LCIA framework and cross-cutting issues guidance within the UNEP-SETAC Life Cycle Initiative

Francesca Verones, Jane Bare, Cécile Bulle, Rolf Frischknecht, Michael Hauschild, Stefanie Hellweg, Andrew Henderson, Olivier Jolliet, Alexis Laurent, Xun Liao, Jan Paul Lindner, Danielle Maia de Souza, Ottar Michelsen, Laure Patouillard, Stephan Pfister, Leo Posthuma, Valentina Prado, Brad Ridoutt, Peter Fantke

Published in

Journal of Cleaner Production

Volume 161, September 2017, Pages 957-967

https://doi.org/10.1016/j.jclepro.2017.05.206

10’974 words including all affiliations, captions, tables and references

LCIA framework and cross-cutting issues guidance within the UNEP-SETAC Life Cycle Initiative

Francesca **Verones**[1]*, Jane **Bare**[2], Cécile **Bulle**[3], Rolf **Frischknecht**[4], Michael **Hauschild**[5], Stefanie **Hellweg**[6], Andrew **Henderson**[7], Olivier **Jolliet**[8], Alexis **Laurent**[5], Xun **Liao**[9], Jan Paul **Lindner**[10], Danielle **Maia de Souza**[11], Ottar **Michelsen**[12], Laure **Patouillard**,[13] Stephan **Pfister**[6], Leo **Posthuma**[14,15], Valentina **Prado**[16], Brad **Ridoutt**[17,18], Ralph K. **Rosenbaum**[19], Serenella **Sala**[20], Cassia **Ugaya**[21], Marisa **Vieira**[22], Peter **Fantke**[5]

* corresponding author: *francesca.verones@ntnu.no*

[1] Industrial Ecology Programme, Department of Energy and Process Engineering, Norwegian University of Science and Technology (NTNU), NO-7491 Trondheim, Norway

[2] US EPA, Office of Research and Development, National Risk Management Research Laboratory, 26 West MLK Dr., Cincinnati, OH 45268

[3] CIRAIG, Ecole des Sciences de la Gestion, Université du Québec À Montréal, 315, rue Sainte-Catherine Est, Montréal, QC, Canada

[4] treeze Ltd, Kanzleistrasse 4, Uster, Switzerland

[5] Division for Quantitative Sustainability Assessment, Department of Management Engineering, Technical University of Denmark, Bygningstorvet 116B, 2800 Kgs. Lyngby, Denmark

[6] ETH Zurich, Institute of Environmental Engineering, 8093 Zürich, Switzerland

[7] Noblis, 16414 San Pedro Ave, San Antonio, TX 78232

[8] School of Public Health, University of Michigan, Ann Arbor, MI, United States

[9] Ecole Polytechnique Fédérale de Lausanne (EPFL), Lausanne, Switzerland

[10] Fraunhofer Institute for Building Physics, Stuttgart, Germany

[11] University of Alberta, Department of Agricultural, Food and Nutritional Science, T6G 2P5, Edmonton, Alberta

[12] NTNU Sustainability, Norwegian University of Science and Technology (NTNU), NO-7491 Trondheim, Norway

[13] CIRAIG, École Polytechnique de Montréal, P.O. Box 6079, Montréal, Québec H3C 3A7, Canada

[14] RIVM (Dutch National Institute for Public Health and the Environment), Centre for Sustainability, Environment and Health, P.O. Box 1, 3720 BA Bilthoven, the Netherlands

[15] Radboud University Nijmegen, Department of Environmental Science, Institute for Water and Wetland Research, Heyendaalseweg 135, 6525 AJ Nijmegen, The Netherlands

[16] Institute of Environmental Sciences CML, Leiden University, Einsteinweg 2, 2333 CC, Leiden

[17] Commonwealth Scientific and Industrial Research Organisation (CSIRO) Agriculture and Food, Private Bag 10, Clayton South, Victoria 3169, Australia

[18] University of the Free State, Department of Agricultural Economics, Bloemfontein 9300, South Africa

[19] IRSTEA, UMR ITAP, ELSA-PACT – Industrial Chair for Environmental and Social Sustainability Assessment, 361 rue Jean-François Breton, BP 5095, 34196 Montpellier, France

[20] European Commission, Joint Research Centre, Directorate D: Sustainable Resource, Bioeconomy unit, Via E. Fermi 2749, Ispra VA, Italy

[21] Federal University of Technology, Avenida Sete de Setembro, Rebouças Curitiba, Paraná, Brazil

[22] PRé Consultants B.V., Stationsplein 121, 3818 LE Amersfoort, Netherlands

Abstract

Increasing needs for decision support and advances in scientific knowledge within life cycle assessment (LCA) led to substantial efforts to provide global guidance on environmental life cycle impact assessment (LCIA) indicators under the auspices of the UNEP-SETAC Life Cycle Initiative. As part of these efforts, a dedicated task force focused on addressing several LCIA cross-cutting issues as aspects spanning several impact categories, including spatiotemporal aspects, reference states, normalization and weighting, and uncertainty assessment. Here, findings of the cross-cutting issues task force are presented along with an update of the existing UNEP-SETAC LCIA emission-to-damage framework. Specific recommendations are provided with respect to metrics for human health (Disability Adjusted Life Years, DALY) and ecosystem quality (Potentially Disappeared Fraction of species, PDF). Additionally, we stress the importance of transparent reporting of characterization models, reference states, and assumptions, in order to facilitate cross-comparison between chosen methods and indicators. We recommend developing spatially regionalized characterization models, whenever the nature of impacts shows spatial variability and related spatial data are available. Standard formats should be used for reporting spatially differentiated models, and choices regarding spatiotemporal scales should be clearly communicated. For normalization, we recommend using external normalization references. Over the next two years, the task force will continue its effort with a focus on providing guidance for LCA practitioners on how to use the UNEP-SETAC LCIA framework as well as for method developers on how to consistently extend and further improve this framework.

Keywords. life cycle impact assessment, characterization framework, uncertainty assessment, human health, ecosystem quality, natural resources

Highlights

- The existing UNEP-SETAC LCIA framework was updated.
- Recommendations were formulated for several LCIA cross-cutting issues.
- Recommendations were provided for specific areas of protection.
- Continuous efforts will focus on further harmonizing cross-cutting issues in LCIA.

1. Introduction

Life Cycle Assessment (LCA) is a method for environmental assessment and management, which has evolved to provide decision support. LCA is used for quantifying potential environmental impacts of products, processes, or services. The adverse impacts are usually assessed for several impact categories, such as acidification, eutrophication, and climate change. LCA is often used for comparative studies to support the selection of environmentally preferable alternatives, for eco-design purposes, and for identification of the potentially largest environmental impacts and trade-offs in a product life cycle (Hellweg et al. 2014). The LCA approach has also recently been extended to assessments of organizations (ISO/TS 14072 2014; UNEP et al. 2015), thereby increasing its range of applications and its reach to high-level decision- and policy-makers. Consequently, LCA-based decisions have become more and more relevant for recognizing and reducing environmental impacts of products and processes.

Triggered by the increasing needs for reliable decision support and by ongoing advances in scientific knowledge, the UNEP-SETAC Life Cycle Initiative (LC Initiative) has been initiated to improve the science and practices in the field of life cycle thinking (UNEP-SETAC 2016). The LC Initiative has established several task forces, aimed at 1) harmonizing current approaches, 2) furthering the development of life cycle impact assessment (LCIA), and 3) providing guidance on recommended models and methods for calculating environmental indicators so that their application provides the best possible transparency, reproducibility, and validity, as well as the best possible support for decision-making.

One of these UNEP-SETAC task forces has been addressing LCIA cross-cutting issues, i.e. topics that are relevant across several, or all, of the existing impact categories. The activities of this task force concentrated on the improvement and harmonization of the LCIA characterization framework, and on aspects such as furthering consensus regarding normalization and weighting, spatial differentiation, uncertainty assessment, endpoint indicators for human health, ecosystem quality, and natural resources, as well as the identification of representative reference states.

In 2004, the LC Initiative published a recommendation for an LCIA framework, embracing an overview of existing impact categories, and the status of their development (Jolliet et al. 2004). Since then, there has been substantial progress in LCIA methods, as well as underlying models and data, both in terms of covered impact pathways, spatial differentiation and resolution, novelties in endpoint indicators, and normalization procedures. It is therefore time to review and evaluate these developments and innovations in a structured way, especially for the damage (endpoint) level, while midpoints are kept as they were described in the 2004 framework. It is the aim of the cross-cutting issues task force to improve the applicability and operationalization of LCIA methods and to integrate scientific advances into the LCIA framework in a compatible and consistent way.

In January 2016, a Pellston workshop (i.e. a workshop hosted by the Society for Environmental Toxicology and Chemistry (SETAC) on critical and urgent topics) was conducted in Valencia, Spain, uniting efforts of the cross-cutting issues and other, topical, task forces, which worked on impacts derived from land and water use, exposure to fine particulate matter, and climate change (Frischknecht et al. 2016a). The workshop participants discussed several cross-cutting issues, such as the need to revise the LCIA framework, in order to include recent advances in LCIA science and achieve a more comprehensive coverage of indicators. In addition, recommendations for harmonization of reference states, spatial differentiation, normalization and weighting, uncertainty assessment across impact categories, as well as specific issues for individual areas of protection (e.g. aggregated metrics for damages on human health and on ecosystem quality) were discussed. This paper provides an overview of the current state of development of the previously mentioned cross-cutting issues, and presents expert recommendations. We deliver recommendations that are currently ready for consideration (section 3), and give an outlook where further research and harmonization are needed (section 4).

2. Approach

The task force on cross-cutting issues was established in January 2015, when it started to work on different issues in individual subtasks, as mentioned in the introduction. In late autumn

2015, all active members of the cross-cutting issues task force consolidated findings from the different subtasks into an internal white paper, which served as starting point for proposing recommendations during the Pellston workshop, to which several members of the cross-cutting issues task force but also members from all other guidance project tasks forces were invited along with various sector experts. Discussions between the workshop participants led to the formulation of recommendations, which were presented and discussed in a workshop plenary session, then finalized and agreed upon, and finally published in the official Pellston workshop report in early 2017, complemented with the main content of the initial cross-cutting issues white paper (Frischknecht et al. 2016b).

For some of the cross-cutting issues subtasks, participants produced and published final recommendations, while for other subtasks it was decided to collate further analytical reports on the current state-of-the-art, as a foundation for ongoing discussions. In the following, a status is given for each of the subtasks in the cross-cutting issues theme, followed by the outlook. The supporting information (SI, Tables S1 to S3) and Table 2 contain case study results for different production and consumption scenarios of 1kg rice, based on Frischknecht et al. (2016a), to exemplify the compliance of the topical indicators to and relevance of recommendations made for cross-cutting issues.

3. Results and recommendations

The discussions on the cross-cutting issues yielded various results, which are summarized below under separate subjects.

3.1. Update to the LCIA framework and damage categories

Currently, LCIA analyses result in outputs for three areas of protection for damages on: human health, ecosystem quality and natural resources. The definition of these areas aims to safeguard the values that are considered important to society (Table 1). For instance, the area of protection “human health” uses aggregated morbidity and mortality impacts as an indicator for measuring damages on human health.

Various methodological developments over the last decade indicate the need for an update of the existing LCIA framework and the harmonization of the different impact categories within and across areas of protection. There are, for example, damage methods published without midpoint indicators because of the lack of linear relationships between these midpoints and elementary flows, as well as between midpoints and observed damages. Also, for some impact categories no good suggestion for midpoints does currently exist (e.g. land use). This makes it necessary to allow for possibilities beyond modeling the impact pathway via midpoints to damages only (e.g. (Chaudhary et al. 2015; Verones et al. 2016b)). Moreover, research is progressing to include other environmental issues, such as ecosystem services, into LCIA (e.g. (Koellner et al. 2013; Cao et al. 2015; Othoniel et al. 2016)). After the scoping phase of the LC Initiative, ecosystem services appeared as a joint area of protection with natural resources (Jolliet et al. 2014). Thus, after analyzing recent developments, we propose to distinguish between two overarching systems (1: natural systems and, 2: humans and man-made systems) with three different types of values, in order to distinguish the reasons for identifying the different areas of protection more clearly. This leads in total to the identification of six potential areas of protection for consideration in LCIA (Table 1). Natural systems are broadly defined and go beyond the concept of ecosystems, including also immaterial assets, such as natural heritage, whereas humans and man-made systems are defined to only relate to anthropocentric values. “Values” in this context refer to aspects society deems worth protecting and are independent of the terms “values” and “value choices” as used in weighting.

The first set of values refers to intrinsic values, i.e. values given for the sake of the existence in itself. For instance, the damage categories human health and ecosystem quality encompass intrinsic values. It is generally recognized that human beings have a right to life on their own, and that non-human species have a value in their existence, i.e., value that would be lost if the species did not exist. A second set of values refers to instrumental values. These encompass values that have a clear utility to humans and are defined from an anthropocentric standpoint. They include, for example, any kind of resource, ecosystem service, or built infrastructure (socio-economic assets) exploitable or otherwise usable by humans. The third set are cultural values. These are again set from a human point of view and refer to spiritual, aesthetic, or recreational dimensions, including cultural and natural heritage. An example is a cultural heritage site (a damage will occur if this site is flooded for a hydropower dam, such as in Turkey, where the damming of the Tigris river risks flooding the ancient city of Hasankeyf (Berkun 2010)).

The cross-cutting issues task force is aware that additional work is required (see section 4 on outlook) to further refine the LCIA framework regarding the consideration of damage categories that have not yet sufficiently been addressed in LCA, such as those addressing ecosystem services and cultural and natural heritage. The inclusion of the latter two borders on social LCA. Recommendations on how to avoid potential double-counting of these values will need to be established (Zimdars et al. 2017) when combining environmental and social life cycle indicators (e.g. also considering the loss of an aesthetically-valued species), once methods for assessing impacts on these values have been developed and are operational. Ecosystem services may also contain cultural values (Millennium Ecosystem Assessment 2005) and therefore also need to be addressed in a way to avoid double-counting. This is a subject for further discussions.

Table 1: Overview of the human societal values and how damages on these values are measured and the respective links to humans/man-made and natural systems.

	Intrinsic values	**Instrumental values**	**Cultural values**
Humans and man-made systems	Human health (measured as damages on humans from morbidity & mortality)	Socio-economic assets (measured as damages on man-made environment such as built infrastructure, loss of cash crops, etc.)	Cultural heritage (measured as damages on buildings, historic monuments, artwork, landscapes, etc.)
Natural systems	Ecosystem quality (measured as damages on ecosystems, i.e. biodiversity loss, by means of species richness & vulnerability)	Natural resources & Ecosystem services (measured as damages on resources, such as exhaustion of mineral primary resources, loss of availability of crops, wood, loss of water flow regulation potentials, etc.)	Natural heritage (measured as damages on flora, fauna, geological elements, etc.)

In the original UNEP-SETAC LCIA framework (Jolliet et al. 2004) two modeling options are distinguished: 1) modeling up to midpoint impact indicators only, 2) modeling up to damage categories *via* midpoint impact indicators. The direct link between life cycle inventory (LCI) and damage category was not foreseen. A midpoint impact indicator was defined as an indicator "*located on the impact pathway at an intermediate position between the LCI results and the ultimate environmental damage*" (Jolliet et al. 2004). However, since then numerous methods, dealing with various impact categories, have been developed that do not contain midpoint impact indicators, but are instead modelled straight to a damage level (e.g. (Souza et al. 2013; Chaudhary et al. 2015; Verones et al. 2016b; Vieira et al. 2016). This is often the case when it is difficult and/or not informative to identify a separately quantifiable midpoint impact indicator for some impact pathways, such as for land use impacts, where in some cases only the area of land being occupied or transformed is provided (inventory parameter) (Vidal-Legaz et al. 2016).

It has been common to provide the linkage between combined impact categories at midpoint level and impact categories at damage level with one constant conversion factor for the whole world. However, since 2004, several impact categories have been developed that take spatial differentiation into account (e.g. land use, water use, and freshwater eutrophication). The consideration of spatial differentiation makes it difficult - or even impossible - to apply constant conversion factors, since the cause-effect model from midpoint impact indicator to damage indicator might vary spatially as well, depending on the impact category.

Even though midpoint impact indicators may be desirable in some circumstances, they are not required for an impact assessment model, nor are damage level indicators necessary. Models stopping at midpoint level, or models going directly to damage, or models encompassing both, are equally appropriate. As mentioned, traditionally, midpoint impact indicators have been converted to damage indicators via constant conversion factors. We assert explicitly that this is not a fixed requirement, but that instead spatially explicit conversion matrices can be used to improve validity, if the impact category in question contains a relevant spatial aspect. This has, for example, been explained for water impacts, where it is acknowledged that differences between regions matter substantially when considering this indicator (e.g. Pfister et al. (2009)). We are aware that non-globally uniform conversion factors may potentially be leading to different conclusions at the midpoint impact versus the damage level due to the introduction of additional information (variability). The discrepancy reflects that modelling beyond the midpoint introduces relevant additional information and hence that the midpoint result is less environmentally relevant than the damage result. We accept, though do not encourage, that, for the case that no relevant midpoint impact indicator can be identified along the impact pathway, proxy indicators can be designed, which are not defined along an impact pathway itself, such as for example water scarcity indicators (Boulay et al. 2016; Boulay et al. in review). These proxies need to be justified, labelled, and documented to avoid confusion. All in all, the proposed extensions to the LCIA framework as triggered by developments in science and societal concerns leads to an increased comprehensiveness, but also potentially more flexibility in the characterization framework (Figure 1). This has the implication that there is an even greater need than before to transparently report which impact pathway has been modelled up to what level, specifying whether (proxy) midpoint levels have been in- or excluded and providing, if possible, a documentation of their uncertainty.

During the Pellston workshop, the topical task forces proposed specific recommendations for indicators and characterization models for land stress, water stress, fine particulate matter formation, and climate change (Frischknecht et al. 2016b). All of these recommendations consistently fit into the recommended updated LCIA framework (Table 1 and Figure 1) and highlight the breadth of options and the need for a more flexible framework. Factors for climate change are recommended for a midpoint level only. While this indicator is on the impact pathway for potentially both human health and ecosystem quality, this is not the case for the recommended water scarcity indicator, which is defined as a proxy midpoint. Impacts from exposure to fine particulate matter on human health are defined at both midpoint and damage level, while water use impacts on human health and land stress impacts on ecosystems are defined on a damage level only. For land stress, no operational midpoint indicator is currently available.

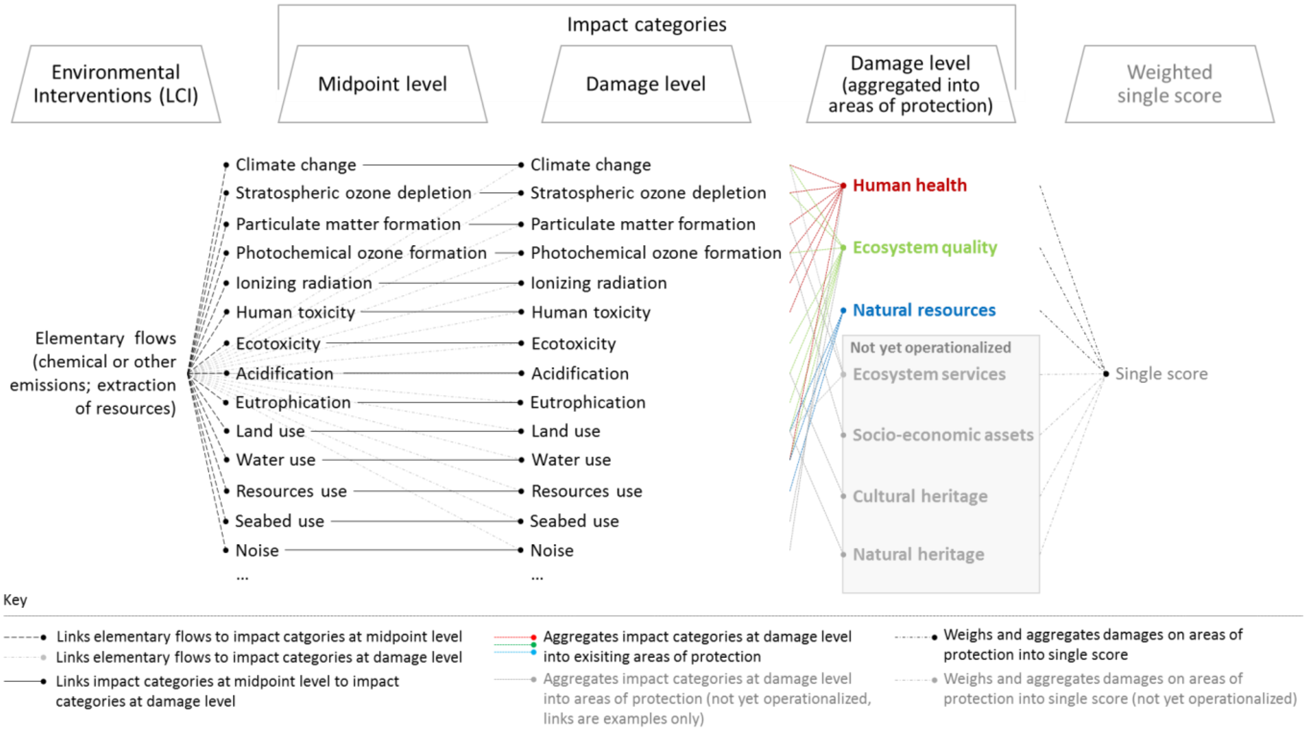

Figure 1: Updated LCIA framework. The lists of impact categories (on midpoint and damage level) are not complete and are meant to be indicative. Impact characterization models can link the Life Cycle Inventory (LCI) to midpoint impact level (column 2, black dashed lines) and stop there or continue to damage level (column 3, solid black lines), or they can go directly from the life cycle inventory (LCI) to damage level (column 3, grey, dotted line). Similar to midpoint modeling, damage modeling is based on natural science and involves assumptions and choices but is not a weighting step. Note that damage categories are available on a disaggregated level (e.g. climate change, land impacts), or they can be aggregated into overarching categories (column 4, colored lines for existing areas of protection, grey lines for not yet operational ones), if wished. Areas of protection that are operational are indicated with colors, those that are not yet fully operational are shown in the grey box. Weighting of damage category scores may include normalization and is an optional step (in grey) distinct from the damage modeling. Normalization and weighting can also be performed on midpoint impact indicator level.

3.2. Specific recommendation for areas of protection

Within each area of protection (aggregated impact categories at damage level), several different impacts may be combined (such as impacts on human health from toxicity, climate change and photochemical ozone formation, *i.e.* aggregation over items in the two left hand side columns in Figure 1). To aggregate, units and metrics need to be consistent among the categories that are aggregated. Thus, our focus here is on recommendations for the damage level, in order to make sure that consistent comparisons within areas of protection are possible. Aggregation into single scores per area of protection may ease the decision-making process and the communication of the results (fewer indicators have to be communicated), but may at the same time decrease transparency with respect to uncertainties and trade-offs among impact categories. Aggregation is a procedure that is commonly applied in LCA practice, and we include it for the sake of completeness, without advocating that assessments at damage level need to be aggregated, as this depends on the goal and scope of the study.

Whenever aggregated damage level results are used, comparability of metrics used and values addressed by the different areas of protection needs to be ensured, which is therefore an important part of the normalization and weighting subtask. Generally, we want to stress that calculating results at a damage level does not necessarily need to entail an aggregation into a single score per area of protection (note that aggregation across areas of protection relates to normalization and weighting processes, addressed in Section 3.5).

In the previous section, we described a potential broadening of areas of protection to consider in environmental decision-making. However, since some of them do not yet exist or are not yet fully evaluated, we will not give recommendations for these at this stage. Instead, we focus on improving the three main established categories, human health, ecosystem quality, as well as natural resources (in color in Figure 1).

Human health: Human health is an area of protection that deals with the intrinsic values of human health, addressing both mortality and morbidity. Several impact categories contribute to damages on human health, covering a wide variety of potential impacts. These range from toxic impacts from exposure to substances (e.g., increasing the incidence of cancer) to malnutrition (e.g., water shortages leading to crop shortages leading to malnutrition) to heat stress-related impacts (cardiovascular diseases) associated with greenhouse gas emissions. To compare impacts of these different categories at a damage level (i.e. the net damages on human health), it is crucial to have a common metric. In this respect, human health impact categories generally build on a well-established and widely adopted metric, which is the disability-adjusted life year (DALY) (Murray et al. 1996; Lopez 2005; Forouzanfar et al. 2015). We recommend to continue using DALYs in LCIA for human health, as proposed and motivated by Fantke et al. (2015). Topical indicators recommended at the damage level by the LC Initiative follow this recommendation (fine particulate matter, impacts of water use on human health; see illustrative rice case study in SI and Table 2). However, it is recommended that methods use the most recent severity weights originating from the Global Burden of Disease (GBD) study series (Salomon et al. 2012; Salomon et al. 2015). This is noteworthy, since the DALYs from the GBD 2010 study (Murray et al. 2012) do not embed age weighting and discounting in their base case anymore (for transparency reasons), which is compatible with the LCIA context. In line with enhancing and moving towards more transparent reporting, we also recommend to document the different components of a DALY separately (e.g., the years of life lost (YLL), the years lived disabled (YLD), and disability weighting).

Table 2 illustrates the usage of DALY in a case study on rice produced in different countries. It brings on the same common DALY scale potential impacts of malnutrition due to water use and impacts due to exposure to primary and secondary fine particulate matter. For India, these impacts per kg cooked rice are of similar order of magnitude, with 2.1×10^{-5} to 3.6×10^{-5} DALY/kg_{rice} for water use impacts, and 1.3×10^{-5} DALY/kg_{rice} for $PM_{2.5}$ related impacts, but are lower than the potential reduction in malnutrition impacts of 1.4×10^{-4} DALY/kg_{rice} associated with the production of one kg rice.

Table 2: Results for the human health impact of the functional unit (FU) of 1 kg of white, cooked rice (cooked at home in rural India, urban China, or Switzerland). The impact is shown at damage level. Further detail of the case study definition can be found in Frischknecht et al. (2016a).

Impact category	**Spatial region/Archetype**			
Water use impacts		**Inventory [m^3/FU]**	**CF [DALY/m^3]**	**Damage[DALY/FU]**
Rural India	Average India	0.78	4.59E-05	3.58E-05
	Ganges		3.80E-05	2.96E-05
	Godavari		2.70E-05	2.11E-05
Urban China	Average China	0.46	7.31E-05	3.36E-05
	Yellow River		1.20E-04	5.38E-05
	Pearl River		4.50E-06	2.07E-06
US/Switzerland	Average US	0.08	5.63E-05	4.51E-06
	Red River		1.30E-06	1.01E-07
	Arkansas River		6.70E-05	5.36E-06
Particulate matter formation (marginal)		**Inventory [kg/FU]**	**CF [DALY/kg]**	**Damage[DALY/FU]**
Rural India	Indoor, primary $PM_{2.5}$	1.71E-03	5.13E-03	8.80E-06
	Rural Outdoor, primary $PM_{2.5}$	4.36E-04	9.65E-05	4.21E-08
	Urban Outdoor, primary $PM_{2.5}$	-	-	-
	Outdoor, secondary $PM_{2.5}$: NH_3	6.07E-03	5.04E-04	3.06E-06
	SO_2	3.32E-03	2.34E-04	7.77E-07
	NO_x	3.49E-03	5.04E-05	1.76E-07
Urban China	Indoor, primary $PM_{2.5}$	-	-	-
	Rural Outdoor, primary $PM_{2.5}$	3.89E-04	9.65E-05	3.76E-08
	Urban Outdoor, primary $PM_{2.5}$	2.25E-04	3.74E-03	8.41E-07
	Outdoor, secondary $PM_{2.5}$: NH_3	6.07E-03	5.04E-04	3.06E-06
	SO_2	3.52E-03	2.34E-04	8.24E-07
	NO_x	3.38E-03	5.04E-05	1.70E-07
US/Switzerland	Indoor, primary $PM_{2.5}$	2.13E-06	1.69E+00	3.60E-06
	Rural Outdoor, primary $PM_{2.5}$	2.64E-04	9.65E-05	2.54E-08
	Urban Outdoor, primary $PM_{2.5}$	1.46E-05	3.74E-03	5.46E-08
	Outdoor, secondary $PM_{2.5}$: NH_3	1.50E-03	5.04E-04	7.56E-07
	SO_2	3.43E-03	2.34E-04	8.04E-07
	NO_x	3.59E-03	5.04E-05	1.81E-07
Particulate matter formation (average)		**Inventory [kg/FU]**	**CF [DALY/kg]**	**Damage[DALY/FU]**
Rural India	Indoor, primary $PM_{2.5}$	1.71E-03	1.66E-02	2.85E-05
	Rural Outdoor, primary $PM_{2.5}$	4.36E-04	2.31E-04	1.01E-07

	Urban Outdoor, primary $PM_{2.5}$		-	-	-
	Outdoor, secondary $PM_{2.5}$:	NH_3	6.07E-03	5.04E-04	3.06E-06
		SO_2	3.32E-03	2.34E-04	7.77E-07
		NO_x	3.49E-03	5.04E-05	1.76E-07
Urban China	Indoor, primary $PM_{2.5}$		-	-	-
	Rural Outdoor, primary $PM_{2.5}$		3.89E-04	2.31E-04	8.97E-08
	Urban Outdoor, primary $PM_{2.5}$		2.25E-04	5.29E-03	1.19E-06
	Outdoor, secondary $PM_{2.5}$:	NH_3	6.07E-03	5.04E-04	3.06E-06
		SO_2	3.52E-03	2.34E-04	8.24E-07
		NO_x	3.38E-03	5.04E-05	1.70E-07
US/Switzerland	Indoor, primary $PM_{2.5}$		2.13E-06	2.32E+00	4.93E-06
	Rural Outdoor, primary $PM_{2.5}$		2.64E-04	2.31E-04	6.08E-08
	Urban Outdoor, primary $PM_{2.5}$		1.46E-05	5.29E-03	7.72E-08
	Outdoor, secondary $PM_{2.5}$:	NH_3	1.50E-03	5.04E-04	7.56E-07
		SO_2	3.43E-03	2.34E-04	8.04E-07
		NO_x	3.59E-03	5.04E-05	1.81E-07

Ecosystem quality: The area of protection “Ecosystem Quality” deals with damages on the intrinsic value of natural ecosystems; to date, most models focus on compositional attributes of biodiversity only, such as species richness (e.g. Goedkoop et al. (2009); (Curran et al. 2016; Teixeira et al. 2016)). This area of protection encompasses diverse drivers and pathways of impacts (e.g., water stress, emissions of chemicals leading to eutrophication or acidification or ecotoxicity). Building consistency across the diverse models in this field is as important as it is challenging (Curran et al. 2011). However, we stress here that further research and developments should by no means be stifled by recommendations based on this paper.

Due to the prevalence of indicators for loss of species richness, we currently recommend the use of potentially disappeared fraction of species (PDF) as a common endpoint metric. However, the currently-used PDFs only seemingly represent a single metric, while representing sometimes (widely) different meanings, e.g., when they have been derived from models based on data from different scales (local, regional, global) or from effects data on different species groups for different stressors (discussed in Curran et al. (2011)). For instance, the action of building a parking lot may lead to a very high local loss of species on the plot occupied (local-scale PDF), but if only regionally and globally abundant species are lost, the regional-scale and global-scale PDF of the same intervention would be negligible. This example illustrates that PDFs of different scales should under no circumstances be mixed without a proper conversion. Also, impacts using different species groups are not to be mixed without proper consideration (first: recognizing possible differences) or conversion (second: handling the difference between groups). If other metrics than PDF are used, we recommend providing (preferably validated) conversion factors to PDF. Transparent reporting is also crucial to document the development of PDFs (e.g., which taxonomic groups or spatial locations were considered). Additionally, we recommend that the model developers report PDFs in a disaggregated way (i.e. separately for freshwater, marine and terrestrial ecosystems), and, if applicable, for specific taxonomic groups (i.e., specifically for plants, or invertebrates, when those were used to define a PDF). If possible, to facilitate application, aggregation procedures across taxonomic groups and ecosystems to one final set of values should be made available. First approaches for this exist (e.g. Verones et al. (2015)), but we recommend putting further efforts into researching options for this aggregation. Until consistent aggregation across taxonomic groups is possible, we recommend developing impact indicators for different taxonomic groups separately. The choice of taxonomic groups and modelling approaches should be documented clearly and transparently to facilitate the understanding by practitioners. Impacts on ecosystems, both at regional and global scales, should be reported whenever possible (global levels reporting on irreversible extinction, regional levels being important for preserving ecosystem functions in places where endemism is low) (see also section 3.3). The indicator recommended for land stress is fully aligned with these recommendations (Chaudhary et al. 2015; Frischknecht et al. 2016b). This PDF indicator quantifies both regional losses and global losses, and clearly does so for a set of taxonomic groups, while, for the ease of application, also providing taxa-aggregated characterization factors. Table S1 (SI) illustrates how this indicator applies to the rice case study for the global PDF impacts of land occupation, showing that three types of land occupation dominate the impact of species, i.e., the production (cultivation) of the rice as could be expected, the intensive forest production of wood for cooking in the India scenario and the use of urban area in the US production/Swiss consumption scenario. Other improvements of this indicator (e.g. regarding intensities of land use) are recommended by the land use task force (Milà i Canals et al. 2016), but do not affect the recommendations related to cross-cutting issues.

<u>Natural resources and ecosystem services:</u> To date, many impact assessment methods (e.g. (Goedkoop et al. 1999; Jolliet et al. 2003; Goedkoop et al. 2009)) consider a third damage category focusing on resources. This is the only category that so far focuses on “instrumental values” (Table 1). We recommend refining the scope of this damage category to “natural resources” (Sonderegger et al. accepted). As of now there are several different definitions of

what should be in- or excluded in such an area of protection (see e.g. the discussion in Dewulf et al. (2015)).

Ecosystem services have an instrumental value for humans, and are defined as "*the benefits people obtain from ecosystems"* (Millennium Ecosystem Assessment 2005). Thus, ecosystem services can also be seen as a part of the natural resources, but are seldom operationalized in LCIA models at this time. However, the LCIA research community has made first steps towards their inclusion (e.g. (Zhang et al. 2010a; Zhang et al. 2010b; Saad et al. 2013)), including the identification of challenges of doing so (Zhang et al. 2010a; Zhang et al. 2010b; Bare 2011; Othoniel et al. 2016), but further efforts are needed to adequately include the different types of ecosystem services (provisioning, regulating, supporting and cultural) in models with global coverage (models covering only a small spatial unit, such as an individual country or part of an ecoregion are often not applicable in other world regions due to differences in present services and environmental conditions. Therefore, models are required that can deliver individual factors for different world regions).

3.3. Guidance on temporal and spatial modelling issues

It is becoming increasingly clear that, in various instances, spatial and temporal issues are of utmost relevance in LCIA (Hauschild 2006). For instance, when evaluating water use impacts, the sensitivity of receiving ecosystems towards impacts can vary significantly, and can therefore lead to spatially different characterization factors (CF) (Boulay et al. 2015). Taking global CFs (averages) may lead to over- or underestimations of impacts. Therefore, introducing spatial differentiation (or regionalization) in LCIA models can help improve the accuracy of LCA results (Mutel et al. 2009). The same is true for aggregation of temporal data in the case of water consumption (e.g. Pfister et al. (2014)) and also for photochemical ozone (Shah and Ries 2009; Huijbregts 1998).

Spatially differentiated LCIA models and CFs are available in various existing LCIA methods, such as LC-Impact (Verones et al. 2016a), TRACI (Bare 2002), IMPACT World+ (Bulle et al. 2012), Ecological Scarcity (Frischknecht et al. 2013), or EDIP (Potting et al. 2004) for either multiple impact categories or single indicators (e.g. water use impacts, eutrophication, land use impacts, toxicity, acidification).

For all recommended impact categories except climate change, some kind of spatial differentiation is included, either through the use of spatial archetypes for capturing at the global level relevant variabilities across various urban and rural areas for particulate matter formation or via full inclusion of spatial details on an ecoregion (land stress) or watershed (water scarcity and water consumption impacts) level. Although these spatial aspects are all clearly reported, the data format of characterization factors is often not consistent. The importance of including spatial differentiation in relation to water stress – the impact category with the largest spatial variation in characterization factors - is highlighted in Table S3 (SI) for the illustrative rice case study: Between the Yellow and Pearl watersheds in urban China, there is almost a factor of 200 difference in terms of how scarce water is, and impacts from water consumption on human health vary more than a factor 25. Using a Chinese or global average would underestimate the impact greatly in one case (Yellow river), while overestimating it in the other case (Pearl River). Moving towards including spatial detail is therefore a crucial recommendation for improving environmental assessments. Still, for the ease of application, all topical indicators recommended in the guidance process provided aggregated CFs (country level, for instance) in addition to regionalized ones to also allow for impact characterization when e.g. emission regions are unknown.

Spatial variation is also high for human impacts from exposure to fine particulate matter due to variation in population density around the locations of emission or the more than 100 times difference in intake fractions between indoor and outdoor releases as function of location. Accounting for such spatial variation based on exact location of emission would require to know the exact emission location and to model the dispersion at a 10 km or higher resolution, which is usually not practical for LCA applications. Table 2 illustrates for the rice case study how such spatial variation can be handled via the definition of characterization factors differentiated by indoor, rural outdoor and urban outdoor archetypes, which can then be linked to present life cycle inventory databases, such as ecoinvent. The exact parameterization of the indoor archetypes can be further customized to the country or continental region of production and consumption, the CFs of Table 2 accounting for regional person density and building tightness in each region. In the case of human health impacts of fine particulate matter exposure, archetypes need to not only reflect spatial variation in population density, but also the level of exposure, since the considered dose-response is non-linear and depends on background exposure of the considered individuals.

If spatial differentiation is meaningful to the nature of the impact category covered, and if data are available, we recommend developing spatial characterization factors for midpoint and damage impact categories. Spatial differentiation is meaningful, if the potentially "impacted entity" shows clear differences in spatial distribution, such as water scarcity or biodiversity. The geographical resolution should ideally reflect the spatial characteristics of the impacted entity (e.g. watersheds for water consumption impacts, ecoregions for land-use impacts, or population density for human toxicity). The recommended topical indicators fulfill these recommendations (Frischknecht et al. 2016b), as shown in the case study results presented in the SI.

In order to facilitate the use of regionalized CF and the interpretation of final LCA results, LCIA method developers should use a standardized format for reporting regionalized CFs. Standards from the Open Geospatial Consortium (OGC 2016) are recommended as a good starting point. For instance, they recommend using the GeoTIFF format for raster data and the GeoPackage Vector format for vector data.

Transparent reporting urges a clear specification of all assumptions related to the inclusion of regionalization in LCIA models (e.g., the level of spatial differentiation of input LCIA parameters, the choice for the resulting spatial resolution for spatially differentiated LCIA methods and the way spatially aggregated CFs have been calculated). This is imperative, even if the chosen model has global resolution without regionalized CFs.

3.4 Reference states

Most impact categories require a baseline scenario, which is commonly referred to as the "reference state." This can be either a historical situation, a (hypothetical) future state of the environment, a situation in absence of human interventions, a political target situation, or the current situation. A reference state, thus, refers to both time and space. Choices in the reference state may influence the outcome of the characterization factors. However, many LCIA methods do not mention explicitly which reference state they use, which makes it hard for researchers and practitioners to judge whether these models are compatible (referring to the same reference state) or not. We therefore recommend that the choice of reference state be reported transparently and explicitly. Table S4 in the SI summarizes the chosen reference states for all topical indicators recommended. Except for land use, all indicators are using current, fixed situations (e.g. a fixed reference year), and represent a pragmatic approach (i.e. constrained by data availability). Land use defines a “natural” situation as baseline and represents a normative approach (i.e. based on desirability).

Regarding modeling procedures, there are also different possibilities, such as modelling marginal or average impacts. Marginal approaches depart from the current situation (i.e. influencing also the choice of reference state) and assess the impact of one additional unit of emission/resource use. Average assessments focus on the difference between the current situation and the background concentration (historical or zero). This also has an implication for the characterization factors and should, for the sake of transparency and user-friendliness for practitioners, be explicitly reported by model developers. Especially regarding emission-based impact categories, we recommend model developers provide both marginal and average characterization factors. The former are useful for practitioners in the case of small changes being assessed (e.g. individual products), while the latter are useful for assessing larger changes in an economy or longer time frames (Huijbregts et al. 2011). The provided CFs for land use and fine particulate matter follow this recommendation, providing both marginal and average CFs. Table 2 compares the marginal and average characterization factors applied in the illustrative rice case study for human health impacts of fine particulate matter exposure. The difference is especially important in the case of indoor emissions from solid fuel combustion with a factor 3 higher average CF than the marginal CF due to the non-linear dose-response with decreasing slope at higher exposure levels. In this particular case of indoor cooking, the average dose-response may be more adequate for LCA decision contexts, since switching to another type of cooking or to low emission cook stoves would reduce exposure by one or several orders of magnitude, which does not correspond any more to a marginal change.

3.5. Normalization and weighting

To date, there is no recommendation for which normalization or weighting approach should be used. According to the ISO standard 14044 both normalization and weighting are optional steps in LCA (ISO 2006). Normalization has three main purposes, namely 1) checking the plausibility of LCA results (i.e. their magnitude of results), 2) setting the results into perspective by comparing the magnitude of every individual impact category, and, optionally, 3) preparing the results for further weighting by translating them into a common unit. The main purpose of weighting is to facilitate aggregation of indicators and to reflect the preferences of decision-maker(s) and stakeholders in the assessment. Weighting factors can be elicited a number of ways: from direct elicitation of preferences to weighting methods based on policy targets (Huppes et al., 2012). In the end, weighting is typically applied to obtain a single score for the assessment. Normalization and weighting may sometimes also be useful when reporting footprints that cover more than one impact pathway (Ridoutt et al. 2015).

A review of the normalization and weighting approaches, including an assessment of their strengths and weaknesses as well as recommendations for their applications and further developments, can be found in Pizzol et al. (2016). Following the outcome of the Pellston workshop, the current recommendation is to favor external normalization approaches in studies that apply normalization, i.e. approaches in which the reference system is independent from or not directly related to the alternatives assessed in the study (e.g. society's background load within a given region or the world). Compared to internal normalization approaches, where the reference system is a function of the assessed alternatives, external approaches are the only ones capable of meeting all three aforementioned purposes. As a subsequent recommendation, wherever possible, LCA practitioners should opt for global instead of regional or national normalization references to avoid the risk of inconsistency between the geographical scopes of the LCI results of the study and that of the inventory behind the normalization references. In a globalized market, LCA studies are typically associated with a geographical scope – and hence LCI results – spread over the entire world. In practice, it is important to note that there are data gaps in current external normalization references, which may lead to biases in the impact results and which the LCA practitioners should be aware of (Heijungs et al. 2006; Laurent et al. 2015; Pizzol et al. 2016; Cucurachi et al. 2017). In all cases, a sensitivity analysis should be performed to test the influence of different weighting and normalization approaches, and sources of uncertainties should be clearly identified, described, and discussed by practitioners.

3.6. Handling of uncertainties

The models underlying each LCIA come with uncertainties, and neglecting these uncertainties may lead to incorrect LCIA interpretations and thus biased decision support. This can be circumvented and made transparent by uncertainty analysis. A complete and fully quantitative uncertainty analysis makes it clear whether predicted median differences for an impact reflect real differences or only reflect a slight (or no) difference (due to overlapping confidence intervals of the items being compared).

In the models and data underlying LCA, there are different types of uncertainty, such as parameter uncertainty, model uncertainty, or value choices (Huijbregts 1998; Hertwich et al. 2001a; Hertwich et al. 2001b). Although it is clear that uncertainties in models and data exist, LCIA methods rarely report uncertainties for their characterization factors. However, first attempts have been made to quantify chemical-specific uncertainty for characterization results related to certain impact pathways, (e.g. Fantke et al. (2016)), or to provide a generic, quantitative uncertainty estimate for characterization results across chemicals, e.g. Rosenbaum et al. (2008), to propagate parameter uncertainty using a Monte Carlo approach (Roy et al. 2014), or to combine model and parameter uncertainty (Henderson et al. 2017). Because of lack of uncertainty information on CFs, uncertainty of LCIA results is rarely included in LCA reports and publications. If sound and transparent decisions are to be supported, reporting of uncertainties should become a routine practice to avoid over-interpretation and biased decisions. Identifying, qualitatively or even quantitatively describing, and finally documenting uncertainties would also allow highlighting assumptions, data and model components for model developers that need special attention to further improve the LCIA methods. We recommend that model developers and practitioners alike report uncertainties at least in a qualitative way (if a quantitative approach is not possible). This advice is followed by the topical indicators who all discuss uncertainty at least in a qualitative way (Frischknecht et al. 2016b). Explicit 95% confidence intervals are given for the land stress impacts, while others, such as the water scarcity indicator reports results of sensitivity analyses or spatial variability (water consumption impacts on human health, particulate matter related impacts).

4. Outlook

Apart from the issues discussed here, there are still multiple cross-cutting issues that need future research and more comprehensive discussion within the UNEP-SETAC cross-cutting issues task force and with external experts and stakeholders. The task force calls for further discussion and development on issues across all areas of protection (especially those not yet developed, see Figure 1), as well as spatial and temporal issues and uncertainty assessment. Below, we discuss some specific, concrete suggestions, without the ambition to be comprehensive, but as a way to stimulate and suggest priority items for research.

Ecosystem quality is an area of protection with a large need for further development. Scientific analyses suggest that a multitude of approaches can be chosen to quantify ecological impacts (e.g., McGill et al. (2015)), warranting close attention to models, metrics and underlying data to define ecological impacts within and across the various impact categories. Apart from completing and improving the coverage of impact pathways, there is a need for increasing the harmonization across impact categories. This includes, for example, thoughts about whether vulnerability measures should be considered. Such measures could include that there are species or ecosystems that are more vulnerable to certain types of interventions than others and that there may be large differences in the importance of different species for the functioning of ecosystems. Impact assessment models that account for several taxonomic groups (e.g. plants, birds and mammals) need to take care to include the differences in species numbers between the groups. Species-rich taxonomic groups tend to dominate the impact assessment, even though they may not be the taxon that is potentially losing the largest fraction of species. Taxonomic groups should not be weighted based on their species richness alone, as this may lead to underestimating impacts on smaller taxonomic groups, whose species may be more threatened. In terms of which species should be used for constructing impact assessment models, we argue that species should be taken into account that are representative for an ecosystem, and its functions and niches, reflecting different levels of threats and endemism.

Damage categories related to natural resources and ecosystem services are in need of further development too. However, there is little consensus on how to model impacts and which endpoint indicators to aspire to. Due to the challenges associated with the damage category of natural resources, from definitions to harmonization and coherence in modelling, a dedicated task force will be in place in the next phase (2016-2017) of the UNEP-SETAC flagship project for guidance on LCIA indicators.

Further research and development is also needed on how temporally and spatially differentiated LCIA methods can be integrated into LCA approaches and how aggregations across different temporal and spatial scales should take place. Uncertainty related to temporal and spatial variability should be reported for temporally and spatially aggregated CFs. Also, future efforts will focus on developing guidance on which uncertainties should and could be reported quantitatively in LCIA. It is suggested to consider the possibility of assigning a generic uncertainty factor to impact assessment methods that do not provide uncertainty values. Such a generic factor is usually much higher than truly quantified uncertainty values to motivate practitioners and developers to report uncertainty values. If such values can be provided (quantitatively or qualitatively, for example through a Pedigree matrix (Weidema et al. 1996; Fantke et al. 2012), this generic factor will be reduced.

For normalization two topics are of interest for further investigation: (i) the Planetary Boundary concept and its integration in LCIA, and (ii) the incorporation of Multi Criteria Decision Analysis (MCDA) methods. The former has recently gained important momentum in environmental assessment and management as it paves the way for developing approaches and tools allowing to benchmark impacts from an analyzed system with absolute thresholds, which should not be exceeded to keep earth systems functioning (Rockstrom et al. 2009). Some early studies have discussed ways of integrating it as part of the characterization, the normalization, or the weighting steps (Fang et al. 2015; Sandin et al. 2015; Bjørn et al. 2016). No consensus currently exists on this aspect and further research that clearly identify the implications of such integration (e.g. uncertainties, applicability to diverse case studies, etc.) are needed before recommendations can be formulated. With respect to Multi Criteria Decision Analysis (MCDA), some methods aiming at improving decision support in comparative LCAs have also been proposed (Benoit et al. 2003; Prado et al. 2012). These methods are typically applied after characterization and require uncertainty information which may not be available to practitioners.

5. Conclusions

The UNEP-SETAC task force on cross-cutting issues in LCIA evaluated an update of the LCIA framework, and worked on harmonizing several other issues, such as regionalization. The evaluations showed latitude for improving LCIA-practices for existing and future indicators. Recommendations are presented with possible improvements on the short and longer term. The improvements will help increase the comprehensiveness as well as the meaningfulness of LCIA outputs for decision-support. The activities of the task force are still ongoing and will focus on further progress towards harmonizing several cross-cutting issues in LCIA. Recommendations made here were followed partly by the topical task forces present at the Pellston workshop (land use, water use, fine particulate matter, climate change) in establishing the consensual indicators. For the LCIA research community our recommendations have three main implications: 1) the call for increased comprehensiveness on the coverage of areas of protection, 2) the call for an improved transparency in model documentation to ease the identification of compatibility among models and indicator results, and 3) an enhanced recognition of the importance of aligning different cross-cutting aspects, such as standards for spatial differentiation and/or how uncertainty is addressed. Recommendations are targeted towards the LCA community in an effort to contribute to improved decision making through the transparent use of LCIA methods.

Disclaimer

The views expressed in this article are those of the authors and do not necessarily represent the views or policies of the organizations to which they belong. The designations employed and the presentation of the material in this publication do not imply the expression of any opinion whatsoever on the part of the UNEP-SETAC Life Cycle Initiative concerning the legal status of any country, territory, city or area or of it authorities, or concerning delimitation of its frontiers or boundaries. Moreover, the views expressed do not necessarily represent the decision or the state policy of the UNEP-SETAC Life Cycle Initiative, nor does citing of trade names or commercial processes constitute endorsement.

Acknowledgements

The authors acknowledge the contribution from the UNEP-SETAC Life Cycle Initiative for funding this activity.

5. References

Bare, J. (2011). "Recommendation for land use impact assessment: first steps into framework, theory, and implementation." Clean Technologies and Environmental Policy **13**(1): 7-18.

Bare, J. C. (2002). "Traci: The Tool for the Reduction and Assessment of Chemical and Other Environmental Impacts." Journal of Industrial Ecology **6**(3-4): 49-78.

Benoit, V. and Rousseaux, P. (2003). "Aid for aggregating the impacts in Life Cycle assessment." The International Journal of Life Cycle Assessment **8**(2): 74-82.

Berkun, M. (2010). "Hydroelectric potential and environmental effects of multidam hydropower projects in Turkey." Energy for Sustainable Development **14**(4): 320-329.

Bjørn, A., Margni, M., Roy, P.-O., Bulle, C. and Hauschild, M. Z. (2016). "A proposal to measure absolute environmental sustainability in life cycle assessment." Ecological Indicators **63**: 1-13.

Boulay, A.-M., Bare, J., Benini, L., Berger, M., Lathuillière, M., Manzardo, A., Margni, M., Motoshita, M., Núñez, M., Pastor, A. V., Ridoutt, B., Oki, T., Worbe, S. and Pfister, S. (in review). "The WULCA consensus characterization model for water scarcity footprints: Assessing impacts of water consumption based on available water remaining (AWARE)." Environ. Sci. Technol.

Boulay, A.-M., Motoshita, M., Pfister, S., Bulle, C., Muñoz, I., Franceschini, H. and Margni, M. (2015). "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., Pfister, S., Motoshita, M., Schencker, U., Benini, L., Gheewala, S., Brito de Figueiredo, M. C. and Harding, K. (2016). Chapter 5: Water use related impacts: Water scarcity and human health effects. Part 1: Water Scharcity. Global Guidance for Life Cycle Impact Assessment Indicators, Volume 1. R. Frischknecht and O. Jolliet. France, Paris, UNEP.

Bulle, C., Jolliet, O., Humbert, S., Rosenbaum, R. and Margni, M. (2012). "IMPACT World +: A new global regionalized life cycle impact assessment method. ." Proceedings of the International Conference on Ecobalance , Yokohama, Japan.

Cao, V., Margni, M., Favis, B. D. and 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., Verones, F., De Baan, L. and Hellweg, S. (2015). "Quantifying Land Use Impacts on Biodiversity: Combining Species-Area Models and Vulnerability Indicators." Environ. Sci. Technol. **49**(16): 9987-9995.

Cucurachi, S., Seager, T. P. and Prado, V. (2017). "Normalization in Comparative Life Cycle Assessment to Support Environmental Decision Making." Journal of Industrial Ecology: n/a-n/a.

Curran, M., De Baan, L., De Schryver, A., Van Zelm, R., Hellweg, S., Koellner, T., Sonnemann, G. and Huijbregts, M. A. J. (2011). "Toward Meaningful End Points of Biodiversity in Life Cycle Assessment." Environ. Sci. Technol. **45**(1): 70-79.

Curran, M., De Souza, D. M., Antón, A., Teixeira, R., Michelsen, O., Vidal-Legaz, B., Sala, S. and Milà i Canals, L. (2016). "How well does LCA model land use impacts on biodiversity?—A comparison with approaches from ecology and conservation " Environ. Sci. Technol. **50**: 2782-2795.

Dewulf, J., Benini, L., Mancini, L., Sala, S., Blengini, G. A., Ardente, F., Recchioni, M., Maes, J., Pant, R. and Pennington, D. (2015). "Rethinking the Area of Protection "Natural Resources" in Life Cycle Assessment." Environmental Science & Technology **49**(9): 5310-5317.

Fang, K., Heijungs, R. and De Snoo, G. R. (2015). "Understanding the complementary linkages between environmental footprints and planetary boundaries in a footprint–boundary environmental sustainability assessment framework." Ecological Economics **114**: 218-226.

Fantke, P. and Jolliet, O. (2016). "Life cycle human health impacts of 875 pesticides." The International Journal of Life Cycle Assessment **21**(5): 722-733.

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. and 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., Wieland, P., Juraske, R., Shaddick, G., Itoiz, E. S., Friedrich, R. and Jolliet, O. (2012). "Parameterization Models for Pesticide Exposure via Crop Consumption." Environmental Science & Technology **46**(23): 12864-12872.

Forouzanfar, M. H., Alexander, L., Anderson, H. R., Bachman, V. F., Biryukov, S., et al. (2015). "Global, regional, and national comparative risk assessment of 79 behavioural, environmental and occupational, and metabolic risks or clusters of risks in 188 countries, 1990–2013: a systematic analysis for the Global Burden of Disease Study 2013." The Lancet **386**(10010): 2287-2323.

Frischknecht, R. and 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, Switzerland.

Frischknecht, R., Fantke, P., Tschümperlin, L., Niero, M., Antón, A., et al. (2016a). "Global guidance on environmental life cycle impact assessment indicators: progress and case study." The International Journal of Life Cycle Assessment: 1-14.

Frischknecht, R. and Jolliet, O. (2016b). Global Guidance for Life Cycle Impact Assessment Indicators, Volume 1. Paris, France, United Nations Environment Programme.

Goedkoop, M., Heijungs, R., Huijbregts, M. A. J., De Schryver, A., Struijs, J. and van Zelm, R. (2009). ReCiPe 2008: A life cycle impact assessment method which comprises harmonised category indicators at the midpoint and endpoint levels. First edition. Report i: Characterization. The Netherlands, Ruimte en Milieu, Ministerie van Volkshuisvesting, Ruimtelijke Ordening en Milieubeheer.

Goedkoop, M. and Spriensma, R. (1999). The Eco-Indicator 99. A damage oriented method for life cycle impact assessment. Methodology report and Annex, Pré Consultants, Amersfoort, The Netherlands. http://www.pre.nl/eco-indicator99/.

Hauschild, M. (2006). "Spatial Differentiation in Life Cycle Impact Assessment: A decade of method development to increase the environmental realism of LCIA." The International Journal of Life Cycle Assessment **11**(1): 11-13.

Heijungs, R., Guinée, J., Kleijn, R. and Rovers, V. (2006). "Bias in normalization: Causes, consequences, detection and remedies." The International Journal of Life Cycle Assessment **12**(4): 211.

Hellweg, S. and Milà i Canals, L. (2014). "Emerging approaches, challenges and opportunities in life cycle assessment." Science **344**(6188): 1109-1113.

Henderson, A. D., Asselin-Balençon, A. C., Heller, M., Lessard, L., Vionnet, S. and Jolliet, O. (2017). "Spatial Variability and Uncertainty of Water Use Impacts from U.S. Feed and Milk Production." Environmental Science & Technology **51**(4): 2382-2391.

Hertwich, E. and Hammitt, J. (2001a). "A decision-analytic framework for impact assessment part I: LCA and decision analysis." The International Journal of Life Cycle Assessment **6**(1): 5-12.

Hertwich, E. G. and Hammitt, J. K. (2001b). "A decision-analytic framework for impact assessment." The International Journal of Life Cycle Assessment **6**(5): 265.

Huijbregts, M. A. J. (1998). "Application of uncertainty and variability in LCA." The International Journal of Life Cycle Assessment **3**(5): 273-280.

Huijbregts, M. A. J., Hellweg, S. and Hertwich, E. (2011). "Do We Need a Paradigm Shift in Life Cycle Assessment?" Environ. Sci. Technol. **45**: 3833-3834.

ISO (2006). Environmental management - Life Cycle Assessment - Requirements and guidelines. International Standard ISO 14044, International Organisation for Standardisation. Geneva, Switzerland.

ISO/TS 14072 (2014). Environmental management -- Life cycle assessment -- Requirements and guidelines for organizational life cycle assessment. International Organisation for Standardisation. Geneva, Switzerland.

Jolliet, O., Frischknecht, R., Bare, J., Boulay, A.-M., Bulle, C., et al. (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.

Jolliet, O., Margni, M., Charles, R., Humbert, S., Payet, J., Rebitzer, G. and 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., Müller-Wenk, R., Bare, J., Brent, A., Goedkoop, M., Heijungs, R., Itsubo, N., Peña, C., Pennington, D., Potting, J., Rebitzer, G., Stewart, M., de Haes, H. and Weidema, B. (2004). "The LCIA midpoint-damage framework of the UNEP/SETAC life cycle initiative." The International Journal of Life Cycle Assessment **9**(6): 394-404.

Koellner, T., De Baan, L., Beck, T., Brandão, M., Civit, B., Margni, M., Milà i Canals, L., Saad, R., de Souza, D. M. and Müller-Wenk, R. (2013). "UNEP-SETAC guideline on global land use impact assessment on biodiversity and ecosystem services in LCA." International Journal of LCA **18**(6): 1188-1202.

Laurent, A. and Hauschild, M. Z. (2015). Normalisation in LCA. Life Cycle Impact Assessment. M. Z. Hauschild and M. A. J. Huijbregts. Springer, Dordrecht NL.

Lopez, A. D. (2005). "The evolution of the Global Burden of Disease framework for disease, injury and risk factor quantification: developing the evidence base for national, regional and global public health action." Globalization and Health **1**(1): 1-8.

McGill, B. J., Dornelas, M., Gotelli, N. J. and Magurran, A. E. (2015). "Fifteen forms of biodiversity trend in the Anthropocene." Trends in Ecology & Evolution **30**(2): 104-113.

Milà i Canals, L., Antón, A., Bauer, C., De Camillis, C., Freiermuth, R., Grant, T., Michelsen, O. and Stevenson, M. (2016). Chapter 6: Land use related impacts on biodiversity. Global Guidance for Life Cycle Impact Assessment Indicators, Volume 1. R. Frischknecht and O. Jolliet. Paris, France, UNEP.

Millennium Ecosystem Assessment (2005). Ecosystems and Human Well-being. Synthesis. Island Press, Washington, DC.

Murray, C. J. L. and Lopez, A. D. (1996). "Evidence-Based Health Policy--Lessons from the Global Burden of Disease Study." Science **274**(5288): 740-743.

Mutel, C. L. and Hellweg, S. (2009). "Regionalized Life Cycle Assessment: Computational Methodology and Application to Inventory Databases." Environ. Sci. Technol. **43**: 5797-5803.

OGC, (Open Geospatial Consortium)). (2016). "OGC® Standards and Supporting Documents." Retrieved 27 January, 2016, from http://www.opengeospatial.org/standards.

Othoniel, B., Rugani, B., Heijungs, R., Benetto, E. and Withagen, C. (2016). "Assessment of Life Cycle Impacts on Ecosystem Services: Promise, Problems, and Prospects." Environmental Science & Technology **50**(3): 1077-1092.

Pfister, S. and 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., Koehler, A. and Hellweg, S. (2009). "Assessing the Environmental Impacts of Freshwater Consumption in LCA." Environ. Sci. Technol. **43**(11): 4098-4104.

Pizzol, M., Laurent, A., Sala, S., Weidema, B., Verones, F. and Koffler, C. (2016). "Normalisation and weighting in life cycle assessment: quo vadis?" The International Journal of Life Cycle Assessment: 1-14.

Potting, J. and Hauschild, M. Z. (2004). "Background for spatial differentiation in life cycle impact assessment. The EDIP2003 methodology.".

Prado, V., Rogers, K. and Seager, T. P. (2012). Integration of MCDA Tools in Valuation of Comparative Life Cycle Assessment. Life Cycle Assessment Handbook, John Wiley & Sons, Inc.**:** 413-431.

Ridoutt, B., Pfister, S., Manzardo, A., Bare, J., Boulay, A.-M., et al. (2015). "Area of concern: a new paradigm in life cycle assessment for the development of footprint metrics." The International Journal of Life Cycle Assessment: 1-5.

Rockstrom, J., Steffen, W., Noone, K., Persson, A., Chapin, F. S., et al. (2009). "A safe operating space for humanity." Nature **461**(7263): 472-475.

Rosenbaum, R. K., Bachmann, T. M., Gold, L. S., Huijbregts, M. A. J., Jolliet, O., Juraske, R., Koehler, A., Larsen, H. F., MacLeod, M., Margni, M., McKone, T. E., Payet, J., Schuhmacher, M., van de Meent, D. and Hauschild, M. Z. (2008). "USEtox - the UNEP-SETAC toxicity model: recommended characterisation factors for human toxicity and freshwater ecotoxicity in life cycle impact assessment." Int. J. Life Cycle Assess. **13**: 532-546.

Roy, P.-O., Deschênes, L. and Margni, M. (2014). "Uncertainty and spatial variability in characterization factors for aquatic acidification at the global scale." Int. J. Life Cycle Assess. **19**: 882-890.

Saad, R., Koellner, T. and 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**(6): 1253-1264.

Salomon, J. A., Haagsma, J. A., Davis, A., de Noordhout, C. M., Polinder, S., Havelaar, A. H., Cassini, A., Devleesschauwer, B., Kretzschmar, M., Speybroeck, N., Murray, C. J. L. and Vos, T. (2015). "Disability weights for the Global Burden of Disease 2013 study." The Lancet Global Health **3**(11): e712-e723.

Salomon, J. A., Vos, T., Hogan, D. R., Gagnon, M., Naghavi, M., et al. (2012). "Common values in assessing health outcomes from disease and injury: disability weights measurement study for the Global Burden of Disease Study 2010." The Lancet **380**(9859): 2129-2143.

Sandin, G., Peters, G. M. and Svanström, M. (2015). "Using the planetary boundaries framework for setting impact-reduction targets in LCA contexts." The International Journal of Life Cycle Assessment **20**(12): 1684-1700.

Sonderegger, T., Dewulf, J., Drielsma, J., Fantke, P., Maia de Souza, D., Pfister, S., Stoessel, F., Verones, F., Vieira, M., Weidema, B. P. and Hellweg, S. (accepted). "Towards harmonizing natural resources as an area of protection in Life Cycle Impact Assessment " International Journal of LCA(DOI:10.1007/s11367-017-1297-8).

Souza, D. M., FLynn, D., Rosenbaum, R. K., de Melo Lisboa, H. and Koellner, T. (2013). "Land use impacts on biodiversity: proposal of characterization factors based on functional diversity." int. J. Life Cycle Assess. **18**(6): 1231-1242.

Teixeira, R. F. M., Maia de Souza, D., Curran, M. P., Antón, A., Michelsen, O. and Milà i Canals, L. (2016). "Towards consensus on land use impacts on biodiversity in LCA: UNEP/SETAC Life Cycle Initiative preliminary recommendations based on expert contributions." Journal of Cleaner Production **112, Part 5**: 4283-4287.

UNEP-SETAC. (2016). "Life Cycle Initiative." Retrieved 21 April, 2016, from http://www.lifecycleinitiative.org/.

UNEP, (United Nations Environment Programme) and SETAC, (Society for Environmental Toxicology and Chemistry) (2015). Guidance on Organizational Life Cycle Assessment, UNEP.

Verones, F., Hellweg, S., Azevedo, L. B., Chaudhary, A., Cosme, N., Fantke, P., Goedkoop, M., Hauschild, M. Z., Laurent, A., Mutel, C. L., Pfister, S., Ponsioen, T., Steinmann, Z., Van Zelm, R., Verones, F., Vieira, M. and Huijbregts, M. A. J. (2016a). "LC-IMPACT Version 0.5- A spatially differentiated life cycle impact assessment approach " Retrieved 28 April, 2017, from http://www.lc-impact.eu/.

Verones, F., Huijbregts, M. A. J., Chaudhary, A., De Baan, L., Koellner, T. and Hellweg, S. (2015). "Harmonizing the Assessment of Biodiversity Effects from Land and Water Use within LCA." Environ. Sci. Technol. **49**(6): 3584-3592.

Verones, F., Pfister, S., van Zelm, R. and Hellweg, S. (2016b). "Biodiversity impacts from water consumption on a global scale for use in life cycle assessment." The International Journal of Life Cycle Assessment: 1-10.

Vidal-Legaz, B., Sala, S., Antón, A., Maia de Souza, D., Nocita, M., Putman, B. and Texeira, R. F. M. (2016). Land-use related environmental indicators for Life Cycle Assessment: Analysis of key aspects in land use modelling. JRC Technical Report. Luxemburg: Publications Office of the European Union.

Vieira, M. D. M., Ponsioen, T. C., Goedkoop, M. J. and Huijbregts, M. A. J. (2016). "Surplus Ore Potential as a Scarcity Indicator for Resource Extraction." Journal of Industrial Ecology.

Weidema, B. P. and Wesnæs, M. S. (1996). "Data quality management for life cycle inventories—an example of using data quality indicators." Journal of Cleaner Production **4**(3–4): 167-174.

Zhang, Y., Baral, A. and Bakshi, B. R. (2010a). "Accounting for Ecosystem Services in Life Cycle Assessment, Part II: Toward an Ecologically-Based LCA." Environ. Sci. Technol. **44**(7): 2624-2631.

Zhang, Y., Singh, S. and Bakshi, B. R. (2010b). "Accounting for Ecosystem Services in Life Cycle Assessment, Part I: A Critical Review." Environmental Science & Technology **44**(7): 2232-2242.

Zimdars, C., Haas, A. and Pfister, S. (2017). "Enhancing comprehensive measures of social impacts in S-LCA by including environmental and economic aspects." International Journal of LCA: DOI: 10.1007/s11367-11017-11305-z.