Assessing biodiversity loss due to land use with Life Cycle Assessment: are we there yet?

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Abstract

Ecosystems are under increasing pressure from human activities, with land use and land-use change at the forefront of the drivers that provoke global and regional biodiversity loss. The first step in addressing the challenge of how to reverse the negative outlook for the coming years starts with measuring environmental loss rates and assigning responsibilities. Pinpointing the global pressures on biodiversity is a task best addressed using holistic models such as Life Cycle Assessment (LCA). LCA is the leading method for calculating cradle-to-grave environmental impacts of products and services; it is actively promoted by many public policies, and integrated as part of environmental information systems within private companies. LCA already deals with the potential biodiversity impacts of land use, but there are significant obstacles to overcome before its models grasp the full reach of the phenomena involved. In this review, we discuss some pressing issues that need to be addressed. LCA mainly introduces biodiversity as an end-point category modeled as a loss in species richness due to the conversion and use of land over time and space. The functional and population effects on biodiversity are mostly absent due to the emphasis on species accumulation with limited geographic and taxonomical reach. Current land-use modeling activities that use biodiversity indicators tend to oversimplify the real dynamics and complexity of the interactions of species among each other and with their habitats. To identify the main areas for improvement, we systematically reviewed LCA studies on land use that had findings related to global change and conservation ecology. We provide suggestions as to how to address some of the issues raised. Our overall objective was to encourage companies to monitor and take concrete steps to address the impacts of land use on biodiversity on a broader geographical scale and along increasingly globalized supply chains.

Keywords: biodiversity indicators, ecological models, global change, land use, life cycle analysis, life cycle impact assessment

Introduction

A large body of science compiled over the past decades shows that biodiversity is directly affected by human activities, very often negatively. This worrisome fact led the recent Rio+20 Summit to shift the focus of the sustainability agenda to biodiversity, which is now the cornerstone of sustainable development (UN, 2012).

According to the Millennium Ecosystem Assessment (MEA, 2005), habitat change brought about by land use and land-use change (LULUC) is one of the five main drivers of terrestrial biodiversity loss, together with climate change, invasive alien species, overexploitation of resources, and pollution. For most ecosystems, habitat change due to LULUC has had the highest impact of all drivers of biodiversity loss over the past century, with most plausible future scenarios predicting that such losses are likely to increase (MEA, 2005). The MEA laid the groundwork for identifying the key drivers – which are mainly anthropogenic – that damage biodiversity and decrease the services it provides. While ecosystem services such as food production are currently well monitored, most other ecosystem services are not (Pereira & Cooper, 2006). The identification of key drivers and monitoring of such services are challenging tasks, but are of paramount importance. The MEA recognized that biodiversity is a multidimensional concept that ‘poses formidable challenges to its measurement’, and that the most common measuring techniques (direct measurement of species richness or genetic diversity) do not capture the whole picture (MEA, 2005). Issues such as...
‘variability, function, quantity, and distribution’ of species are usually not taken into consideration. The concept of biodiversity is, in itself, elusive. Better metrics for biodiversity loss assessment must be interdisciplinary, inclusive (respecting the specificity of biodiversity while maintaining methodological accuracy), comprehensive (including direct and indirect effects), and replicable.

Life Cycle Assessment (LCA) meets these four criteria for good biodiversity metrics. LCA methodology can be used to quantitatively model the potential environmental impacts of a product, process or service over all stages of its life cycle, from raw material acquisition, production and use, to its final disposal (Curran et al., 2011). Data on a product’s life cycle, such as its raw material consumption and emissions, are compiled and used to assess its potential environmental impacts, referred to as its ‘impact categories’. The latter may range from global (e.g. stratospheric ozone depletion, global warming) to regional (e.g. acidification) and local (e.g. aquatic ecotoxicity). In particular, impacts on biodiversity can be assessed within the LCA framework using different impact pathways, for instance habitat conversion and change in species composition. Figure 1 shows an example of various impact pathways that may lead to loss of species diversity and ecosystem services, thereby impacting the overall quality of the ecosystem.

The manufacturing of any product or group of products may have global effects, as increasingly globalized markets allow companies to obtain materials and services from complex networks of suppliers distributed over many regions of the world. LCA covers whole supply chains, comprehensively accounting for all impacts that occur at different steps and in different locations along the life cycle, regardless of the physical location of the particular process. Using LCA, pressures on biodiversity caused by local land use can be traced back to global causes (e.g. shifts in consumption patterns, technological improvements during production). LCA assigns ultimate responsibility for downstream and upstream impacts within production chains to a single unit (usually the final product – a consumer good). Given the importance of LCA today and its integration in policy instruments and in private company’s environmental information systems, an impact category included in the framework will necessarily become a priority issue for the main actors in the supply chain. Decisions taken by final producers can then trickle down the supply chain and be implemented in the field. It is therefore of the utmost importance that accurate measurements of the state of biodiversity be defined in LCA.

Models that aim to assess land-use impacts using LCA have been continuously improved during the past two decades. While many methodologies have been

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**Fig. 1** Impact pathway representing some of the midpoint and endpoint impacts, caused by environmental interventions (land use and land-use change) on biodiversity and related ecosystem services (adapted from Koellner et al., 2013b).
proposed (Müller-Wenk, 1998; Koellner, 2003; Schmidt, 2008a; de Baan et al., 2013b; Souza et al., 2013), the assessment of the impacts of production systems using biodiversity indicator approaches remains a challenge (Geyer et al., 2010a; de Baan et al., 2013a; Koellner et al., 2013b).

There is a clear need to devise consensual approaches. This is the objective of a flagship project being carried out by the UNEP/SETAC Life Cycle Initiative, entitled ‘Environmental Life Cycle Impact Assessment Indicators’, which aims to provide global guidance and build consensus on life cycle impact indicators, such as biodiversity loss. Among the challenges to be addressed are the spatial variability and availability of LULUC and related biological diversity data, such as species richness, abundance, distribution, and functions (Geyer et al., 2010a). A better understanding of the dynamics of ecosystem complexity is also needed to quantify the spatial and temporal variability in changes in biodiversity (Reap et al., 2008; Geyer et al., 2010a).

This review synthesizes how current impact assessment methodologies in LCA take into account the impacts of LULUC on biodiversity, and makes some suggestions to help improve modeling activities and reduce some of the uncertainties involved. For the benefit of readers that are less familiar with LCA, we start with a brief description of the basic methods and key-related concepts, including a review of the current methods and the framework used to assess land-use impacts. We then describe relevant aspects on which we consider further research should be carried out, and propose some insights on issues to be considered. We conclude with indications collected from recent reviews of the topic (Curran et al., 2011; Koellner et al., 2013a), to which we add further insights, and make suggestions as to how to address some of the main areas that require further research.

Impact assessment in LCA

LCA can be used to assess the environmental impacts of systems through part or all the stages of its life cycle. It is an important methodological tool that can help to ensure environmentally sound choices in the context of decision-making based on scientific evidence (EEA, 1997). Its application may range from single products, with a short life cycle, to large-scale policy decision issues.

LCA comprises four main phases: the definition of goal and scope of the study, including the description of the product system, process or service; the collection of input and output data; impact assessment; and the interpretation of the results in terms of damages to the environment. The first phase – goal and scope definition – consists of the description of the aim of the study and all related activities and processes in the life cycle of the product that impact the environment. This is followed by the life cycle inventory (LCI) phase, which involves data collection of all input (raw-materials, energy) and output (waste, emissions) flows during the product’s lifecycle. LCI results are then translated, i.e. classified and characterized, into specific environmental impact categories (e.g. global warming, human toxicity, land use) at the life cycle impact assessment (LCIA) phase (ISO, 2006b). The last phase consists of the interpretation of results, whereby the information obtained is checked and evaluated, and the results are reported clearly (ISO, 2006a).

The potential environmental impact of the product's inventory flows are assessed during the LCIA phase. These potential impacts may be expressed at the midpoint or endpoint level, within a cause–effect chain or environmental mechanism (Fig. 2). Midpoints (the impact categories) lie between the life cycle inventory phase and the endpoints (the damage categories). Due to the use of more accurate environmental data and relevant indicators, and less subjective evaluations of the relative importance of each indicator (Finnveden et al., 2009), midpoint results are well-characterized in terms of physical units, as shown in Fig 2. Endpoints are located beyond midpoints and represent quality changes aggregated in three main areas of protection: human health, ecosystem quality, and natural resources. Endpoints are more concise and easy to understand for decision-making purposes. However, the uncertainty of endpoint models is higher, due to their complexity and the need for reliable data to carry out robust modeling (Bare et al., 2000). Characterization factors, at mid- or endpoint levels (e.g. loss of species richness in a defined area), are the values that translate life cycle inventory data into their damage impacts. They allow the magnitude of potential impacts of each of the substances related to each category to be evaluated (Curran, 1996).

LCA does not assess the changing state of biodiversity or ecosystems, but rather the relative environmental impact of anthropogenic production systems. It links units produced to their marginal/average impacts on the environment. These impacts are assumed to grow linearly with the quantity produced, assuming that the total system is not otherwise affected by the change in production. The total system, or state of biodiversity, is mainly assessed within a static framework when evaluating the marginal/average unit impacts. Ecological interventions and resources were introduced into this static framework on an ad hoc basis. There is no normative or scientific definition of the scale of
impacts to the health or resilience of the ecosystem (unlike concepts such as the ecological footprint).

Conceptual framework for land-use impact assessment in LCA

Although research on the development of biodiversity indicators to assess land-use impacts in LCA has been ongoing for more than 15 years, such indicators are not yet widely applied (Koellner et al., 2013b). Table 1 lists ecological models applied in the literature, specifying their geographic reach, and the biodiversity indicators they use. Most models use ‘species richness’ as the indicator of the state of biodiversity, and use the Species–Area Relationship (SAR) (Sarkar & Margules, 2002) ecological model to calculate species loss due to land use. Other studies use different models and meta-analysis (Michelsen, 2008; de Souza et al., 2013; Coelho & Michelsen, 2014), or apply Species Distribution Models (SDM) (de Baan et al., 2013a) to assess local environmental impacts based on empirical patterns observed between land-use classes. Geyer et al. (2010a,b) calculated regional biodiversity impact indicators based on wildlife species and their habitats. Indicators based on ecosystem characteristics (Brentrup et al., 2002) have also been used in LCA, as have those based on the concept of exergy (Wagendorp et al., 2006). In the following paragraphs, we explain the current general...
framework for land-use impact assessment in LCA, before moving on to the limitations of the choices made in modeling.

The current land-use framework consists of an area-time model, which combines spatial and temporal dimensions. Biodiversity levels are measured according to a chosen indicator – species richness being the most commonly used unit – and assigned to different land-use classes, assuming a defined land area is used for a specific time period.

To consider the temporal dimension of land use/cover in LCA (Fig. 3), land-use impacts are associated with two interventions: land occupation (land use, LU), and land transformation (land-use change, LUC), also referred to as land conversion (Lindeijer, 2000; Koellner, 2003; Liu et al., 2003; Schmidt, 2008a).

![Fig. 3](image)

**Fig. 3** Scheme of current framework for land-use impact assessment, displaying land quality change according to each land intervention: land-use change/transformation (a) and land-use/occupation (b) impacts (adapted from Schmidt, 2008a). The figure shows the initial drop in quality (due to land-use change) from $Q_0$ (land quality just before land conversion) to $Q_i$ (land quality right after land conversion), the point at which land use may start, $t_{occ}$ (duration of land use), up to $Q_f$ (land quality right after land use). At this point, land is set aside and regenerates, during $t_{rec}$ (duration of land recovery) to a potential quality state, $Q_{PNV}$ (potential quality after land recovery, i.e. after reaching the state of Potential Natural Vegetation, PNV). From these interventions, two types of impacts may result: temporary (conversion $I_{trans}$ and land use $I_{occ}$) and permanent ($I_{perm}$) impacts.
simplification, in LCA, land occupation and transformation impacts are not separated when representing land quality changes (Milà i Canals et al., 2007). It is assumed that land transformation causes a sudden change in the land quality indicator (e.g. a drop in quality from $Q_i$ (land quality just before LUC) to $Q_f$ (land quality right after LUC)) at the point in time $t$, in Fig. 3), while land occupation is simply defined (in the context of LCA) as the postponement of the recovery of land to its potential natural vegetation state (Schmidt, 2008a). Land quality is considered to be constant during its use, as seen in Fig. 3, in which $Q_i = Q_f$, where $Q_i$ represents the land quality right after LU. The impacts of land occupation and transformation are calculated by integrating the overall impacts on land quality over time (Milà i Canals et al., 2007), on a per unit of area basis.

According to the calculation procedure set by Milà i Canals et al. (2007), impacts resulting from land occupation ($I_{occ}$), represented by the area B in Fig. 3, are determined as the product of the area occupied ($A_{occ}$), the period of use, or land occupation ($t_{occ}$), and the quality difference between the actual and potential quality ($Q_{PNV}$) after land recovery. PNV refers to the Potential Natural Vegetation assumed to be reached after ecosystem recovery. The impact of land occupation (with land quality level starting in $Q_i$ and ending in $Q_f$, considering $Q_i = Q_f$) is determined as:

$$ I_{occ} = A_{occ} \times t_{occ} \times \Delta Q = A_{occ} \times t_{occ} \times (Q_{PNV} - Q_i) \quad (1) $$

The change in land quality over a defined area ($\Delta Q$) is considered to be reversible. In Fig. 3, the transformation impacts ($I_{trans}$) correspond to the area of a trapezium (base $t_{rec}$ and height $\Delta Q$) and can be derived from the following equation:

$$ I_{trans} = A_{occ} \times t_{rec} \times \frac{1}{2} \Delta Q = A_{occ} \times t_{rec} \times \frac{1}{2} (Q_{PNV} - Q_i) \quad (2) $$

The recovery time of land is represented by $t_{rec}$.

Permanent impacts ($I_{perm}$), for example, the irreversible damage caused to land use in terms of species loss ($\Delta Q_{perm}$), are calculated as

$$ I_{perm} = A_{occ} \times t_{occ} \times \Delta Q_{perm} = A_{occ} \times t_{occ} \times (Q_N - Q_{PNV}) \quad (3) $$

We turn next to a discussion of the substantial gaps in the impact assessment methods and framework.

Gaps in modeling

The biodiversity metrics currently integrated into LCA are not yet sufficiently inclusive and comprehensive, and leave ample room for improvement. Albeit a useful abstraction, the land quality model is of doubtful realism as its framework is a linearization of transient and dynamic ecosystem processes. Furthermore, it is based on assumptions that are at odds with observations and can compromise the accuracy of results.

We identified and summarized some of the existing gaps in modeling in Table 2 and divided them into four classes: conceptual issues, inventory, definition of indicators, and refinement of impact assessment methods. In the following sections, we highlight the most important issues, providing a thorough explanation of the major limitations identified and suggestions of how to address the identified shortcomings.

Our discussion of the limitations posed by the methodological framework of LCA (Curran et al., 2011) and related ecological models begins by addressing some of the limitations of using the SAR to predict species extinctions resulting from land use. We further assess the potential of other ecological methods such as species distribution models (de Baan et al., 2013b), including habitat suitability models (Geyer et al., 2010a,b) that are also applied in LCA to model damages to biodiversity. We then explore the use of surrogate species and their limitations as predictors of changes in ecosystems, and introduce the concepts of immigration credit and extinction debt to discuss the frailness of considering land interventions as being static. This brings the discussion back to the overall conceptual framework and we draw considerations on the choice of baseline scenario and assumptions to calculate occupation and transformation impacts (e.g. instantaneous drop in quality during land conversion). The issue of data demand at the global scale is also touched upon in these sections.

The limitations of using species–area relationships (SARs) and species distribution models (SDMs)

The starting point of our discussion is the Species–Area Relationship (SAR) model, which is much used in LCA for land-use impact assessments. According to SAR, the larger an area of land is, the greater variety of species it will contain. This model was first described by Arrhenius (1921) as

$$ S(A) = (A/A_0)^z S_o $$

where $S(A)$ is the number of species in a changed patch of area $A$, $S_o$ is the number of species found in an entire biome of area $A_0$, prior to the change, and $z$ – the gradient of the increasing number of species with increasing area (Rybicki & Hanski, 2013) – is a constant typically in the range of 0.10–0.35, depending on the taxon, and the land-use class and scale.

The SAR model is frequently used in conservation biology to estimate species extinctions due to changes...
| Aspect                  | Suggestion/limitation                                                                 | Milà i Canals et al. (2007) | Curran et al. (2011) | Koellner et al. (2013a) | Koellner et al. (2013b) | Penman et al. (2010) | This study | Addressed in LCA land-use models? |
|------------------------|---------------------------------------------------------------------------------------|------------------------------|----------------------|------------------------|------------------------|---------------------|------------|----------------------------------|
| Conceptual issues      | Consider the (1) existence value and/or the (2) natural resource aspects of biodiversity| X                            | X                    | X                      | X                      | This study          | (1) yes, (2) no |
| Conceptual issues      | Discuss the quantification of biodiversity using data or expert judgment              |                               | X                    | X                      | X                      |                     | No, but the LCA community is increasingly turning to data-intensive methods |
| Inventory              | Improve land use and cover classifications in inventory flows                         | X                            | X                    | X                      | X                      | This study          | A proposal has been made (Koellner et al., 2013a), but it is yet to be implemented in LCA inventories |
| Inventory              | Allow a finer flow specification in inventory, according to spatial scale            |                               | X                    | X                      | X                      | This study          | A proposal has been made (Koellner et al., 2013a), but it is yet to be implemented in LCA inventories |
| Definition of indicators | Discuss the inclusion of biodiversity as a midpoint or endpoint indicator (position in the cause–effect chain) | X                            |                      |                        |                        |                     | No – no consensus exists on the inclusion of biodiversity as midpoint or endpoint indicator |
| Definition of indicators | Include other levels of biodiversity (genes, communities, ecosystems, and landscapes) besides species | X                            | X                    | X                      | X                      | This study          | Partially (de Baan et al., 2013a) |
| Definition of indicators | Include other attributes of biodiversity (diversity, function, structure)           | X                            | X                    | X                      | X                      | This study          | Partially (for functional diversity – Souza et al., 2013) |
| Definition of indicators | Include differentiation of species by threat status                                   | X                            |                      | X                      | X                      | This study          | Partially (Müller-Wenk, 1998) |
| Definition of indicators | Discussion on the surrogate species to be included for modeling and consequences of this choice | X                            |                      | X                      | X                      | This study          | Partially (Schmidt, 2008a) |
| Definition of indicators | Include landscape aspects (habitat fragmentation, connectivity of ecosystems, etc.) | X                            |                      | X                      | X                      | This study          | No |
| Definition of indicators | Include bio-geographical differentiation (regionalization), increasing taxonomic and geographic coverage | X                            | X                    | X                      | X                      | This study          | Partially – global factors are available today for most taxa, but the differentiation may not be sufficient |
| Aspect                                                                 | Suggestion/limitation:                                                                 | Milà i Canals et al. (2007) | Curran et al. (2011) | Koellner et al. (2013a) | Koellner et al. (2013b) | Penman et al. (2010) | This study | Addressed in LCA land-use models? |
|-----------------------------------------------------------------------|----------------------------------------------------------------------------------------|-----------------------------|----------------------|-------------------------|------------------------|----------------------|-----------|----------------------------------|
| Definition of indicators                                             | Propose multi-indicators, each representing distinct aspects of biodiversity           | X                           | X                    | X                       | No                     | No                   | No        | Partially (for species richness and functional diversity – Souza et al., 2013) |
| Definition of indicators                                             | Discuss the suitability of SAR ecological models                                       |                             |                      |                         |                        | No                   | No        |                                   |
| Definition of indicators                                             | Include dynamic modeling of biodiversity and discuss the consequences of changes in land quality | X                           | X                    | No                      |                        |                      | No        |                                   |
| Refinement of impact assessment model – conceptual framework – link to inventory | Replace linear damage relation between biodiversity loss and land-use area            | X                           | X                    | X                       | No                     |                      | No        |                                   |
| Refinement of impact assessment model – conceptual framework – land interventions | Review drop in quality during land conversion, which should not always be considered to be instantaneous |                             |                      |                         |                        |                      | No        |                                   |
| Refinement of impact assessment model – conceptual framework – land interventions | Ecosystem quality does not necessarily remain constant over land-use duration         | X                           | X                    | No                      |                        |                      | No        |                                   |
| Refinement of impact assessment model – conceptual framework – land interventions | Time and area of land use must not be interchangeable                                 | X                           | X                    | No                      |                        |                      | No        |                                   |
| Refinement of impact assessment model – conceptual framework – cartography | Replace land cover maps with continuous environmental information                   | X                           |                      |                         |                        |                      | No        |                                   |
| Refinement of impact assessment model – conceptual framework – scale | Develop characterization factors for different scales (local, regional, global)      | X                           | X                    | X                       | No                     |                      | No        |                                   |
| Refinement of impact assessment model – conceptual framework – baseline scenario | Consider alternatives to natural land cover and potential natural vegetation as reference (baseline) land use/cover and consider active restoration as an alternative to natural ecosystem recovery |                             |                      |                         |                        |                      | No        |                                   |
| Refinement of impact assessment model – conceptual framework – quality recovery | Regeneration must not be linear and independent of the land-use history              | X                           | X                    | No                      |                        |                      | No        | (de Baan et al., 2013b)           |
| Refinement of impact assessment model – conceptual framework – quality recovery | Recovery times based on field data and not expert judgment                           | X                           | X                    | X                       | No                     |                      | No        |                                   |
in the landscape. However, the SAR model is often used without the consideration of key factors that influence its application, and it has been criticized for its simplicity and its tendency to make extrapolation mistakes (Smith, 2010). For example, the model is not time- and scale-invariant (Sizling et al., 2009), and its variance for small-scale observations is very high, probably due to sampling effects (Dengler, 2008). The number of parameters is low [only one parameter, $z$, controls relative extinction dynamics, as shown in Eqn (4)] and while it has been shown to work particularly well in islands, other functional models for SARs are better suited in continental situations (Dengler, 2009; Rybicki & Hanski, 2013).

Some shortcomings of the SAR model have been discussed in recent studies, such as its dependence on scale, which seems poorly understood (Dolnik & Breuer, 2008), and which may determine the degree of differences among the species–area curves (Scheiner, 2003). Moreover, other factors should be taken into account (Turner & Tjörve, 2005), such as habitat heterogeneity (Triantis et al., 2003). Specific spatial scales associated with relevant environmental factors should be defined. Fattorini & Borges (2012) discuss the inappropriateness of using the SAR model when the species loss does not take place as a consequence of habitat reduction, or when the shape, fragmentation level or contiguous character of the area is not identified (Dengler, 2008). Despite these considerations, the SAR model as described in Eqn (4) has proved to be robust. Over time, many changes have been tested and proposed to the generic SAR framework (Triantis et al., 2003, 2012; Adler et al., 2005; Dengler, 2008; Koh & Gazoul, 2010), but so far no model has consistently replaced the Arrhenius (1921) SAR.

As an alternative, Species Distribution Models (SDMs) can describe patterns and predict the natural geographical distribution of species, based on observations of species occurrence and/or abundance (Austin, 2007; Elith & Leathwick, 2009). Abundance and species distribution data reduce the problems associated with the spatial accuracy of data and changes in scale (He & Legendre, 2002) but, as with the SAR model, the application of SDMs may still represent a challenge for time- and scale-dependent assessments. Guisan & Thuiller (2004) summarized some of the potential problems to be considered, following aspects such as target species, scale, selection of predictors, data model, and model uncertainties. A spatial scale, for example, has several components, such as the grain, i.e. the smallest spatial unit used in the analysis, and the extent of the geographic area (domain), which are relevant to species distributions and their environment (Scheiner, 2003; Elith & Leathwick, 2009). The spatial accuracy and characteristics of the data applied will have implications on the model’s performance (Ferrier & Watson, 1997), and the use of mixed models may be required (Dormann et al., 2007). Guidance on the application of relevant methods still needs to be clearly provided (Elith & Graham, 2009).

The discussion as to which macro-ecological model should be used to estimate species loss transcends LCA and invites scientists with different backgrounds to explore a more broad cooperation and exchange of knowledge. The topic is still heavily debated. For LCA impact assessments, the question is whether the chosen model assumptions fit the goals of the studies.

**The challenges in using species metadata**

In addition to the use of SARs and SDMs, a common practice in land-use impact assessment in LCA is to use the metadata of species richness for different species, to grasp the change from one land use to another. Some problems exist in applying this procedure to LCA. Firstly, some studies do not define the state of successional of certain land cover types (e.g. secondary forest), and it is difficult to define the correct classification of that land cover type with specific flow categories defined for LCA (Koellner et al., 2013a), as suggested by Souza et al. (2013). Secondly, the maximum number of different species – e.g. bird populations or small mammals (Ostfeld, 1997) – will appear at different moments in the succession time (Odum & Barret, 2005), i.e. populations change according to the successional stage. Identifying the successional stage at the time of data collection is important, as population size per species in a region is not necessarily directly proportional to species diversity in the same region. In general, increases in populations may occur initially and stabilize or even decrease in later stages (McKinney, 2002). Ecological succession also often reduces the diversity of nonnative (alien) species in an area (Gibson et al., 2011), many of which rely on disturbance to sustain their populations (Luken, 1997). Further, different taxonomic and trophic groups may exhibit different patterns of diversity change in response to similar pressures (Pronča & Pereira, 2013).

In the current framework, urban areas are usually considered to be areas of poor species diversity; however, this might not always be true. Pyšek et al. (2004) emphasize that urban areas have a particular spatial pattern of species distribution, behavior, and population dynamics. While some authors argue that land cover types such as urban areas may contain a particularly high species richness, including both native and alien species (Kuhn et al., 2004), it is recognized that urbanization is one of the main threats to biodiversity.
influence on biodiversity, focusing on conservation, monitoring, and management actions, and in a political context, to direct attention to specific environmental aspects (Caro & O’Doherty, 1999). Surrogate species are generally used due to the impossibility of measuring the overall biodiversity in a certain area, given the difficulty of collecting sufficient relevant data (Groves et al., 2002; Margules et al., 2002).

Despite the apparent usefulness of surrogate species in impact assessment, no consensus exists on the use and consequences of the specific species chosen (Favreau et al., 2006). Some authors contend that the use of a surrogate, without verifying effective environmental responses, might compromise the effectiveness of the conservation aims (Andelman & Fagan, 2000; Cushman et al., 2010; Santi et al., 2010; Murphy et al., 2011), mainly because the cross-taxon surrogacy (i.e. of different taxa and species) may be poor (Kremen, 1992; Negi & Gadgil, 2002; Santi et al., 2010). Favreau et al. (2006) argue that the use of surrogate species should be based on good knowledge of the affected species and how they interact with their environment. They might be a single or a group of species, chosen for various criteria such as body size, which is correlated with sensitivity to disturbances and lifetime (Caro & O’Doherty, 1999). Most ecological processes are dependent on the scale and time (Wiens et al., 2008), and these two factors will also influence the ability of one species to reflect the behavior of others (Cushman et al., 2010).

A meta-analysis of 138 different studies has shown that the use of different taxonomic groups for the assessment of land-use impacts might lead to different results, due to differences in sensitivity to land-use change (Gibson et al., 2011). It was found that mammals were less sensitive to disturbances in the landscape, while birds were shown to be particularly sensitive, especially to the land-use change from forest into agriculture. Within some groups, different taxonomic orders were found to be impacted differently. Santi et al. (2010) investigated the degree of concordance among species richness and compositional patterns of different taxonomic groups, and found little cross-taxon congruence among most taxa.

The choice of surrogate species is still debated and is barely discussed in current applications of land-use impact assessment in LCA, although it is recognized that different taxonomic groups should be used (de Baan et al., 2013a; Souza et al., 2013). Vascular plants have been used in the assessment of the magnitude of land-use impacts (Müller-Wenk, 1998; Koellner, 2000; Weidema & Lindeijer, 2001; Vogtländer et al., 2004; Michelsen, 2008; Schmidt, 2008a), without further verification of the cross-taxon congruence/surrogacy of species diversity. Vascular plants have historically been
used in land-use impact assessment in LCA mainly due to data availability (Schmidt, 2008a) and the belief that they might correlate well with other species (Koellner, 2000; Weidema & Lindeijer, 2001; Vogtlaender et al., 2004), even though some authors argue that vascular plant species are an inappropriate surrogate indicator for biodiversity (Michelsen, 2008; Schmidt, 2008a). Besides these issues, current discussions barely touch the importance of the choice of surrogate species. One complicating factor is that LCA requires models that can better represent a more globalized characterization of impacts, due to the global or cross-regional coverage of different production systems.

**Biodiversity dynamics: does it matter?**

In this section, we discuss the influence of ecological phenomena such as extinction debt and immigration credit on the assumptions of the impact assessment framework. We assess how compromising the disregard of related phenomena such as landscape interactions and species assemblage is to land-use impact assessment in LCA.

The treatment of land interventions as being static, as in the current framework, is inadequate for reflecting changes in species diversity. In reality, these changes are not simply a result of the postponement of the recovery of land (occupation, LU), or a unique change in unit time (transformation, LUC), but the consequence of continuous anthropogenic activities and other environmental conditions, such as climate change. The land impacts that are considered do not reflect the cumulative effect of practices actually taking place.

A fundamental issue with this model is that, for some land-use changes, extreme drops in the quality after land transformation are unrealistic if the metrics implemented refer to biodiversity (Kuussaari et al., 2009). With regard to species richness, and to use an extreme example, the conversion of land from natural vegetation to intensive agriculture will not instantaneously extinguish all local species and reduce the level of biodiversity as soon as the new land use takes effect.

This concept of a delayed drop in quality (expressed in species diversity), and related temporary diversity surplus, is relevant to the issue because if losses are progressive and occupation is short enough, land-use impacts may be lower than expected. Conversely, if a patch of land is occupied over a relatively long period of time, the total impacts over time may lead to a threshold limit of biodiversity quality (Gibson et al., 2011) being met, leading to the collapse of the ecosystem. In this case, the difference between the initial and final steady-states should be higher. Documented examples show that extinction debts may persist for more than a century (Vellend et al., 2006; Cousins & Vanhoenacker, 2011), and this may lead to an underestimation of the potential impacts of land-use change on biodiversity.

The ecological concepts of ‘extinction debt’ (Tilman et al., 1994) and ‘immigration credit’ (Jackson & Sax, 2010) were established to address the reasons why it is empirically observed that species levels decay at a slower rate than predicted by models. An immigration credit is the number of new species that immigrate to a certain area, following a disturbance, during a time elapsed (immigration lag) between the disturbing event and the establishment of the species (Jackson & Sax, 2010). The extinction debt is the time-delayed loss of species as a result of habitat destruction and fragmentation which may occur generations later in the undisurbed fragments that remained after destruction. A surplus in the number of species is observed, during a definite time (extinction lag), after a forcing event takes place, in comparison to the final equilibrium level. It should be considered that, as the model applies to each unit of land area, what happens to the vicinity of that area is also crucial – highly fragmented areas may experience accelerated decay, while land areas surrounded by natural areas may see no change at all for some taxa.

The need to accommodate landscape effects has been suggested in recent reviews (Curran et al., 2011; Koellner & Geyer, 2013), but has rarely been fully implemented in assessments. Some impact assessment methods, such as ReCiPe (De Schryver & Goedkoop, 2013), include regional effects in calculations of local impacts in an attempt to include the contextual circumstances of the area of land (De Schryver et al., 2010). However, the quantification of regional effects is not bio-geographically discrete.

Extinction debt and immigration credit also depend on the characteristics of species and habitats. Individual species respond differently to events, depending on characteristics such as their sensitivity to change (Hylander & Ehrlén, 2013). For example, species whose populations are near their extinction threshold are most likely to experience an extinction debt (Hanski & Ovaskainen, 2002; Kuussaari et al., 2009), and the same is true for species with resistant life stages. Changes in habitat, including biotic interactions, habitat size, and related traits and specializations are also determinant factors.

**Reference state: are the natural and potential natural vegetation the correct alternatives?**

Another strong limitation of the current formulation is the difficulty in comparing the quality of natural (Q_n)
and potential ($Q_{PNV}$) vegetation (Fig. 3). The concept of Potential Natural Vegetation (PNV) has been recently introduced in LCA and extensively mentioned in current developments (de Baan et al., 2013a; Koellner et al., 2013b; Núñez et al., 2013; Souza et al., 2013). According to Chytrý (1998), PNV, as introduced by Túxen (1956), is the vegetation that would develop in the absence of any human influence. It is assumed that the vegetation should be in equilibrium with climatic conditions (Chiarucci et al., 2010), i.e. that the state is reached without any effect of climatic changes and succession, and the influence of time (Zerbe, 1998). However, during the expected period of recovery of species diversity (10–100 years) or endemism rates (1000–10 000 years), climatic conditions are bound to have an influence on the revegetation process, and should therefore be taken into account. Therefore, the baseline may not represent a unique static level.

In LCA, $Q_{PNV}$ is taken to be the level of quality reached after ecosystem recovery (Michelsen, 2008; de Baan et al., 2013a; Souza et al., 2013), while the difference between $Q_o$ and $Q_{PNV}$ is used to calculate the permanent impacts of land use and land-use change, as shown in Eqs (1), (2), and (3). But there may be multiple potential equilibria for the final state. It is not clear how to measure land quality, if it is defined as the capacity to sustain biodiversity (Dale & Beyeler, 2001). Firstly, the two states (PNV and natural) can hardly be compared on the basis of species richness, as the species that are supported at the end of the process may be different from the original species (Curran et al., 2013). The recovery process is dynamic and may have multiple equilibrium states, depending on the climatic and geomorphologic characteristics of the region (Chen et al., 2011; Bellard et al., 2012; Mokany et al., 2012), the existence of species and genetic resources available in the vicinity to recolonize the area, and the previous land-use history of the land area. Secondly, the final state is not necessarily equal to the original, even if the two states are similar in terms of ecosystem functioning (McKinney, 2002). For example, keystone, endangered or endemic species may have been replaced by other species that fulfill the same ecological role. Land quality can rarely be defined by a single criterion. We will return to the issue of using aggregate biodiversity metrics to define land quality in the conclusion.

Building on this point, it might appear to be useful to consider the natural vegetation as the starting point of the discussion (Weidema & Lindeijer, 2001). However, in practice, as recovery times are calculated as a function of the present and recent past land uses and the natural vegetation of the region, the indicators are calculated as there were a leap between $Q_{PNV}$ and $Q_N$. However, given the history of the land area, the land quality just before land conversion ($Q_o$) may be very different from that calculated for a situation where natural vegetation had just been converted. It is the case of a crop recently sown in a recently converted rainforest area vs. a crop planted over decades or centuries in an area that used to be a rainforest. The dotted regions in Fig. 3 are important if some type of natural vegetation is the reference. How far back in time one goes to find ‘natural vegetation’ is also often ill-defined. It remains unclear as to how far back in time one needs to go to discover PNV characteristics, as for most areas this is empirically impossible to observe, and to assess what it really represents for species recovery, especially when taxonomic groups other than plants are used as indicators.

Figure 3 also suggests that the indicator takes PNV as the maximum possible level of land quality that can be achieved within the recovery time of the ecosystem. This is not necessarily the case, as it fails to address active land restoration (which can reintroduce some lost species and help the ecosystem to recover in a shorter period of time), and the cessation of anthropogenic land management activities (which would lead to a decrease in the richness of species and habitats that are dependent on such activities) – as is the case, for example, of Mediterranean cork oak savannahs (Bugalho et al., 2011). However, even in the cases of active restoration and critical dependence on land management, hardly any nonnaturalized area can compete with biodiversity levels in the abstract, idealized baseline. This strongly suggests that calculations based on indicators built using this model will always show a loss of quality. No matter how much the state of biodiversity is improved by any intervention or management type, human activity always results in an effective loss of land quality, as it converts land areas to uses that are not natural. For policy-makers, the comparison of an action with a conceptual seminatural baseline may be of little interest (Koellner et al., 2013b). Other alternatives for a baseline, such as land use in the year 2000 (Koellner & Geyer, 2013), have not yet been tested. The Potential Replacement Vegetation (PRV) concept could be a more suitable alternative (Chytrý, 1998), as it takes into account climate, soil, and anthropogenic influences (such as management practices) on the habitat. However, it cannot help predict future changes.

Issues identified and contribution toward their resolution

Future developments in making informed choices of indicators should be linked to the valuation of biodiversity by its existence or by its value to society (or the
value society attributes to the species/ecosystem services it provides). This topic is properly addressed by Duelli & Obrist (2003), who discuss the difference between indicators that define biodiversity itself and the ones that use biodiversity to define other environmental and social attributes.

Given the complexity of biodiversity assessment, all the different aspects cannot be understood by applying one single indicator. Several aspects of biodiversity will probably need to be addressed to do so – at midpoint and endpoint levels. This implies, for instance, a more in-depth assessment of the different issues in the cause-effect chain (Fig. 1), which may lead to a better clarification of ecological mechanisms and biodiversity dynamics. It also requires a better definition or clear statement of ‘what is to be sustained’, to appropriately orient the decision-making process. For example, the choice of indicators will depend on whether the goal is to preserve specific ecosystem services or a group of targeted species.

The LCA community is already moving beyond the use of species richness as an indicator of biodiversity, toward the inclusion of other types of diversity, such as functional diversity (Souza et al., 2013). Other levels of analysis, such as populations and landscapes, should also be included. However, if multiple indicators are to be used to determine impacts at each level of the cause-effect chain, further research should be carried out to determine how each indicator should be evaluated, and how results should be interpreted and communicated to avoid double counting. If indicators are used, for example, in eco-labeling, it must be decided as to which and how much information regarding biodiversity loss should be passed along to consumers and the public to help them make informed decisions.

The conventional approach of using species richness as a biodiversity indicator can be enhanced by applying different macro-ecological models, while taking their limitations into account.

Several factors need to be considered when using surrogates in species models, such as knowledge of the biota, number and geographic range (spatial scale) of the species in the area, and temporal changes. The correlation of temporal and spatial dynamics with the surrogate species and the area of assessment also need to be taken into account. Extinction debt and immigration credit can be included using a mix of multivariate spatial models. Gibson et al. (2011) use a meta-analytic approach that integrates variables that determine biodiversity dynamics to infer immigration credit and time since conversion as an estimate of extinction debt. These biodiversity dynamics variables are indirect measures of both effects, and they can also be modeled directly. Unfortunately, this topic is still poorly understood in the ecological literature and the way to address it transcends LCA (Jackson & Sax, 2010).

The issue of temporal dynamics encompasses more than these two effects. Failing & Gregory (2003) mention the importance of specifying the time scale of assessment of a given indicator, to consider cross-temporal trade-offs. One alternative to be explored would be to define indicators at different scales (local, regional, national, global) or at ecosystem-specific levels, which are able to grasp the particularities of several species groups and their dynamic interactions with their habitats. To an extent, there has been an effort in this sense (de Baan et al., 2013b), but it is likely that different levels of biodiversity (genes, species, ecosystems) will be impacted differently depending on their scale, and they should thus be modeled separately. Therefore, future proposals of characterization factors should be scale-dependent, and not simply an extrapolation from local to global or vice-versa. Characterization factors must be carefully validated against field data and local/national case studies.

The issues regarding the impact assessment model will require revamping the ideas of impacts of LULUC, as well as their dynamic representation. In some cases, no sudden loss in biodiversity occurs due to land conversion, as the response of ecosystems is often gradual. Improved SAR models should also take the spatial and temporal dimensions of biodiversity loss into consideration. Similarly, the use of PNV concept needs to be re-re-evaluated. After anthropogenic changes, natural systems go back to a near-equilibrium state (O’Neill, 1999), but the condition reached after recovery may not reflect the original natural or close-to-natural conditions – as currently assumed in LCIA. Moreover, built-up areas such as urban or industrial areas can hardly be set aside for land recovery.

The choice of the baseline is not an arbitrary topic of discussion in LCA land-use impact assessment. Weidema & Lindeijer (2001) led part of this discussion, presenting the different choices of the baseline land quality level: original state before any human intervention (Blond et al., 1997), state immediately before land use (Baity et al., 1998), and state immediately after the land-use activity (Koellner, 2000). The topic has also been discussed by Milà i Canals et al. (2007), de Baan et al. (2013a), and Souza et al. (2013). Rather than using PNV, the baseline may also be a reference year with globally identified land cover types (e.g. Global Land Cover 2000). However, this approach also has some drawbacks. First, future assessments would need to conform to the (often poor) level of accuracy and differentiation of land cover information as presented in the reference dataset, ignoring improvements in land cover data. This is also the case with regard to the use of PNV as a
baseline reference, as land cover is measured and PNV is estimated. Second, if at the moment of the assessment, the land use/cover in a specific area was the same as the land use/cover of the reference year, no impact would be assigned to the activity taking place in that area, even though the area may have undergone changes. Alternatives exist, such as taking local or regional conservation values into account. The choice of baseline reference involves value choices that are beyond the scope of the scientific method, and must be carefully assessed before any specific assessment method is adopted. This is a matter that should be comprehensively investigated and further discussed.

Conclusions

The current modeling of land use using biodiversity indicators significantly simplifies the real transient dynamics and complexity of natural processes and interactions among and between species and their habitats. In this paper, we explained the main limitations of LCA in grasping the scale and extent of the impacts of land use on biodiversity. We compiled some of the limitations reported in the specialized literature and proposed several issues that have yet to be addressed.

Assigning responsibilities for biodiversity loss in supply chains is an important first step toward reaching conservation goals. LCA is a suitable top-down method that can contribute toward this goal. However, such an effort is multidisciplinary and LCA would greatly benefit from a stronger engagement of experts in related fields, such as biologists and ecologists, who could help organize and integrate knowledge based on their extended experience in understanding and modeling complex ecological systems (e.g. modeling population dynamics). LCA calculates potential impacts rather than making accurate depictions of field-level dynamics, but it must provide practitioners with accurate and consensual measures of biodiversity. This target, though within reach, has not yet been met.

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Author contributions

D.M.S conceived and designed the research review. D.M.S and R.F.M.T. carried out the review and wrote the manuscript, contributing equally to this work. O.P.O. proofread the final version of the manuscript.

References

Adler PB, White EP, Laueenroth WK, Kaufman DM, Rassweiler A, Rusak JA (2005) Evidence for a general species-time-area relationship. Ecology, 86, 2032–2039.

Andelman SJ, Fagan WF (2000) Umbrellas and flagships: efficient conservation surrogates or expensive mistakes? Proceedings of the National Academy of Sciences, 97, 5954–5959.

Antin A, Castello F, Montero JI (2007) Land use indicators in Life Cycle Assessment. Case study: the environmental impact of Mediterranean greenhouses. Journal of Cleaner Production, 15, 432–438.

Arrhenius O (1921) Species and area. Journal of Ecology, 9, 95–99.

Austin M (2007) Species distribution models and ecological theory: a critical assessment and some possible new approaches. Ecological Modelling, 200, 1–19.

de Baan L, Alkemade R, Koellner T (2013a) Land use impacts on biodiversity in LCA: a global approach. The International Journal of Life Cycle Assessment, 18, 1216–1230.

de Baan L, Mutel CL, Curran M, Hellweg S, Koellner T (2013b) Land use in Life Cycle Assessment: global characterization factors based on regional and global potential species extinction. Environmental Science & Technology, 47, 9281–9290.

Baizt M, Kreissig J, Schöch C (1998) Methode zur Integration der Naturnorm-Ansprüche in Ökobilanzen. IKP, Universität Stuttgart, Stuttgart.

Bare JC, Hofstetter P, Pennington DW, Ludo de Haes HA (2000) Midpoints versus endpoints: the sacrifices and benefits. The International Journal of Life Cycle Assessment, 5, 319–326.

Bell HL (1986) Occupation of Urban Habitats by Birds in Papua New Guinea. Western Foundation ofVertebrate Zoology, Los Angeles.

Bellard C, Bertelsmeier C, Leadley P, Thullier W, Courchamp F (2012) Impacts of climate change on the future of biodiversity. Ecology Letters, 15, 365–377.

Blich H, Lindeijer E, Broers J (1997) Towards a methodology for taking physical degradation of ecosystems into account in LCA. The International Journal of Life Cycle Assessment, 2, 91–98.

Brentrup F, Kusters J, Lammel J, Kuhlmann H (2002) Life cycle impact assessment of land use based on the Hemonoby concept. The International Journal of Life Cycle Assessment, 7, 339–348.

Bugalho MN, Caldeira MC, Pereira JS, Caldeira MC, Aronson J, Pausas JG (2011) Mediterranean cork oak savannas require human use to sustain biodiversity and ecosystem services. Frontiers in Ecology and the Environment, 9, 278–286.

Caro TM, O’Doherty G (1999) On the use of surrogate species in conservation biology. Conservation Biology, 13, 805–814.

Chen IC, Hill JK, Ohlemüller R, Roy DB, Thomas CD (2011) Rapid range shifts of species associated with high levels of climate warming. Science, 333, 1024–1026.

Ciuracci A, Araújo MB, Decoq G, Beierkuhnlein C, Fernandez-Palacios JM (2010) The concept of potential natural vegetation: an epitaph? Journal of Vegetation Science, 21, 1172–1178.

Chytrý M (1998) Potential replacement vegetation: an approach to vegetation mapping of cultural landscapes. Applied Vegetation Science, 1, 177–188.

Coelho CRV, Michelsen O (2014) Land use impacts on biodiversity from kiwi/fruit production in New Zealand assessed with global and national datasets. The International Journal of Life Cycle Assessment, 19, 285–296.

Cousins SAO, Vanhovenacker D (2011) Detection of extinction debt depends on scale and specialisation. Biological Conservation, 144, 782–787.

Curran M (1996) Environmental Life-Cycle Assessment. McGraw-Hill, New York. pp. 432.

Curran M, de Baan L, De Schryver AM et al. (2011) Toward meaningful end points of biodiversity in life cycle assessment. Environmental Science & Technology, 45, 70–79.

Curran M, Hellweg S, Beck J (2013) Is there any empirical support for biodiversity offset policy? Ecological Applications, 24, 617–622.

Cushman SA, Mc Kelvey KS, Noon BR, McGregor K (2010) Use of abundance of one species as a surrogate for abundance of others. Conservation Biology, 24, 830–840.

Dale VH, Beyeler SC (2003) Challenges in the development and use of ecological indicators. Ecological Indicators, 3, 3–10.

De Schryver A, Goedkoop M (2013) Impacts of land use. In: ReCiPe 2008: A Life Cycle Impact Assessment method which Comprises Harmonised Category Indicators at the
