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LCIA OF IMPACTS ON HUMAN HEALTH AND ECOSYSTEMS

Ecosystem quality in LCIA: status quo, harmonization, and suggestions for the way forward

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Abstract

Purpose Life cycle impact assessment (LCIA) results are used to assess potential environmental impacts of different products and services. As part of the UNEP-SETAC life cycle initiative flagship project that aims to harmonize indicators of potential environmental impacts, we provide a consensus viewpoint and recommendations for future developments in LCIA related to the ecosystem quality area of protection (AoP). Through our recommendations, we aim to encourage LCIA developments that improve the usefulness and global acceptability of LCIA results.

Methods We analyze current ecosystem quality metrics and provide recommendations to the LCIA research community for achieving further developments towards comparable and more ecologically relevant metrics addressing ecosystem quality.

Results and discussion We recommend that LCIA development for ecosystem quality should tend towards species-richness-related metrics, with efforts made towards improved inclusion of ecosystem complexity. Impact indicators—which result from a range of modeling approaches that differ, for example, according to spatial and temporal scale, taxonomic coverage, and whether the indicator produces a relative or absolute measure of loss—should be framed to facilitate their final expression in a single, aggregated metric. This would also improve comparability with other LCIA damage-level indicators. Furthermore, to allow for a broader inclusion of ecosystem quality perspectives, the development of an additional indicator related to ecosystem function is recommended. Having two complementary metrics would give a broader coverage of ecosystem attributes while remaining simple enough to enable an intuitive interpretation of the results.

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Conclusions We call for the LCIA research community to make progress towards enabling harmonization of damage-level indicators within the ecosystem quality AoP and, further, to improve the ecological relevance of impact indicators.

Keywords Biodiversity · Damage-level · Endpoint · Functions · Harmonization · LCIA · Species · UNEP-SETAC

1 Introduction

It is important to have decision support tools that allow for quantifying environmental impacts of human activities. Life cycle assessment (LCA) was developed to provide such decision support by identifying products or services with comparatively lower environmental impacts compared on a functional basis (Hellweg and Mila i Canals 2014). The LCA framework enables characterization of inventories of emissions and resource use over the life cycle of modelled products or services in terms of potential impacts on various areas of protection (AoPs): humans, ecosystems, and natural resources. With respect to impacts related to ecosystems, the life cycle impact assessment (LCIA) framework distinguishes between impacts affecting intrinsic values and instrumental values for humans (Verones et al. 2017a). The former mostly relates to current concerns for biodiversity loss, which is covered under the area of protection (AoP) "ecosystem quality". Ecosystem services, the instrumental benefits people obtain from ecosystems, are foreseen as a future, additional AoP (Verones et al. 2017a). In this paper, we focus on recommendations for ecosystem quality, with ecosystem services outside the scope of our recommendations.

Ideally, LCA should encompass models and indicators that directly and unequivocally provide insights into the potential impact on ecosystem quality of product- or service life cycles. However, problems related to the conceptual, technical, and data aspects that define ecosystem quality impact indicators remain (Curran et al. 2011; McGill et al. 2015). Furthermore, the implementation of potentially useful indicators requires considerations of practical applicability and consistency with the evolving LCIA framework (Jolliet et al. 2014; Verones et al. 2017a).

Curran et al. (2011) provide the most recent cross-cutting review on models to assess damage to ecosystem quality in life cycle impact assessment (LCIA). Note that more recent reviews exist for specific impact categories, especially land use (Curran et al. 2016; Koellner et al. 2013; Maia de Souza et al. 2015; Michelsen and Lindner 2015; Teixeira et al. 2016). Curran et al. (2011) provided two overarching research recommendations. The first addressed conceptual shortcomings, with specific emphasis on increasing spatial detail within LCIA models. The second advocated expanding the use of globally available biodiversity data and developing new impact factors reflecting additional attributes of biodiversity, such as phylogenetic diversity, trait-space distance, and indicators of structure at the community and ecosystem level (Curran et al. 2011). These recommendations were derived with a focus on improving the representation of biodiversity, as a whole, in ecosystem quality impact indicators. An idealized set of indicators would comprehensively represent damage to ecosystem quality, and be useful, i.e., straightforward to apply and interpret by LCIA practitioners and decision-makers.

Advancing LCIA in the context of global biodiversity threats, attributed to human activities, constitutes a challenge for LCIA model developers. More specifically, the challenge is to improve the assessment of potential impacts on ecosystem quality in LCIA in a manner consistent with the larger LCA framework. The international LCA community has created the UNEP-SETAC flagship project "Global Guidance on Environmental Life Cycle Impact Assessment Indicators" (Frischknecht et al. 2016; Jolliet et al. 2014) to address this challenge. This paper presents the consensus viewpoint of the UNEP-SETAC life cycle initiative working group on ecosystem quality, a part of the cross-cutting issues task force. Recommendations of the working group are based on discussions and recommendations from the Pellston workshop (Verones et al. 2017), which included participants of various disciplines and backgrounds.

In this paper, we present the result of these discussions, balancing current scientific knowledge with the practical needs of LCA, aiming towards transparent, reproducible, and operational decision support. While there is the need to reach consensus on specific models for individual impact categories, e.g., land use and water use, the cross-cutting issues task force aims for harmonization and comparability across impact categories. For that purpose, we highlight the need to improve the comparability and ecological relevance of damage-level impact metrics within the AoP "ecosystem quality." Herein, we explore the current state of ecosystem quality within the LCA framework, highlight limitations of current ecosystem quality metrics (building on the foundation of Curran et al. (2011)), and propose pragmatic, cross-cutting options for improving the assessment of damage to ecosystem quality in LCA. With this paper, we hope to encourage and stimulate model developers to work towards solving cross-cutting issues, i.e., issues applicable across impact categories, to contribute to more harmonized, and thus useful, LCIA models.

2 Ecosystem quality within the life cycle impact assessment framework

The "ecosystem quality" AoP encompasses multiple, independent impact categories (Maia de Souza et al. 2015), such as eutrophication, acidification, ecotoxicity, land use, and water use, each linked to distinct stressors, i.e., emissions and resource use. These stressors initiate one or more impact pathways, crossing different environmental compartments. For example, an emission of NH_2 or NO_x may cross from air to surface water, soil, and/or marine water compartment, creating acidification and eutrophication impacts. Modelling of these pathways may reach to the midpoint level (environmental damage, e.g., presence of phosphorus, a eutrophying substance, in the environment) or damage level (previously referred to as endpoint level; this is the ecosystem damage, i.e., the consequence of environmental problems on ecosystem quality (Verones et al. 2017a)).

In recent years, substantial effort has been directed at an all-embracing coverage of various measures of ecosystem damage in LCIA. In that sense, there has been an increasing use of different indicators of damage to ecosystem quality, most prominently the potentially disappeared fraction of species (PDF) (Goedkoop and Spriensma 2001), the net primary production (NPP) loss (Pfister et al. 2009; Taelman et al. 2016), and the expected increase in number of extinct species (EINES) (Itsubo and Inaba 2012). However, in order to allow for comparison across the various impact categories, and/or to provide an aggregated indicator of potential damage to ecosystem quality, indicators at damage level need to be comparable, regardless of stressors or impact pathways. In agriculture, for example, the application of pesticides and fertilizers, the occupation of land areas, and the use of irrigation water will all affect terrestrial and aquatic ecosystems, though through different impact pathways, such as ecotoxicity, eutrophication, and habitat changes.

Defining "ecosystem quality", however, as a self-contained and comprehensive AoP requires a significant effort (Paetzold et al. 2010), as the concept is multifaceted (as highlighted by Curran et al. (2011)) and encompasses various biological features and different levels of organization (Noss 1990). According to the convention on biological diversity, an ecosystem is defined as "a dynamic complex of plant, animal and micro-organism communities and their non-living environment interacting as a functional unit" (CBD 1992). The condition of an ecosystem is defined by the interrelated status of its biotic and abiotic components, processes (functions), and structure (Cardinale et al. 2012), which interact in complex and non-linear ways (Chapin III et al. 2000). Indicators of potential ecosystem damage should embody this complexity, where possible, for improved decision support.

In LCIA, ecosystem quality refers to the condition of an ecosystem relative to a reference state. The reference state can be a past, present, or potential future situation/condition of an ecosystem. However, defining reference states is a complex matter (Stoddard et al. 2006); the feasibility and definition of a harmonized reference state is currently under discussion in the UNEP-SETAC flagship project but is outside the scope of this paper.

Ecosystem complexity and dynamics need to be carefully incorporated in LCIA models such that an increase in model complexity is accompanied by an increase in the relevance and accuracy of the results (Van Zelm and Huijbregts 2013). This task is even more imperative as LCIA results are to be interpreted by decision-makers and LCA practitioners, who often do not have domain knowledge about ecosystem dynamics. Deriving metrics that embrace a wide range of relevant ecosystem components remains a challenge (Maia de Souza et al. 2015), leading to a trade-off between having multiple indicators to cover damage to multiple ecosystem attributes and maintaining ease of interpretation, i.e., having fewer and comparable metrics (Teixeira et al. 2016).

3 Ecosystem quality impact metrics in LCIA: state of the art and limitations

Anthropogenically induced ecosystem changes such as a decline in species richness, loss of functional diversity, and reduced ecosystem biomass, are metrics used to indicate damage to ecosystem quality. Recently, a study identified at least 15 different metrics for describing trends in biodiversity (McGill et al. 2015), which is a key component of ecosystem quality. Additionally, the existence of non-linear ecological responses to stressors is a challenging issue in the context of current LCIA models, which are based on the assumption of linear stress-response functions (Huijbregts et al. 2011). LCA studies often consider comparative emissions of small magnitudes of various individual stressors, and the error introduced by reducing stressor-response relationships to linear functions is considered small or negligible. In addition, data on more complex (i.e., non-linear) relationships are often unavailable. Various LCIA linearization approaches exist, including the marginal approach, which calculates the marginal change from a background condition, and the average approach, which calculates the average change between the background condition and a preferred environmental state (Huijbregts et al. 2011).

Non-linear stress-response relationships for modelling damage to ecosystem quality originate from the application of species sensitivity distribution (SSD) models to exposure data of various species (Posthuma et al. 2002; van Straalen and Denneman 1989). An SSD model describes the statistical relationship between the intensity of a stressor, e.g., concentration of a pollutant, and the potentially affected fraction of species (PAF). The SSD model is constructed based on the stress response of individual species at a certain level of adverse effects, for example, effect concentration-50 (EC50), which is the concentration of a chemical causing a reduction of 50% in the performance of a life history trait of a tested species. In the LCA context, this PAF metric has been the basis for a metric of ecosystem damage in terms of species loss, by a conversion of PAF (at midpoint level) into PDF (at damage level) (Rosenbaum 2015). Originally used to quantify multi-species impacts for individual pollutants (within the field of ecotoxicity), an SSD can potentially be constructed for any stress factor (Posthuma and De Zwart 2014). Laboratory or field data, especially (bio)monitoring data sets that contain a stressor variable of interest which is sufficiently independent from other stressors, can be used to derive species distribution models (SDMs), which are used to describe and predict the probability of species occurrence in relation to environmental factors, e.g., Schipper et al. (2014). SDMs can be seen as the field-based equivalents of SSDs, as shown by, e.g., Struijs et al. (2011). A multi-species metric can, thereby, be derived using stacked SDMs of the species occurring in a region, e.g., as shown by Schipper et al. (2014). Beyond SSDs and (stacked) SDMs for various stressor variables, other non-linear modelling approaches have been applied. For example, various authors model land use impacts using empirical species-area relationships (SAR) (e.g., Chaudhary et al. (2015)), habitat-suitability models (e.g., Geyer et al. (2010)), or meta-analysis of field-monitoring studies (Elshout et al. 2014). Similarly, species-discharge relationships are used to model potential impacts of water consumption on fish species in rivers (Hanafiah et al. 2011; Tendall et al. 2014).

Despite the development of a suite of LCIA impact metrics over the last decade, covering several impact categories, limitations still exist. Curran et al. (2011) found that most indicators concentrate on species richness only, thereby neglecting the characterization of potential damage to ecosystem function, i.e. ecosystem processes, and/or structure, i.e., the physical attributes of ecosystems. However, even within the suite of damage-level metrics focused on species richness, a multitude of different units and modelling approaches is available that would require harmonization to support comparability and final aggregation to overall damage estimates. Curran et al. (2011) identified conceptual limitations associated with LCIA modelling approaches, many of which remain and are summarized (with some additional limitations) in Table 1.

The current PDF-impact metric in this context seems to be an example of a well-designed metric, useful for aggregation towards ecosystem damage quantification across impact categories in LCIA. However, there are hidden differences in the underlying modelling approach with this metric. For example, in Table 1, we demonstrate the hidden non-comparability of two land occupation impact scores that indicate damage to ecosystem quality in terms of PDF*years. Considering only variation in temporal and spatial considerations for simplicity, the seemingly uniform PDF impact indicator could pertain to four different spatio-temporal scales, i.e., combinations of sub-global or global and temporary or permanent. A stressor may cause species extirpation (i.e., loss of a species within a spatially defined, "sub-global," compartment) or extinction (i.e., loss of a species globally) or sub-extinction effects at a sub-global or global scale (e.g., reduced species abundance). The global loss of a species is irreversible and related to aspects of vulnerability, as outlined below and discussed in Section 4. Otherwise, the distinction between a "temporary" and a "permanent" loss is related to the time horizon and impact to be considered. In LCIA, it is recommended (Verones et al. 2017b) to distinguish between impacts that occur within 100 years or longer, in line with recommendations from the IPCC (IPPC 2014). Thus, if an ecosystem recovers within 100 years, we would consider the impact to be temporary, and would consider the impact permanent if recovery takes longer. Therefore, even the seemingly uniform PDF metric may mask differences across models. Furthermore, in the absence of harmonized metrics, aggregation of impact indicators across impact categories has potential for double counting, which may arise, for example, due to different temporal scales and taxonomic coverage.

Another aspect of non-comparability relates to aspects of vulnerability (see also discussion in the next section). Some impact categories have developmental metrics that include information highlighting that some species may be more likely than others to disappear because of an environmental stressor (e.g., Chaudhary et al. (2015) and Verones et al. (2013)). Comparing vulnerability-related impacts with those that do not contain such information may bias the comparison. Furthermore, there is currently not a consistent way to take such information into account.

4 Towards harmonized ecosystem quality damage metrics

Based on the diversity and structure of existing ecosystem quality metrics, the harmonization of damage-level metrics within the ecosystem quality AoP requires agreement on two conditions: (i) choice of ecosystem attribute, i.e., harmonized "units"; and, (ii) sufficient accounting of the ecosystem complexity in estimating potential impacts to those attributes, i.e., the context to which these units apply.

In the following sections, we recommend potential cross-cutting developments for improving ecosystem quality metrics to increase the usefulness of indicators of damage to ecosystem quality for LCA practitioners and decision-makers. These recommendations, summarized in Table 2, target development within LCIA, rather than life cycle inventory.

Table 1 Summary of conceptual limitations of LCIA damage modelling in the Ecosystem Quality AoP, building on the work of Curran et al. (2011) and exemplified by two land occupation impact scores (PDF*years) with modelling dissimilarities (**bold** text). The characterization factors we apply are available from http://www.lifecycleinitiative.org/applying-LCA/LCIA-CF and explained by Chaudhary et al. (2015)

Numerical example. Inventory flow = 1 m^2*years of annual croplands. Impact category = land stress (occupation). Location = Albania within the Balkan mixed forests ecoregion (PA0404)

Modelling (dis)similarity criteria	Description	CF = 5.98E-15 PDF*m^{-2} Impact score = 5.98E-15 PDF*years	CF = 1.30E-10 PDF*m^{-2} Impact score = 1.30E-10 PDF*years
Ecosystem attribute	Impact indicators reflect damage to ecosystem *composition, function, or structure*	Species richness	Species richness
Biodiversity scale	Impact indicators apply to *species, species assemblages, or ecosystems*	Ecosystems	Ecosystems
Spatial scale	Impacts are modelled at a *local, regional or global scale*	**Regional (Country; Albania)**	**Regional (Ecoregion; Balkan mixed forests)**
Temporal scale	Indicators reflect either *temporary or permanent* damage to ecosystem quality, and potential impacts are modelled over *different time horizons*	Time horizon determined by occupation period	Time horizon determined by occupation period
Vulnerability coverage	Coverage of *vulnerability* aspects, which include species or ecosystem, sensitivity, adaptive capacity and recoverability, *varies* between LCIA models	**Yes, a vulnerability score is included. Impact score represents a global species loss**	**No vulnerability score is included. The impact score represents a regional species loss**
Sensitivity measure	Impact modelling concerns species, assemblage or ecosystem responses to a stressor based on either *lab-scale testing* or *field-based observations*	Countryside SAR-modelled	Countryside SAR-modelled
Taxonomic coverage	*Taxonomic coverage* (typically determined by data availability, sensitivity of a taxonomic group to a particular stressor, and perceived representativeness of a taxonomic group of overall ecosystem quality) *varies* considerably between LCIA models	**Mammals, birds, reptiles, amphibians, plants**	**Mammals**
Relative or absolute	Damage to ecosystem quality is indicated by either *relative*, e.g., PDF, *or absolute*, e.g., species equivalents, measures	Relative	Relative
Marginal or average	Non-linear stress-response relationships are typically linearized using a marginal or average approach	Average	Average

4.1 Potential cross-cutting developments

4.1.1 Consolidation towards relative species-richness-related ecosystem metrics

There is scientific consensus that biodiversity (variation in genes, species and functional traits) underpins ecosystem function (Cardinale et al. 2012). Differing ecological interactions among species, i.e., additive, keystone or redundant interactions, co-determine whether and how biodiversity loss results in impacts (Valiente-Banuet et al. 2015). More specifically, the loss of a functionally redundant species has a smaller influence on ecosystem function than the loss of a keystone species (Valiente-Banuet et al. 2015). As such, a decrease in species richness does not wholly reflect ecosystem function loss nor damage to ecosystem quality. Similarly, at the local scale, the presence of genetically diverse populations of the same species is an important indicator of the ecosystem's capacity to provide its benefits to human communities (Hughes et al. 1997). This is the case regardless of species extinction at the global scale. However, the current ability to measure and map genetic diversity globally is limited by the availability of georeferenced molecular markers and thus by the choice of genetic polymorphisms that can be statistically correlated to human pressures in a mechanistic impact pathway (Miraldo et al. 2016), as would be required for LCIA. Given the existing need for a harmonized damage-level metric for ecosystem quality, the lack of operational approaches for genetic diversity in LCA (Curran et al. 2011), and the current prevalence of species-related metrics in operational LCIA methods, LCIA model development could best move, as a first step, towards a harmonized species-related damage-level metric. This damage-level metric should indicate potential relative species loss, thereby accounting for the uneven distribution of species richness globally. An additional development would be the proposal of a complementary damage-level metric, related to functional diversity (see section: Potential for furthering ecosystem quality metrics). In parallel, and with a more long-term

Table 2 Summary of recommendations emerging from the UNEP-SETAC task force on ecosystem quality. Priorities are based on the opinion of task force members. High: established current research theme or key first step for harmonization. Moderate: explorative research in progress, moderate contribution towards harmonization. Low: little or no research ongoing, smaller contribution to harmonization. The basis and scientific rationale for each recommendation is explained in the relevant section

Section	Recommendation summary	Priority
Potential cross-cutting developments		
Consolidation towards relative species-richness-related ecosystem metrics	A species-related damage-level metric that indicates potential relative species loss (such as the potentially disappeared fraction of species)	High
Improved recognition of spatial and temporal scale issues	Time- and space-integrated impact metrics at a resolution consistent with the stressor type and variability in ecosystem vulnerability	High
Improved recognition of spatial and temporal scale issues	Addition of a time-dependent parameter, such as one related to recoverability, as a first step towards vulnerability-adjusted indicators	High
Taxonomic coverage	Incorporate new data as availability improves, thereby using the best available data, and broadening taxonomic coverage over time	Moderate
Taxonomic coverage	Further research into potential species weighting options and the appropriateness of comparing impacts across different compartments, e.g., terrestrial, freshwater and marine.	Moderate
Conversion between spatio-temporal scales	Clearly distinguish between permanent and temporary impacts	High
Conversion between spatio-temporal scales	Further research with respect to potential temporary and permanent impacts and discounting, time horizons and potential secondary species losses	Low
Conversion between measurement types	Converting between relative and absolute metrics using species density data at the native spatial scale of the original impact metric, or, if species density data are limited, the best available spatial resolution	High
Conversion between levels of effect	For PAF metrics, provide a factor for converting to a PDF based on best available evidence	High
Potential for furthering ecosystem quality metrics		
Further development of species-richness-related metrics	Fully develop the concept of vulnerability in LCIA, especially when the LCIA would be serving decision processes for non-global, but regional purposes	Moderate
Operationalization of ecosystem function-related metrics	Further research to develop a complementary damage-level metric, related to functional diversity. Two complementary metrics for the ecosystem quality AoP; capturing species loss and functional diversity respectively	Low

perspective, further research regarding the potential and complementarity of genetic diversity-based indicators for LCIA purposes is encouraged in order to broaden the scope and representativeness of LCIA results.

4.1.2 Improved recognition of spatial and temporal scale issues

Ecological impacts are scale- and time-dependent. Damage-level impacts are influenced by time-related components of environmental stressors, such as the timing of occurrence, different life stages of affected species, and exposure time (Verones et al. 2010). The ecological response over time then depends on the sensitivity, adaptive capacity, and recoverability of individual species or communities (Zijp et al. 2017). These vulnerability parameters can vary, depending for example on spatial distribution, such as in the recovery time of forests (Müller-Wenk and Brandão 2010), and species endemism, such as the endemic species richness of plants (Kier et al. 2009). Impact indicators should therefore be calculated using characterization factors that are time- and space-integrated. Furthermore, the spatial resolution of CFs should be consistent with the type of stressor and variable ecosystem vulnerability, as indicated by the sensitivity and vulnerability of its component species and the emergent characteristics of the complex of inter-species interactions in the food web (Zijp et al. 2017).

Recoverability over time is one defining aspect of ecosystem vulnerability (Mumby et al. 2014). Addition of such a time-dependent parameter to LCA impact scores that are usually based on species sensitivity to stress exposure would pave the way for the development of vulnerability-adjusted indicators, i.e., metrics that reflect that vulnerable species are less able to recover than resilient species. This would introduce additional information into LCA for identifying "hotspots" of potentially high damage to ecosystem quality. We recommend incorporation of such parameters across a wider range of impact categories. Options include the use of proxies such as ecosystem scarcity and species threat level, as determined by the International Union for Conservation of Nature and Natural Resources (IUCN), to indicate the capacity for recovery from local damage (Verones et al. 2015). We provide further discussion of including vulnerability aspects in LCIA impact metrics for potential damage to ecosystem quality below, in the section "Further development of species-based metrics." The task force on cross-cutting issues currently has the mandate to further explore options to include these aspects of vulnerability and recoverability into assessments of ecosystem quality. This is done in discussion with experts from various disciplines (such as ecology and industrial ecology) and affiliations (such as academia, private sectors, and government agencies).

4.1.3 Taxonomic coverage

The choice or availability of taxa sensitivity data is currently different for each stressor for which a stress-response curve is constructed (including SSDs, SDM-type models, and SARs). The taxonomic coverage represented by "species richness" in LCA is therefore specific to individual impact pathways, e.g., (Azevedo et al. 2013a; Azevedo et al. 2013b; Verones et al. 2013). Accounting for the whole number of species that populate an ecosystem is not feasible due to lack of data.

With a few exceptions, e.g., Verones et al. (2013), at present, all species included within a particular model are assumed to have the same intrinsic value, no matter whether the species is range-restricted and rare or widespread and common. In addition, the numbers of species in different compartments, e.g., rivers or lakes, or coastal regions or oceans, and within different taxa, e.g., arthropods, fish and mammals differ. Therefore, aggregation across different species groups requires a weighting method: Some options for aggregation have been proposed based on equal weight per species or taxonomic groups or according to species' vulnerability (Verones et al. 2015). These estimation methods eventually lead back to a final expression of damage as PDF, a fraction of species, weighted over affected compartments and species groups. We recommend further research into potential species weighting options before reaching consensus. In addition, comparison of impacts among different compartments (e.g., terrestrial, freshwater and marine) has the additional difficulty of whether it is appropriate to compare and/or sum impacts across different spatial units/ecosystems.

Functional roles of individual species and functional diversity differ between ecosystems and taxonomic groups, and lead to non-linear effects of species loss on reduced ecosystem quality (Valiente-Banuet et al. 2015). The complexity of measuring functional diversity and the potential to include aspects of functional diversity in LCIA modelling is discussed below in the section "Operationalization of ecosystem-function-related metrics."

4.1.4 Enabling comparison of dissimilar metrics: conversion options

Similarity between impact metrics may be in kind, e.g., a PDF, or context, e.g., the same spatial and temporal scale. To obtain comparable impact metrics, there is a need to first consider dissimilarity, and, second, to apply an appropriate conversion approach. Conversion approaches are particularly required in cases of dissimilarity pertaining to impacts at different spatial scales (e.g., local vs. global), measurement type (i.e., relative vs. absolute), and levels of effect (e.g., PAF vs. PDF).

Conversion between spatio-temporal scales Converting impact metrics pertaining to a sub-global context to an indicator of permanent impacts at a global scale requires care with respect to temporary impacts. One option is to produce two impact indicators, i.e., to represent potential temporary and permanent impacts separately. The result would still be simple enough to allow interpretation and comparison. However, difficulties may arise when comparing a large temporary impact with a small permanent impact. This leads to the question of impact weighting. A permanent impact without extinction would still increase the likelihood of an extinction through increased threat level, decreased geographic range size, and increased geographic isolation and fragmentation. We recommend that potential permanent and temporary impacts, as defined above, should be interpreted separately to support decision-making. Future harmonization of indicators pertaining to potential temporary and permanent impacts requires further research with respect to discounting, time horizons and potential secondary species losses, e.g., Brodie et al. (2014).

Conversion between measurement types Conversion between relative metrics that express impacts as a fraction of potentially impacted species and absolute metrics that express the potential number of species (or other ecosystem characteristic) lost has already been implemented in the endpoint method ReCiPe (Huijbregts et al. 2016). ReCiPe uses global average species density data to convert from PDF to species-equivalents. This ignores the large amount of variation in species density globally. Optimally, the conversion between relative and absolute metrics can be refined when needed, and this could be achieved using species density data at the native spatial scale of the original impact metric. In cases of empirical data (estimating species density) limiting the potential spatial resolution of conversion factors, the best available spatial resolution should be adopted.

Conversion between levels of effect Some LCIA models yield impact metrics that are representative of potential damage not directly related to species loss. For example, a PAF can theoretically be derived from an SSD-model that is based on No-Observed Effect Concentrations in ecotoxicity; such a PAF-NOEC identifies the fraction of species experiencing some kind of harm, but does not directly relate to species loss. Converting from such a no effect-based PAF to an effect-based PDF (representing lost species) does not seem straightforward, as the step implies the extrapolation from sub-lethally affected species (a fraction affected or possibly not affected if based on NOECs) to "ecological damage" (a fraction affected at the level of at least extirpation). However, conversion from "affected" to "disappeared" implies considering a substantial body of ecological knowledge. In addition, conversion factors are dependent on stressor type, impact pathway and the affected ecosystem. Various methods have been proposed. In the recent past, a fixed factor for this conversion has been suggested as being 1:1 (Goedkoop et al. 2009). Jolliet et al. (2003) proposed dividing estimated PAFs by a factor of two, and Goedkoop and Spriensma (2001) suggested dividing by a factor of 10. Recent research has shed new light on this step. Eco-epidemiological analyses, using large-scale monitoring data to find habitat-response relationships for a stressor of interest (like the SDMs), have been shown to yield useful insights with respect to relating predicted PAF to PDF conversion factors. In a study on sediment contamination, Posthuma and de Zwart (2012) demonstrated a nearly linear association between predicted impacts (a PAF derived from an SSD-model constructed from EC50-data) and disappeared species ($PAF_{EC50} \cong$ PDF). Although this is fully in line with expectations, it is reasonable to check whether a relationship that holds regarding "conclusion *in kind*" within an impact category (higher predicted PAF implies higher PDF) also implies similarity of "conclusion *in magnitude*" across impact categories, as the basis for various PDF-models may be different. Ground-truthing and cross-comparability of PAF and PDF metrics can be reached via field-based, eco-epidemiological research, as mentioned above. In such studies, all impact pathways can be studied for one or more representative regions and species groups, in order to generate a uniform basis for all PAF- and PDF outputs across impact pathways, e.g., Goussen et al. (2016).

Given the complexity of converting from different effects on ecosystems to species loss, we recommend that whenever a PAF is available, model developers should also provide a factor they consider appropriate for converting to a respective PDF, based on best available evidence.

4.2 Potential for furthering ecosystem quality metrics

To address damage to ecosystem quality more comprehensively, species-based metrics and function-based metrics could be further developed. This is similar to the approach taken within the Japanese LCA method LIME (life-cycle impact assessment method based on endpoint modelling; Itsubo and Inaba (2003)).

4.2.1 Further development of species-richness-related metrics

To consider vulnerability more fully, as mentioned in Section 4, and more specifically the potential for recovery of extirpated species, LCIA models need to consider biogeographical concepts, such as spatial distributions of species in meta-populations (average values for spatial compartments because individual species are not identified in traditional methods), along with standard exposure and sensitivity concepts. Species-specific threat levels should be considered in the light of species occurrence and distribution at local and global scales, as well as the differing ability that species have to respond to environmental stress, e.g., dispersal capacity. These concepts have not yet been sufficiently addressed for applicability (conceptually as well as practically) in LCA studies. First steps towards inclusion of these concepts have been taken for the impact categories addressing land use, e.g., Michelsen (2008) and Chaudhary et al. (2015), and water use, e.g., Verones et al. (2013) and Tendall et al. (2014). Further research is needed to fully develop the concept of vulnerability in LCIA, especially when the LCIA would be supporting decision processes for non-global, regional purposes. In this respect, vulnerability metrics would allow for the differentiation of local and global effects of species loss, ensuring increased comparability between environmental impacts at the same spatial scale.

4.2.2 Operationalization of ecosystem function-related metrics

Functional diversity (FD) is an ecosystem attribute that considers the functional attributes (or traits) of organisms to predict the mechanistic relationship between species and their ecosystem (Petchey and Gaston 2006). These traits are numerous and may be morphological, structural, phenological or even behavioral characteristics of organisms (Díaz et al. 2013). In comparison to taxonomic indicators such as species richness, FD is able to reflect responses to changes in the ecosystem function more accurately and would be a more appropriate link to impacts on ecosystem services than species richness (de Bello et al. 2010). However, some challenges exist in the development of operational models for LCA using functional measures, especially at a global scale. Firstly, FD metrics may initially be more data demanding than existing species-related metrics, as trait data on each species present in the ecosystem needs to be gathered (Maia De Souza et al. 2013). Data sets to this end are, however, available (see e.g., http://traitnet.ecoinformatics.org/), and this makes the approach conceptually feasible. Secondly, there are diverse ways to measure functional diversity, such as continuous, e.g., specific leaf area, and categorical, e.g., does or does not fix nitrogen, measures (Petchey and Gaston 2006). Currently, there is no consensus on a single method to quantify functional diversity (Maire et al. 2015; Mouchet et al. 2010). Additionally, the choice of what types of traits and which traits to use in modelling influence the results of the biodiversity loss assessment.

To date, a sole model has been proposed to explicitly address functional diversity in LCA, mainly associated with land use impacts on ecosystem quality (Maia De Souza et al. 2013). In ecology, many studies use functional diversity metrics to assess changes in ecosystem function due to anthropogenic pressures. For example, de Bello et al. (2010) explored the links between functional traits and ecosystem services in different ecosystems and across various trophic levels. Similarly, Brown and Milner (2012) investigated the relation between functional diversity and species richness in a changing environment, i.e., areas with glacial retreat, touching upon issues such as functional redundancy. Insight from ecological research on different ecosystems could serve as a basis for including functional diversity as an ecosystem quality indicator in LCA. Moreover, this transdisciplinary exercise could provide a better understanding of the relation between the rate of species loss and the variation in ecological functions (Valiente-Banuet et al. 2015) in LCA. Ongoing work in the UNEP-SETAC life cycle initiative task force on ecosystem quality aims to further investigate the feasibility of cross-cutting functional diversity metrics in LCIA.

5 The way forward

In their review of indicators of damage to ecosystem quality in LCA, Curran et al. (2011) recommended having multiple impact factors to better reflect the complexity of ecosystems, and applying regionally specific models to generate output with local relevance when needed. This expansion of indicators may have an unintended consequence of making LCA results less comparable and more difficult to interpret. Applying a two-part ecosystem quality AoP to global LCA frameworks, as done in LIME for Japan (Itsubo and Inaba 2012), may improve both coverage of ecosystem complexity and clarity of interpretation. Such a split ecosystem quality AoP would focus on the intrinsic value of biodiversity conservation using two distinct damage-level metrics: one based on species loss and the other on functional diversity, thereby covering both compositional and functional aspects of ecosystems. Comparability of impact-category specific contributions to the overall potential impact within an AoP would be facilitated, and having two complementary ecosystem quality damage-level indicators would allow for a more holistic assessment of potential damage to ecosystem quality without overly complicating interpretation.

Cross-cutting developments should focus on vulnerability-adjusted impact indicators applicable at a global scale. More specifically, given that vulnerability is a function of exposure and sensitivity modified by a recovery capacity and various other traits, and that damage-level indicators already incorporate exposure and sensitivity, further development of impact modelling approaches requires refinement or inclusion of species/ecosystem recoverability. Recoverability data comes in a variety of forms, including species traits (IUCN data), species richness (at ecosystem level), and ecosystem scarcity (ecosystem area relative to potential natural).

Ecosystems have characteristics that are not solely predicted by the sensitivity of their species, but by a vast number of characteristics related to, for example, biogeography and trait-related aspects, such as reproductive strategy. The usefulness of impact indicators based on species loss, therefore, could be improved by recognition of aspects of ecosystem complexity. While we acknowledge that this likely leads to an increase in model complexity and therefore greater uncertainty (Van Zelm and Huijbregts 2013), this uncertainty is likely to be reducible through further refinement of the indicators. Furthermore, while we advocate cross-cutting development towards a harmonized species-loss-based impact indicator, ascertaining the appropriateness of such an indicator for representing damage to ecosystem quality requires further discussion within the ecology research community (e.g., Mace et al. (2014)). Verones et al. (2015) suggested several options for harmonizing results between a land and a water consumption impact category. However, further work is required to ensure that compatibility is extended to all ecosystem quality-related impact categories. In addition, additional discussion about which harmonization approach should consistently be used (e.g., based on species richness, based on vulnerability or based on number of taxonomic groups, etc.) are needed.

6 Conclusions and recommendations

To date, development efforts in LCIA have delivered a variety of valuable metrics for impacts on species and ecosystems.

Because LCIA covers multiple impact categories, there is a motive for harmonization. Our overview of the current potential for a harmonization effort shows that there is scope to apply and improve on the PDF-type approach, with an emphasis on harmonization of modelling context and clear reporting of this context. Further LCIA model development should aim towards the following:

1. improved modelling of ecosystem complexity within a species-loss based indicator, and
2. broadened coverage of aspects of ecosystem damage to address damage to ecosystem function.

However, we would like to stress that these are recommendations based on current LCIA research. We wish—under no circumstances—to stifle research for further metrics and impact categories that may prove relevant and useful in future.

Compliance with ethical standards

Disclaimer This paper has been reviewed in accordance with Agency policy and approved for publication. The views expressed in this article are those of the authors and do not necessarily represent the views or policies of the U.S. Environmental Protection Agency.

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