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status quo, harmonization, and suggestions for the way forward

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Abstract (max 400 words)

Purpose:

Life cycle impact assessment (LCIA) results are used to assess potential environmental impacts of different products and services. The usefulness of LCIA results is dependent on the comparability and environmental relevance of the impact indicators used.

As part of the UNEP-SETAC Life Cycle Initiative flagship project that aims to harmonize indicators for environmental impacts, we highlight the necessity for improving comparability and environmental relevance of damage-level metrics within the ecosystem quality area of protection.

Methods:

We analyze current ecosystem quality metrics and provide suggestions to the LCIA research community for achieving near- and long-term progress towards comparable and more environmentally relevant metrics addressing ecosystem quality.

Results and discussion:

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

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We suggest that near-term LCIA development should tend towards species-assemblage-related impact metrics. Impact indicators, which result from a range of modelling approaches that differ *inter alia* according to spatial and temporal scale, taxonomic coverage and whether it produces a relative or absolute measure of loss, should be convertible to facilitate their final expression in a single indicator unit. This would improve comparability of damage-level indicators.

In the longer term, environmental relevance of species-assemblage-related metrics should be improved through appropriate inclusion of aspects capturing greater ecosystem complexity within models. Furthermore, to allow for a broader inclusion of ecosystem quality perspectives, the development of an additional ecosystem function-related indicator is recommended, as well as one based on ecosystem services. These distinct indicators would give a broader coverage of ecosystem attributes and valuation viewpoints.

**Conclusion and recommendations:**

We call for the LCIA research community to make progress towards enabling harmonization of damage level indicators within the ecosystem quality area of protection and, in the longer term, improve the environmental relevance of impact indicators.

**Keywords**

LCIA, harmonization, UNEP-SETAC, endpoint, damage-level, species, functions, ecosystem services

**1. Introduction**

Human activities on earth have resulted in substantial pressures on the natural environment, such that the term Anthropocene was recently coined as a Geological Epoch in which human impacts are considered to yield observable earth-scale system impacts (Crutzen and Steffen 2003; Lewis and Maslin 2015; Waters et al. 2016), which include substantial changes occurring in ecosystems and biological diversity (e.g. Barnosky et al. 2011; Waters et al. 2016). Concurrently, the dependence of human life on the services of ecosystems is increasingly recognized (Millennium Ecosystem Assessment 2003), and policies to halt, counter and report on adverse impacts have been adopted at the global scale in the Convention on Biological Diversity (CBD, [http://www.cbd.int/](http://www.cbd.int/); Secretariat of the Convention on Biological Diversity 2014).

Life Cycle Assessment (LCA) was developed to provide decision support in the selection of products or services with lower environmental impacts compared on a functional basis (Hellweg and Mila I Canals 2014). The LCA
method characterizes inventories of emissions and resources used over the life cycle of modeled products or services in terms of potential impacts on humans, ecosystems and natural resources within a suite of environmentally relevant impact categories. With respect to ecosystems, the LCA framework directly relates to current global concerns for biodiversity loss, as reflected by one of the three areas of protection (AoP) that are currently considered in LCA, namely the ‘ecosystem quality’ AoP. Ideally, LCA encompasses models and indicators that directly and unequivocally provide valid insights into the expected impact on ecosystem quality of alternative product- or service life cycles. However, as Curran et al. (2011) highlight, this is not yet the case. This problem relates to conceptual, technical and data aspects that define ecosystem quality impact indicators (Curran et al. 2011; McGill et al. 2015) and the use of these indicators in the context of the developing LCA framework (Jolliet et al. 2014; Verones et al. In Preparation). Frequently, ecosystem quality impacts quantify impacts via species-assemblage level metrics, such as the Potentially Disappeared Fraction of species (PDF).

Curran et al. (2011) provide the most recent cross-cutting review on encompassing biodiversity in life cycle impact assessment (LCIA) impact metrics (more recent impact-category specific reviews exist, e.g. Curran et al. 2016) and derived two overarching research recommendations. The first addresses conceptual shortcomings, with specific emphasis on increasing spatial detail within LCIA models. Spatial differentiation is an increasingly recognized need in LCA, as impacts from human activities relate to specific spatial scales, e.g. impacts from water use relate to local water availability, extraction and mass flows (e.g., Gerten et al. 2013; Mutel et al. 2012; Pastor et al. 2014). The second, specific to species- or species-assemblage-based indicators, advocates expanding on the use of globally available biodiversity data to improve current LCIA characterization modelling. Greater incorporation of biodiversity data would alleviate a limitation of some current LCIA ecosystem quality metrics i.e. impact indicators that imply completeness (such as PDF), even though the LCIA models are trained on data from a single taxonomic group, or geographic area.

Advancing LCIA, in the context of global biodiversity threats (attributed to human activities) and the status and proposed development of the LCIA framework to support minimum-impact choices in the anthroposphere, constitutes a challenge for LCIA model developers. More specifically, the challenge is to improve the assessment of potential impacts on ecosystem quality in LCIA, whilst the LCA framework itself is refined, and advances in measuring actual impacts on ecosystem quality (outside LCA) are made. This challenge has been adopted by the international LCA community in the UNEP-SETAC flagship project “Global Guidance on Environmental Life
Cycle Impact Assessment Indicators” (Frischknecht et al. 2016; Jolliet et al. 2014), which encompasses activities to improve modelling of biodiversity impacts in the context of LCA.

In this paper, we try to balance the use of current scientific knowledge with the practical needs of LCA-based, transparent, reproducible, meaningful, and operational decision-support. While there is the need to find consensus on specific models for individual impact categories (e.g. land use and water use), whenever scientific knowledge alone is not able to address all relevant problems and questions at hand, it is also important that harmonization and comparability across impact categories is ensured. For that purpose, we aim to highlight the need for improving the comparability and environmental relevance of damage-level impact category metrics within the AoP ‘ecosystem quality’. Herein, we explore the current state of ecosystem quality within the LCA framework, highlight limitations of ecosystem quality-related metrics (building on the foundation of Curran et al. (2011)), and posit pragmatic short-term options to improve the assessment of impacts on ecosystem quality in LCA, as well as outline longer-term improvement options.

2. Ecosystem Quality within the Life Cycle 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) to which distinct stressors (emissions and used resources) are contributing. These stressors initiate one or more impact pathways, crossing different environmental compartments. Impact magnitude and type eventually becomes apparent in the format of predicted potential impacts on one or more ecosystems. Potential impacts are indicated at midpoint level (environmental problems, e.g. change in concentration of phosphorus, a eutrophying substance, in the environment) or damage level (previously referred to as endpoint level; ecosystem damage, i.e. the consequence of environmental problems on ecosystem quality).

Over recent years, substantial development efforts in LCIA have aimed to reach an all-embracing coverage of various measures of ecosystem damage. There has been an increasing use of different indicators of damage to ecosystem quality, most prominently the PDF (Goedkoop and Spriensma 2001), 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 or provide an aggregated indicator of potential overall ecosystem quality damage, indicators at damage level need to be comparable and able to accommodate the multitude of different stressors that operate via different impact
pathways. In agriculture, for example, the use of pesticides and fertilizers, and the use of land and irrigation water will all affect the terrestrial and aquatic ecosystems, but via different impact pathways (such as ecotoxicity, eutrophication, and habitat changes). Whilst impact types and magnitudes may be determined according to different processes in time and place, comparable damage indicators would facilitate decision-making processes.

Defining ‘ecosystem quality’, however, as a self-contained and comprehensive area of protection requires a significant effort (Paetzold et al. 2010) as the concept is multifaceted (as highlighted by Curran et al. (2011)), encompassing 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). In LCIA, ecosystem quality refers to the condition of an ecosystem relative to a reference state. The condition of an ecosystem is defined by the interrelated status of its components (biotic and abiotic), processes (functions) and structure. These ecosystem attributes interact in complex and non-linear ways. The predicted ecosystem damage (i.e. impact indicators at damage level) need to consider these matters where possible (model and data availability) and needed (for the decision quality). The reference state can be a past, present or potential future situation.

However, defining reference states (or minimally disturbed sites) is a complex matter (Stoddard et al. 2006), and a fully adopted and harmonized reference state definition across impact pathways is currently lacking in LCIA. This is currently a topic for discussion in the UNEP-SETAC flagship project, but outside the scope of this paper.

The complexity and dynamics of ecosystems need to be modelled parsimoniously in LCIA, which essentially means that an increase in model complexity should come with an increase in relevance for the decision at hand. This task is even more imperative as impact assessment results are to be interpretable by decision makers and LCA practitioners. Deriving metrics that encompass 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).

The general LCIA framework is being updated (Verones et al. In Preparation). Updates include a widening in the scope of damage types included and a call for consistency across spatial scales. The update to the structure of the LCIA framework (Verones et al. In Preparation) proposes to introduce ecosystem services as an additional AoP separate from ecosystem quality. Ecosystem services focus on the instrumental value of ecosystems for humans (i.e. valued for the services provided to humans), thus broadening the information on the natural environment that is included in the assessments. The ecosystem quality AoP will continue to focus on the intrinsic value of
ecosystems (i.e. valued as an entity in itself). Recommendations made in this paper are consistent with current updates to the LCIA framework and LCIA structure.

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, reduced ecosystem biomass, and increased risk of species extinctions, are all formats of metrics to characterize damage to ecosystem quality. Recently, a summary overview of possible metric types yielded at least 15 metric types that can be considered (McGill et al. 2015). For LCIA, it is key to select metrics that meaningfully summarize impacts within impact categories, and allow for aggregation or comparison across impact categories. Furthermore, the choice and construction of a damage-level metric influences the sensitivity of the impact scale, and therefore interpretation, of impact scores generated in an LCIA. The existence of non-linear ecological responses to stressors are challenging in the context of current LCIA models, which are based on the assumption of linear dose-response. This is a preferred principle as LCAs often consider comparative emissions of typically small magnitudes of individual stressors, whereby the uncertainty generated from current approaches to linearize non-linear stressor-response relationships (e.g. marginal, average; Huijbregts et al. (2011)) is accepted. Overall, the choice of impact metrics that are sensitive enough and meaningful, is complex.

Currently, many species-impact related metrics for ecosystem quality (for example within the ecotoxicity, freshwater eutrophication and terrestrial acidification impact categories) exist. They 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) at a certain level of adverse effects, for example, effect concentration-50 (EC50): the concentration level 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). Whilst originally used for quantifying 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). Distributions of sensitivities across species can be derived from laboratory or field data, especially when a (bio)monitoring data set contains a stressor variable of interest which is sufficiently independent from other stressors to derive Species Distributions Models (SDMs, for specific species; Struijs et al. (2011)) and associated stacked SDMs (to represent a biodiversity metric; Schipper et al. (2014)). Other impact categories, such
as biodiversity loss as a result of land use, mainly make use of empirical species-area relationships (SAR, e.g. Chaudhary et al. (2015)), but also habitat-suitability models (e.g. Geyer et al. 2010) and meta-analysis (Elshout et al. 2014). Similarly, species-discharge relationships are used to model potential impacts of water consumption on fish species in rivers (Hanafiah 2011; Tendall et al. 2014).

Impact indicators therefore usually pertain to modelled parameters (typically average values or proxies derived from statistical models) rather than to detailed site-level knowledge of ecological effect-phenomena (types and magnitudes) in affected ecosystems, although SDM-based proxies are derived from field phenomena. Impact indicators thus often apply to ecosystems generalized at the resolution of modeled parameters, life cycle inventory data and the spatial resolution of characterization factors. By virtue of their derivation protocol, the comparability of impact scores between impact categories is determined by, i) the ecosystem attribute represented by the impact indicator (species composition, or other structural or functional ecosystem impact metric) and, ii) modelling approach considerations (such as temporal and spatial scale, or laboratory- versus field-based). That is, dissimilar options and choices in the derivation of impact models, the current suite of CFs covering the range of ecosystem quality impact categories has not, so far, been derived in the context of a single-theory, single-metric framework.

Despite the development of relevant and meaningful metrics over the last decade, there are limitations. Curran et al. (2011) found that most indicators concentrate on species richness only, thereby neglecting characterization of potential functional and/or structural ecosystem damage. However, even within the suite of damage-level metrics focused on species richness there is a multitude of different units and modelling approaches available that may be optimized further in the context of harmonization to support comparability and 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 apparently uniform PDF-impact metric is an example of how hidden differences in the underlying modelling approach can imply a hidden non-comparability of impact metrics, even though they are expressed using the ‘same’ unit Considering only variation in temporal and spatial considerations for simplicity, the seemingly uniform PDF impact indicator could pertain to four different spatio-temporal scales (Table 2). A stressor may cause species extirpation (i.e. loss of a species within a spatially defined, ‘local’, compartment, on a temporary or permanent basis depending on recovery following cessation of the stressor) or extinction (i.e. loss of a species globally) or sub-extinction effects at a global scale.
Overall, even impact indicators such as the seemingly uniform PDF metric are not intrinsically coherent/transparent, as they seem to cover something uniform (i.e. the potentially disappeared fraction of species), while their meaning is different because the PDFs were ‘trained’ on a specific timeframe, regional data or selected taxonomic groups only. Thus, making comparisons between, or aggregating, impact indicators across impact categories should be based on checked (sufficient) similarity of the underlying models and data.

4. Towards harmonized and more environmentally relevant damage metrics

We provide suggestions to LCIA model developers with a view of improving LCA as a decision-making method. That is, we focus on the usability and usefulness of impact indicators at damage level for LCA purposes with an awareness of a trade-off between the feasibility and complexity of fully capturing the concept of biodiversity and maintaining (and improving) the interpretability of LCA results.

Based on the diversity and structure of existing ecosystem quality metrics, harmonization is not only needed, but also possible. Taking a general approach, harmonization of damage-level metrics within the ecosystem quality AoP requires convergence on two condition: i) choice of ecosystem attribute (i.e. harmonized units); and, ii) modeled environmental complexity (i.e. the context to which these units apply).

In the following sections, we suggest potential near- and longer-term developments towards improving the usefulness of impact indicators for LCA practitioners and decision makers.

Potential near-term developments

Consolidation towards species-related ecosystem metrics

Species richness data is more readily available, with respect to application in LCIA, than for other ecosystem attributes. In addition, species-related metrics (such as species loss, reflected e.g. in PDF, PAF, EINES, species.yr) are often more understandable to non-specialists than, for example, structural or functional diversity metrics, such as net primary production. In addition, there is an emerging consensus that biodiversity underpins ecosystem functioning (Cardinale et al. 2012). This is also reflected in the order in which ecosystem damage happens: most commonly, initial ecosystem damage occurs according to the sensitivity of exposed species to a stressor. Loss of ecosystem function then occurs according to the functionality of affected species within the affected ecosystem. Due to functional redundancy, not all species loss results in loss of ecosystem function. However, protection of species, regardless of functionality, also protects ecosystem functions and structures. It seems logical, therefore,
that near-term LCIA model development should move towards harmonizing on a common species-based damage-level unit.

**Improved recognition of spatial and temporal scale issues**

In reality, 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, with reference to different life stages of affected species, and exposure time. The ecological response over time then depends on the sensitivity, adaptive capacity and recoverability of the single species or communities, which can be different depending on their spatial distribution. Impact scores 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).

Recoverability is an aspect of vulnerability. Addition of a recoverability parameter would generate a vulnerability-adjusted indicator, i.e. the metric would 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 biodiversity damage. In the near-term, we recommend incorporation of recoverability parameters across a wider range of impact categories. Options for recoverability parameters 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).

**Taxonomic coverage**

The choice or availability of species and taxa sensitivity data is currently different for each stressor for which a sensitivity curve is constructed (including both SSDs and SARs). The use of “species” in LCA is therefore specific to individual impact pathways (e.g. (Azevedo et al. 2013a; Azevedo et al. 2013b; Verones et al. 2013)). Although modelling is becoming more inclusive, accounting for the whole number of species that populate an ecosystem is currently 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 value, no matter whether the species is range-restricted and rare or widespread and common. In reality, 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. Furthermore, aggregation across
different species groups requires a weighting method. This has been done based on richness, vulnerability, or quality of species, to mention some of many options (Verones et al. 2015). From the starting point of the PDF, such estimation methods eventually lead back to a final expression of damage as PDF, a fraction of species, weighted over affected compartments and species groups.

Conversion between metrics

Modelling impact pathways may or may not result in similar, method-specific impact metrics, and similarity may be in kind (e.g. PDF) or contents (e.g. same spatial and temporal meaning). Prior to obtaining comparable impact indicators, there is thus a need to, firstly, consider metrics that are, in fact, dissimilar, and, secondly, to convert these metrics. Converting between units allows for comparison of relative importance between impact categories with dissimilar impact units. Conversion methods could be developed to aid comparison between impact indicators representing impacts at different spatial scales (e.g. local vs. global), levels of effect (e.g. PAF vs. PDF) and measurement type (i.e. relative vs. absolute).

Converting impact indicators that represent impacts within a spatial compartment to permanent impacts at a global scale requires care with respect to temporary impacts. If all impact categories had two impact indicators, i.e. temporary and permanent, the result would still be simple enough to allow 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, increased geographic isolation and fragmentation.

Conversion between relative metrics (that express impacts as a fraction of potentially impacted species, e.g. PDF) and absolute metrics (the actual number of species (or other ecosystem characteristic) lost, e.g. species-equivalents and global species-equivalents) is already implemented in the endpoint method ReCiPe (see Goedkoop et al. 2009). However, this method uses global average species density data, which ignores the large amount of variation that exists around such an average. Such data would be needed to perform the conversion and the approach could thus be improved by using such data for scales lower than the global scale. A more realistic estimate of species loss per year could be generated using spatially differentiated effect and species density factors, which would represent spatial compartments that are more ecologically homogenous than global scale compartments. The potential spatial resolution, however, will remain to be limited by the constraints imposed by the availability of empirical data for estimating the number of existing species per area or compartment. Furthermore, comparison of impacts between
different compartments (e.g. terrestrial, freshwater and marine) will encounter difficulties with respect to whether it is appropriate to compare and / or sum impacts across different spatial units / ecosystems and need further research.

Some LCIA models produce impact indicators that may not yet be representative of the potential or actual damage on ecosystems. A PAF based on No-observed Effect Concentrations in ecotoxicity, for example, identifies the fraction of species experiencing some kind of harm, but not species loss. Converting from such a PAF to a PDF (representing lost species) seems straightforward, as the step implies the extrapolation from ‘statistical damage’ (a fraction affected at a sub-lethal level based on statistical modelling) 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 and conversion factors would be dependent on stressor type, impact pathway and affected ecosystem. Various methods have been proposed. In the recent past, a fixed factor for this step 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. The differences between the three proposed factors relate, in part, to the type of sensitivity data used in modelling, or the field-studies in which damage was quantified for different species groups.

When the relationship between the statistically predicted PAF, and its damage-related derivative the PDF, and the real impacted fraction of species in affected ecosystems is known, PAF or PDF can be extrapolated to ecosystem quality damage. Eco-epidemiological analyses, i.e. exploring large-scale monitoring data to find habitat-response relationships for any stressor of interest, have been shown to yield useful insights with respect to relating predicted PAF to PDF conversion factors. In a study on sediment contamination, Posthuma et al. (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} \approx PDF). Although this is fully in line with expectations, it is again logical 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.

Potential longer-term developments
In the longer-term, to cover ecosystem attributes more comprehensively, species-based metrics could be further developed, and additional metrics leading to an ecosystem-function-related indicator and an ecosystem-service-related indicator could be worked towards. 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).

Further development of species-based metrics

In the longer term, to consider vulnerability more fully, 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), next to standard exposure and sensitivity concepts. This is a novel concept in LCA and few approaches already include this. Further research is needed to fully include the concept of vulnerability in LCIA, especially when the LCIA would be serving decision processes for non-global, but regional purposes. This discussion will be part of the following phase of the flagship project.

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 functioning more accurately and would be more appropriate for linking to impacts on ecosystem services than species diversity (de Bello et al. 2010). However, some challenges exist in the development of globally operational models for LCA using functional measures. Firstly, FD metrics may initially be more data demanding than existing species-related metrics, as trait data on each species present in the ecosystem need 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 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 (Mouchet et al. 2010). Finally, the choice of what types of traits and which traits to use in modelling may influence the results of the biodiversity loss assessment.

Operationalization of ecosystem services metrics
Ecosystem services (ES) are those outputs of ecosystems that benefit society. The outputs may come in the form of material flows (e.g., crop and fisheries yields), or in terms of regulation or mediation of stresses or changes (e.g., disease regulation and organic matter decomposition and nutrient cycling). Services described by the Millennium Ecosystem Assessment (2005), and set out in the Common International Classification of Ecosystem Services (CICES) framework (Haines-Young and Potschin 2013), are considered essential to sustaining life, including human health and well-being. The economic terms “use” and “non-use value” are also generally accepted and preferred to distinctions, such as indirect utility that can be a matter of perspective.

A variety of models exist to quantify ecosystem services, including economic valuation, although some are still under development (e.g., InVEST, LUCI, Co$ting nature, ARIES, etc.; see Bagstad et al. 2013). The ongoing challenge is to go beyond qualitative association of biodiversity and ecosystem function to provide predictive, quantitative relationships between the two. InVEST, for example, uses inputs such as land use and land cover, combined with empirical biophysical models to forecast system dynamics under alternative management scenarios, and includes economic methods for valuation of some ES. Example ES include timber harvest, carbon sequestration, water yield, coastal protection, offshore wind energy, fishery production, and crop production.

Double counting, especially when bundling multiple services, can frequently be a problematic issue. Using final ecosystem services and a beneficiary perspective are considered best practices to avoid this. Although ES models admittedly do not capture all final goods and services, in the context of decision making, they provide useful information. This is consistent with LCA, which also acknowledges it does not capture all impacts of products and services (e.g. Bare and Gloria 2008).

There will be challenges with linking these models to LCA inputs (see the recent review by Othoniel et al. 2016). Typical ES models use globally available datasets at a 1km² resolution for quantities such as depth of soil root zone, vegetated cover type, and elevation but do not utilize contaminant emissions and exposure concentrations. There is the potential to link LCI emissions of ecotoxicologically relevant substances to changes in the quality of fisheries yield. In addition, emissions of acidifying substances to the atmosphere could be linked to potentially affected crop yield and forest carbon storage, but these stressors are not yet captured in ES models. Alternative ES modelling approaches that include toxins in addition to climate change and land use change drivers are fewer (e.g. Johnston et al. 2011).
There will also be challenges with linking ES model outputs to LCA. Typical ES outputs include total water yield for a catchment, crop yield, edge of field sediment retention, etc., and these outputs can also be linked to economic value provided (e.g., tourism revenue and commercial fishery catch) or avoided costs of engineered systems (e.g., drinking water treatment). These costs could be weighted to create a single ES impact category score. An alternative approach for some metrics, such as water purification, would be to consider the avoided engineered systems (such as a water treatment plant) provided by the ecosystem services. However, this approach presents the challenge of requiring significant extra work (e.g., doing an LCA of the avoided system), and there are not well-defined engineered systems to cover all ecosystem services (e.g., pollination or climate regulation).

5. The way forward

In their review of biodiversity indicators in LCA, Curran et al. (2011) recommended having multiple impact factors to better reflect the complexity of ecosystems, and (when needed) apply regionally specific models to generate output with local relevance. This may have an unintended consequence of making LCA results less comparable and more difficult to interpret, when used in too much detail. Applying a ‘split’ ecosystem quality area of protection, as done in LIME (Itsubo and Inaba 2012), to global LCA frameworks may improve both coverage of ecosystem complexity and clarity of interpretation. The two parts of a split ecosystem-quality area of protection 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. Potential damage to the instrumental value of ecosystems would be captured by a separate, third, AoP incorporating ecosystem services. These three discrete ecosystem quality related damage-level impact categories would cover intrinsic and instrumental value as well as 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 three diverse ecosystem-related damage-level indicators would allow for a more holistic assessment of potential ecosystem damage whilst at the same time being few enough to allow for simple interpretation.

Future development in the near-term should focus on the intrinsic value of species, particularly vulnerability-adjusted impact indicators applicable at a global scale. 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 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 species-loss-based impact indicators, therefore, could be improved by incorporating greater ecosystem complexity in the development of characterization factors. Whilst we acknowledge that an increase in model complexity may generate greater uncertainty, this uncertainty is likely to be reducible through further development and impact indicators would ultimately be more representative of real-world impacts. Furthermore, whilst we advocate near-term development towards a harmonized species-loss-based impact indicator, ascertaining the appropriateness of such an indicator for representing biodiversity requires further discussion with the ecology research community (e.g. Mace et al. 2014). In the longer-term, however, complementary, partially overlapping, damage-level indicators pertaining to ecosystem functions and services would broaden coverage of ecosystem attributes and minimize potential misrepresentation of impacts on ecosystem quality.

Eventually, LCA needs operational models to quantify ecological impacts for a suite of product life cycle stressors. It is evident that the current operational approaches require further theoretical improvements and a sufficient amount of high quality data. We believe that there is scope to develop improved and better applicable methods in assessment approaches regarding ecosystem quality, focusing on (variants of the concept of) modelling Species Sensitivity Distributions and field-based approaches such as Species Distribution Models with the addition of a recoverability parameter.

6. Conclusion and recommendations

To date, development efforts in LCIA has delivered valuable metrics for impacts on species, species assemblages and ecosystems of different kinds so far. Due to the intrinsic needs of LCIA to allow for stressor-midpoint-damage modelling for multiple impact categories, there is a motive for harmonisation. The harmonisation effort shows that there is, in the shorter term, scope to apply and improve on the PDF-type approach, with an emphasis on harmonisation of modelling context and clear reporting of this context. In the longer term, LCIA model development should improved capture of ecosystem complexity within a species-loss based indicator and could broaden coverage of aspects of ecosystem damage to cover damage to ecosystem functioning and the supply of ecosystem services.
Disclaimer

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.

7. References


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8. Tables

Table 1. Summary of conceptual limitations of LCIA damage modelling in the Ecosystem Quality AoP. Based on (Curran et al. 2011).

<table>
<thead>
<tr>
<th>Modelling (dis)similarity criteria</th>
<th>Description</th>
</tr>
</thead>
<tbody>
<tr>
<td>Ecosystem attribute</td>
<td>Impact indicators reflect damage to either ecosystem composition, function or structure</td>
</tr>
<tr>
<td>Biodiversity scale</td>
<td>Impact indicators apply to either species, species assemblages or ecosystems</td>
</tr>
<tr>
<td>Spatial scale</td>
<td>Impacts are modelled at either a local, regional or global scale</td>
</tr>
<tr>
<td>Temporal scale</td>
<td>Indicators reflect either temporary or permanent biodiversity loss. Recoverability (of impacted species, assemblages or ecosystems) is either modelled or neglected</td>
</tr>
<tr>
<td>Sensitivity measure</td>
<td>Impact modelling concerns species, assemblage or ecosystem responses to a stressor based on either lab-scale testing or field-based observations</td>
</tr>
</tbody>
</table>
**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.

<table>
<thead>
<tr>
<th>Relative or absolute</th>
<th>Potential biodiversity loss is indicated either <em>relative to biodiversity richness</em> <em>(e.g. species richness)</em> or <em>absolute</em> measures.</th>
</tr>
</thead>
<tbody>
<tr>
<td>Valuation approach</td>
<td>Biodiversity loss is typically measured taking an <em>intrinsic valuation approach</em> – <em>instrumental value</em> of biodiversity neglected.</td>
</tr>
</tbody>
</table>

**Table 2.** Spatio-temporal scales of biodiversity impact.

<table>
<thead>
<tr>
<th>Spatial Scale</th>
<th>Time scale</th>
<th>Temporary</th>
<th>Permanent</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Local</strong></td>
<td>Temporary removal of species from a specific area</td>
<td>Permanent removal from a local area – but the species exists elsewhere</td>
<td></td>
</tr>
<tr>
<td><strong>Global</strong></td>
<td>Temporary effects at the global level</td>
<td>Permanent removal from the globe (i.e., extinction)</td>
<td></td>
</tr>
</tbody>
</table>