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Towards inclusion of Biodiversity in Life Cycle Assessment

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TOWARDS INCLUSION OF BIODIVERSITY IN LIFE CYCLE ASSESSMENT

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ABSTRACT

Biodiversity- crucial for ecosystem health and its products and services – is being lost at an alarming rate. While it is clear that human consumption is the main driving force of the considerable losses, conversion of natural habitats for production is continuing and the subsequent intensification of those systems is likely to cause even further biodiversity decay. Insights in consumption-based biodiversity loss, or biodiversity footprints, offer starting points for policy to reduce global biodiversity loss. To assess environmental performances of production systems, Life Cycle Assessment (LCA) can be applied. This is an internationally recognised methodology to map the environmental impact of a product, chain or activity from the beginning to the end of the life cycle. While certain impacts, such as those of climate change, are readily incorporated into standardised LCAs, methodologies for impacts on biodiversity are still being developed.

This thesis aims at contributing to improved methodology for assessing the impacts of land use and land use change on terrestrial biodiversity in LCA. Particular attention is given to what is needed to enable more societally and ecologically relevant assessments of impacts on biodiversity and how biodiversity indicators can be developed that quantify impacts on biodiversity accordingly. The results show that in current life cycle impact assessment (LCIA) models, the use of so-called baseline references dominates, implicitly striving for 'naturalness' and how this contrasts the aim of biodiversity conservation frameworks. Furthermore, it is discussed how inclusion of genetic attributes could increase the relevance of current assessments and it is tested how LCIA modelers could make use of genetic data generated by metabarcoding approaches of environmental DNA.

Recommendations given include the development of reference situations in LCIA models based on biodiversity targets aligned with society's conservation frameworks. This means that impact on biodiversity will be defined as a distance to a target measure, rather than impact on the 'natural'. In addition, next steps are identified that are needed to include genetic biodiversity metrics in LCIA models.

Keywords: Biodiversity, Life Cycle Assessment, LCA, Land Use, Biodiversity conservation, Indicators, Impact assessment, Sustainability, Reference situation, eDNA, Metabarcoding

LIST OF INCLUDED PAPERS

I. Vrasdonk, E., Palme, U. & Lennartsson, T. (2019) Reference situations for biodiversity in life cycle assessments: conceptual bridging between LCA and conservation biology. *Int J Life Cycle Assess* **24,** 1631–1642. doi:10.1007/s11367-019-01594-x

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RELATED CONTRIBUTIONS

A. Vrasdonk, E., Palme, U., Lennartsson, T., Antonelli, A., Berg, S., Jonsson, A., Cederberg, C. (2016) Defining the reference situation for biodiversity in life cycle assessments: Review and recommendation. *Proceedings of the 10th International Conference on Life Cycle Assessment of Food,* Dublin, Ireland, October 19-21, 2016.

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"Taste" is a noun and a verb: We all have it and we all do it. But we don't all have a language or a system for understanding and expressing that experience… I knew chocolate was something I didn't want to lose, but I didn't have the words to communicate why it was so important to me, or the knowledge on how best to save it. Now I do."

— Simran Sethi

1. INTRODUCTION

Life on earth is threatened. During the last centuries and more dramatically in the last four decades, natural habitats were destroyed at a fast pace and subject to high levels of pollution, overexploitation, invasive species and climate change mainly provoked by man-induced greenhouse gas emissions (IPBES 2019). As a result, natural ecosystems have declined by 47 per cent on average, approximately 25 per cent of species are threatened with extinction and the global biomass of wild mammals has fallen by 82 per cent (IPBES 2019). It is generally accepted that the diversity of life is in need of protection. To denote the diversity of the somewhere between 1 and 6 billion species inhabiting the earth (Larsen et al. 2017) the term "biodiversity" was introduced. As a contraction of “biological diversity”, the word was first mentioned by Walter Rosen for the 1986 National Forum on BioDiversity (Wilson 1988). It encompasses all forms, levels and combinations of natural variation and should therefore be regarded as a broad unifying concept. Biodiversity is defined 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" (United Nations 1992). This means that biodiversity not only includes species lists, but also the pattern and processes associated with life on earth, including the diversity of all biotic systems: organisms, populations, communities, ecosystems, landscapes and biomes.

In the race to combat further loss of biodiversity, there is urgency for increasing conservation worldwide. In cooperation with the United Nations Environment Program (UNEP), the Convention on Biological Diversity (CBD) set out the ‘Strategic Plan for Biodiversity 2011-2020’, in which the vision is to restore, value and conserve biodiversity for the benefit of all people by 2050 (UN CBD 2010). Embedded within this plan are 20 so called ‘Aichi Biodiversity Targets’, which are organised under five major strategic goals:

- Strategic Goal A: Address the underlying causes of biodiversity loss by mainstreaming biodiversity across government and society
- Strategic Goal B: Reduce the direct pressures on biodiversity and promote sustainable use
- Strategic Goal C: Improve the status of biodiversity by safeguarding ecosystems, species and genetic diversity
- Strategic Goal D: Enhance the benefits to all from biodiversity and ecosystem services
- Strategic Goal E: Enhance implementation through participatory planning, knowledge management and capacity building

With the end of the Strategic Plan for Biodiversity in sight, the strategic plan and associated targets will officially be reviewed in 2020. Evidence so far suggests however, that most of the targets will not be met due to insufficient progress towards- or even movement away from them (Tittensor et al. 2014; CBD 2019; Green et al. 2019). The limited number of government parties implementing biodiversity strategies and action plans shows the urgency to significantly accelerate efforts to implement the Strategic Plan for Biodiversity across governments (CBD 2019). CBD therefore stresses the need for not only political and financial support, but also technical support, technology transfer and capacity-building to help parties in taking action (CBD 2019). Knowledge generation has a central role in this process and is considered a direct action to accelerate progress towards achievement of the targets. It includes, inter alia, the generation of, and access to, biodiversity information, a better integration of social sciences to account for different visions and knowledge systems, and a better consideration of direct and indirect impacts of policies and production and consumption patterns on biodiversity.

Life Cycle Assessment (LCA) presents a practical tool to generate part of the knowledge requested by CBD and directly addresses Strategic Goal B of the Aichi Biodiversity Targets (Winter et al. 2017). It is an internationally recognized methodology to map the potential environmental impact of products, production chains or activities from the beginning to the end of their life cycle (ISO 2006). LCA could provide answers in comparative analyses ("is product A better than product B?") and allows the identification of hotspots (in LCA terminology defined as the part of the product life cycle with particularly high impacts) along the life cycle of a product. In this way, LCA results enable better informed decisions related to environmental performance of products, and is utilized by product designers, consumers and policy makers. Examples of uses include environmental optimisation of production systems, environmental labelling of products and the development of more targeted policies to diminish adverse impacts on the environment, including biodiversity (Winter et al. 2017).

Although LCA is commonly used to assess e.g., climate change, human toxicity, acidification and eutrophication, impact on biodiversity is as yet seldomly included in standard LCA due to methodological limitations and data scarcity (Helin et al. 2014). Given the urge of biodiversity conservation and the importance of LCA as an environmental decision support tool, the scientific community is pressed to provide methods in order to examine how production systems impact biodiversity and define how biodiversity should be measured in this kind of assessments. Many authors developed indicators and methods to include different attributes of biodiversity in LCA (See inter alia Michelsen 2008a; Schmidt 2008; Curran et al. 2011; Souza et al. 2013; de Baan et al. 2013a, c; Jeanneret et al. 2014; Chaudhary et al. 2015; De Baan et al. 2015; Teixeira et al. 2016a; Winter et al. 2018; Lindner et al. 2019). Despite the progress made in the research field, these models are still under development and due to lack of consensus on which model to use, many LCA studies do not consider impacts on biodiversity (Winter et al. 2017; Tang et al. 2018).

The main challenge in the development of Life Cycle Impact Assessment (LCIA) models for biodiversity remains to describe biodiversity in an ecologically relevant manner, including measures of multiple biodiversity attributes and levels. So far, LCIA models on biodiversity mainly address the impacts on terrestrial diversity caused by land use and land-use change. The dependency on widely available biodiversity data is an important bottleneck in this context, arising from the need of LCA models to support assessments of production systems which are increasingly globalized (Frischknecht et al. 2016). This has led to the inclusion of often simplified biodiversity metrics in these kinds of models, mostly based on numbers of species (species richness or alpha (α) diversity), (Winter et al. 2017). Considering the wide range of biological features with distinct attributes (ecological composition, function, and structure) and the multiple levels of organisation of biodiversity (genetic, species, population, community, and ecosystem), solely compositional measures such as the number of species do not adequately portray biodiversity. The ideal biodiversity indicator for LCIA models would however not only catch its complexity and the spatial and temporal characteristics of its attributes; it would also be easy to measure and simple to communicate (Curran et al. 2011a; Frischknecht et al. 2016). It may be clear that this is a considerable challenge and some simplification is inevitable. Although it must be acknowledged that all aspects may not be possible to include, a multidimensional approach would contribute to a more ecologically relevant and 'meaningful' biodiversity assessment in LCA. Of the models currently available, indicators at community and ecosystem level are most common, although there are several models addressing species-level effects (such as population size) and vulnerability. Most of these indicators address the composition of the level they are assessing, in the form of species richness. Models addressing functional attributes are rare, two examples that were found include trophic relationships (Jeanneret et al. 2014) and functional diversity (de Souza et al. 2013). Structure is integrated in the 'hemeroby' concept; a measure of the 'naturalness' of land (Brentrup et al. 2002). Apart from these, the model by Michelsen (2008a) builds on dead-wood as a prerequisite for biodiversity. Models including genetic components of biodiversity are virtually absent (Curran et al. 2016; Winter et al. 2017; Maier et al. 2019).

Besides a more ecologically relevant measure of biodiversity, Teillard et al. (2016b) point out that LCIA models also need to better link land use to biodiversity, as the models covering a large geographical scale often lack precision in terms of spatial differentiation of biodiversity and management practices. In addition, impact assessment models need to include a wider range of pressures on biodiversity. Although habitat change due to land use is being considered the most important driver of terrestrial biodiversity loss (Millennium Ecosystem Assessment 2005; IPBES 2019), climate change, invasive species, overexploitation of resources and pollution are also known to cause considerable impacts (Millennium Ecosystem Assessment 2005). Some examples of efforts made to include other drivers than habitat change include the model of Verones et al. (2017) assessing the impacts of water use on biodiversity, van Zelm et al. (2007) addressing acidification, De Schryver et al. (2009) linking climate change to damage on ecosystems, and Hanafiah et al. (2013) proposing a model to assess impacts of invasive species. More examples can be found in the review by Winter et al. (2017).

Many authors have emphasised the need for an extended interdisciplinary approach to involve LCA and ecology in the development of LCA methods for biodiversity (Zhang et al. 2010; de Baan et al. 2013a; FAO 2016; Teixeira et al. 2016; Teillard et al. 2016b), to increase the ecological relevance of LCA models for biodiversity and align them with conservation targets. This would make LCA results more meaningful to society as a whole. 5 specific domains of collaboration, as identified by Teillard et al. (2016b), would include:

(1) ecological concepts, such as landscape level processes (effects of habitat fragmentation or landscape heterogeneity), reference situations (which are needed to quantitatively measure impacts of pressures on biodiversity) and functional ecology (such as species' functional traits that define species in terms of their ecological roles – how they interact with the environment and with other species)

(2) local designation frameworks and information, which should be used to reflect how biodiversity and impacts vary with location and indicate that conservation value and priorities can differ between locations

(3) data, as many datasets of different kinds are currently not used by LCIA models' developers, but could lead to improvements of biodiversity integration

(4) models, many LCIA models already rely on ecological models such as species-area relationships or habitat suitability models. There are other models used in (agro-)ecology, such as models linking the effect of agricultural intensity to biodiversity changes, that could strengthen impact assessment modelling in LCA

(5) interpretation; as (over)simplified biodiversity metrics might lead to ill-informed decision making, calling for guidance in the interpretation of biodiversity LCA results

1.1. RESEARCH QUESTIONS AND PROJECT OBJECTIVES

The general purpose of this thesis is to contribute to an improved methodology for assessing terrestrial biodiversity impacts in LCA, limited to the assessment of land use and land use change impacts. As a step towards achieving the general purpose of this thesis, I aim to investigate what would be needed to enable more relevant assessments of impacts on biodiversity and how we can develop biodiversity indicators that quantify impacts on biodiversity accordingly. I approach this aim by answering two research questions and related sub-questions:

1) How does current use of reference situations relate to the purpose of the assessment?
    a) How does the reference situation relate to the rationale of the assessment?
    b) How are reference situations currently used in LCIA models?
    c) What does the current use of reference situations mean for the rationale of LCIA models?
2) How can LCIA models be improved to make them societally and ecologically more relevant?
    a) How to align models with the purpose of biodiversity conservation?
    b) Why would genetic aspect be important for ecological relevance?
    c) What would be needed to include genetic aspects in LCIA models?

Given the aim of this thesis, I intend to make a conceptual contribution by assessing how LCIA models could be developed to better fit biodiversity conservation practices. I also aim to develop and test new biodiversity indicators which would allow for more inclusive and complete biodiversity assessments.

My research is presented in two appended articles. Article 1 is about reference situations for biodiversity in LCA. It explores the first research question by identifying the current usage of reference situations both within and outside LCA, how the reference situation relates to the rationale of the assessment. Furthermore, we argue which goal biodiversity assessments in LCA would preferably serve and provide recommendations accordingly on how reference situations should be developed to align with the purpose of biodiversity conservation.

Article 2 aims to contribute to the need for more inclusive biodiversity indicators, in particular the genetic aspect of biodiversity. The article is an explorative study on how LCIA modelers could make use of genetic data generated by metabarcoding approaches of environmental DNA, and what is needed to be able to include genetic biodiversity metrics in LCIA models.

2. THEORETICAL AND METHODOLOGICAL FOUNDATIONS

2.1. GENERAL FRAMEWORK OF LIFE-CYCLE ASSESSMENT

The increased awareness of the importance of environmental protection, and more specifically the possible impacts associated with the manufacturing and use of products, has led to the development of tools to assess and understand these impacts. Life Cycle Assessment (LCA) is a widely used tool nowadays, designed to evaluate the potential environmental impacts of a product system throughout its life cycle (e.g. the acquisition of raw materials, through production, use of the product, end-of-life treatment such as recycling and final disposal) (ISO 2006). The LCA procedure and requirements for conducting LCA are standardized and outlined in International Standards 14040 and 14044, established by the International Standard Organization (ISO) (ISO 2006). LCA consists of four phases:

- Goal and scope definition
- Inventory analysis
- Life cycle impact assessment
- Interpretation

At the core of the LCA is the functional unit, which expresses the product(s) function(s) and serves as a common quantified reference unit that enables comparison between alternative systems fulfilling the same function. The functional unit, as well as the reference flow (the amount of product(s) required to fulfil the functional unit) and the system boundaries are established in the first phase: goal and scope definition. In the Life Cycle Inventory analysis (LCI), inputs and outputs to or from the environment, called elementary flows, of the product system, are quantified through data collection (such as energy inputs, waste, emissions to air) and data calculation related to the reference flow. In the Life Cycle Impact Assessment phase (LCIA), the results of the LCI are assigned to particular areas of environmental concern called impact categories, which are quantifiably represented by an impact category indicator (**Fig. 1)**. For each elementary flow assigned to an impact category, the amount is multiplied with a so called characterisation factor (Hauschild and Huijbregts 2015). This characterisation factor is expressed in a metric common to all contributions within a specific impact category and gives the quantitative representation of its importance within this category (Hauschild and Huijbregts 2015). It is thus an equivalency factor which provides a common ground for comparison of different elementary flows within an impact category and allows us to sum up all the different contributions of different elementary flows into one final impact result. These characterisation factors are derived from characterisation models, which are often complex models developed separately for each impact category. In short, characterisation factors are equivalency factors used to convert the LCI results into indicator results summed up for each impact category. In this way, the LCIA maps the potential life cycle environmental impacts of the production system (ISO. 2006).

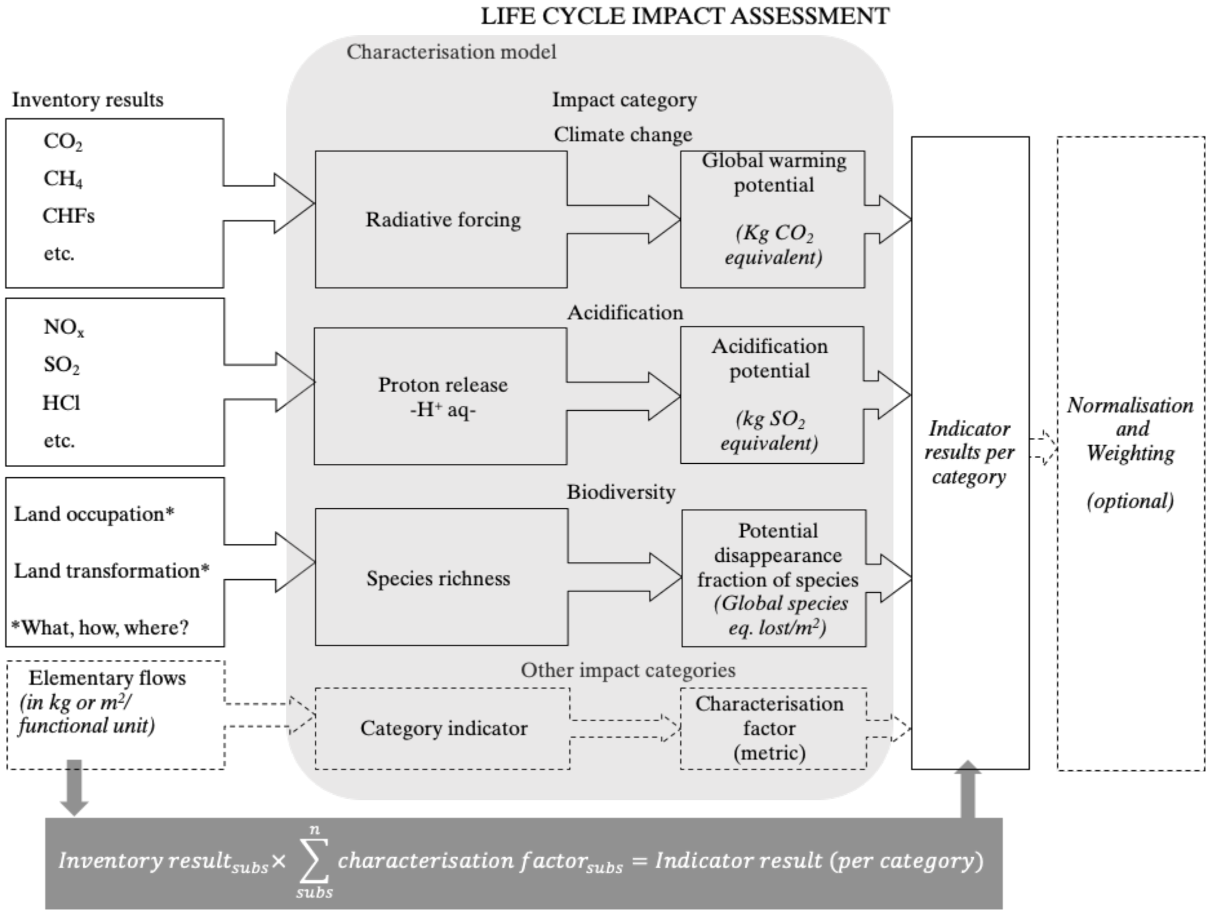

Figure 1: The structure of the LCIA. Note that for biodiversity, no consensus exists on which model to use.

2.2. INTEGRATING LAND USE IMPACTS ON BIODIVERSITY IN LCA

Regardless the definitions of biodiversity or indicator, four steps are required to be able to translate the impact of land use on biodiversity to a method suitable for LCA (Lindeijer et al. 2002). These steps can be treated in different ways, as will be explained below.

- A specification of the interventions in the LCA inventory (how to define the elementary flows impacting biodiversity)
- The choice of indicators to measure biodiversity

- The assessment of the impacts on biodiversity of different kinds of land use
- The translation of these impacts into characterization factors

2.2.1 Interventions in the LCA inventory

As a starting point for including land use impacts in LCA, a flagship project entitled 'Environmental life Cycle Impact Assessment Indicators' carried out by the UNEP-SETAC Life Cycle Initiative, established a general framework for incorporating land use impacts in LCA (Koellner et al. 2013). In accordance with this UNEP-SETAC framework, two types of land use interventions are usually considered in life cycle inventories and impact assessments; land transformation (also called Land Use Change, LUC) and land occupation (Land Use, LU) (Lindeijer 2000; Milà I Canals et al. 2007 among others). During land transformation, a piece of land ('natural' or already in use) is transformed to another cover type to suit the envisioned use. The time for transformation itself is usually very short, thus the time it takes to transform the area is often set to zero. Occupation of land is defined as the use of the land as envisioned and during which reversal or regrowth of the (semi-) natural status of the land is postponed (Weidema and Lindeijer 2001; Koellner and Geyer 2013).

Assumptions of the framework are as follows: If land after transformation is not occupied and thus not maintained, a state more similar to the original will slowly develop due to natural processes (Milà i Canals et al. 2007). It could be assumed that after a certain regeneration time, the land quality of the initial situation would be fully re-established. Both earlier natural states and earlier land-use-generated states, may however be difficult to reach in land that has been heavily changed. It is more likely to reach a steady state of which the quality deviates from the earlier one, or the regeneration time could be longer than the modelling horizon of the LCIA. In both of these cases the changes in quality may be regarded as permanent impacts (Weidema and Lindeijer 2001; Koellner and Geyer 2013; Koellner et al. 2013). The theoretical framework for assessing impacts of land use interventions on land quality as a function of time is shown in **Fig. 2** (note that in this case, the quality refers to biodiversity, but this framework has been used for other land-use related impacts as well- such as impacts on soil organic carbon). The figure should be interpreted as follows: At time *t1* land is transformed to another land use type. This change results in a drop of the quality of the land from *Qref* to *QLU1*. At time *t2*, the area is transformed again for another purpose and the quality drops from *QLU1* to *QLU2*. At *t4*, the land lies fallow since it is not in use anymore and recovers. After the recovery time *tLU2*, reg, there will be established a new steady state at *t5*. This new steady state *Qref2* is not necessarily the same as the original quality and could be either of lower or higher quality than before. In this example, the quality *Qref2* of the land does not reach the initial reference quality (Lindeijer et al. 2002; Koellner et al. 2013).

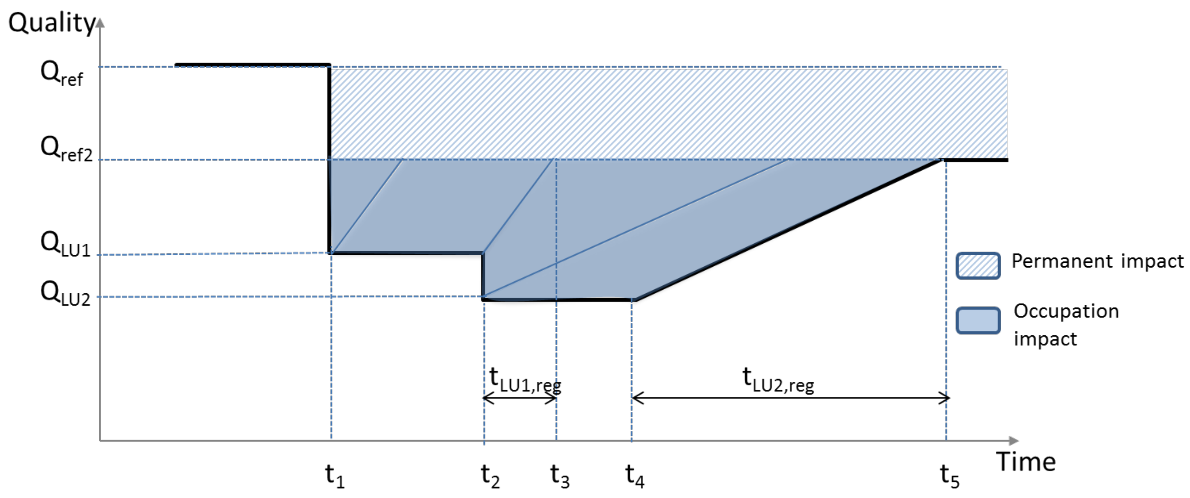

Figure 2: Evolution of land quality with land use interventions (adapted from Koellner et al. (2013).

2.2.2 The choice of indicators to measure biodiversity

Biodiversity levels are measured according to a chosen indicator, which can be used to calculate characterisation factors for an impact category. Since biodiversity manifests itself at several levels (species, genes and ecosystems) and through different attributes (variety within the levels, quantity or distribution) (EASAC 2005), indicators can differ since they address different aspects of biodiversity. They could be based on physical, chemical, or biological parameters of organisms, ecosystem functions, or structures (Miller et al. 2012). α diversity or species richness, which is the number of different species in a habitat or region, is the most frequently used aspect for biodiversity indicators (Winter et al. 2017). Some other measurements include β diversity (or ecological diversity), which is the extent to which different locations or habitats differ from one another and says something about inter alia the connectivity and history of the locations. Y diversity, finally, is the overall species diversity of several habitats or locations together, on a larger scale (Whittaker 1972; Souza et al. 2015).

Except from well-known taxa at very well-known locations, it is almost impossible to count every species in a certain area. Measurements of biodiversity must therefore rely on adequate sampling to estimate the actual or 'true' biodiversity. For this reason, it's common to use surrogate species, which are species which should represent the total species richness of an area (Souza et al. 2015). The first methods developed for the assessment of biodiversity in LCA mainly employed the number of vascular plant species as an indicator, mainly because the data availability of vascular plant species is one of the highest (Lindeijer et al. 2002). While the awareness grew that just one taxonomic group is not sufficient as an indicator for total species richness (Lindeijer et al. 2002; Michelsen 2008b; Schmidt 2008), the diversity of indicators for capturing biodiversity increased.

Not only do models currently include multiple taxonomic groups like amphibians, birds, mammals and reptiles, they also have the tendency to recommend the use of multiple indicators simultaneously (Vrasdonk et al. 2019). The list of suggested indicators for biodiversity in LCIA now includes ecological scarcity (ES), ecological vulnerability (EV) and structural indicators (Michelsen 2008a), functional diversity (de Souza et al. 2013), and indicators based on expert knowledge (e.g., Jeanneret et al. 2014; Lindner et al. 2014; Lindqvist et al. 2015). Indicators capturing the genetic level of biodiversity remain few (Winter et al. 2017).

Despite the variety of indicators proposed, only few models have been tested as data availability limits the practical implementation of many indicators. This has for example been a hindrance for the implementation of functional diversity as an indicator, as data collection on functional traits (those that define species in terms of their ecological roles - how they interact with the environment and with other species) and species composition across land use types is very difficult. Moreover, global databases which include these functional traits do not yet exist (Souza et al. 2013). Data availability for species richness is greater than for other biodiversity indicators (de Baan et al. 2013a) and has pragmatically led to the most commonly used metric for biodiversity in LCA. It includes limited information on many aspects of biodiversity though. For example, species richness does not account for species abundance (the number of individuals per species) and weights species equally regardless of their ecosystem function. As an indicator it also lacks information on differences in species composition, i.e. what kind of species that are found in different habitats. Moreover, the results of this method are highly dependent on sampling effort, which leads to uncertainty regarding the robustness of results.

2.2.3 Assessment of impacts on biodiversity of different kinds of land use

To be able to define the decrease or increase in biodiversity as a result of land use, the land should be compared with an earlier defined reference **(Fig. 2)**. The term *reference situation* itself is a broad concept, describing any starting point against which we can quantitatively compare another situation (Frischknecht et al. 2016). In the case of LCIA of land use, the biodiversity impacts of an intervention can be quantified as the difference between the quality of the land as a result of its use, and the reference situation. However, this reference situation could be a point in the past, present, or future (de Baan et al. 2013a), and its choice could imply different results and conclusions (Michelsen et al. 2014; de Baan et al. 2015; Nordborg et al. 2017).

The most commonly used reference situations so far are so-called baseline (or 'natural') references (Vrasdonk et al. 2019; **Paper I**), i.e., a situation that describes a human-free situation. Examples are historical "natural" situations, natural counterfactual situations, or re-naturalisation situations. An often used concept in LCIA models for biodiversity is the PNV (potential natural vegetation), which is a state of mature vegetation in its expected form, when the land is not being maintained or intervened upon by human society (Chiarucci et al. 2010). There are however many options and the goal of an LCA may lead to different value judgments as to which reference situation is best (Koellner et al. 2012) (as also described in **Paper I**). Still, some form of consensus would be highly desirable as the non-alignment of models complicates comparison between characterisation factors derived by different LCIA models.

2.2.4 The translation of impacts into characterisation factors

For impacts of the land use intervention to be defined in relation to the reference situation, characterisation factors need to be defined (See section 2.1 and **Fig. 1).** The factors can translate land use interventions into contributions to impact categories (Guinée and Heijungs 2005). As a simple example, part of the inventory results of 1 kg of hypothetical product X could be;

40 m^2 pastureland, extensively managed, in Sweden

25 m^2 agricultural land, intensively managed, in Sweden

2 m^2 agricultural land, intensively managed, in Brazil

If the characterisation factor for pasture land, extensively managed, in Sweden per m^2 = 0.02, that of agricultural land, intensively managed, in Sweden per m^2 = 0.04, and agricultural land, intensively managed, in Brazil per m^2 = 0.05, the total impact is 1.9 – possibly expressed in Potential Disappearance Fraction (PDF) of species (40x0.02+25x0.04+2x0.05).

This structure comes with certain challenges for LCA characterisation models that link land use to biodiversity loss. Trade-offs exist between the models' geographical coverage (e.g. local, regional or global), their degree of spatial differentiation (e.g. ecoregions or biomes) and their definition of land-use classes and management practices (Maier et al. 2019).

2.2.5 Summarising

In the last couple of years, great effort is seen in the development of new LCIA methods to include biodiversity or impacts from land use in LCA. Although most methods solely rely on species richness as an indicator, a trend is visible towards more encompassing assessment methods which include more taxonomic groups and accounts for inter alia functional traits of species. However, global data availability has shown to be too poor and is a major limiting factor in the choice of impact assessment model.

2.3 SOLVING THE 'RIGHT' PROBLEM – SUBJECTIVITY OF THE PROBLEM DEFINITION

Biodiversity is composed of a myriad of components in dynamic relationships. The more relationships and the more dynamic the relationships, the healthier the system. This contrasts the need for simplification inherent to LCA and toughens efforts to capture biodiversity in a way that suits fast and global assessments. A central question that has to be solved to include biodiversity in LCA is: how to combine impacts on biodiversity occurring at different steps of the supply chain and at different geographical locations? If considering the transformation or use of one hectare of habitat, the biodiversity impact strongly depends not only on the species composition and habitat type, but also on how biodiversity is measured. As more extensively discussed in **Paper I**, measures of what is considered 'high' biodiversity depend on a human-derived value system that gives a higher conservation priority to some specific components of biodiversity, species and habitats (FAO 2016; Teillard et al. 2016a). Priorities could for example differ depending on the history of the region, the local use of specific species under threat or one's perspective on biodiversity. That given, a model builder aiming at including biodiversity impacts in LCA, has to look at the political and societal context. Doing so, he or she will realize that no single biodiversity 'optimum' exists and by trying to establish one, trade-offs related to different value perceptions of biodiversity are inevitable.

The subjectivity of the problem definition, the absence of a 'true' solution and the difficulties establishing cause-effect relations between anthropogenic activities and biodiversity are only a few signs of the wicked nature of the problem. 'Wicked' problems, as first described by Rittel and Webber (1973), describe a set of problems that are complex, poorly understood and resisting clear definition. They involve many stakeholders which perceive the problem differently, partly due to different sets of values and they often have conflicting interests with other stakeholders. As new information becomes available, or conditions change which influence the problem, the understanding of the problem by stakeholders will change accordingly. As a result, beliefs about possible solutions depend not only on what is already known, but also on every new question that is asked about the problem – and each question is a function of the current understanding. Although there is no quick fix, - no glib formula about 'Seven Steps to Crush Complexity' or 'Tame Your Way to the Top' as Conklin and Christensen (2009) put it so aptly, there might be some ways to handle wickedness. One should however be careful not to treat the problem as 'just like' a previous problem that has been solved. Although simplifying problems is a natural and common way of coping with it, attempts to tame a wicked problem will always fail in the long run (Stichler 2009).

It should be clear that every model that is developed reflects the modelers' understanding of the problem and their values of nature and biodiversity. This poses a risk when models which become well known and accepted are used in other contexts, not considering the purpose for which is was originally developed. If neglecting to acknowledge the purpose as an important characteristic of a model, the risk increases that one also neglects to consider the reason for including or excluding certain assumptions that constitute the backbone of the model. Thus, for an LCIA indicator to be effective, a clear definition of the perspective taken in an LCIA model is important for transparency. The perspective on biodiversity refers to the role and context in which biodiversity is seen, and the motivation behind protecting biodiversity. Equally important to a specification of the perspective underlying a model, is a model that reflects the chosen perspective.

3 RESEARCH APPROACH

This thesis was based on two studies presented in the form of appended articles. Although both studies are framed within the research project's general aim and explore the possibilities to develop a life cycle impact assessment model for land use impacts on biodiversity, they differ in research focus. The analysis in **Paper I** not only aimed to contribute to 'one piece of the puzzle', the reference situation, but also to express -and contribute to the understanding of- part of the fundamental value system of the authors, used in subsequent research efforts. **Paper II**, in contrast, was a more practical study, in the form of an explorative pilot study using genetic data for constructing indicators for biodiversity. This thesis is a synthesis of what was learned from both studies, informing the conclusions and suggestions for future research.

Paper I appended to this thesis addresses one of the current research gaps in LCIA models for biodiversity; the reference situation, with the objective to contribute to an improved methodology for assessing biodiversity impacts in attributional LCA. The article was designed as a conceptual study, based on a descriptive review of the current use of reference situations for biodiversity in LCIA models as well as how such are defined in biodiversity conservation. To evaluate the conceptual understanding of reference situations and their implications, (1) strengths and weaknesses of currently used reference situations in LCA were defined, (2) the relationships between the use of reference situations and the aims of biodiversity conservation were explored, and (3) an analysis of how LCA could be developed to better align with biodiversity conservation practices was performed.

Paper II is an explorative study on how LCIA modelers could make use of genetic data generated by metabarcoding approaches of environmental DNA. The aim was to investigate its potential for use in LCA, identify research needs and provide recommendations on how to take the inclusion of genetic biodiversity metrics in LCIA models forward. For this purpose, a case study was set up in Sweden where environmental DNA samples were taken and analysed. Besides a taxonomic assessment, different biodiversity indices were tested on the data, including both α diversity and β diversity indexes. More detailed descriptions of the methods used in Paper I and II can be found in the papers appended.

4 RESULTS AND DISCUSSION

In line with the overall aim of this thesis, two studies have been conducted, focusing on the reference situation for biodiversity in LCIA models and the development of genetic biodiversity indicators. In the following text, the aim and the research questions of this licentiate thesis will be further explored by drawing on the abovementioned papers. Research question 1 (a,b and c) and 2a are answered with the main results from **Paper I**. To answer research question 2b, the discussion drew on, and was extended beyond, findings in the paper. Lastly, under 2c the results of **Paper II** added a more practical view and recommendations towards the implementation of genetic indicators in LCA.

4.1 RQ1: HOW DOES CURRENT USE OF REFERENCE SITUATIONS RELATE TO THE PURPOSE OF THE ASSESSMENT?

a) How does the reference situation relate to the rationale of the assessment?

The reference situation represents a situation against which land use and land transformation interventions can be evaluated through some measure of change in biodiversity. Thus, the biodiversity impacts of an intervention are quantified as the difference between the quality of the land as a result of the intended use and the reference situation (See also **Fig. 2**). The more the intervention under study departs from the reference situation, the higher the impact, either in positive or negative directions. To facilitate the comparison between biodiversity situations in this quantitative way, the reference situation has to be translated into a set of functional reference conditions, i.e. a set of attribute values or quantifiable characteristics (Miller et al. 2012). These attributes will depend on the indicators chosen to assess biodiversity and can be physical, chemical, or biological parameters of organisms, ecosystem functions, or structures, and could be represented by single values or a distribution (Miller et al. 2012). Considering that some LCA practitioners in some situations may favour impact results which are less negative, or as positive as possible, the reference situation acts as a direct reflection of what is considered a 'good' direction of the indicator values. Although the exact indicator values acting as the reference situation will be different for different ecosystems and circumstances, they are the quantifiable form of the rationale of the assessment: whether this is "naturalness", "functional anthropogenic ecosystem", or something else.

b) How are reference situations in LCIA models currently used?

Results of **Paper I** showed that most of the LCIA models addressing biodiversity apply a so-called *baseline reference situation*. Baseline reference situations represent land in its 'original', ‘unmanaged’ state and are derived from time periods or locations in the absence of human intervention. Examples of different baseline references are *historical ‘natural’ baseline references*, for which usually a point in time is selected to compare current conditions to; *re-naturalisation situations* -such as Potential natural vegetation (PNV, Ricotta et al.,2002)- which is a future hypothetical situation which will establish after all human interventions have stopped (if **fig 2.** Shown as Qref2), and *counterfactual reference situation*, which is a hypothetical natural situation that could be outlined for the present time (or a certain earlier period depending on the assessment) imagining that the area would never had been subjected to human interventions (Tüxen 1956). Re-naturalisation has been identified as the most suitable reference situation for use in attributional LCA by several authors (Milà i Canals et al. 2007; Soimakallio et al. 2015; Curran et al. 2016; Koponen et al. 2018) as land occupation postpones natural regeneration of the land and the fact that the resulting final ecosystem could be of a different biodiversity ‘value’ than the original. These permanent impacts need to be accounted for (Weidema and Lindeijer 2001). In contrast, we showed in **paper I**, that a re-naturalisation reference situation is unsuitable in assessments using only a single reference situation as a re-naturalisation reference situation by principle only catches occupational impacts and neglects permanent impacts. This could lead to distorted impact results, for example in the plausible case that certain land use management causes further degradation of the land which limits the capability of the land to regenerate. This will result in a lower estimated impact (as its distance to the reference becomes smaller)(Soimakallio et al. 2015). Instead, complying to the established UNEP-SETAC framework for including land use and land use change impacts and account for permanent impacts, would require the use a double reference situation. Considering the many practical difficulties and uncertainties involved in applying reference situations as described in **Paper I**, the use of two reference situations to calculate impacts on biodiversity has however rarely been seen. Having said that, there are some examples of LCIA models where the second reference situation is implicitly present. For example, in the model by Curran et al. (2016) based on conservation status of species (the IUCN red list of threatened species), a baseline reference situation is applied due to the use of contemporary threat/rarity data. In this case, the application of a re-naturalisation reference situation allows for assessing the potential permanent impacts expressed as the extinction risk of species.

Besides inconsistencies seen in the application of the UNEP-SETAC framework for land use LCIA and the major practical difficulties in the application of baseline situations (see **Paper I**), the LCA approach to measure impacts relative to “the natural”, is in general doubtful as discussed in the next sub question 1c.

c) What does the current use of reference situations mean for the rationale of LCIA models?

To include impacts on biodiversity in LCA, we not only need to ask ourselves the question what to assess, but also -and probably at first- what to preserve? Considering the fact that LCA-models are powerful decision aids, they can be helpful but also harmful if the results point decision makers in a sub-optimal direction. The distinction of what is considered 'good' and 'bad', 'high' and 'low' or 'positive' and 'negative' biodiversity is dependent on circumstances like the ecosystem under study, the biodiversity attribute and one's perspective on biodiversity. Yet, it is possible to establish some general rules. As a general rule, the fundamental goal of LCA is to map anthropogenic impacts on the "natural" environment. To achieve this goal, LCA divides the world into a technosphere and an ecosphere. The ecosphere is made up of the thin skin of the earth; the air, water and soil. That is where all living things are found, and we are part of that. But unlike every other living thing, we produce things that otherwise do not occur in nature. We have created, by technical means, another area in which we live, called the technosphere. Changes to the ecosphere (as a result of activities in the technosphere) can be considered unintentional "man-made" consequences (Hauschild et al. 2018) which should be avoided. In the case of agricultural systems however, the location of the boundary of these two spheres is quite abstract and therefore often debated (Hauschild et al. 2017). Yet, as this is the way LCA is constructed, we have seen in **Paper I** that this approach, i.e. measuring biodiversity impacts of land use as a departure from a "natural" or "human-free" situation, is currently also embedded in the UNEP-SETAC framework for land use LCIA. This approach is however in contrast to most of the world's conservation efforts that serve the double purpose of preserving the Earth's diversity or organisms in functional ecosystems and *using* biodiversity sustainably. The global Convention on Biodiversity (CBD) for instance, explicitly recognize humans -and their land use management- as an integral component of these systems. With its vision "Living in Harmony with Nature", it does not aim predominantly at naturalness but emphasizes the sustainable coexistence of humans and biodiversity (UN CBD 2010, p. 8). The same goes for the European Species and Habitats Directive (Council Directive 92/43 EEC) that aims at maintaining or restoring European protected habitats and species listed in the Annexes at a favourable conservation status. Many of the habitats listed are anthropogenic and their species depend on continued land use (The Council of the European Communities 2013). The wish to conserve anthropogenic habitats and species reliant on it, not only originates from the utilisation dimension of conservation; many species have the ability to adapt to man-made habitats both phenotypically and genetically (Hunter 2007). Recent examples include songbirds adjusting various aspects of their songs to overcome noise pollution in cities (Slabbekoorn and den Boer-Visser 2006) and the house sparrow (*Passer domesticus*), being evolved to produce higher levels of the enzyme amylase, to be able to digest the high-starch diets associated with human food waste (Ravinet et al. 2018). It is therefore not difficult to imagine that the biodiversity we find today co-evolved with a few million years of human involvement (Faurby et al. 2020). Many important habitats for biodiversity and wildlife have been created and maintained by millennia of extensive, low-intensity land use such as mowing, grazing, or burning, making many cultural landscapes of utmost importance for biodiversity today (Lennartsson and Helldin 2007; Norderhaug 2007; Oppermann et al. 2012). By using reference situations solely reflecting "naturalness" or some kind of "human-free" situation in LCIA models for biodiversity -implying a human-free world as what should be striven for- would thus provide support for a part, but certainly not all of the biodiversity highly valued today. Instead, impact assessment models should address biodiversity impacts in a way that make the assessment compatible with current conservation strategies and policies, in a manner that can account for long-term human occupancy. This includes both natural and cultural landscapes.

4.2 RQ2: HOW CAN LCIA MODELS BE IMPROVED TO MAKE THEM SOCIETALLY AND ECOLOGICALLY MORE RELEVANT?

a) How do we translate the definition of 'meaningful' biodiversity impacts into recommendations for LCIA models?

If results are considered not to match existing policies or established targets and strategies, there might be a danger that results will not be paid attention to or, in a worse case, point decision makers in sub-optimal directions. Therefore, in order to be societally useful, LCIA models aiming to include biodiversity impacts need to address such biodiversity impacts that are applicable for current conservation strategies and policies (Teixeira et al. 2016). Conservation strategies, either on regional, national or global level, have the advantage of reflecting society's preferences, and values related to biodiversity and bring these together at an aggregated level. Because of this, they also include biodiversity that depends on a certain level of management. We concluded therefore in **Paper I**, that LCIA models need to assess impacts against reference situations that include both natural and anthropogenic ecosystem states and processes. To reach this, we have to let go of the thought that LCA should be used to quantify the impact on 'the natural', but rather as the impact on a desirable future. The reference situation should thus be seen as a target, always reflecting a desired situation.

b) Why would the genetic aspect be important for ecological relevance?

In the previous paragraph, I argued that a target reference situation should be used as the reference situation for LCA, as this reflects the societal value of biodiversity and gives room to different motivations for the conservation of biodiversity. I argued that societal conservation frameworks should guide target setting as these reflect the greater societal opinion of how biodiversity is valued.

These frameworks may include intrinsic value, but also instrumental value of biodiversity. Not only the value perceptions can change (i.e. how society looks upon biodiversity), the environment is constantly subjected to changes as well, either human induced or ‘natural’. This underpins the importance of biodiversity metrics that respect the plurality of different values related to biodiversity as well as uncertainty about how best to secure the different values and possible future changes both in the environments and in human value systems (Lean and Maclaurin 2016). In short, we thus need measures that captures the ‘security’ of biodiversity in the long run, thus resilience and stability of ecosystems.

The stability and function of the ecosystems we rely on is not just a function of biodiversity by itself, but ecosystem stability depends on functional diversity capable of differential response to environmental disturbances (McCann 2000). Not only will a broader diversity of functions lead to the presence of more pathways for energy flow and nutrient recycling, different species also react differently towards environmental changes. In case one species is lost, another will be able to do well under changed conditions. Also, an ecological function will not disappear until all the species performing that function disappeared from an ecosystem (McCann 2000). This recognition have led to the proposal of functional diversity as a more appropriate indicator of biodiversity loss in LCIA models in comparison to taxonomic indicators because of the association between species traits and ecosystem functioning (de Souza et al. 2013; Souza et al. 2015). By measuring functional diversity, however, we still not arrive at the core of ecosystem resilience. One of the issues arising from this approach is that groups of taxa with similar traits are often considered as a significant unit in terms of function. In this view, particular taxa within functional groups are considered redundant and interchangeable (Geeta et al. 2014). Such measures necessarily ignore subtle or unrecognised differences within groups that may have crucial roles in the community (Geeta et al. 2014). Even if these small differences do not play a significant role at the moment, they could do so in the future as the environment changes and evolutionary processes drive further diversification. Indeed, it is evolution that is the ultimate core of resilience, as all biodiversity is a product of evolutionary processes (Darwin 1859).

The idea that it is critical to consider evolutionary processes when valuing biodiversity for the sake of sustainable development goals is not new and explicitly included in the concept of “evosystem services” (Faith et al. 2010). “**Evo**-system services”, not to be confused with the well-known concept of “**eco**-system services” (it intentionally prompts comparisons and contrasts with the latter concept), is broadly conceived as “the benefits to society stemming from evolutionary processes in the past, present and future" (Faith et al. 2010, 2017). It came to live to counterbalance the distinction between ecosystem services and biodiversity made by The Millennium Ecosystem Assessment (2005), in which biodiversity is not considered a service per se, but rather a prerequisite for ecosystem services. The distinction herewith undermines the value of biodiversity that extend beyond currently perceived ecosystem services and the need for the measurement of overall biodiversity (Faith et al. 2010). In addition, it allows for the valuation of biodiversity other than purely instrumental, whereas intrinsic values and non-human interests are per definition excluded from the ecosystem services concept.

Evosystem services not only include the huge reservoir of current biodiversity that results from past evolution (on which current attempts to conserve biodiversity mainly focus), but also the 'option' on future biodiversity, constituted by evolutionary processes themselves such as mutation, selection and diversification which produce new biodiversity for new environments, solutions, or problems (Faith et al. 2010; Geeta et al. 2014). One way to quantify both the current evolutionary information as well as the future potential of evolutionary processes is phylogenetic diversity (PD) (Vane-Wright et al. 1991; Faith 1992). PD, is often referred to as 'evolutionary diversity' based on phylogeny (the tree of life). The most widely used phylogenetic metric is Faith's PD (Faith 1992), in which the PD of a set of species is equal to the sum of the lengths of all those branches on the tree that span the members of the set (**Fig. 3**).

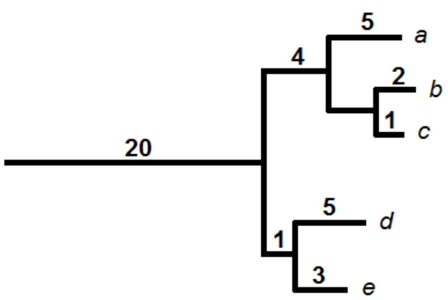

Figure 3: A hypothetical phylogenetic tree for species a through e. Branch lengths are shown above branches. The PD is 41 for this set of species (20+5+4+2+1+5+1+3). Note that the branch lengths on the tree are informative because they count the relative number of new features arising along that part of the tree (Faith and Richards 2012).

Although there might be other ways and indicators to approach evosystem services or evolutionary diversity, it must be clear that measurements are primarily based on genetic data. Despite the acknowledgment that the inclusion of genetic metrics are an important aspect for LCIA models for Biodiversity (Curran et al. 2016; Frischknecht et al. 2016), they are as of now, still lacking.

c) What would be needed to include genetic aspects in LCIA models?

Genetic data in LCA would allow us to calculate indices such as phylogenetic diversity. It is increasingly acknowledged that genetic samples taken from the environment (environmental DNA or simply eDNA) contains rich information about a large diversity of organisms found in ecosystems and will cover a much larger proportion of the biodiversity when compared to conventional biodiversity methods (Bohmann et al. 2014). eDNA, which is extracted from environmental samples such as water, air, soil, sediment or faeces, allows us to explore diversity data of entire communities across the whole tree of life (Baird and Hajibabaei 2012; Burki 2014). This brings new opportunities for the generation of quantitative indices for analysing species, community diversity and dynamics and for exploring ecosystem-level processes (Bohmann et al. 2014; Goldberg et al. 2016).

Considering the recent advances in technology and the new opportunities genetic data would bring for the generation of quantitative indices for biodiversity, we tested the process of data gathering and the construction of diversity indexes of genetic data in the form of metabarcoding of eDNA samples (**Paper II**), in which metabarcoding refers to the method used to process the eDNA and identify the DNA from the multiple species within the sample. With this study, we answer the questions:

a) Would it be possible to construct meaningful indicators for biodiversity based on e-DNA?
b) If so, is there enough data available in databases for such indicators?
c) If e-DNA appears to be a promising option, what further research and methodological development is needed?

The potential has shown to be large with a great variety of different indicators possible to construct from the eDNA metabarcoding data, which would be suitable for use in LCIA models. Due to the variety of indicators, it is possible to include indicators on different levels without the need for extra sampling efforts. They could include measures varying from indicators which are used already for a long time (species richness), on different levels (genetic, species, community) to indicators which gives us 'new' information about the state of biodiversity of which some are impossibly to construct with conventional biodiversity methods (such as phylogenetic trees of whole communities and information on (microbial) taxa not earlier included). At the same time, ongoing eDNA metabarcoding monitoring programs will feed into the growing availability of global data. Global databases are expanding fast, giving confidence that soon an acceptable spatial coverage is reached to support LCIA modelers to calculate characterisation factors for different regions and at different spatial scales.

Yet, some progress is to be made in the standardisation of protocols for sampling strategies (as elaborated on in **Paper II**). This needs time and expertise, but should not hinder LCIA modellers using the data.

As a concluding remark, it should be noted that, despite the potential, metabarcoding data of eDNA is not the holy grail of biodiversity assessments. Just as in conventional biodiversity inventories, false positive and false negative occurrences of species will be present and different ways of sampling and handling the data, will give different results. However, every LCIA modeler or ecologists should, in all cases, be aware of the shortcomings in his or her data and acknowledge it if the uncertainty could alter the conclusions. Metabarcoding of eDNA is now an established technique, with the capacity to improve the quality and utility of LCIA models for biodiversity. Understanding its shortcomings will help LCIA modelers, preferably together with ecologists, to design, test and review metabarcoding approaches and the construction of biodiversity metrics based on this.

5 CONCLUSION

In the last couple of years, great effort has been seen in the development of new LCIA methods to include biodiversity in LCA. Yet, there are challenges to overcome in the development of ecologically relevant LCIA models for biodiversity. One of them is the alignment with current conservation strategies. In order to be societally useful, LCIA models aiming to include biodiversity impacts need to address such biodiversity impacts that are applicable for current conservation strategies and policies. This calls for a reconsideration of the way in land use impacts on biodiversity are currently assessed and a shift towards distance to target measures, rather than quantifications of impact on the natural.

Conservation strategies, either on national, global or regional level, have the advantage of reflecting society's preferences and values related to biodiversity, and bring these together at an aggregated level. Herewith, they give room to different motivations for the conservation of biodiversity. This underpins the importance of biodiversity metrics that respect the plurality of instrumental and intrinsic values in biodiversity as well as uncertainty about how best to secure the different values and possible future changes both in the environments and in human value systems. Quantification of genetic diversity is one of the missing links, currently not included in LCIA models for biodiversity, with the potential to not only increase the ecological relevance of current biodiversity assessments, but also to account for future security of biodiversity as an important conservation target. Genetic approaches certainly seem to have a bright future ahead, but also a few obstacles yet to overcome.

6 FURTHER RESEARCH

As discussed, for reliability of results and applicability of LCIA models addressing biodiversity, social and ecological relevance are key features. In order to further enhance those features, the current scope of investigation needs to be deepened and expanded. With half of the doctoral time left, there are several possible and interesting research directions which could bring this project further.

Studies on the use of genetic indicators in LCIA models could be deepened, with in particular the inclusion of functional diversity and phylogenetic diversity. This would pick up on results of Paper II, which showed that data based on metabarcoding of eDNA samples would suit global assessments of biodiversity and its inclusion in LCIA modelling. Furthermore, functional diversity and phylogenetic diversity were identified as relevant and well-equipped indicators for biodiversity resilience. Practical considerations to continue this work include the current availability of data (generated during the study on which Paper II is based and upcoming global data bases), and expertise, creating the right circumstances to elaborate on this part of research.

To broaden the scope, a second option is to explore if and how landscape scale processes such as habitat fragmentation and landscape heterogeneity could be included in LCIA modelling. Species-area relationships, on which many LCIA models for biodiversity are based, describe the importance of the area to sustain biodiversity. It is however not only loss of habitat that major implications for biodiversity: very much related is the habitat fragmentation it causes and how well the connectivity of remaining habitat patches is (Levins 1969; Wilson et al. 2016) Empirical studies have also shown that biodiversity increases with increasing number of biotopes in a landscape (Benton et al. 2003; Katayama et al. 2014), pointing out that landscape heterogeneity and connectivity of the landscape are important aspects of a biodiversity indicator. Landscape level processes are currently not well captured in LCAs, which has probably a lot to do with the way in which inventory flows are registered (in m^2) without the spatial information required to consider landscape structures. Yet, it might be very interesting to investigate if, and in what form, it would be possible to catch impacts on these landscape structures.

Apart from research needs directly connected to the research questions of this thesis, there is a need for the testing of different approaches and indicators in LCA case studies. Preliminary frameworks consist mainly of articulated 'preconceptions', which over time, need to be developed according to what is discovered through empirical fieldwork, analysis and interpretation (no theory can be understood without empirical observation and vice versa; Dubois and Gadde 2002). In other words: case studies provide indispensable proofs of concept, or calls for adjustments in the development of LCIA models (Arvidsson et al. 2018). As for now, two case studies are planned, comprising forest and agricultural production systems, to simultaneously test and develop the indicators and concepts discussed.

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