2012). Environmental relevance—the key to understanding water footprints. Proceedings of the National Academy of Sciences, 109(22), E1424-E1424.
- Rockström, J., Steffen, W., Noone, K., Persson, Å., Chapin, F. S., Lambin, E. F., ... & Nykvist, B. (2009). A safe operating space for humanity. Nature, 461(7263), 472-475.
- Rost, S., Gerten, D., Bondeau, A., Lucht, W., Rohwer, J., & Schaphoff, S. (2008). Agricultural green and blue water consumption and its influence on the global water system. Water Resources Research, 44(9).
- Rugani, B., Huijbregts, M. A., Mutel, C., Bastianoni, S., & Hellweg, S. (2011). Solar energy demand (SED) of commodity life cycles. Environmental science & technology, 45(12), 5426-5433.

- Müller Schmied, H., Eisner, S., Franz, D., Wattenbach, M., Portmann, F. T., Flörke, M., & Döll, P. (2014). Sensitivity of simulated global-scale freshwater fluxes and storages to input data, hydrological model structure, human water use and calibration. Hydrology and Earth System Sciences, 18(9), 3511-3538.
- Schwegler, R., Iten, R., Grünig, M., Boteler, B., Känzig, J., Hauser, A. (2011). Quality Requirements for Environmental Information – Development, Definition and Application of Quality Requirements for Reporting on the Environmental Impacts of Consumption and Production. Federal Office for the Environment, Bern. Environmental studies no. 1119: 13 pp.
- Schyns, J. F., Hoekstra, A. Y., Booij, M. J. (2015). Review and classification of indicators of green water availability and scarcity. Hydrol. Earth Syst. Sci. Discuss., 12:5519–5564.
- Seckler, D., Amarasinghe, U., Molden, D., de Silva, R. Barker, R. (1998). World water demand and supply, 1990 to 2025: Scenarios and issues. , p.50.
- Smakhtin V. U. (2001). Low flow hydrology: a review J. Hydrol. 240 147–86
- Smakhtin, V., Revenga, C., Döll, P. (2004). A pilot global assessment of environmental water requirements and scarcity. Water International, 29(3), 307-317.
- Sonderegger T., Pfister S., Hellweg S. (2015). Criticality of water: aligning water and mineral resources assessment Environ. Sci. Technol. 49 12315-12323
- Stewart, M., Weidema, B. (2005). A consistent framework for assessing the impacts from resource use - A focus on resource functionality. Int. J. Life Cycle Assess. 10 (4), 240–247.
- Sullivan, C.A. (2002). Calculating a Water Poverty Index. World Development, 30(7), pp.1195–1211.
- Sullivan, C.A. (2010). Quantifying water vulnerability: a multi-dimensional approach. Stoch Environ Res Risk Assess 25:627–640
- Szargut, J., Morris, D. R., Steward, F. R. (1988). Exergy Analysis of Thermal, Chemical and Metallurgical Processes; Hemisphere/ Springer: Berlin.
- UNEP. (2012). Measuring water use in a green economy, A Report of the Working Group on Water Efficiency to the International Resource Panel. McGlade, J., Werner, B., Young, M., Matlock, M., Jefferies, D., Sonnemann, G., Aldaya, M., Pfister, S., Berger, M., Farell, C., Hyde, K., Wackernagel, M., Hoekstra, A., Mathews, R., Liu, J., Ercin, E., Weber, J.L., Alfieri, A., Martinez-Lagunes, R., Edens, B., Schulte, P., von Wirén-Lehr, S., Gee, D.
- United Nations. (2015). Transforming our World: the 2030 agenda for sustainable development: A/RES/70/1.
- Unep (2016) Global guidance for life cycle impact assessment indicators. Volume 1. ISBN: 978-92-807-3630-4. Available at: http://www.lifecycleinitiative.org/life-cycle-impact-assessment-indicators-and-characterization-factors/
- Van Beek, L. P. H., Wada, Y., Bierkens, M. F. (2011). Global monthly water stress: 1. Water balance and water availability. Water Resources Research, 47(7).
- Van Oers, L., de Koning, A., Guinee, J., Huppes, G. (2002). Abiotic Resource Depletion in LCA, Amsterdam.
- Van Zelm, R., Schipper, A. M., Rombouts, M., Snepvangers, J., & Huijbregts, M. A. (2010). Implementing groundwater extraction in life cycle impact assessment:

Characterization factors based on plant species richness for the Netherlands. Environmental science & technology, 45(2), 629-635.

- Veldkamp, T. I. E., Wada, Y., Aerts, J. C. J. H., & Ward, P. J. (2016). Towards a global water scarcity risk assessment framework: incorporation of probability distributions and hydro-climatic variability. Environmental Research Letters, 11(2), 024006.
- Verones, F., Saner, D., Pfister S., Baisero, D., Rondinini, C., Hellweg, S. (2013). Effects of Consumptive Water Use on Biodiversity in Wetlands of International Importance. Environ. Sci. Technol.47:12248-12257
- Vörösmarty, C. J., McIntyre, P. B., Gessner, M. O., Dudgeon, D., Prusevich, A., Green, P., ... & Davies, P. M. (2010). Global threats to human water security and river biodiversity. Nature, 467(7315), 555-561.
- Wada, Y., van Beek, L. P., van Kempen, C. M., Reckman, J. W., Vasak, S., & Bierkens, M. F. (2010). Global depletion of groundwater resources. Geophysical research letters, 37(20).
- Wada, Y., Van Beek, L. P., Wanders, N., & Bierkens, M. F. (2013). Human water consumption intensifies hydrological drought worldwide. Environmental Research Letters, 8(3), 034036.
- Wada, Y., & Bierkens, M. F. (2014). Sustainability of global water use: past reconstruction and future projections. Environmental Research Letters, 9(10), 104003.
- Wada, Y., Wisser, D., & Bierkens, M. F. P. (2014). Global modeling of withdrawal, allocation and consumptive use of surface water and groundwater resources. Earth System Dynamics, 5(1), 15.
- Yano S., Hanasaki N., Itsubo N. Oki T. (2015). Water Scarcity Footprints by Considering the Differences in Water Sources. Sustainability 7:9753-9772; doi:10.3390/su7089753

6 Impact due to Particulate matter

6.1 Introduction

Several human activities such as those related to combustion of fossil fuels and biomass (either for heating, transport and industrial process) are leading to an increase emission of particulate matter. As part of the World Health Organization (WHO) Global Burden of Disease Comparative Risk Assessment, particulate air pollution is consistently and independently related to the most serious health effects, including lung cancer and other cardiopulmonary mortality (Cohen et al 2005 and GBD, 2017).

The default impact category for PM and related models and characterization factors for the Environmental Footprint (EC 2013) is based on the recommendations of the International Reference Life Cycle Data System (ILCD) Handbook (EC-JRC, 2011) and their preceding analyses (EC-JRC 2010a, b).

The current model recommended in ILCD (EC-JRC, 2011) has been identified as in need of updates, hence, the results of the UNEP-SETAC activities are taken as reference for the present evaluation, because they can be considered the state-of-the art in the field of particulate matter impact assessment in LCA.

Respiratory inorganics' impacts expressed as health effects from PM2.5 exposure were selected as one of the initial impact categories to undergo review with the goal of providing global guidance for implementation in life cycle impact assessment (LCIA) by the UNEP/SETAC Life Cycle Initiative.

Within the UNEP/SETAC task force on human health impacts, an initial Guidance Workshop was organized in Basel, Switzerland, in August 2013. Based on a literature review and expert input, the workshop organizers reached out to a broad range of internationally recognized experts in PM exposure and health effects.

The specific objectives of the workshop were to first identify and discuss the main scientific questions and challenges for quantifying human health effects from PM2.5 exposure and then to provide initial guidance to the impact quantification process. Three main topics were addressed at the workshop:

a. the general assessment framework as proposed by Humbert et al. (2011),

b. approaches and data to determine human exposure to PM2.5 expressed as intake fractions, and

c. approaches and data to determine exposure-response functions (ERFs) for PM2.5 along with disease severity.

For these topics, the workshop participants discussed a set of key questions. These questions are reported in Fantke et al (2015), where a deeper discussion on the topics is also summarized.

The main needs emerged from the literature review and the discussion within the Basel workshop, as improvements of the models available at the time of previous EC-JRC recommendation (EC-JRC, 2011), were:

- to consider also secondary PM in addition to primary PM
- to develop archetypes able to model different conditions: outdoor and indoor emission, urban/rural areas, ground level/stack emission and low/high ventilation rate of buildings (for indoor)

- to identify the most suitable model for characterizing the intake and the effects on human health in different conditions (based on archetypes).

6.2 Framework and scope of the evaluation

The previous ILCD recommendation about the impact category Particulate matter/Respiratory inorganics considered several models to derive characterization factors at midpoint and endpoint (EC-JRC, 2011). As explained in EC-JRC (2012), the CFs for fate and intake (referred as midpoint level) and effect and severity (referred as endpoint level) are the result of the combination of different models, reported in Humbert (2009).

The recommended models in EC-JRC 2011 have been used for calculating CFs but they were complemented as in Humbert 2009, where a consistent explanation on the combination of different models for calculating CFs is provided. For fate and intake, the CFs were based on RiskPoll (Rabl and Spadaro, 2004), Greco et al. (2007), USEtox (Rosenbaum et al. 2008), Van Zelm et al. (2008). Effect and severity factors are calculated starting from the work of van Zelm et al. (2008) that provides a clear framework, but using the most recent version of Pope et al. (2002) for chronic long-term mortality and including effects from chronic bronchitis as identified significant by Hofstetter (1998) and Humbert (2009).

At the time of the recommendation, the model recommended was not published in a scientific journal. This has been done later, with some improvements, in Humbert et al. (2011). Therefore, Humbert (2009) was taken as the main reference source for developing the CFs.

As mentioned before, the work done by the PM task force within the UNEP-SETAC LC initiative is taken as reference because it builds on the state-of-the-art in the field of LCIA for impacts generated by particulate matter emissions. Therefore, some of the preliminary steps undertaken for the evaluation of models in the other impact categories under revision (resources, land use and water) were not included in the evaluation done for the impact category particulate matter. These steps are supposed to be already covered by the work of the task force and the results of the Basel workshop mentioned before (Fantke et al., 2015), which are taken as starting point for the evaluation.

Indeed, the whole procedure followed by the UNPE-SETAC LC initiative in the task force for PM is in line with the procedure for recommendation within the ILCD framework. It includes (among others) the following steps:

1) Development of/customization of set of criteria of good practice in assessment approaches and modeling
2) Inventory analysis of available assessment approaches and models
3) Pre-selection of assessment approaches/models based on qualitative evaluation
4) Quantitative models and factors comparison (limited to a real example defined in the first stages)
5) Identification of recommended assessment approaches and models
6) Determination of recommended factors for each archetype worldwide
7) Preparation of report with recommendations for 2015 Pellston Workshop

Moreover, the approach adopted builds upon the general framework proposed by Humbert et al. (2011), which is an update of the document taken as reference for the previous ILCD recommendation on PM (Humbert, 2009).

Therefore, the only model considered for the evaluation in view of a possible recommendation in the ILCD is the one developed in the UNEP-SETAC process for consensus building and related recommendation on PM life cycle impact assessment (Fantke et al, 2016).

6.3 Environmental mechanism (cause-effect chain)

The cause-effect chain taken as reference for the evaluation of LCIA models about particulate matter is reported in Figure 6.1. The figure depicts the cause-effect chain from the human intervention (which define the border between the ecosphere and the technosphere) to the final effect on the Areas of Protection (AoPs).

Figure 6.2 provides more details on the most relevant aspects for each step of the chain and the LCIA metrics used to calculate the midpoint and endpoint indicators.

As explained in Fantke et al. (2015), the impact pathway presented by Humbert et al. (2011) starts from emissions of primary $PM_{2.5}$ and secondary $PM_{2.5}$ precursors into the environment (mass emitted), and multiplies these emissions with:

- intake fractions, iF (mass of $PM_{2.5}$ inhaled by the affected population per mass of primary $PM_{2.5}$ or secondary $PM_{2.5}$ precursor emitted, respectively),
- an exposure-response factor derived from epidemiological studies linking health effects in the affected population to ambient $PM_{2.5}$ concentrations, ERF (disease rate per unit mass concentration), and
- a severity factor, SF (disability-adjusted life years (DALY) per disease case), to calculate a human health-related impact score, expressed in DALY.

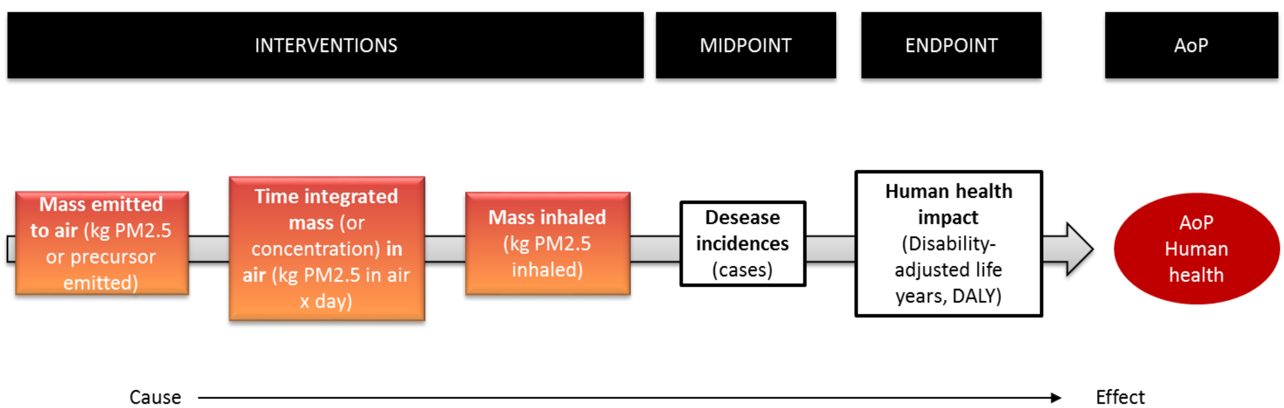


Figure 6.1 Impact pathway (cause-effect chain) for particulate matter (modified from Fantke et al., 2015).

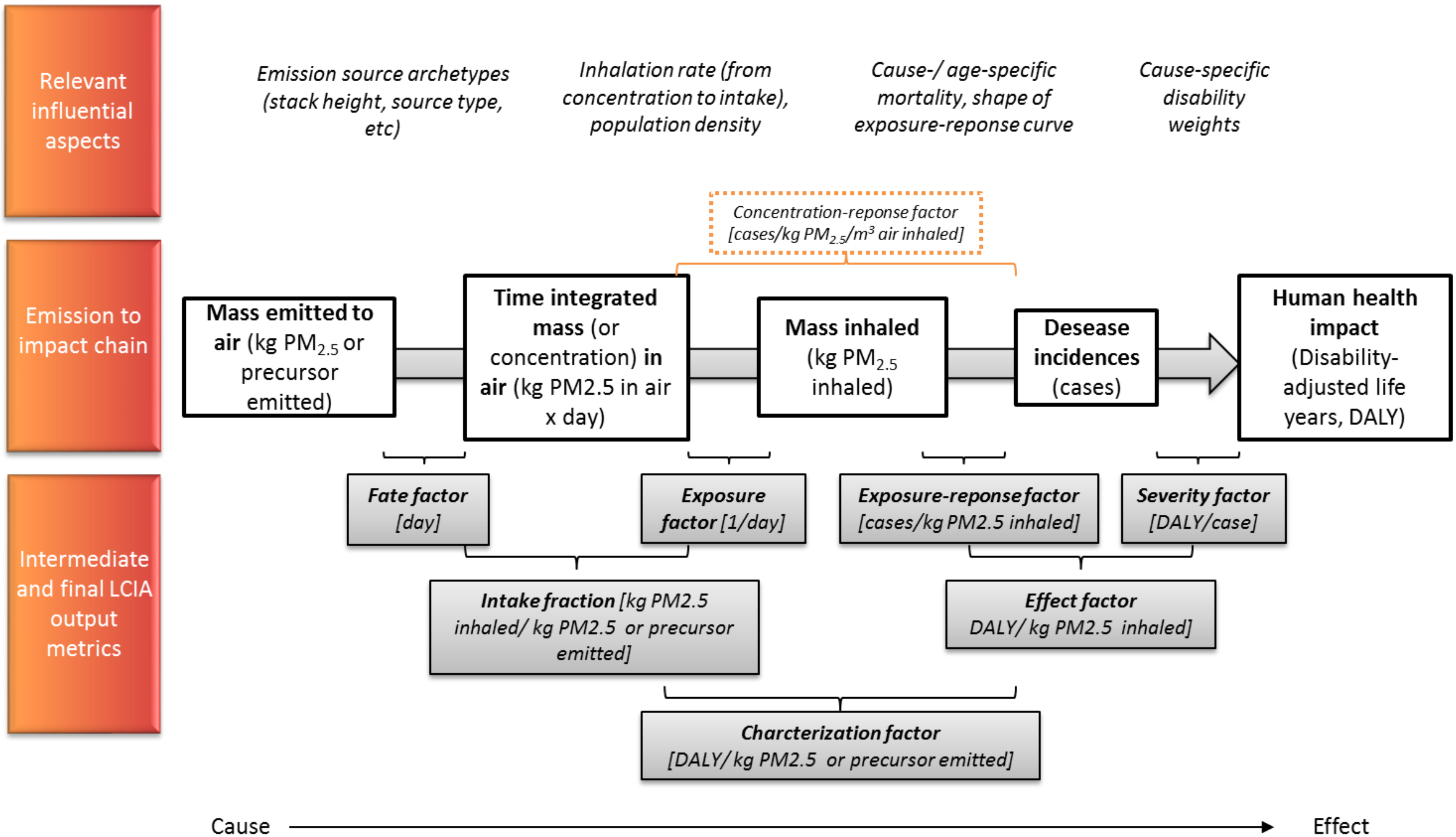


Figure 6.2 Detail of relevant influential aspects and LCIA metrics with reference to the impact pathway depicted in figure 6.1 (modified from Fantke et al., 2015).

6.4 Criteria for the evaluation of this impact category

In line with what was done for the other impact categories under revision in the EF context (resources, water and land use), in addition to the general criteria defined for the evaluation of all the impact categories, some other criteria have been selected to take into account specific features of the impact category particulate matter. These additional criteria are described below.

Environmental relevance.

Inclusion of PM precursors. The criterion is aimed at checking if the model includes the contribution of PM precursors, and to what extent the precursors included in the ILCD elementary flow list are covered by the CFs.

Differentiation between urban and rural areas and other archetypes. The purpose of this criterion is to evaluate the suitability of the model to differentiate the impact of emissions in different conditions (e.g. urban or rural areas, ground level or stack height, etc). The current ILCD recommendation includes 4 archetypes; therefore, the highest score is given to models that include 5 archetypes of more (i.e. improvements over the current recommendation).

Characterisation factors.

Availability of a CF for the elementary flow "PM10". Some of the existing models do not provide a CF for PM_{10}, because the $PM_{2.5}$ fraction is considered the main responsible of impacts on human health. However, some inventories report only PM_{10} and not $PM_{2.5}$. Hence, an assumption of the impact coming from emissions of PM_{10} (i.e. a related CF) helps to avoid disregarding some of the emissions included in the inventory.

Annex 6.1 (separate file) reports all the criteria used for the evaluation of models in the impact category particulate matter.

6.5 Preselection of models for further evaluation

Following the reasoning presented in section 6.2, the model recommended by the UNEP-SETAC initiative (Fantke et al., 2016) is the only one pre-selected for evaluation.

6.5.1 Description of the UNEP-SETAC model

The model developed by the UNEP-SETAC Task Force (TF) on PM aims at assessing damage to human health from outdoor and indoor emissions of primary and secondary $PM_{2.5}$ in urban and rural areas. The model follows the impact pathway described in section 6.3.

The framework adopted for the model involves three stages: i) analyzing $PM_{2.5}$ fate and exposure (including indoor and outdoor urban/rural environments), ii) modeling exposure-response, and iii) the integration of exposure-response and $PM_{2.5}$ exposure reflecting population and location characteristics. The exposure model is organized as a mass balance matrix that tracks the global fate of primary $PM_{2.5}$ and secondary $PM_{2.5}$ precursor emissions (both indoors and outdoors) as an embedded system of compartments including urban environments, rural environments, and indoor environments within urban and rural areas. In order to account for all the factors that contribute globally to the variation of iF values for ambient emissions, a set of archetypes was developed, taking into account source characteristics, population density relative to source location, and meteorological conditions (Fantke et al., 2016).

The main sources of data and background models to calculate the CFs are the following:

- Apte et al. (2012): data on iF for outdoor urban environment
- Brauer et al. (2016): data on iF for outdoor rural environment
- Hodas et al. (2015): iF for indoor environments

- Burnett et al. (2014): risk function for calculating the ERF slope (with data from Apte et al. 2015).

The model is built to calculate an endpoint indicator, damage to human health, expressed in DALY/kgPM2.$5_{emitted.}$

The related midpoint indicator is the change in mortality due to PM, expressed in disease incidents/kgPM2.$5_{emitted}$.

Two sets of CFs are provided. The first one ("Marginal") is calculated using the marginal slope at the background concentration working point on the ERF for total mortality due to PM exposure. The second one ("Average") is derived considering the average between the background concentration working point on the ERF and the theoretical minimum-risk level of 5.8 µg/m^3 for total mortality due to PM exposure.

As stated by the authors, "the marginal approach ideally takes the current situation as the working point […] an is most appropriately when informing decision that affect short-term and restricted changes in overall emissions, while the average approach may be relevant when larger and longer term changes are expected[…]" (Fantke et al., 2016). Therefore, the average approach (and related set of CFs) is taken into account for the present evaluation.

6.5.2 Characterization factors at midpoint

The list of characterization factors provided by the model for the average ERF were collected as they are published by model developers and then mapped to the ILCD elementary flow list. Name correspondence and the similarity in the description of the archetype represented by the flow were the main criteria used. For the flows of unspecified emissions, a precautionary approach was applied, by assigning the highest CF among those available for that kind of particle. The model assessed does not provide a CF for the elementary flow "PM10", because the PM 2.5 fraction is considered the main responsible of impacts on human health. However, as explained in section 6.4, some inventories include only PM10 and not PM2.5. Hence, an assumption of the impact coming from emissions of PM10 (i.e. a related CF) is made, to avoid disregarding some of the emissions included in the inventory. In line with what was done for the previous recommendation, the CF for PM10 is calculated by multiplying the CF for $PM_{2.5}$ by 23% (i.e. by the fraction of $PM_{2.5}$ over the total amount of PM10). The elementary flows "Particles (PM0.2)" and "Particles (PM 0.2-2.5)" were not included in the original model. However, they could be part of the inventories currently used. Therefore, to avoid disregarding the emission of very small particles, the CF for PM2.5 is assigned as a proxy to these flows (and related sub-compartments).

The mapped CFs were than compared to the current ILCD recommendation in terms of flow coverage and range of values. The environmental relevance and scientific robustness of CFs is not discussed in this section and have been assessed in the evaluation of the model (a summary of results is reported in section 6.8).

The model by UNEP-SETAC provides 22 CFs at midpoint (including indoor emissions), whereas the current ILCD recommendation includes 43 CFs. However, in the current ILCD recommendation some assumptions were made to map the original list of CFs (Humbert 2009) to the ILCD elementary flows and the same CFs is assigned to more than one flow, in case they are considered equivalent with respect to the model approach (e.g. for groups of substances, like nitrogen oxides, the same CF of nitrogen dioxide is applied). If the same reasoning is applied to the UNEP-SETAC set of CFs, the number of flows covered by the new model is 71. The main difference between the two is the lack of CFs for CO as precursor of PM in the UNEP-SETAC recommendation.

The range of CFs' values is larger for the model by UNEP-SETAC than for the current ILCD recommendation (6 orders of magnitude instead of 4).

The UNEP-SETAC model includes CFs also for characterizing indoor emissions, which are currently not taken into account in ILCD.

6.6 Model evaluation

		UNEP-SETAC (Fantke et al., 2016) Average	
Science-based criteria	**Summary information (descriptive)**	The model developed by the UNEP-SETAC Task Force (TF) on PM aims at assessing damage to human health from outdoor and indoor emissions of primary and secondary PM2.5 in urban and rural areas. The exposure model is organized as a mass balance matrix that tracks the global fate of primary PM2.5 and secondary PM2.5 precursor emissions as an embedded system of compartments including urban environments, rural environments, and indoor environments within urban and rural areas.	
	Completeness of the scope	B	Good completeness of scope
	Environmental relevance	C	The environmental relevance of the model is quite high, but the underestimation of the impact due to the impossibility to use both outdoor and indoor factors is a drawback
	Scientific robustness & Uncertainty	B	The model reflects the state-of-the art and derives from a consensus building exercise of a group of experts. Uncertainty and distribution not provided at the moment.
	Documentation, Transparency & Reproducibility	A-B	The documentation is complete and transparent. The model is not accessible in an oprational format.
	Applicability	C	Quite good level of applicability. Lack of indoor emissions in the background datasets is an issue
	Characterization factors	B	Good relevance of the CFs, but still not tested in real conditions (only fictive case studies)
Overall evaluation of science-based criteria		B	The model reflects the state-of-the-art and has a quite good level of applicability. The underestimation of the impact due to the impossibility to use both outdoor and indoor factors is a drawback
Overall evaluation of stakeholders acceptance		B	Model coming from an international consensus building exercise involving a group of experts
Final evaluation		B	The model reflects the state-of-the art and derives from an international consensus building of a group of experts. Limit for applicability is the lack of data on indoor emissions in the existing background datasets.

The extended version of this table, with the detailed scores for each model is reported in Annex 6.1

6.7 Discussion on model evaluation

As discussed before, the model is considered as an improvement of the previous ILCD recommendation in terms of advancement in the implementation of state-of-the art knowledge in the field of health impacts due to PM emissions. Moreover, the model was developed in the context of an international consensus building exercise, involving some experts in the field of LCIA for PM, and has already been recommended by UNEP-SETAC.

If the original characterization factors are mapped to the ILCD elementary flows, following the same criteria applied for the previous ILCD recommendation, the coverage of substances is quite good and the range of factors is higher than the previous one. The CFs have been tested in fictive case studies, and still not applied in real cases. However, the applicability looks to be easy, due to conformity with other existing models and the structure of the elementary flows in existing inventories and in the ILCD flow list.

A limit of the model is the lack of data about indoor emissions in current practice (especially for background datasets) and in the ILCD elementary flow list. Limiting the use of CFs to the outdoor compartment only can lead to an underestimation of the impacts (because in the original model the fraction of outdoor emission that goes into the indoor environment is accounted for only in the CFs for indoor). However, there is room for improvement in the future, first of all by introducing indoor emission flows in the ILCD elementary flow list, and the recommendation of a model that can be applied also to indoor emissions can push the collection of data for indoor emissions in the inventories.

Therefore, the model assessed is considered suitable to be recommended.

6.8 Recommended default model for midpoint

The recommended default model for midpoint assessment in the impact category particulate matter is the model developed by UNEP-SETAC and documented in Fantke et al. (2016).

6.9 Model for endpoint

The recommended default model for endpoint is the same as for midpoint, i.e. the UNEP-SETAC model developed by UNEP-SETAC and documented in Fantke et al. (2016). Endpoint indicator is damage to human health, expressed in DALY/kgPM2.5$_{emitted}$.

6.10 Consistency between midpoint and endpoint models

As the recommended CFs at the midpoint level and the interim model at the endpoint level are derived from the same model, there is a high level of consistency between the two levels.

6.11 Classification of the recommended default models

The model is recommended as level I.

6.12 Recommended characterization factors

Characterisation factors are available to be downloaded at the EPLCA website at http://eplca.jrc.ec.europa.eu/LCDN/developerEF.xhtml

6.13 Normalisation factors

Source and data used to calculate the normalisation factors are available in Crenna et al. 2019. The EF normalisation factors to be used are available at http://eplca.jrc.ec.europa.eu/LCDN/developerEF.xhtml.

6.14 Research needs

The research needs have been identified by the expert group that developed the preliminary version of the model. In Fantke et al. (2016), they define a roadmap for further improvement and for completing the model and the set of CFs based on spatially explicit models. The roadmap is reported below, as presented by the authors themselves.

"A roadmap has been established for updating secondary $PM_{2.5}$ characterization factors, based on spatially explicit models. This includes the following steps: a) Perform a systematic sensitivity study over the entire US to analyse the spatial variation of the formation rate of secondary $PM_{2.5}$ and intake fractions using the Intervention Model for Air Pollution, InMAP (Tessum et al., 2015), and compare it to outputs of the Community Multiscale Air Quality (CMAQ) model with decoupled direct methods (DDM), isolating the contribution of individual precursors (Buonocore et al., 2014). b) Identify archetypes for secondary $PM_{2.5}$ as a function of population density and main limiting substance in the considered region (NH_3, SO_2 and organic carbon). c) Extend the analysis to world level. Provide characterization factors for emissions of secondary $PM_{2.5}$ precursors based on both marginal and average responses, using a tiered approach corresponding to different levels of spatialization.

The process for assessing secondary $PM_{2.5}$ formation, both outdoors and indoors, requires continuing monitoring of the $PM_{2.5}$ health effects literature to assure an adequate set of *case studies globally for evaluating the reliability and representativeness of secondary $PM_{2.5}$ CFs.*

There remains a need in this effort to assess uncertainty by reviewing the emissions to impact factors that have significant data gaps and/or lack mechanistic understanding. This effort will be supported by a sensitivity analysis that flags parameters that have a strong influence on model the CF analysis outcome". (Fantke et al., 2016 p.94)

While not an improvement potential for the LCIA model, it has to be noted that, for the time being, most of the available Life Cycle Inventory datasets do not include information about indoor emissions, so this improvement on the LCIA side has only limited immediate applicability in LCAs using secondary datasets. However, once this information becomes available in mainstream life cycle inventory databases, the ILCD flow list should be updated to include new flows for indoor emissions.

6.15 References the chapter on particulate matter

- Apte JS, Bombrun E, Marshall JD, Nazaroff WW. (2012). Global intraurban intake fractions for primary air pollutants from vehicles and other distributed sources. Environmental Science and Technology 46:3415-3423.
- Apte JS, Marshall JD, Cohen AJ, Brauer M. (2015). Addressing global mortality from ambient PM2.5. Environmental Science and Technology 49:8057-8066.
- Brauer M, Hirtle R, Lang B, Ott W. (2000). Assessment of indoor fine aerosol contributions from environmental tobacco smoke and cooking with a portable nephelometer. Journal of Exposure Analysis and Environmental Epidemiology 10:136-144.
- Buonocore JJ, Dong X, Spengler JD, Fu JS, Levy JI. (2014). Using the Community Multiscale Air Quality (CMAQ) model to estimate public health impacts of PM2.5 from individual power plants. Environment International 68:200-208.
- Burnett RT, Pope III CA, Ezzati M, Olives C, Lim SS, Mehta S, et al. (2014). An integrated risk function for estimating the global burden of disease attributable to ambient fine particulate matter exposure. Environmental Health Perspectives 122:397-403.
- Cohen, A. J., Ross Anderson, H., Ostro, B., Pandey, K. D., Krzyzanowski, M., Künzli, N., ... & Smith, K. (2005). The global burden of disease due to outdoor air pollution. Journal of Toxicology and Environmental Health, Part A, 68(13-14), 1301-1307.
- Crenna E, Secchi M, Benini L, Sala S (2019) Global environmental impacts: data sources and methodological choices for calculating normalisation factors. International Journal of LCA, doi.org/10.1007/s11367-018-1534-9
- EC (2013): European Commission. COMMISSION RECOMMENDATION of 9 April 2013 on the use of common models to measure and communicate the life cycle environmental performance of products and organisations (2013/179/EU)
- EC-JRC (2010a). International Reference Life Cycle Data System (ILCD) Handbook - Analysis of existing Environmental Impact Assessment methodologies for use in LCA - Background Document to the International Reference Life Cycle Data System (ILCD). Luxemburg. Publications Office of the European Union.
- EC-JRC (2010b). International Reference Life Cycle Data System (ILCD) Handbook - Framework and Requirements for Life Cycle Impact Assessment Models and Indicators. EUR 24586 EN. Luxembourg. Publications Office of the European Union.
- EC-JRC (2011). International Reference Life Cycle Data System (ILCD) Handbook - Recommendations for Life Cycle Impact Assessment in the European context. EUR 24571 EN. Luxemburg. Publications Office of the European Union.

- EC-JRC (2012). Characterisation factors of the ILCD Recommended Life Cycle Impact Assessment models. Database and Supporting Information. First edition. February 2012. EUR 25167. Luxembourg. Publications Office of the European Union; 2012.
- Fantke, P., Jolliet, O., Evans, J.S., Apte, J.S., Cohen, A.J., Hänninen, O.O., Hurley, F., Jantunen, M.J., Jerrett, M., Levy, J.I., Loh, M.M., Marshall, J.D., Miller, B.G., Preiss, P., Spadaro, J.V., Tainio, M., Tuomisto, J.T., Weschler, C.J., McKone, T.E. (2015). Health effects of fine particulate matter in life cycle impact assessment: findings from the Basel Guidance Workshop. The International Journal of Life Cycle Assessment, 20(2), 276-288.
- Fantke, P., Evans, J., Hodas, N., Apte, J., Jantunen, M., Jolliet, O., McKone, T.E. (2016). Health impacts of fine particulate matter. In: Frischknecht, R., Jolliet, O. (Eds.), Global Guidance for Life Cycle Impact Assessment Indicators: Volume 1. UNEP/SETAC Life Cycle Initiative, Paris, pp. 76-99 (available at: www.lifecycleinitiative.org/applying-lca/lcia-cf/, accessed January 2017).
- GBD (2017) Global Burden of Disease. Accessed at http://ghdx.healthdata.org/gbd-results-tool (March 2017)
- Greco, S., Wilson, A., Spengler, J. and Levy, J. (2007). Spatial patterns of mobile source particulate matter emissions-to-exposure relationships across the United States. Atmospheric Environment, Volume 41, Issue 5, Pages 1011-1025.
- Hodas N, Loh M, Shin H-M, Li D, Bennett D, McKone TE, et al. (2015). Indoor inhalation intake fractions of fine particulate matter. Review of influencing factors. Indoor Air:doi:10.1111/ina.12268.
- Hofstetter, P. (1998). Perspectives in Life Cycle Impact Assessment. A Structure Approach to Combine Models of the Technosphere, Ecosphere and Valuesphere. Kluwer Academic Publishers, Dordrecht, The Netherlands, 484 pp.
- Humbert S (2009) Geographically differentiated life-cycle impact assessment of human health. Doctoral dissertation, University of California, Berkeley, California, USA.
- Humbert S, Marshall JD, Shaked S, Spadaro JV, Nishioka Y, Preiss P, et al. (2011). Intake fraction for particulate matter: Recommendations for life cycle impact assessment. Environmental Science and Technology 45:4808-4816.
- Pope, C.A., Burnett R.T., Thun, M.J., Calle, E.E., Krewski,D., Ito, K., Thurston, G.D. (2002). Lung cancer, cardiopulmonary mortality, and long-term exposure to fine particulate air pollution. Journal of the American Medical Association 287, 1132-1141.
- Rabl, A. and Spadaro, J.V. (2004). The RiskPoll software, version is 1.051 (dated August 2004). www.arirabl.com
- Rosenbaum, R.K., Bachmann, T.M., Gold, L.S., Huijbregts, M.A.J., Jolliet, O., Juraske, R., Köhler, A., Larsen, H.F., MacLeod, M., Margni, M., McKone, T.E., Payet, J., Schuhmacher, M., van de Meent, D., Hauschild, M.Z. (2008): USEtox - The UNEP-SETAC toxicity model: recommended characterisation factors for human toxicity and freshwater ecotoxicity in Life Cycle Impact Assessment. International Journal of Life Cycle Assessment, 13(7), 532-546, 2008.
- Tessum CW, Hill JD, Marshall JD. (2015). InMAP: a new model for air pollution interventions. Geoscientific Model Development 8:9281-9321.
- van Zelm R, Huijbregts MAJ, den Hollander HA, van Jaarsveld HA, Sauter FJ, Struijs J, van Wijnen HJ, van de Meent D (2008) European characterization factors for human health damage of PM10 and ozone in life cycle impact assessment. Atmos Environ 42:441–453.

Acronyms and definitions

CFs	Characterisation Factors
DALY	Disability-Adjusted Life Years
EC	European Commission
EC-JRC	European Commission, Joint Research Centre
EF	Environmental Footprint
EPLCA	European Platform on Life Cycle Assessment
ILCD	International Reference Life Cycle Data System
iF	Intake Fraction
ILCD	International Reference Life Cycle Data System
ISO	International Organisation for Standardisation
LCA	Life Cycle Assessment
LCDN	Life Cycle Data Network
LCI	Life Cycle Inventory
LCIA	Life Cycle Impact Assessment
OEF	Organisation Environmental Footprint
PEF	Product Environmental Footprint
PDF	Potentially disappeared fraction of species
SOC	Soil Organic Carbon
SOM	Soil organic matter
TAB	Technical Advisory Board
SQI	Soil Quality Index
UNEP	United Nations Environment Programme
UNEP-SETAC life cycle initiative	United Nations Environment Programme (UNEP) and the Society for Environmental Toxicology and Chemistry (SETAC) Life Cycle Initiative
UUID	Universally Unique Identifier
WULCA	Water Use in LCA (name of the working group on water use related impact assessment)

Annexes

The following excel files are available as annexes to this report at http://eplca.jrc.ec.europa.eu/LCDN/developerEF.xhtml (the first number refer to the chapter the Annex pertains to):

- Annex 1.1 Environmental Footprint (EF) versioning
- Annex 3.1 Evaluation of characterisation models for resources use
- Annex 4.1 Evaluation of characterisation models for land use
- Annex 4.2 LANCA® model aggregation for calculating the soil quality index: list of cases excluded by the cut-off criteria
- Annex 5.1 Evaluation of characterisation models for water use
- Annex 6.1 Evaluation of characterisation models for particulate matter

KJ-NA-28636-EN-N

JRC Mission

As the science and knowledge service of the European Commission, the Joint Research Centre's mission is to support EU policies with independent evidence throughout the whole policy cycle.

EU Science Hub
ec.europa.eu/jrc

@EU_ScienceHub

EU Science Hub - Joint Research Centre

Joint Research Centre

EU Science Hub

Publications Office

doi:10.2760/78072

ISBN 978-92-79-69335-9