

Article

Why Include Impacts on Biodiversity from Land Use in LCIA and How to Select Useful Indicators?

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Abstract: Loss of biodiversity is one of the most severe threats to sustainability, and land use and land use changes are still the single most important factor. Still, there is no sign of any consensus on how to include impacts on biodiversity from land use and land use changes in LCIA. In this paper, different characteristics of biodiversity are discussed and related to proposals on how to include land use and land use changes in LCIA. We identify the question of why we should care about biodiversity as a key question, since different motivations will result in different choices for the indicators, and we call for more openness in the motivation for indicator selection. We find a promising trend in combining pressure indicators with geographic weighting and regard this as a promising way ahead. More knowledge on the consequences of different choices, such as the selection of a reference state, is still needed.

Keywords: LCIA; biodiversity; land use; land use changes; species diversity; structural indicators

1. Introduction

Loss of biodiversity is one of the most severe threats to environmental sustainability [1], and despite the recent focus due to the Convention on Biological Diversity [2], there are few signs of reductions in the rate of loss of biodiversity [3]. The pressure is in fact increasing [3,4].

Life cycle impact assessment (LCIA) is used to quantify the potential environmental impacts throughout the life cycle of a product or a service [5], but impacts on biodiversity are only partly included [6]. Changes in land use are often the most important driver for loss of biodiversity [7–11], but climate change, pollution, invasive species and overexploitation are also of importance [3,9,12–14]. Barnes [15] and Haines-Young [10] point out that not only changes where one land use activity is replaced by another are of importance; small modifications in land use, such as changes in intensity, changes in use of fertilizers and watering, changes in crop, *etc.*, might be equally important as transformations, even if such modifications are not identified when land use changes are in focus.

The importance of land use and land use changes (LULUC) has long been recognized as a topic to be treated in LCIA [16]. Of the main drivers for biodiversity loss, LULUC is at least partly included [6] in LCA. Still, the question of how LULUC should be included is debated; a range of proposals have been put forward [6,17,18], but no consensus has been reached. Most proposals are based on the premise that land use impacts should be assessed in three dimensions—area, quality and time—where the basic idea is that a stretch of land (area) is altered from its current state (quality) to adapt it to the intended use for a period of time [18,19]. While the area affected by a certain impact is relatively easily determined, quality changes and the time scale of the alteration are much harder to assess. The inequalities in different approaches give rise to different results and recommendations [20,21], and inevitably, the results are questionable for decision support [22]. United Nations Environmental Programme and Society of Environmental Toxicology and Chemistry have a joint effort to enable a global use of credible life cycle knowledge and in the latest guidelines from UNEP-SETAC, a call for “more consistency” is put forward [18], and this work is taken further by the UNEP-SETAC Life Cycle Initiative “Environmental Life Cycle Impact Indicators”, which aim to build consensus within the field.

Due to the complexity of biodiversity, it is argued that a universal indicator for biodiversity is not likely to be found [9,23–25], and using the structure from Noss [23], Curran *et al.* [6] show that biodiversity indicators can be divided into 12 categories based on hierarchical components and biological attributes. Still, most proposals for including impacts on biodiversity from LULUC in LCIA try to capture the impact in a single indicator.

Curran *et al.* [6] provide the state-of-the art of the present status for impacts on biodiversity in LCIA, and Koellner *et al.* [17] and Souza *et al.* [26] provide the latest overview of different approaches. Koellner *et al.* [18] propose a guideline for building a land use impact assessment method. The guideline is full of choices need to be made, and what is still lacking in this picture is a debate on the value choices, whether deliberately made or not, made when the complexity of biodiversity is transformed to, in most cases, a single indicator.

In this paper, we focus on methods for the inclusion of land use in LCIA. Land use change is the most important driver for the loss of biodiversity, and it is the area where a diverging set of approaches are put forward. Still, value choices for indicator selection are relevant for other aspects, as well.

In this paper, we first discuss different characteristics of biodiversity as a starting point for biodiversity indicator selection in LCIA. We relate these to existing proposals and discuss the relationship between the methodologies and the theoretical foundation. Numerous reviews on these topics have been published, and what we will highlight here is the value choices behind the different proposals. These are seldom explicitly addressed. We see this as a problem, since the motivation for the different choices is consequently not openly discussed. More attention to these value-based choices will

give a better understanding for the motivation behind the different methodological approaches, which aspects of biodiversity will be highlighted following the different approaches and what consequences does this have. Based on this, we give our recommendations on how we see this development being brought a step forward.

2. Biodiversity and Protection of Biodiversity

The Convention on Biological Diversity (CBD) defines biological diversity as “the variability among living organisms from all sources including, *inter alia*, terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are part; this includes diversity within species, between species and of ecosystems” [2]. There is a widespread agreement that biodiversity is a prerequisite for human survival and needs protection, but this broad, top-level definition does not explicitly state what and how to protect.

In the convention preamble, it is stated that the contracting parties are “affirming that the conservation of biological diversity is a common concern of humankind” and are “conscious of the intrinsic value of biological diversity and of the ecological, genetic, social, economic, scientific, educational, cultural, recreational and aesthetic values of biological diversity and its components”.

As identified in the CBD, the ways in which humanity benefits from biodiversity are manifold, and the Millennium Ecosystem Assessment [12] identifies four types of ecosystem services to which biodiversity contributes, namely provisioning, regulatory, supporting and cultural services. Provisioning services include the provision of water, food, fibers, energy sources, as well as genetic resources. Regulatory services include climate regulation, disease control and other feedbacks. Supporting services are indirect provisioning services, such as soil formation, that provide a basis for other services and people to generate wealth. Cultural services are the least tangible, including aesthetic values, recreation opportunities, a sense of belonging, and the like. By all practical means, these services are highly interrelated and can hardly be separated from one another; however, all are dependent on a certain level of biodiversity, and ambitious plans for stopping the loss of biodiversity have been launched [27–29].

Due to this complexity, it is argued that indicators for biodiversity should be selected based on the purpose of the assessment and initial conditions [30–32]. Still, there are quite a few initiatives for the construction of extensive biodiversity indicator sets and biodiversity indexes based on a high number of indicators [9,24,25]. The European Environment Agency (EEA) framework [33] is a typical example of the first, consisting of 26 different indicators, where eight regard the status and trends for biodiversity. The German indicator set of the national biodiversity strategy consists of 19 individual indicators following the pressure-state-response system [34]. Examples of biodiversity indexes are the Natural Capital Index [35], the Living Planet Index (LPI) [36], the Biodiversity Intactness Index (BII) [37], the Global Biodiversity Model (GloBio) [13] and the Norwegian Nature Index (NNI) [25]. Some of these, such as the NNI, are a combination of rather different indicators based on, e.g., the size of populations, changes in habitats and expert judgements, while LPI, on the other hand, is solely based on vertebrate population data. Both NNI and BII are constructed in a way that allows for further inclusion (or exclusion) of indicators, without changing the range of the index (from zero to one, where one represents no impact). It is important to have in mind that some of these indexes are designed to assess changes in, and not necessarily the actual state of, biodiversity [25,35].

The question of how much biodiversity is actually needed to provide the ecosystem services identified in Millennium Ecosystem Assessment [12] is still open. Lawton [38] summarizes four different hypotheses for the relationship between species diversity and ecosystem processes that provide services. The redundant species hypothesis [39] suggests that most species are redundant; only a minimum diversity is needed for proper ecosystem functioning. The rivet hypothesis [40] suggests that all species contribute, but that the magnitude of change is dependent on which other species are present. The idiosyncratic response hypothesis [38] suggests that functioning changes when the number of species is changed, but neither the direction nor magnitude is predictable. The null hypothesis postulates that ecosystem functioning is insensitive to changes in biodiversity. Bengtsson *et al.* [41] and Loreau *et al.* [42] summarize attempts to test these hypotheses and conclude that none of them so far can be generally supported nor rejected. As examples, Cardinale *et al.* [43] identified declining ecosystem functioning from the very first losses of species diversity, while, e.g., Wardle *et al.* [44] found a higher productivity with lower species diversity.

These hypotheses address the provision of ecosystem services at a given time, but the stability of their provision is also of interest [1,7,41,45–48]. For this purpose, the diversity-stability hypothesis [49] and the insurance hypothesis [41] are formulated. Chapin *et al.* [7] postulate that high species diversity reduces the risk of large changes in ecosystem functions in response to directional or stochastic variation in the environment, e.g., climate change and invasive species and pathogens. Based on this, Bengtsson *et al.* [41] claim that the succeeding insurance hypothesis consequently is one of the strongest arguments for maintaining biodiversity; seemingly redundant species might be important for ecosystem functioning under altered future conditions (in a general sustainability context, this is also known as the precautionary principle). Species that seemingly are redundant today might be important for ecosystem resilience [10].

Traditionally, conservation targets on biodiversity have focused on the number of species [31,50,51] with an additional focus on threatened species through the preparation of red lists [52], lately also with red lists of communities/ecosystems as a supplement [53]. The focus is now shifting towards ecosystem services, and Macfadyen *et al.* [54] claim that there is a tendency to link the conservation of biodiversity to maintaining ecosystem services; the (economic) value of these are increasingly emphasized [55–57]. The question is then how to translate these targets into conservation targets that are possible to assess when it is not known how much biodiversity is needed to ensure the different ecosystem services. The correlation between species diversity and ecosystem services is weak [10,14,32,56,58–61], and the increased focus on ecosystem services thus reduces the focus on species diversity as a value in itself. Some of the ecosystem services are also weakly correlated with each other [4,56,60,62,63], meaning that a strong focus on selected ecosystem services might be at the sacrifice of others.

In order to identify the “right” (purpose dependent) indicators for assessing biodiversity, it is consequently necessary to identify the motivation (case specific) for protecting biodiversity. Different motivations might give rise to different indicators and targets. If the focus is on ecosystem services, the basic challenge is to identify and conserve the right and enough species [14,49]. Even though this is difficult to identify [64], it does have the implication that species-rich ecosystems are not more valuable than species-poor ecosystems *per se* [23] and that one species can potentially substitute another [54]. Depending on the ecosystem service or the aspects of biodiversity in focus, the valuation of different species may be very different and even contradictory. For example, the bulk of many provisioning

services is provided by a few dominant species (e.g., plants processing carbon/oxygen). In this context, common species are more valuable than rare species.

On the other hand, if the motivation is to protect species and biodiversity in general, irrespective of whether the motivation is rooted in intrinsic values or the potential for (genetic) commodities (*cf.*, [56,65]), the primary concern will be to conserve as many species as possible, and an area with high species diversity would consequently be judged to have higher quality than a species-poor area. In addition, threatened species could be given extra weight, since these are most in danger of being permanently lost as potential sources [56]. Distribution in time and space is also important; we obviously would not perceive rhinoceroses in central Europe or flowering dandelions in winter as a sign of intact biodiversity.

In order to identify the “right” biodiversity indicators, that is indicators actually focusing on aspects that are of identified interest, the questions mentioned above must be addressed. Indicators for biodiversity impacts from LULUC in LCIA are no exception.

3. Impacts from Land Use on Biodiversity in the LCA Literature

The debate on how to include LULUC in LCIA has been going on since the early 1990s [16], but despite methodological progress [17–19,66], it has been hard to find a clear converging trend. Two main directions can be identified: proposals based on species richness as an indicator for biodiversity and proposals based on other indicators, here collectively denoted as structural and indirect biodiversity indicators [67].

The main goal with this section is to describe different proposals of the indicators for LULUC in LCIA, to identify what value choices, explicit or implicit, lie behind the choice of indicators and to identify to what degree existing frameworks for biodiversity assessments are implemented in the methodological development for the implementation of LULUC in LCIA. Since the focus is on the value choices, this is not a full review of the methods, as such; this can be found in, e.g., Koellner *et al.* [17] and Souza *et al.* [26]. The main focus here is on the assumptions and choices behind the methods.

In order to identify the relevant papers, we performed a rather wide search in Scopus (www.scopus.com) for the combination of “life cycle assessment” and “land use”. The papers were examined to identify papers dealing with methodological development and proposals on how to include biodiversity issues related to land use and/or land use changes in LCIA. Included papers are listed in Table 1.

Table 1. Identified papers on methodological development for including impacts on biodiversity from land use and land use changes in LCIA. See the text for selection criteria.

| Paper | Indicator | Spatial resolution | Geographic origin of data | Biodiversity valuation |
|-----------------------|---|--------------------|---------------------------|---|
| Köllner (2000) [68] | Relative species richness (vascular plants) | Corine classes | Switzerland and Germany | All species are weighted equally, regional factor for extra focus on scarce regions |
| Lindeijer (2000) [69] | Vascular plant species diversity | Not stated | Global (meta-analysis) | Intrinsic value of nature, species diversity, since this is available |
| | Free net primary biomass production | Not stated | Global (meta-analysis) | Not for biodiversity, but for “life support function”, <i>i.e.</i> , ecosystem services |

Table 1. Cont.

| Paper | Indicator | Spatial resolution | Geographic origin of data | Biodiversity valuation |
|--|---|---|----------------------------------|--|
| Mattsson <i>et al.</i> (2000) [70] | “Biodiversity” qualitatively | Not stated | Sweden, Brazil, Malaysia | “Preserve landscape values and biological diversity”; the number of species focused on |
| Brentrup <i>et al.</i> (2002) [71] | Deviation from naturalness [0,1] | Biogeographic regions (11 in Europe identified) | | “Naturalness” is a resource |
| Vogtländer <i>et al.</i> (2004) [72] | Species richness (finally: eco-costs) | 1 km ² | Netherlands | Many species increase the chance of capturing valuable species |
| | Ecosystem rarity (finally: eco-costs) | 1 km ² | Netherlands | Rareness = valuable |
| Bare <i>et al.</i> (2003) [73] | Absolute number of threatened and endangered species | Not stated, county mentioned as an example | Formula given for calculation | Assumes that an area with a higher number of threatened and endangered species is inherently more valuable due to a higher risk of species extinction |
| Wagendorp <i>et al.</i> (2006) [74] | Thermal indicators/exergy | Must be developed—case specific | Must be developed; case specific | Not stated, but “climax/undisturbed nature” is the reference state |
| Garcia-Quijano <i>et al.</i> (2007) [75] | Range of indicators, among other relative changes in species number, canopy cover of exotic plants, <i>etc.</i> | Case-specific data | Case specific | Not stated, includes a range of indicators to use a “best compilation of universally-available indicators”. Only focuses on “on-site” effects, e.g., not the impact of water stress elsewhere. |
| Koellner and Scholz (2007, 2008) [76,77] | Ecosystem damage potential based on relative changes of (vascular) species diversity, one version adjusted with a focus on threatened species. Some additional data on mosses and mollusks. | 53 land use types (Corine Plus) | Switzerland and Germany | States that biodiversity is a key element for ecosystem functioning and claims that “Obviously the occurrence of those [threatened] species should be weighted more” and “Obviously the impact of occupying a plot of land should be assessed relative to the region where the occupation takes place” |
| Toffoletto <i>et al.</i> (2007) [78] | Species richness, ecosystem scarcity and ecosystem vulnerability | Canadian ecozones (15) | Canada | Based on the work by Weidema and Lindeijer (2001). An area is given higher value based on inherent scarcity (here the ecozone), vulnerability and the number of vascular plant species. |
| Burke <i>et al.</i> (2008) [79] | Number of endemic and red listed plant species | Case specific site, here 140 km ² | Namibia | Only red listed and endemic species considered, given a score based on rareness |
| Held and Bos (2008) [80] | Net primary biomass production (NP(B)P), soil parameters | Not stated | Not stated | Does not give any value; this is a focus on aspects necessary for biodiversity, actually different ecosystem services (term not used) |

Table 1. Cont.

| Paper | Indicator | Spatial resolution | Geographic origin of data | Biodiversity valuation |
|---|--|--|---------------------------------|--|
| Jeanneret <i>et al.</i> (2008) [81] | Vascular plants, birds, mammals, amphibians, snails, spiders, carabids, butterflies, wild bees, grasshoppers as “indicator species groups” (ISG) | Agricultural field, size not stated | Switzerland | Number of species in each ISG, and the importance of the ISG as food for other indicator species groups is given a higher weight |
| Kløverpris <i>et al.</i> (2008, 2010) [82,83] | Area | 8 biomes | Global | Not stated |
| Michelsen (2008) [84] | Ecosystem scarcity, ecosystem vulnerability and ecosystem specific structural indicators | 867 ecoregions | Global + Norway | Increased weight on rare and vulnerable ecoregions |
| Schmidt (2008) [85] | Absolute vascular plant species richness | Multiple land use categories for 9 habitats for different (undefined) regions, 19 combinations are shown | Denmark, Malaysia and Indonesia | Not explicitly stated, but focuses on species richness. All species equal (native, endemic, invasive). |
| Glendining <i>et al.</i> (2009) [86] | (Monetary) value of ecosystem services | Biomes, 11 main classes | Global | No focus on ecosystem services and the value of these (based on Costanza <i>et al.</i> 1997) |
| Mila i Canals <i>et al.</i> (2009) [87] | Water availability | River basins | Global | Not stated |
| Geyer <i>et al.</i> (2010) [88,89] | Vertebrate species richness and abundance (based on presence/absence in habitats). | 59 potential local habitats (29 occur in the included case) | Case study, California | “Biodiversity protection”. All native species equal (invasive not included). |
| Coelho and Michelsen (2014) [90] | Ecosystem scarcity, ecosystem vulnerability and naturalness | 867 ecoregions | Global | Increased weight on rare and vulnerable ecoregions |
| de Baan <i>et al.</i> (2013) [91] | Biodiversity Damage Potential (BDP), primarily based on relative changes in species composition (several taxonomic groups) | 14 biogeographic regions | Global, but skewed | Not stated, but discusses that different views exists. All species equal (native, endemic, invasive). |
| de Baan <i>et al.</i> (2013) [92] | Absolute species richness for mammals, birds, amphibians, reptiles and plants | 804 ecoregions | Global | All species equal (native, endemic, invasive). Separates local and global extinction (reversible and irreversible impacts). |

Table 1. Cont.

| Paper | Indicator | Spatial resolution | Geographic origin of data | Biodiversity valuation |
|-----------------------------------|---|--------------------|--|---|
| Souza <i>et al.</i> (2013) [93] | Functional diversity index of mammals, birds and plants | Ecoregion | 21 Ecoregions in North and South America | Functional traits valued; species regarded as redundant if other species in the area have similar functional traits. Ecosystem processes and stability regarded as more important than species loss <i>per se</i> . |
| Elshout <i>et al.</i> (2014) [94] | Relative species richness | Biome | Global, limited to agricultural cases | Not explicitly stated, but relies on species diversity as an acceptable proxy for biodiversity. Equal weight to all species within a study area. |
| de Baan <i>et al.</i> (2015) [95] | Habitat suitability (mammals) | 900 × 900 m | East Africa, global available dataset | Avoid global extinction, extra weight to rare and threatened species |

3.1. Species Richness as a Quality Indicator

3.1.1. Species Diversity as a Proxy for Biodiversity

Methodologies using species diversity all have the starting point that biodiversity, or at least important aspects of biodiversity, can be assessed based on species diversity. This assumption should be of no surprise, since species diversity is the most commonly-used indicator on biodiversity in general [31,50,51], and the use of changes in species richness as an indicator can be traced back to some of the first attempts to come up with methodologies to include impacts from LULUC on biodiversity in LCIA, such as Müller-Wenk [96] and Köllner [68,97]. Furthermore, in the recommendations in the UNEP-SETAC guidelines, species diversity is the preferred measure (together with functional diversity) for assessing damage to biodiversity [18]. The question of whether species diversity captures the preferred or focused aspects of biodiversity is not well addressed in the identified papers. Köllner [68] briefly argues that the occurrence of species can be considered as a proxy for ecological value in terms of sustaining ecological functions and services and consequently argues that all species should be equally weighted. Müller-Wenk [96] argues in the same direction, while, e.g., Lindeijer [69], Schmidt [85] and de Baan *et al.* [91] have a more pragmatic approach and acknowledge that the presence of a species is the best available dataset.

The earliest of these proposals also assumed, explicitly or implicitly, that one taxonomic group can be used as an indicator for species diversity in general. It is argued that the correlation is fairly good and that vascular plant diversity is an adequate proxy, since data are readily available and the diversity is fairly correlated with other taxonomic groups (*cf.* [77]). This is supported in some studies (see [24]), while most studies show very weak or no correlations at all [7,26,50,98–105]. Grenyer *et al.* [106] show that correlations between taxonomic groups are even weaker if only rare and endangered species are the focus.

Schmidt [85] addressed this as a serious drawback, but concluded that vascular plants still are the best available alternative due to data availability. More recent proposals and studies include more taxonomic

groups in their proposals, and in most of these, significant differences in how taxonomic groups respond to different impacts are identified [81,91–94]. Still, as clearly shown in de Baan *et al.* [91,95], very few datasets are available on a global scale for numerous taxonomic groups.

Characterization factors based on species diversity depend on species-area relationships, since changes in species richness must be related to a standardized area unit. A species-area relationship is given by $S = cA^z$, where S is the number of species, A is the area, c is a species richness factor and z the species accumulation rate [107]. These relationships are not generally established and vary depending on taxonomic group, size of area and land use, among others [108,109]. In some of the proposed methods, fixed species accumulation rates are still used (e.g., [77,85]), and de Schryver *et al.* [110] show how this might influence the final results and increase the uncertainties. de Baan *et al.* [95] use data on mammal species richness, but as an alternative to actual species counts and species-area relationship, they use habitat suitability models and overcome some of these challenges.

There are some additional drawbacks with the focus on the number of species that are not addressed in the papers. The increase or decrease of species within one taxonomic group gives in itself little information about changes in the pressure on the present state of the ecosystem [9,111]. Gottfried *et al.* [112] have documented that an early response to a long-term negative impact might in fact be a short-term increase in species richness. A naturally changing number of species during a succession where the final and “mature” ecosystem is not the most species rich is a similar challenge [113], raising the question of how many species are there “naturally” in a specific habitat. This makes the selection of a reference point for species diversity challenging.

Related to this is also the question of how to treat invasive species. Methodologies using mere species counts treat invasive species as a positive impact, even though it is well documented that invasive species also are a major threat to diversity [3,6,10]. Some of the proposals that are site specific avoid this problem by excluding invasive species from the assessments [79,89], but this requires detailed knowledge of the original species composition, increasing the data demand. Koellner *et al.* [18] give no recommendations for this question.

3.1.2. Absolute or Relative Changes

A core question when focusing on changes in species diversity is whether the focus should be on absolute or relative changes [67]. Based on the motivation for biodiversity protection as described above, this is an important question, but motivation for the choices made is hardly reflected in the literature. Koellner *et al.* [18] highlight this as a value choice; relative impacts give equal weight to ecosystems, while absolute impacts give equal weight to species. Koellner *et al.* [18] recommend focusing on absolute changes, since these are easier to interpret in their opinion. Furthermore, de Baan *et al.* [95] raise this question and argue that absolute species richness is the preferred choice for avoiding species extinction. Most suggestions do however focus on relative changes and use the term “potentially affected/disappeared fraction” (PAF/PDF) of species (e.g., [65,75–77,91,94,96]). The PAF/PDF approach is also implemented in LCIA tools, such as EcoIndicator 99 [114], ReCiPe [115] and IMPACT2002+ [116].

Exceptions to this are the proposals from Lindeijer [69], Weidema and Lindeijer [117], Schmidt [85] and de Baan *et al.* [92], who focus on the absolute number of species. Furthermore, de Baan *et al.* [95]

indirectly focus on the absolute number of species through their use of habitat suitability models. Mattsson *et al.* [70] also do this, but without a quantified impact assessment. With the exception of de Baan *et al.* [95], there is no thorough discussion in either of the papers on the pros and cons, but all have an underlying “more is better” approach. This choice could be related to the diversity-stability hypothesis; species-rich ecosystems are more stable, but this is not mentioned in any of the proposals using absolute changes. Lindeijer [69] argues that local biodiversity expresses the intrinsic value of a region, and Schmidt [85] suggests a link between the number of species and evolutionary age and that areas with a longer evolutionary time should be given higher scores on biodiversity value. Vogtländer *et al.* [72] argue that the main issue is to preserve “valuable” species, and that a higher quantity might be one strategy to achieve this, while Bare *et al.* [73] only include the number of threatened and endangered species, since their focus is on the potential of species extinction, and thus, they do not take common species into account. de Baan *et al.* [92,95] also focus on potential species extinction and have chosen absolute numbers for global comparisons. All of these implicitly argue that the number of species *per se* is the most important biodiversity quality.

3.1.3. Functional Diversity

An alternative to species diversity is functional diversity [18,93]. Koellner *et al.* [18] argue that functional diversity is the first step towards ecosystem services. Measuring functional diversity as an alternative to species diversity is based on the assumption that loss of species does not necessarily result in loss of functions, since other species with similar functions will fill the potential gap, *cf.*, the redundancy hypothesis [39,42,93]. As a consequence, ecosystem processes might be unaffected by the loss of certain species.

3.1.4. Double Counting of Impacts

Common for almost all methods based on species counts is that they focus on changes in species diversity as a result of changes in land use types or classes (*cf.* [17]). The impact from the (physical) land use change is thus not separated from changes in, e.g., chemical impacts as a result of the use of pesticides, fertilizers, and so on. If these chemical impacts are included in the impact assessment, this involves the danger of double counting, which is barely addressed in the literature.

3.2. Structural and Indirect Biodiversity Indicators as Quality Indicators

As identified in the previous section, there are several challenges in using species counts for assessing changes in biodiversity. A growing body of literature argues that biodiversity can be measured by other methods than counting species, e.g., by structural indicators [23,33,103,118,119]. This is also used in some proposals for the LCIA methodology, which can be seen as a response to the fact that the complexity of biodiversity is not captured by just counting species.

The general assumption here is that it is possible to assess total impact on an ecosystem; how far an area is from being “undisturbed” or “natural” and that this “naturalness” is a desired state. Basically, this is the same idea behind the use of the PDF of species, but without counting species. Wagendorp *et al.* [74] argue that exergy indicators can be used and provide a proposal for how this can

be calculated for different cases. Brentrup *et al.* [71] propose using hemeroby classes and provide scores from zero (no impact) to one (purely artificial) for a number of land use classes. Michelsen [84] suggests an ecosystem-based approach to construct an index for “conditions for maintained biodiversity” where a range of indicators in sum will identify the degree of impact on biodiversity. He suggests combining this with data on ecosystem scarcity and ecosystem vulnerability to get the ecoregion-specific scores (*cf.* [120]), while Brentrup *et al.* [71] suggest using European biogeographic regions for similar purposes.

Such measures on the degree of intactness can also be found in other proposals. The Norwegian Nature Index (NNI) and the indicator set of the German national biodiversity strategy have a similar approach, where pressure on different ecosystems is assessed [25,34], and also, in GloBio3, a mean species abundance factor is used to assess pressure [13]. Certain *et al.* [25] stress that the index should be used to measure changes and that zero impact is not necessarily the goal, but the index nevertheless refers to a “natural” state.

There are also a number of proposals that focus on changes in aspects known to be important for biodiversity and consequently give values for changes in the pressure on biodiversity. Mila i Canals *et al.* [87] focus on water availability as a typical example (included in Table 1). Other examples are Maes *et al.* [55] focusing on water supply, Nuñez *et al.* [121] addressing desertification and Mila i Canals *et al.* [122] focusing on soil organic matter. Even though several of these aspects are relevant for biodiversity [123], no link between these aspects and the actual rate of pressure on biodiversity is proposed in the papers, and consequently, these proposals cannot be used as they are to quantify changes in biodiversity. Other proposals include Garcia-Quijano *et al.* [75], who suggest a number of indicators, including pressure, structural and species indicators, without any further discussion of the relative importance, and Kløverpris *et al.* [82,83], who identify in which biome the impact occurs, but without trying to quantify this impact in terms of biodiversity. Two exceptions to this are Hanafiah *et al.* [124] and Verones *et al.* [125], who both calculate characterization factors based on species loss caused by water use.

A general problem with all of these indicators is to scale the actual impact on biodiversity. Michelsen [84] stresses this problem: how much is needed of different structural indicators before it can be stated that the impact is at a certain level. This is an issue with little scientific background [64], which is a drawback for a direct link to actual impact on biodiversity.

3.3. Geographic Validity

Most proposed methods are based on data from a limited geographic area (Table 1). The methods proposed by de Baan *et al.* [91,92] and Coelho and Michelsen [90] all use global datasets. Furthermore, de Baan *et al.*'s work [95] is based on a global dataset, even though only data for East Africa is given in the paper. However, de Baan *et al.* [91] intend to develop characterization factors for seven land use classes for 14 biomes, and even if this gives very broad classes with high internal variation, the lack of data is still a severe problem; from a theoretical maximum of 98 combinations of land use classes and biomes, characterization factors for only 38 are given, while a substantial number of potential characterization factors potentially provided by de Baan *et al.* [92] is either not available or found not to be significantly different from zero. Coelho and Michelsen [90] provide a method that can be used to generate characterization factors for all areas, but this is based on hemeroby values that are criticized for

a weak empirical basis. Other methods based on species richness are based on data from a restricted part of the world, and it is questionable whether these are applicable outside this area (*cf.* [115]). Other methods [79,81,88,89,93,94], again, are case- or site-specific studies, and even if the methodologies most likely could be used elsewhere, this would require the same access to species distribution data as in these studies.

Both Michelsen [84] and Brentrup *et al.* [71] suggest combining an intactness indicator with a geographic weighting, assigning a higher value (and, if damaged, a higher impact) to rare ecosystems. Brentrup *et al.* [71] only provide relevant data for Europe, while Michelsen [84] has a case-specific intactness indicator that hampers global application of the method. This is suggested by Coelho and Michelsen [90] to be overcome by actually combining the two methods using the generic values from Brentrup *et al.* [71] on intactness and the geographic valuation from Michelsen [84]. de Baan *et al.* [92] and Verones *et al.* [125] use changes in species diversity to assess impact, but also, they include a geographic weighting factor for calculating characterization factors, making the methods similar in structure as the method proposed by Coelho and Michelsen [90].

3.4. Assumptions on Linear Cause-Effect Relations

Most characterization models in LCIA assume a linear cause-effect relationship. For impacts on biodiversity and ecosystem functioning related to biodiversity, this is generally not true [38]; the impact on biodiversity will in most cases be non-linear, and above a certain threshold value, an increase in pressure will cause a higher impact than a linear cause-effect relationship predicts. The importance of such cumulative effects is not always known in advance. Most of the proposed methods use fixed numbers for characterization factors (e.g., [73,76,77,86,91]). With small land use changes, this is most likely not a problem, but with larger changes, this would almost certainly give rise to wrong conclusions. As an example, both de Baan *et al.* [91] and Koellner and Scholz [77] operate with negative values, *i.e.*, regarded as a net benefit, for some land use alternatives, indicating that if all areas in a region were transferred to such activities, this would be beneficial, a fact most would disagree with.

Some proposals focus on rarity (e.g., [72,84,95,125]) and give an increasing weight to more rare species and/or structures, but if characterization factors are to change, this means that these have to be continuously updated or calculated for each case. The latter is done by Coelho and Michelsen [90], who calculated values for ecosystem vulnerability based on present levels of native vegetation in the areas in focus. To some degree, this solves the problem of cumulative effects, but at the same time requires data that often will be hardly accessible.

3.5. Land Use Classes

The focus on land use classes in most of the proposed methods fails to incorporate what Lambin [126] describes as land cover modifications, *i.e.*, changes in land use not large enough to be noted as a land use change, but that still might be equally important as land use change [32,62]. In addition, when the impact is based on species counts, the total impact might in addition be underestimated due to the phenomenon called the extinction debt, *i.e.*, the phenomenon where populations are doomed to extinction due to events that already happened and that are still present [115,127,128]. This might be of particular importance in ecosystems with a high number of long-lived species. The same holds for populations (or

functions in Souza *et al.* [93]) that are so low in number that they are no longer functionally relevant, but still present [129].

4. Discussion

In this paper, we have identified some assumptions made when different biodiversity indicators are chosen. These are also relevant for assessing impacts on biodiversity from LULUC in LCIA, and we have reviewed how these assumptions and choices, implicitly or explicitly, are treated in the LCIA literature.

Even though the body of literature discussing LULUC in LCIA is increasing, there is still a high number of publications avoiding the topic. Cherubini and Strømman [130] found that land use and/or land use changes were included in 28 out of 94 studies on bioenergy, most often as changes in carbon stocks (e.g., [131,132]). Some papers acknowledge the importance of LULUC, but have not included this due to the immature methodology (e.g., [133]), and still, some include it as an assessment of area used without any quality judgment (e.g., [134–136]). Lack of clear recommendations and diverging methodological proposals might be some of the reasons for this.

A problem in the methodological proposals seems to be a lack of focus on the question of why biodiversity should be considered. As discussed in this paper, the different perspectives on this question should give rise to different indicators for biodiversity. This problem is hardly addressed in the LCIA literature. At least the main question of richness *vs.* rareness should be addressed. It can be broken down into two questions.

First, is more always better? There are a few proposals based on this premise, using absolute numbers of selected taxonomic groups as indicators.

Second, is rareness to be valued higher? Here, some of the proposals using species diversity [73,79,92,95] and indirect indicators [84,90] make this assumption, while others do not. This can be seen as a response to the insurance approach [10,41,46,48]: as long as we do not know what is most important, we should try to protect as much as possible of the biodiversity in the broad sense, but this is not explicitly addressed in the papers. When these questions are answered, it is time to start discussing which indicators are possible with present knowledge and data.

Surprisingly little of the ongoing work on biodiversity indicators in general is mentioned in the LCIA literature. International proposals, such as the European Environment Agency proposal [33] and the LPI [36], and similar national initiatives should be valuable inputs in the debate on how to assess biodiversity impacts from LULUC in LCIA, but these are hardly mentioned in the LCIA literature. It might be that inspiration from such a more general biodiversity indicator development is underreported in the papers, but it seems obvious that LCIA development should incorporate the recommendations from this growing body of literature. The NNI [25], the BII [37] and the “Habitat Hectare Approach” [137] all provide values in the range from zero to one and should thus be useful at least if relative changes are in focus.

A choice between different motivations for biodiversity conservation is no doubt a value-laden question. Furthermore, other choices that have to be made are also value laden. The result of this is clearly illustrated if the methods proposed by Schmidt [85] and Michelsen [84] are compared. In a case where a naturally species-poor ecosystem is transformed to a managed ecosystem with higher species richness, this will be identified as a positive impact following Schmidt [85], while the opposite will be

the outcome following Michelsen [84]. Which one of these gives the “right” answer depends on some initial choices [21]. Vogtländer *et al.* [72] argue that “rareness” is a better indicator for botanical value than “richness”, but again, this is not a universally valid argument; it is context dependent.

Due to the different perspectives, it is not likely that it is possible to develop one universal biodiversity indicator covering all aspects; the choice of indicator is dependent on the purpose and initial conditions (*cf.* [30–32]). Not surprising is also that most of the proposed frameworks for biodiversity assessments are a mix of indicators, often a combination of structural indicators and selected indicator species or taxonomic groups [9,25,33]. Due to the lack of correlation between, in particular, species diversity and ecosystem services, it must also be expected that conflicting results will be found in LCA if such aspects are included, just like conflicting results between any other impact categories frequently found in comparative LCA studies [138]. Irrespective of the perspective taken, we will recommend more focus on emerging biodiversity indicator sets, such as the European Environment Agency [33] proposal, potentially supplemented with national initiatives.

The possibility to differentiate between different intensities in land use is a question that must be addressed. Koellner *et al.* [17] recommend using land use/cover classes. As discussed above, this ignores what is normally classified as land cover modification, not large enough to be classified as land use change, but still of importance for a changed impact on biodiversity [32,62]. A few methods, e.g., Michelsen [84] and Mila i Canals *et al.* [87], include such changes, but this will inevitably increase data demand. Coelho and Michelsen [90] demonstrate a trade-off between accuracy and data requirements for a more local adjustment of land use data. At present, no methods with global available data give the opportunity for including land cover modifications; all are based on a pre-defined list of land use classes.

The selection of a reference state has long been recognized as an important question [18]. To our knowledge, very few studies look into the effect of the different choices. Michelsen *et al.* [21] used both natural and present state as the reference for plantation forestry in New Zealand and documented large differences. Whether historic impacts should be included or not is a clear value choice. de Baan *et al.* [95] comes to the same result for agricultural production in East Africa when different reference situations are compared.

In particular, methods based on species counts face another problem: How many species are there actually in the reference state? Which state in a succession should be used; the climax or the most species-rich state? Should species present, but in so low abundance that they are functionally not present, be included? Interestingly, when Elshout *et al.* [94] calculated characterization factors, they could not find any significant difference whether they used data from the same site, the same ecoregions or just the same biome as the reference. Intuitively, this sounds strange, since the variation within a biome is enormous, and the question is thus whether the variation even at the same site is so large that further variation does not influence the final result. This is an obvious question for further research and should be addressed if changes in species diversity are used as an indicator. It should also be mentioned that Chiarucci *et al.* [139] characterize potential natural vegetation (PNV) as a concept that is hard to define and with limited predictive power; still, PNV is often recommended as a reference situation. See also Souza [26] for a thorough discussion of the reference state.

To some degree, the new approach by de Baan *et al.* [95] avoids these challenges. By using habitat suitability models and not species counts as such, the questions of, e.g., abundances and successional stage are less relevant, since it is the habitat, and not the actual presence, that is addressed.

Koellner *et al.* [18] argued that functional diversity could be the first step toward assessing ecosystem services. The concept of functionality does, however, rely on the theory of redundancy, and Loreau [42] questions whether redundancy, even in theory, can exist in a stable ecosystem. The methodology proposed by Souza *et al.* [93] is so far the only one using functional diversity. They compare characterization factors derived from functional diversity and species diversity and get the surprising result that for multiple situations, the characterization factors based on species diversity is smaller than the characterization factors based on functional diversity. If there were a redundancy, this should not be a possible outcome, since loss of function always should be related to loss of species, while loss of species should not always result in loss of function. This question is not addressed by Souza *et al.* [93], and it seems that the concept of functionality still is too immature to be used [42].

The issue of “invasive species” is addressed above, but we lack a clear definition of what an invasive species actually is in order to operationalize this issue. Defining a species as “invasive”, *i.e.*, unwanted in a certain context, is a normative decision. In addition, species that have invaded ecosystems within human history are accepted today (e.g., storks, which could only invade Central Europe after large stretches had been cleared). Others have been deliberately introduced and achieved widespread acceptance over time (e.g., raccoons, corn and potatoes in Europe). It is also an arbitrary decision, since species are invasive by nature, but this insight does not limit the value of having such a definition. This might also be a problem of limited magnitude. It could be of importance in areas with few species and in highly artificial areas (gardens, *etc.*), where most species will be invasive, but in other ecosystems, there are reasons to believe that the new species will be low in number compared to species that will disappear due to the invasion [12]; the consequence of whether they are counted or not might be more of a theoretical interest.

Wagendorp *et al.* [74] claimed that the proposed indicators seemed to be chosen more or less arbitrarily with little ecological basis, and Geyer *et al.* [89] used four different methods to score the included habitats based on case-specific data on terrestrial vertebrates, focusing on naturalness, species richness, abundance and evenness. They found that even if they used the same species, the choice of method influenced the conclusions of land use impact and concluded that at present, none of the proposed methods can be used as a “suitable, meaningful, and scientific (...) biodiversity indicator”.

We will argue that there has been significant progress over the last few years, and it can no longer be claimed that the chosen indicators are arbitrary. However, the methods still give contradicting results [21,95], which is not acceptable if the reason for this cannot be identified. We have tried to argue that the reason is, at least partly, buried in the choices made for the selection of indicators, but these are in most cases not explicitly addressed in the literature.

Based on the complexity in biodiversity, it is unlikely that all aspects of biodiversity can be captured in a single indicator. This also holds true when land use impacts on biodiversity are to be included in LCIA. The absolute number of species is, in a given situation, important, but this cannot be reflected with the same indicators that assess biodiversity for providing ecosystem services and indicators for assessing rareness which also are important aspects [13]. The question then is: what is to be included, what is not to be included and which value choices are guiding these decisions? We call for more

openness and awareness on this. We also advise for a much higher focus on the development of biodiversity indicators and indexes outside the LCIA community.

We find it interesting that an increasing number of papers try to combine a pressure indicator, whether it is based on species counts [92,125] or structures [84,90], with a geographic weighting factor. It is also interesting that Michelsen *et al.* [21] find a clear correlation in the results based on the proposal from de Baan *et al.* [92] and Coelho and Michelsen [90] on a case study on forestry plantations. These methods are based on different indicators, but are similar in structure, with a pressure factor and a geographic weighting factor. It is too early to tell whether this is a general pattern, but a very clear recommendation from an expert workshop stresses this combination of pressure and geographic weighting as important for further methodology development [140]. As an example, Freudenberger *et al.* [141] show different aspects that can be included for geographic priority settings.

We have not included the growing body of literature concerning the economic value of ecosystem services and biodiversity (e.g., [57,142,143]). It is far too early to judge, but in time, monetization of aspects related to biodiversity might be a possible approach for comparing different aspects of biodiversity in indicator development; however, due to its immaturity, this is not included here.

5. Conclusions

In this paper, we review and compare the indicators proposed for the inclusion of land use impacts on biodiversity in LCIA. We try to understand the underlying normative assertions and highlight the relevance of indicator choices for the result of the impact assessment. Following from that, we would like to close with these points:

- (1) Progress has been made in method development for including impacts on biodiversity from LULUC in LCIA over the last few years, but still, no consensus has been reached.
- (2) It is unlikely that all aspects of biodiversity can be included in a single indicator. More focus is needed on the choices made for selecting the identified indicators; different indicators will assess different aspects of biodiversity, and the transparency of these choices and the rationale for them are needed.
- (3) We encourage the LCA community to pay more attention to indicator development for biodiversity in general and to try to use these in LCIA instead of generating new indicators.
- (4) A number of methodologies combine a pressure indicator with a geographical weighting factor. We find this promising, since it combines important aspects for biodiversity.
- (5) The consequences of selecting different reference states are not understood well enough and should be a prioritized area for further research.

Biodiversity is much less tangible than other impact categories. Any act of perception, any definition of an object is subjective, but there is a fair amount of consensus concerning other impact categories in LCIA, e.g., the relevance of greenhouse gas emissions for Earth's energy balance is hardly questioned anymore. Biodiversity, *au contraire*, is more of a mind construct: there are organisms, and we group them into species, then wonder about the multitude of species. We may interpret an ensemble as "species rich" or "species poor", but this will be based on a chosen reference situation. This is not a problem, but we as scientists should acknowledge this special characteristic of biodiversity.

Author Contributions

The manuscript was written through contributions from both authors. Both authors have given approval to the final version of the manuscript.

Conflicts of Interest

The authors declare no conflict of interest.

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