Research on methodologies for impact assessment on biodiversity
in LCA
The case of biobased products

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THESIS FOR THE DEGREE OF LICENTIATE OF PHILOSOPHY

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ABSTRACT

The topic of this thesis is impact assessment on biodiversity in life cycle assessment (LCA).

With humans in focus, we need the earth's resources and its biodiversity for our existence and human well-being may depend on the management of the earth's resources. The earth, however, does not need humans to continue its existence.

Considering the last fifty years, humans have contributed to land degradation and changes in ecosystems, including biodiversity loss, at a speed never experienced before (MA, 2005). The acquisition of earth's resources is often accompanied with impact on e.g. land used, which may have positive or negative impact on biodiversity. One way to assess impact on land used and hence biodiversity, is by the use of life cycle impact assessment (LCIA). This thesis presents a literature overview on LCIA methods on biodiversity with potential use in LCAs for bio based products and discusses challenges met when applying two selected LCIA methods on biodiversity on local and regional level.

Focusing on the two case studies being part of my thesis, on forestry in southern Sweden and the results gained. The results showed that a number of methodological adjustments and decisions had to be made pertaining available data and suitable format on data sets. Also, the interpretation of definitions on reference situations showed to be problematic, since ecosystems are dynamic but the definitions on reference situations are not. The choice of reference situation, and when in the production cycle assessment was made, led to differences in the categorization factors generated. This indicate that biodiversity may not remain constant during the occupational land use phase and that the choice of reference situation is important.

The results of the literature overview indicate that we see a clear trend towards diversification in the field, choice of indicator, modelling and reference situation. The diversification can be seen as an attempt to better match the diversity of the biodiversity concept itself. However, there is a risk that the level of methodological diversity needed for this purpose will lead to an overwhelming need for on the one hand decisions to be made by the LCA practitioner, and on the other hand for data accessible databases.

Keywords: *life cycle impact assessment, biodiversity, land use, forestry*

This thesis is based on the following appended papers

Paper I

Palme Ulrika; Lindqvist Maria, Røyne Frida

Biodiversity in life cycle impact assessment: trends, challenges and potential

Paper II

Lindqvist Maria; Lindner Jan Paul (2014)
Potential mutual benefit between renewable energy resources and biodiversity
– a case study of biodiversity impact assessment in LCA

Presented at the LCAXIV 2014 conference
San Francisco, U.S.A October 6 to 8, 2014

Paper III

Lindqvist Maria; Lindner Jan Paul; Palme Ulrika,
Tillman Anne-Marie (2015).
A comparison of two different biodiversity assessment methods in LCA
– a case study on Swedish spruce forest

Submitted manuscript

Other publications by the author

Janssen Matty; Nyström Claesson Anna; Lindqvist Maria (2015)
Design and early development of a MOOC on
”Sustainability in everyday life”: role of the teachers

Accepted for oral presentation at the 7th International Conference on Engineering Education for Sustainable Development, EESD15 Vancouver, Canada June 9 to 12, 2015

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Maria Lindqvist

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1 Introduction

The importance of biodiversity is an often highlighted topic and has been widely recognized among scientists. It is well known that the development of human societies including human activities have effects on the earths systems and can threaten the resilience of these. One such system for which the planetary boundary currently is exceeded is the biosphere integrity, in which biodiversity is included, see Figure 1. According to Steffen et al. (2015) the biosphere integrity can be divided into genetic diversity and functional diversity, for which the latter the boundaries cannot yet be quantified.

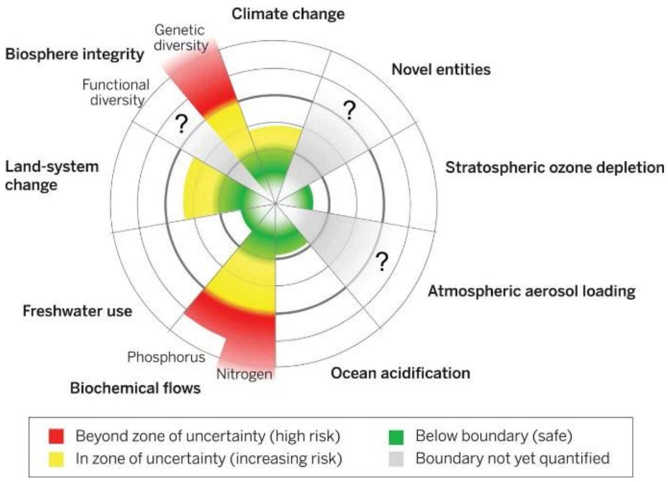


Figure 1 Picture illustrating the current status of the control variables for seven of the planetary boundaries (Will Steffen et al. 2015).

One of the major drivers of biodiversity loss is land use, i.e. forestry, mining, house-building or industry. The resource demand in today's society is largely met by consumption of fossil resources, the use of which is one of the major contributors to climate change through emissions of CO_2 to the atmosphere (IPCC 2014). Furthermore, the substitution of fossil resources by renewable ones called for by the Intergovernmental Panel on Climate Change (IPCC 2014) to mitigate the effects of climate change implies increased and intensified land use and hence increased impact on biodiversity. The substitution of fossil resources with biobased resources is one important step towards a biobased economy. A sustainable such transition, however, requires that impact on biodiversity is minimised, and hence that environmental assessments can be made that include impacts on biodiversity.

Stakeholders such as companies and authorities express an interest in business practices that benefits environmental issues including biodiversity (BBOP 2014; BBP 2003; Biodiversity in Good Company 2008). In industry, life cycle assessment (LCA) is a well-known tool designed to analyse the environmental impacts that a product generates from cradle to grave. Life cycle impact assessment (LCIA) is the step within the procedure of life cycle assessment (LCA) which quantifies the contribution to different types of environmental impact (e.g. global warming, acidification, impact on biodiversity) of the environmental loads (e.g. amount of carbon dioxide or nitrogen oxide emissions or amount of used land) quantified in presceeding inventory step. This is done using different models for different impact categories. These models result in characterisation factors (CFs) which the LCA practioner uses to multiply with the quantified environmental load, in order to assess the contribution of the load in questinon to the type of environmental impact in question. Biodiversity may be impacted via different routes, e.g. indirectly by impacts usually accounted for in LCA such as global warming, eutrophication and acidification. It may also be direclty affected as a result of land use practices, which is the topic of this thesis.

The environmental importance of biodiversity is increasingly recognized within the research of LCA, but there are challenges regarding how to include biodiversity in LCIA. Since the start of the millenium, different methods of LCIA on biodiversity have been suggested (Coelho et al. 2014; de Baan et al. 2013b; de Souza et al. 2013; Köllner 1999; Lindeijer 2000; Lindner et al. 2014; Michelsen Ottar 2008; Quijano 2002; Schmidt 2008; Weidema and Lindeijer 2001). Further, the United Nations Environment Programme (UNEP) - Society of Environmental

Toxicology and Chemistry (SETAC) Life Cycle Initiative has published a framework including guidelines and recommendations for assessment of land use impacts on biodiversity in LCA. However, the number of cases where these methods have been applied is limited. Further, because of the many proposals of LCIA methods on biodiversity there is not yet a consensus on how inclusion of impact assessment on biodiversity in LCIA should be conducted. The many proposals of approaches of impact assessment on biodiversity of this research area has worked as an incentive in the development of the research aims of my thesis, which are described in the following section.

1.1 Research aims

Based on the above description of the current state of biodiversity and the need for a sustainable transition to a biobased society the research aims of my thesis are:

1.) To map and analyze currently existing and most commonly used LCIA methods on biodiversity with potential use in LCAs of bio based products.

2.) To perform two case studies on biodiversity impacts from land use, with the aim to explore challenges, difficulties and feasibility pertaining inclusion of biodiversity in LCIA. Specifically the biodiversity indicators, time resolution, reference situation and data availability for implementing the methods on a regional scale were investigated.

1.2 Outline of the thesis

In chapter 1, the background of the research area is described and is followed by a description of the research aims of my thesis. Chapter 2, the current state of biodiversity and definitions needed to facilitate the understanding of impact assessment on biodiversity are described. In chapter 3, I describe the method used to perform the literature overview on LCIA methods on biodiversity. This is followed by a description of the two selected LCIA methods on biodiversity applied in Case study I and II and a description of the methodological adjustments required in order to apply the two methods tested. Chapter 4 includes the results generated by the three studies being part of my thesis and is followed by chapter 5 where the results generated are discussed. In the closing chapter 6, conclusions from all studies being part of my thesis are presented.

2 Background

2.1 Biodiversity

Living organisms are part of complex ecosystems, which convey and provide many fundamental prerequisites for other organisms. In biology, organisms are organized and categorized into taxons pertaining their distinct and recognizable characteristics e.g. the taxonomic group of vascular plants. The variety of living organisms e.g. the number of different species on different hierarchical levels are often described as biodiversity. Below follows a description of the hierarchical components which biodiversity can be divided into and the accompanying biological attributes each hierarchical component of biodiversity can possess, see Figure 2.

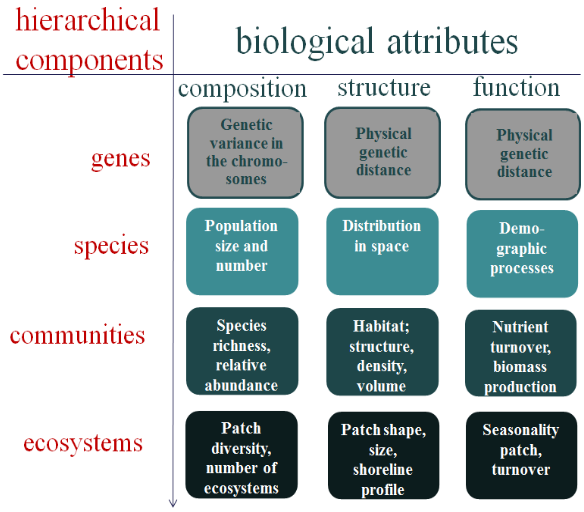


Figure 2 A simplified version of biodiversity indicators pertaining hierarchical components (genes, species, communities and ecosystems) and biological attributes (composition, structure and function). This is a modified version of the original developed by Curran et al.(2011).

2.1.1 Assessment of biodiversity

Below follows a selection of key concepts pertaining the components of species and ecosystems and methods commonly used to assess biodiversity.

2.1.1.1 *Species indices*

Species richness is a definition frequently used which refers to the number of species in a sample, community or a taxonomic group (McGinley 2014a). Not all species are equally abundant and this is refered to as relative abundance. If species are distributed more or less equally, this is refered to as species evenness. One way to quantify numbers of species is by the use of indices, which are commonly used to track changes over time or describe ecological relationships among species, habitats and ecological communities.

There are three basic species indices commonly used to assess biodiversity. α-diversity refers to the actual number of species within a given area, e.g. 1m^2 or a defined ecosystem. β-diversity refers to the similarity in species composition when investigating a possible species overlap between two assemblages of species, e.g. in two different ecosystems. Y-diversity, finally, refers to all species within a larger geographical region (Whittaker 1972). In addition to the three basics species indices described above, a selection of additional commonly used species indices are described below.

One commonly used species index is the *Shannon index,* which assesses the relative abundance and refers to species richness and species evenness (Shannon 1948). A second additional commonly used index is the *Simpson diversity index* which includes the number of species, the abundance of species and species evenness (Simpson 1949). The generated value refers to the probability that two randomly selected individuals will belong to the same species. The Simpson index considers the relative contribution of each type of species. A third additional species index is the *Sörensens dissimilarity index* (Sörensen 1948) which compares the number of species in the area of study and a reference area. This species index assesses to what extent species in the two areas of study overlap. A fourth additional index is the *mean species abundance* (MSA) developed for the GLOBIO3 framework model by (Alkemade et al. 2009). In this index a mean abundance of species in the area of study is divided by the species abundance in a reference area.

2.1.1.2 *Species estimation*

Several of the LCIA methods rely on estimations of species richness. There are several ways to make such estimations, of which the main ones are presented in the following.

It is well known that the number of species is greater on a larger area compared to the number of species on a smaller area, with similar ecological prerequisites (Arrhenius 1921; Rosenzweig 2003). It would take a substantial effort, time and money to monitor *all* species of an area and it is difficult to tell when a survey of species is complete. In order to bring a biodiversity survey to completeness, different types of statistical calculations can be used to estimate the maximum number of species in an area (Colwell 1994).

Statistics of species estimation can be based on species information from a limited number of sample plots located in a small area and give information on the estimated number of species in a large area. The sample plots can for example be monitored for information on whether a species is present or absent and is referred to as presence-absence data (Colwell 2013). The presence-absence data from each sample plot is recorded, plot by plot, and compiled into a data sheet. The compiled information can be used in different types of statistical species estimations, in which the estimated numbers of species of an area is calculated by extrapolation (Colwell 2013).

The history of species estimation is long and many different metrics exists (Arrhenius 1921; Chao 1984; Connor et al. 1979; Hurlbert 1971). To construct metrics of species estimation there are two types of empirical indata which can be used, which are those from sample plots of equal sizes or those from plots of of different sizes (Köllner 1999). Below follow three examples of species estimation types. One which is based on differently sized plots and two based on equally sized plots.

One species estimation method is developed by Arrhenius (1921) and is based on presence-absence data collected from sample plots of different sizes and is called the species area relationship (SAR). SAR is based on the "island theory" developed by MacArthur et al. (1967), which is a theory of colonization and extinction of species in relation to the size of a habitat and geographical distance to other habitats. This theory can be described by Equation 1, where S is the number of species, A is the area, c is the lowest number of species in one sample plot and z is a constant for the accumulation rate of species (Rosenzweig 2003).

$$S=cA^z \qquad \text{Eq.1}$$

Another method for species estimation, which also relies on presence-absence data, is rarefaction. This metric is, however, based on species data from equally sized sample plots. An example of this metrics of species estimation is the rarefaction equation developed by Hurlbert (1971) in which the expected number of species is calculated through Equation 2 (below) in which E stands for the estimated number of species and S stands for the plot size from which data on species are collected. .N is the total number of sample plots in a collection, N_i is the number of plots where species i is present, and n is the number of species from a collection of sample plots.

$$E(S_n) = \left[1 - \frac{\sum_{i=1}^{S} \binom{N-N\,i}{n}}{\binom{N}{n}}\right] \qquad \text{Eq. 2}$$

Another example of species estimation, which also relies on presence-absence data from equally sized plots is the rarefaction Chao2 developed by Chao (1984). This metric is based on the frequency of rare species in a sample (Chao et al. 2014). In the Chao2 equation, see Equation 3, S_{obs} is the total numbers of species observed in all samples, m is the total number of samples, Q_1 is the number of species that occur in only one sample and Q_2 is the number of species that occur in exactly two samples. Further, the Chao2 equation allows for comparison of species between surveys.

$$S_{Chao2} = S_{obs} + \left(\frac{m-1}{m}\right)\frac{{Q_1}^2}{2Q_2} \qquad \text{Eq. 3}$$

2.1.1.3 *Ecosystem indicators*

The use of species based indices only represent a limited part of biodiversity, that of a particular species. However, one taxonomic group may be used as a proxy for all biodiversity. On the other hand ecosystem indicators aims to capture biodiversity as a whole in contradiction to the species based indices, which often focus on one taxonomic group. Ecosystem indicators, which also are called indirect indicators, are based on parameters known to be important for biodiversity (Michelsen Ottar 2008). These indicators can for example be designed to capture the prerequisites for biodiversity, such as structural components, a process, amount of area set aside or amount of dead wood. The prerequisites for biodiversity are often different for different ecosystems, which is why specific knowledge on a specific ecosystem is required in order to identify parameters of importance for biodiversity. Sources of information often used are literature studies or interviews with ecological experts.

2.1.1.4 *Ecosystem scarcity and ecosystem vulnerability indicators*

Ecosystem scarcity and vulnerability are two indicators that were developed by Weidema & Lindeijer (2001) to give information about the intrinsic value of biodiversity of an area in for example a biome or ecoregion. The indicator of ecosystem scarcity generates a value for the smallness of an area. The smaller an area is the scarcer it is considered to be. The area of an ecosystem is not always the same as the potential area that could be possible with respect to ecological prerequisites. If the area belongs to an ecosystem with a small potential area it is assigned a higher value than if it would have belonged to an ecosystem with a large potential area. However, two equally scarce areas may not be equally vulnerable. For this reason, the indicator of ecosystem vulnerability is developed that describes to what extent the area of study is stressed. If the area of land used of a potential ecosystem is dominating. The remaining area of a potential ecosystem becomes smaller and hence more stressed.

2.2 Life cycle assessment

There are several examples of tools for environmental assessment e.g. Life cycle assessment (LCA), Strategic Environmental Assessment (SEA), Environmental Impact Assessment (EIA), Environmental Risk Assessment (ERA) and Ecological Footprint (Finnveden and Moberg 2005; Ness et al. 2007). LCA is a useful tool for environmental assessment and increasingly one of the most influential methods (Cooper et al. 2008) and aspires to assess a wide range of environmental impacts that a product generates from the e.g. cradle to the grave. The result generated by an LCA can provide useful information and guidence for stakeholders in their efforts towards an environmentally sustainable product manufacturing.

The history of LCA started during 1980ies. During the 1990ies LCA became an became an accpeted enviromnetal assessment tool within industry and standardized principles and frameworks, the ISO 14040 series were developed. The ISO 14040 was later (2006) revised into ISO14044.

An LCA can be divided into a four step procedure, which constitutes an iterative procedure, see Figure 3 which is described in the following sections. To receive a description in detail of an LCA see Baumann and Tillman (2004).

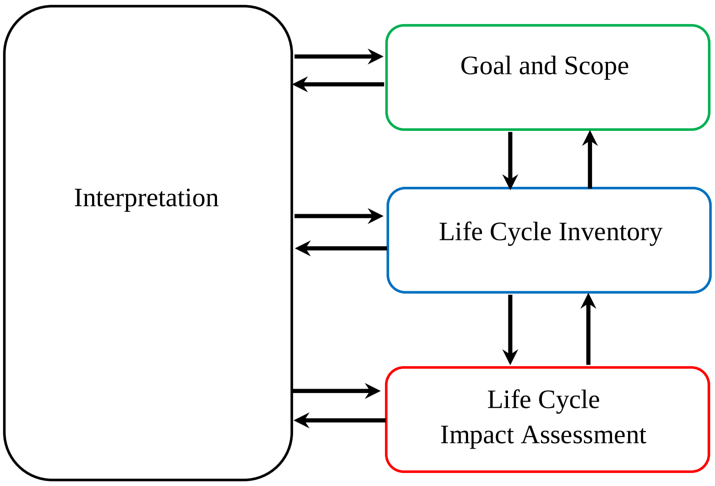


Figure 3. The four steps of LCA accoring to the ISO 14040 and 14044.

The first step constitutes of setting aim with the study, goal and scope, describing the context of the study and to who the results are going to be communicated. In doing so the question “who wants to know what and for what reason?” is answered. The answers to these questions also influence the scope of the study, in which the product system (system boundary) is described in a flow chart and the relevant environmental information is chosen and explained. The scope includes methodological choices including the functional unit that defines what is being studied e.g. the function delivered from $10m^2$ painted house facade for ten years. The functional unit allows for comparison between products with identical functions.

The second step is constituted by the life cycle inventory (LCI) in which the environmental loads, i.e. the inputs and outputs through the life cycle of a product is measured. The result is an inventory of all relevant environmental flows to and from the technical system e.g. energy and emissions and systems outside the chosen system boundary e.g. the environment or another product system (Tosca 2011). Relevant environmental data is collected and the flows are further quantified in accordance to the functional unit.

Life cycle impact assessment (LCIA) is the third step, in which the flows are categorized into different environmental impact categories, which are contributing to different environmental impact e.g. acidification, global warming, ozone depletion, photo oxidant formation This is done by means of multiplying the inventory flow, e.g. land use, with the relevant characterisation factor (CF), e.g. the one for impact on biodiversity.

The indicators from impacts assessment are fewer in number than those from the inventory and express a potential environmental impact. Some types of flows such as emission of toxic compounds or land use causes an impact wich depends on the location of the flow, because the impact is dependent on local parameters e.g. soil structure and/or existing prerequisites for biodiversity which is why such impact assessments include high uncertainty.

Biodiversity may be impacted via different routes. It may be impacted indirectly from other effects usually accounted for in LCA such as eutrophication and acidification. It may also be direclty affected as a result of land use practices, which is the topic of this thesis. The cause effect chains from land use are depicted in Figure 4.

The indicators used in LCIA can be divided into midpoint or endpoint categories. Midpoint categories are impacts which occur early in the cause effect chain. Sometimes it is difficult to compare different midpoint indicators such as radiative forcing with acidification because the indicators are expressed very abstract. End-point indicators describe effects that occur later in the cause effect chain. The use of endpoint indicators makes the results easier to interpret as they express the effect as a damage to damage to ecosystems such as a forest see endpoint level in Figure 4. Generally, the endpoints, areas of protection considered in LCA are natural resources, human health and ecosystem quality (Goedkoop 2008). LCIA can also include weighting or normalisation. Weighting can be conducted in accordance with e.g. political goals, and normalisation in which the value of e.g. a particular environmental impact is related to a reference e.g. that particular environmental impact in the region.

The fourth step consists of interpretation of the results, drawing conclusions from the study. For this reason the results can be presented in bar diagrams or by weighting the results to increase the feasibility of the interpretations. Additional ways to interpret the results can be to conduct a dominance analysis, which is an analysis of identifying the activities that generate the most dominant environmental impact, or contribution analysis, which is an analysis of identifying the environmental loads e.g. greenhouse gases that generate the most dominant environmental impact. Uncertainty and sensitivity analysis can also be conducted.

In the following section the methodological principles of land use impact assessment are described.

2.2.1 Methodological principles of land use impact assessment in LCIA

This section provides a description of concepts often used in the context of LCIA on biodiversity, those of geographical scale, key concepts and methodological principles of land use impact assessment in LCIA, recommendations made by the UNEP-SETAC framework (Köllner et al. 2013a), which is followed by suggested reference situations and land cover systems.

2.2.1.1 *Geographical scale*

Assessment of biodiversity can be made with respect to different geographic scales. Biome is the largest delineation and is based on climate such as precipitation and temperature which have a strong influence on organisms and vegetation. Organisms and vegetation within each biome have evolved similar characteristics through adaptation to the climate. (McGinley 2014b). There is no agreement on the exact number of biomes on earth but they are commonly organized into 14 different types that include grasslands, forests, deserts, aquatic and tundra (WWF 2014), which gradually merge in to one another (Reece et al. 2012). The next lower level of delineation are ecoregions, which are areas with relative similarity in climate, species of flora and fauna and ecological communities. Ecoregions have an average size of 50 000km^2 (Fund 2014). Ecoregions can further be divided into regions. The concept of region is not defined as region can refer to different sizes and also to different latitudinal and altitudinal levels. However, for practical reasons in an LCA context administrative boarders of nations are often used for defining regions (Köllner et al. 2013b). The final delineation is a site specific geographical location.

2.2.1.2 *Key concepts and methodological principles*

With impact assessment on biodiversity in LCA in focus, a selection of common key concepts and methodological principles are described in this section. Land cover is defined by Di Gregorio and Jansen (2005) as "the observed (bio) physical cover on the earth's surface, including the vegetation (natural or planted) and human constructions (buildings, road, etc.) which cover the earth's surface". Land use is defined by Köllner et al. (2013b) as the arrangements and activities by human interventions on a specific land cover type in order to produce, change or maintain it. As proposed by Köllner et al. (2013b), land use can be classified on several different levels ranging from global land cover classes which are divided into more refined categories and further into very specific categories of different land use intensities.The UNEP-SETAC framework (Köllner et al. 2013a) gives recommendations on how to assess impact on biodiversity from land use in LCA and the following two types of land use are included in the framework. Transformational land use, is a short phase where a piece of land is changed considerably, e.g. through drainage to make it fit for agriculture or forestry. Occupational land use, refers to the period of time when a piece of land is used for a specific purpose, see Figure 4. In the framework, the quality on biodiversity is assumed to remain constant from the start to the end of the occupational land use phase. The impact from occupational land use is calculated as the difference between the quality on biodiversity of the land occupied and a reference situation. When modelling biodiversity in LCA, data is collected from the land used for transformational or occupational land use and the LCIA method on biodiversity generates a characterization factor, which is expressed as a biodiversity damage potential (see Köllner et al. 2013 for more details).

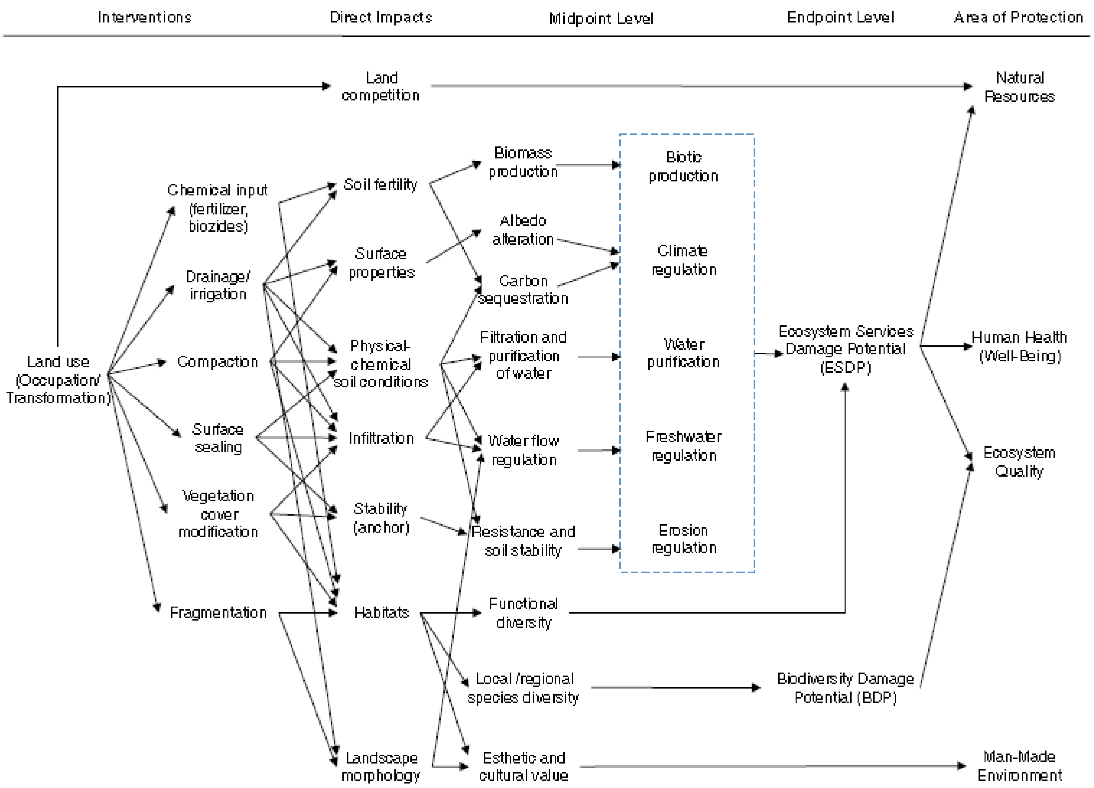


Figure 4 Cause effect chains for land use impacts on biodiversity and ecosystem services, according to UNEP-SETAC (2013).

2.2.1.3 *Reference situation*

As the biodiversity damage potential is recommended by the UNEP-SETAC guidelines to be calculated as the difference in ecosystem quality between land used and a reference situation, a reference situation requires to be defined. The guidelines suggest three different reference situations and these are 1) the potential natural vegetation (PNV), which a hypothetical description of what a landscape would look like if all anthropogenic interventions would stop, 2) the quasi natural land cover in each biome or ecoregion i.e. the current land use mix, e.g. the natural mix of forests, wetlands, shrublands, grassland, bare area, snow and ice, lakes and rivers and 3) average measure for current mix of land uses as proposed by (Köllner and Scholtz. 2008). Other definitions on reference situations exist, for example the semi natural reference situation proposed by de Baan et al. (2013b) , which represents current late succession habitat stages of forests, grasslands, wetlands, bare areas and water bodies, i.e. areas that are often used as targets for restoration ecology. Another proposal made by Lindner et al. (2014) is a hypothetic reference situation represented by the maximum quality of biodiversity in the region based on “desired state of biodiversity as defined in national strategy documents”. Further, different reference situations are recommended to be used for different types of LCA, attributional or consequential (Köllner et al. 2013a). In attributional LCA, the impact caused by occupational land use should be assessed against a state of natural relaxation, i.e. PNV (Milà i Canals et al. 2007). In consequential LCA, the difference between the land use resulting from a change in the system in relation to an alternative activity (Milà i Canals et al. 2007) is focused. The alternative activity e.g. another type of land use is then the reference. Among other things this implies that the impact on biodiversity may be either positive or negative.

2.2.1.4 *Land cover system*

According the guidelines proposed by Köllner et al. (2013) the quality on biodiversity on the land used and on the reference situation is required to be assessed. Generating CFs by e.g. for different geographical locations, requires identification of the type of land cover. Land cover classification systems can serve as an instrument for defining land use types as the characteristics of different land covers are relevant in order to compare for example coniferous forest on two different continents. One example of many is the Coordination of Information on the Environment (CORINE), which is a European programme coordinating data sets of land use types. The CORINE data base includes information on land cover classification of 38 countries with a total area of 5.8 Mkm^2 (European Environment Agency 2007).

2.2.2 Biodiversity in LCIA

The topic of this thesis, research on LCIA methods on biodiversity in LCA worked as an incentive to investigate to what extent impact assessment on biodiversity have been included in LCAs on wood based products. A preparatory search was made by the use of the Scopus database using the keywords**, “**forest”, **“**wood”, "life cycle analysis", "life cycle assessment", and “LCA”. The results showed that twenty percentage of a total of 90 articles published between 1997 and 2013 included quantitatively biodiversity assessments made by ready-made impact assessment packages. To meet new requirements made by e.g. political goals and international standards, existing ready-made impact assessment packages in LCA have over a period of time developed to include impact assessment on biodiversity. The historical development of dealing with biodiversity issues in LCA up to current is described in the following sections.

One of the first LCA initiatives to include biodiversity concerns was the EPS system (Environmental Priorities Strategies in product development) (Steen 1999). In EPS, biodiversity is one out of five or impact categories, or areas of protection as they are called. Assessment of impact on biodiversity is in EPS based on contribution to species extinction; the category indicator defined as ‘the normalized extinction of species’.

In contrast to the EPS system, the LCA guidelines from Centre of Environmental Science – Leiden University (CML) (Guinée et al. 2002) and that from The United States Environmental Protection Agency (US EPA) (Curran 2006) contained very little information on how to deal with biodiversity issues. The recommendation in the CML guideline from 2002 was to not include biodiversity impacts due to methodological restrictions. However, from 2010 the CML guideline includes additional characterization factors from e.g. Eco-indicator 99 and EPS, in which characterization factors for biodiversity are incorporated (CML 2012).

Over the last ten years the interest in how to capture biodiversity issues in LCA has increased dramatically and there are now several proposals for how to include biodiversity in LCA, most often related to land use. The proposed biodiversity indicators predominantly assess specific species or taxa, with diversity of vascular plants being the most commonly proposed indicator (see e.g. Köllner and Scholtz 2008). The existing guidelines from UNEP-SETAC on land use impact assessment on biodiversity and ecosystem services in LCA (Köllner et al. 2013a), recommend assessment at two levels: species (de Baan et al. 2013b) and functional diversity (Souza et al. 2013), each for which characterization factors are proposed. Additionally, methods based on assessment by means of identified key factors have been developed (Lindner et al. 2012; Michelsen 2008) and combinations of different levels such as species and potential ecosystem areas (Weidema P Bo and Lindeijer Erwin 2001).

The UNEP-SETAC guidelines (Köllner et al. 2013a) recommend two main impact pathways. Figure 4, illustrates cause-effect chains with the linking pathways between impact categories from land use interventions to areas of protection: natural resources, human well-being and ecosystem quality. Various pathways can be defined towards two endpoints, ecosystem services damage potential (ESDP) and biodiversity damage potential (BDP). Notably, neither of these are termed "areas of protection", indicating that ecosystem services as well as biodiversity are regarded as instrumental values contributing to human well-being, ecosystem quality and man-made environment.

3 Methods

This chapter describes the research approach of the three studies that are part of my thesis, see Figure 5. The first step was a literature overview pertaining current LCIA methods on biodiversity with potential use for bio based products. Two different LCIA methods were selected for further application in two case studies on forestry. The methods used for conducting the studies reported in papers I, II and III are described in the following sections.

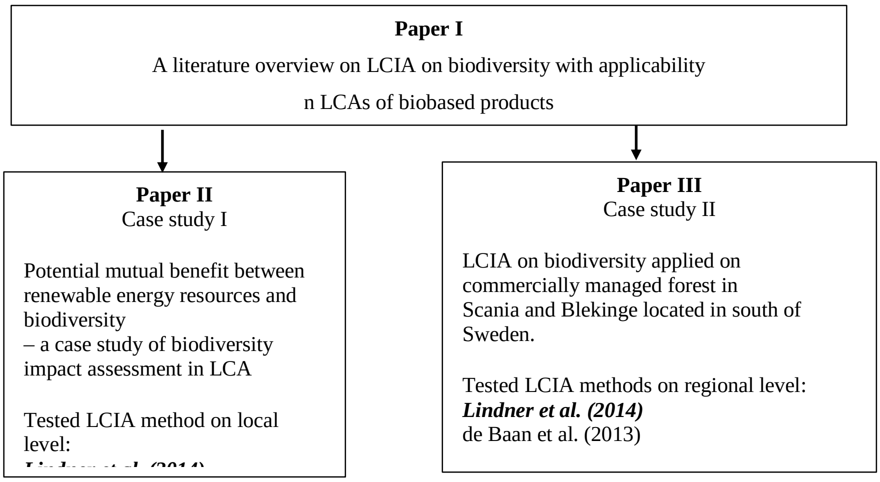


Figure 5 Simplified illustration of the research approach

3.1 Method for literature overview

A literature study was conducted with the aim to find LCIA methods capturing biodiversity impacts from land use. The selection of methods was based on the criteria that the LCIA method was developed for bio based products or with potential applicability within forestry, and that the impact assessment method provided an apparent characterization factor, which can be integrated into an LCA. The Summon search engine[1] was used for search with the keywords "impact assessment +biodiversity +LCA +land use", combined with "life cycle assessment". For cross references, Scopus and Google Scholar were used. This resulted in a total number of eighteen LCIA methods on biodiversity.

The total numbers of LCIA methods were analyzed with respect to a number of methodological characteristics as they reappeared in all or most of the methods studied, except for feasibility which was included for the opposite reason, i.e. that discussion on the topic is lacking, see Table 1.

[1] Summons is a search engine covering all articles, books, e-books and other types of materials held by Chalmers' library

Table 1 Framework for analysis of methodological characteristics found in the literature overview

Findings from the literature overview pertaining methodological characteristics	**Explanation of methodological characteristics**
Biodiversity aspect and target taxonomic group (if relevant)	Explains the biological process(es) captured
Biodiversity indicator and underlying statistical model	Metrics used in the method
Reference state	Creates understanding of the impact scale and the relation to the potential original species pool
Spatial scale	For which geographical scale the method is adjusted
Production system or land use type investigated	
Type of impact Occupational Transformational Permanent	Land use type
CF (per m^2 for T and per m^2 and year for O)	Abbreviation of the damage factor
Signification of CF	Explanation of the CF
Feasibility	The degree of possibility to apply the method

3.2 Case study methods

The findings in the literature overview resulted in a variety of different LCIA methods on biodiversity. Based on the framework, see Table 1, the methods were organized into two main categories of approaches, those based on of relative species and those based on ecosystem indicators.

For the case studies the intention was to test one method of each type, one based on of relative species and one based on ecosystem indicators. This was done for case study II. However, for case study I, the data proves not to be available in a format which allowed for the use of a species based method, in spite of the case study area being unusually well inventoried. As a result, in case study I only one method, which was based on ecosystem indictors, was applied.

3.2.1 The selected methods

The method developed by Lindner et al. (2014) is based on the WWFs definitions on ecoregions by multiplying the impact calculated with an ecoregion factor (EF). EF is a description of a combination of ecosystem scarcity (smallness) and ecosystem vulnerability (area left with no anthropogenic impact). The Lindner et al. (2014) method was selected for three reasons. Firstly, the impact assessment on biodiversity was based on ecosystem indicators identified by an ecological expert on the ecoregion in the study, see Figure 6. Secondly, the characterization factor was obtained by the use of a multivariate function, which generates a total biodiversity contribution from the ecosystem indicators, see box *ii* in Figure 6. Thirdly, the method included a reference situation representing a hypothetical maximum quality biodiversity in the ecoregion according to expert knowledge and in agreement with existing policy documents.

The species based method developed by de Baan et al. (2013b) was selected for two reasons. The first reason was because the method generates a characterization factor by the use of a relative measure for species richness, see box *v* in Figure 6. The second reason was that the method developed by de Baan et al. (2013b) suggests the use of semi natural reference situations, which requires a regional specific interpretation.

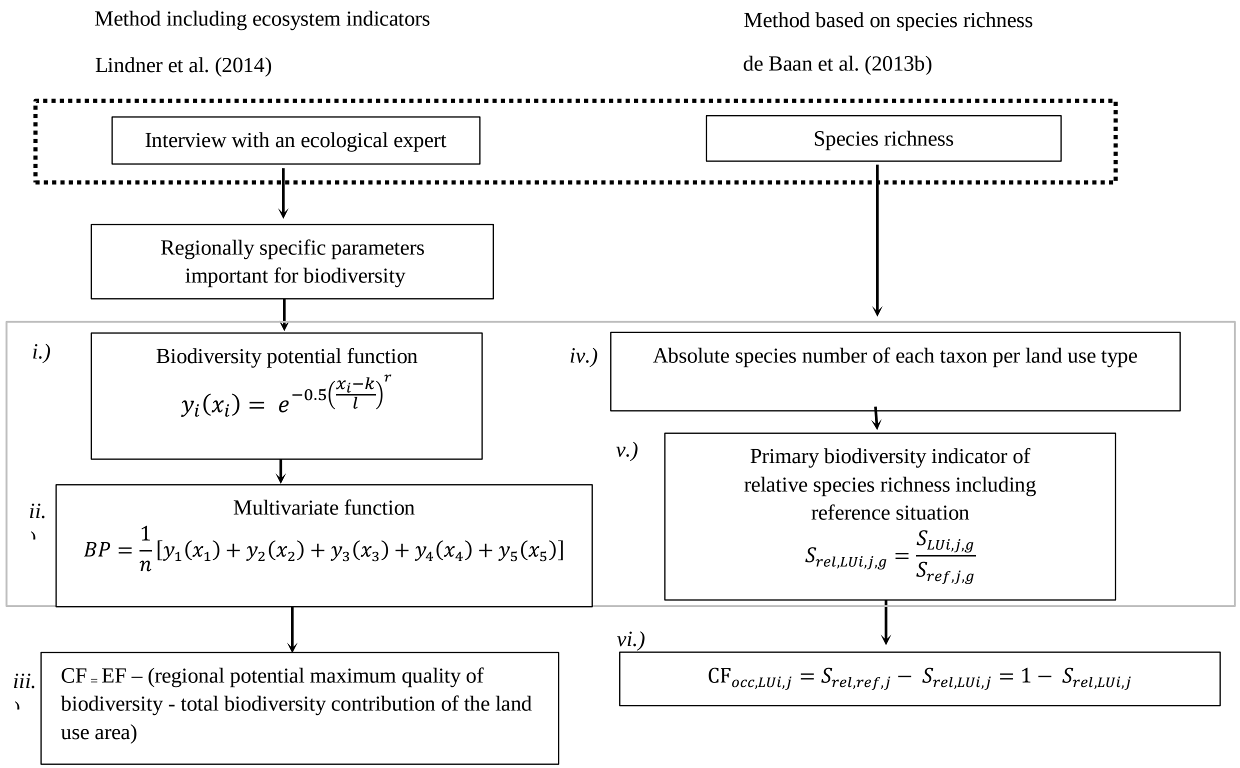


Figure 6 Description of the methods used in case study I and II as proposed by the methods.

	Input data	r	Fitted constant based on expert opinion	$S_{rel,\,LU}$	Relative species richness on the land used
	Metrics	BP	Biodiversity potential	i	Land use type
y_i	Contribution of parameter	n	Number of contributions	j	Region
x_i	Parameter important for biodiversity	CF	Characterization factor	g	Taxonomic group
k	Fitted constant based on expert opinion	S_{LU}	Species richness on land used	$S_{rel,ref}$	Relative species richness in the reference situation
l	Fitted constant based on expert opinion	S_{ref}	Species richness in reference situation	$S_{rel,\,LU}$	Relative species richness on the land used
				i	Land use type

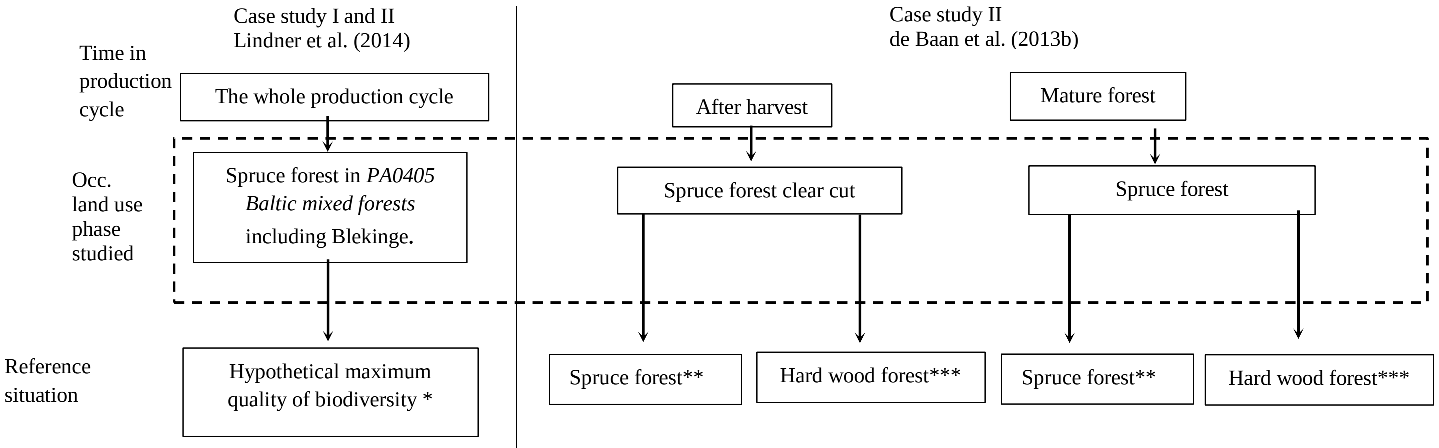
3.3 The case studies

Two case studies on two different types of forestry were made. Case study I applied the method developed by Lindner et al. (2014) locally on a 1 ha coppice forest growing on a hay producing meadow, located in southern part of Sweden, see Case study I in Paper II. In case study II, two different LCIA methods on biodiversity were tested on a regional level on commercially managed spruce forest, also located in southern part of Sweden. The two methods tested were the methods developed by Lindner et al. (2014) and de Baan et al. (2013b) .Both methods have been developed in line with the UNEP-SETAC framework for land use impact assessment on biodiversity (Köllner et al. 2013a).

3.3.1 Adjustments made in the applied methods

In case study I and II the selected LCIA methods on biodiversity where tested to assess impact on biodiversity on commercially managed forest in southern part of Sweden. The two selected methods are distinguished from one another by their choice of biodiversity indicator, difference in metrics and suggested choice of reference situation.

When applying the LCIA methods on the case studies adjustments pertaining time frames, choice of reference situations and calculation of species richness were required. An overview of the adjustments made in case study I and II are given in Figure 7, which is followed by a description of data sources used and methodological adjustments made for the two methods



after harvest	start of the production cycle	*	Reference situation representing a hypothetical maximum quality biodiversity in the ecoregion according to expert knowledge and in agreement with existing policy documents
mature forest	end of the production cycle	**	Reference situation representing PNV consisting of planted 97%, spruce forest with a stand age of 55-105 years
		***	Reference situation representing PNV consisting of 99%, mixed hard wood forest with a stand age of approximately 110 years

Figure 7 Schematic figure showing the methodological adjustments pertaining time frames and reference situations made when operationalizing the two selected LCIA methods.

3.3.1.1 *The method developed by Lindner et al. (2014)*

The method developed by Lindner et al. (2014) was applied in both case study I and II and is developed to assess impact on biodiversity in the ecoregions described in the WWF Wildfinder (2015), based on prerequisites for biodiversity identified by ecological expert[2,3]. The characterization factors obtained in case study I and II do not include EF but follow the four step procedure according to the Lindner et al. (2014) methodology. In accordance with the methodology, four questions were asked, to achieve a description of the regional biodiversity with the objective to identify prerequisites important for biodiversity, see Table 3. These questions are developed to capture qualitative and quantitative aspects of different prerequisites and their specific impact on biodiversity. The qualitative questions included a discussion of the importance of each prerequisite, e.g. dead wood, and its relation to biodiversity. Following this, quantitative questions were designed to identify the intensity and amount of specific prerequisites for biodiversity. The experts were asked to give their assessments of the production cycle as a whole. The reference situation used was taken to be the maximum biodiversity quality possible of natural land cover in the ecoregion according to expert knowledge and in agreement with existing policy documents.

Table 3 Interview questions according to the Lindner et al. (2014) methodology

	Interview questions
1	How would you describe biodiversity in the WWF ecoregion?
2	How do you identify a place with high biodiversity in the region?
3	What land management parameters are important for biodiversity?
4	In what way do structural parameters influence biodiversity in more or less wooded areas?

In the first step of the interview with the experts, the answers generated by the interviewees were compiled into a written description of the identified prerequisites for the regional biodiversity. The compiled answers generated were returned to the experts to verify the validity. Step one was followed by an iterative process between the experts, method developer and author in order to create biodiversity contribution curves for each prerequisite, in which the amount and intensity of the identified prerequisites and their relative importance for biodiversity were expressed. In the third step the biodiversity contribution from each prerequisite was assessed according to a multivariate function in order to generate the total biodiversity contribution. The fourth step that of the calculation of the characterization factor was conducted through subtracting the total biodiversity contribution from that of the reference situation. Sensitivity analysis was conducted though varying the four prerequisites obtained from the experts +/- 10 percent in a systematic manner.

3.3.1.2 *Selected reference situations and period of time in production cycle for impact assessment*

Based on the suggestion made by Milà i Canals et al. (2013) to investigate the use of different reference situations, in our cases we assessed biodiversity using three different reference situations. Two of these were different assumptions of what the potential natural vegetation (PNV) can possibly be like, i.e. hypothetical descriptions of what a landscape would look like if all anthropogenic interventions would stop. In other words, if the studied land, in this study commercially managed spruce forest, is left unoccupied, the PNV describes what it, could possibly evolve into in the future. One option is that that the commercially managed spruce forest, if left unmanaged, will

[2] For case study I, Professor Urban Emanuelsson, plant ecologist with focus on cultural landscapes and historical ecology, Centre for Biodiversity with affiliation at the Swedish University of Agricultural Sciences (SLU), Uppsala University (UU).

[3] For case study II, Professor Jörg Brunet, ecologist with affiliation at SLU in Alnarp. An additional expert, Professor Lena Gustafsson, conservation biologist with affiliation at SLU in Uppsala, was asked to give a second opinion.

simply evolve into an older spruce forest. The oldest spruce forest for which data were available for southern Sweden had a stand age of 55-105 years, and was hence chosen as a reference, in spite of being commercially managed, 97%, spruce forest. The other option is that the spruce forest will evolve into a mixed hard wood forest, more similar to what once grew in the studied area. Data were available for a 99% mixed hard wood forest with a stand age of approximate 110 years, which was used as a second reference. Both these possible descriptions if a PNV were used when applying the method developed by de Baans et al. (2013b). The third choice of reference situation, applied when the method developed by Lindner (2014) used, was that of a hypothetical maximum quality biodiversity in the ecoregion according to expert knowledge and in agreement with existing policy documents.

Both methods tested in this study, de Baan et al. (2013b), Lindner et al. (2014) are based on the framework for LCIA of land use, developed by UNEP-SETAC (Köllner et al., 2013b). In the framework land quality is assumed to be constant during occupation. However, for forestry this assumption proved to be difficult to apply, since biodiversity varies extensively over the production cycle. For this reason, a higher temporal resolution was used in this study. For the method developed by de Baan et al. (2013b), impact on biodiversity was assessed at two points in the production cycle, after harvest, and in the mature forest, see Figure 7. In addition, this was done using two different reference situation, hence four characterization factors were calculated see Figure 7.

The characterization factor generated by the method developed by Lindner et al. (2014), which is based on ecosystem indicators, assessed impact on biodiversity by the use of a reference situation representing a hypothetical description of the maximum quality on biodiversity in Scania and Blekinge, according to expert knowledge and in agreement with existing policy documents, see Figure 7. The generated characterization factor represented impact on biodiversity over the whole production cycle.

3.3.1.3 *Quantification of species richness*

With case study II and the method developed by de Baan et al. (2013b) in focus, quantification of species richness was conducted by the use of presence-absence data in rarefaction type Chao2 species estimator by extrapolation, see equation 3 described in section 2.1.1.2, using the statistical analysis software PRIMER (Clarke & Gorley 2001 & 2006). Presence-absence data on vascular plants was taken from the Swedish National Forest Inventory (SNFI) (2010). In order to receive a sufficient data as required by species estimation methods used, forests growing in Scania and Blekinge were chosen. Data sets based on presence- absence data from plots of equal size of 100m^2 where compiled. A limited number of sample plots from the total area of 89853 hectare commercially managed forest in Scania and Blekinge were included in the study. SNFI provided information on 87 plots from after felling, 94 plots from before felling on the occupational land use, 137 plots from the reference situation of spruce and 73 plots from the reference situation of mixed hardwood forest.

4 Results

4.1 Compilation of the results from the literature overview

The purpose of the literature overview was to analyze a selection of LCIA methods on biodiversity. The results are compiled in Table 4 and the results are described in the following sections below.

4.1.1 Biodiversity aspect and target taxonomic group

Table 4 shows that the species level of biodiversity is the biodiversity aspect captured in a vast majority of the methods, and most often expressed as species richness. In the earliest methods, this is the only aspect captured. Ecological scarcity and ecological vulnerability appear in the LCIA methods in 2001. Lately (towards the top of Table 4) the range of biodiversity aspects captured has broadened further with the inclusion of prerequisites for biodiversity and functional diversity. The six ready-made impact assessment packages (EPS, Eco-Indicator, Impact, Lime, ReCiPe, Ecological Scarcity; marked with an asterisk in Table 4) all rely on species richness.

The most commonly used species is the taxonomic group vascular plants, which is often used as a proxy for all biodiversity. Here there is a diversification towards the top of the table with more frequent occurrence of other groups, such as mammals, birds and amphibians, in later years. A couple of methods combine data on species with other parameters, such as ecosystem vulnerability and ecosystem scarcity, or cost for conservation of the ecosystem.

4.1.2 Biodiversity indicator and underlying statistical model

There is a wide span in the indicators listed in Table 4, from globally applicable indicators such as EV, to the regionally or locally specific.

The by far most frequently used biodiversity indicator in the methods investigated is alpha diversity (number of species in an area) calculated by use of some kind of species estimation method. The most frequently recommended such method is that by Hurlbert (1971), but also Arrhenius (1921), Matsuda (2003) and Koh & Ghazoul (2010) are being used. Lately, reliance on expert knowledge has been introduced as a completely different approach to assessing biodiversity e.g. Michelsen (2008), Jeanneret et al. (2014) and Lindner et al. (2014). The indicators ecological scarcity (ES) and ecological vulnerability (EV) are only used as a complement to other indicators and the two are normally recommended together, except by Schmidt (2008) who uses EV only as a weighing factor in combination with species richness. Generally, the diversification of indicators for capturing biodiversity is increasing, as is the tendency to recommend multiple indicators.

Within the species based indicators there is room for variation. Among the methods in Table 4 are recommendations mostly on species in general, or on threatened species. Extinction of a species is an obvious loss and an example of a permanent impact from land use. Data on threatened species is furthermore available through the IUCN red data list (IUCN Red List 2014).

4.1.3 Reference state

In the UNEP-SETCs guidelines, Köllner et al. (2013b) propose three different reference situations: 1) potential natural vegetation (PNV; what would become if human intervention stopped), 2) quasi natural land cover in e.g. each ecoregion or biome (presently existing most authentic vegetation) and 3) current regional average number of species. Sometimes the reference state is represented by an actual situation in an ecosystem where biodiversity assessments have been made, but sometimes it's a hypothetical situation, like PNV. Most of the reference situations found in Table 4 belong to one of these categories but there are some exceptions. Weidema and Lindeijer (2001), Michelsen (2008) and Coelho and Michelsen (2014) define their reference situation as the natural state given by ES and EV. In Lindner et al. (2014), the reference situation is defined as "the desired state of biodiversity as defined in national strategy documents", which implies a hypothetical highest level of biodiversity, including states of above natural biodiversity due to management of landscapes in line with e.g. cultural heritage.

Naturally, this could be the same kind of reference situation as the quasi natural situation described above, but we prefer to treat it separately as it obviously includes also managed landscapes in the reference situation. Notably, the descriptions of the different reference situations in the guidelines (Köllner et al. 2013a) are not very clear and it is obvious that in the methods studied, the authors have made their own interpretation of the reference situations introduced in the previous section.

The reference situation of PNV appears to be problematic to define because of the difficulty in predicting how today's ecosystems will be affected by major drivers such as large mammals, forest management, wild and cultural fires, soil, climate (change) and invasive species. The definition of PNV is based on the definition by Chiarucci et al. (2010). These authors, however, argue that it is too problematic to define and model PNV and suggest that the concept should be abandoned.

The definition of the semi natural reference situation goes back to current late succession habitat stages often used as targets for restoration ecology. The definition on semi natural leaves great room for interpretations as it could be represented by different types of vegetation and vary between regions. It is also unclear whether managed ecosystems should be included or not. In order to use a semi natural reference situation in a study, an interpretation of the definition in general, and in the specific region, in particular, has to be made.

Ten out of eighteen LCIA methods on biodiversity suggest the use of a regional reference situation. Seven of these ten methods rely on species average from the Swiss lowlands. All the ready-made impact assessment packages use the data base Ecoinvent, which imply that the data on numbers of species used reside in Switzerland. Few methods, those of Michelsen (2008), de Souza et al. (2013), LIME (2005) use a larger geographical scale as reference situation.

4.1.4 Spatial cover

The global and (eco) regional scales are captured in many methods; local in very few. The most common ambition is obviously to develop methods with a wide geographic applicability, and avoiding the need for local or case specific data. As a consequence these methods will be rather coarse grained. Among the methods with a (potentially) higher spatial resolution, the challenge of acquiring local data has been met either by using a global species list on vascular plants, geographic information systems (GIS) for inventory modelling (Geyer et al. 2010), or by making use of expert knowledge rather than relying on detailed monitoring of species (Lindner et al. 2014 and Jeanneret et al. 2014).

4.1.5 Operationalized spatial cover

This section leaves out the ready-made impact assessment packages as it was beyond our scope to investigate all cases that these have been applied to. It can be noted however that only one of the ready-made packages, the Japanese LIME method, was developed for non-European conditions. Also when looking at the other methods studied, and the affiliation of the authors, the entire biodiversity-in-LCIA-project is strikingly European. Most of the methods analysed have been operationalized in Europe, followed by South America and a limited number of studies in North America and Asia. Africa is left out entirely for the time being.

4.1.6 Production system or type of land use studied

Looking at production systems and land use types captured by the methods studied, we find methods covering "everything", e.g. de Baan et al. 2013b, at the one end of the spectrum, and those specifically developed for a specific land use type such as forestry (e.g. Michelsen et al. 2008) or agriculture (e.g. Jeanneret et al. 2014), at the other end. Generally, the methods with large spatial covers also include many different kinds of land use types, while methods designed for a more limited spatial cover, and a more detailed scale, are more specific also with regard to types of land use or production systems covered.

Notably, all recent methodological development is largely in line with the UNEP-SETAC guidelines (Köllner et al. 2013a), so in spite of some apparent differences between the methods they all follow the same basic methodology.

4.1.7 Type of impact

All methods studied were designed to assess occupational impacts from land use. Four methods on addition to this assess transformational impacts, and one (de Baan et al. 2013a) also captures permanent impacts.

4.1.8 Characterization factor generated and its signification

The list of characterization factors (CFs) in Table 4 may appear heterogeneous at a first glance, but can be grouped into two categories: those that refer to changes in species diversity in one way or another, which is the majority, and those that express the change in ecosystem quality as captured by ecological indicators, ES and EV included. Two methods deliver CFs that include both species richness and EV and ES, (Weidema and Lindeijer 2001) or species richness and EV (Schmidt 2008).

Most of the methods generating species based characterization factors deliver relative values for species loss. One such characterization factor is ecosystem damage potential (EDP) which calculates for the anticipated number of species compared with actual encountered number of species. The values can be extracted from calculating α diversity and βdiversity. The first characterization factor of $EDP^{sp\text{-}div}$ was developed by (Köllner. 2002), to be further developed by (Köllner and Scholtz. 2007, 2008) and finally implemented in Ecological Scarcity in 2009. An additional characterization factor based on relative species richness is the biodiversity damage potential (BDP), developed by de Baan et al. (2013b) which is one of the two methods recommended by the UNEP-SETAC guidelines (Köllner et al. 2013b). The other one is de Souza et al. (2013) where the focus is on functional diversity, which is also expressed as a relative loss.

Another characterization factor, based on relative species richness is the potential disappeared fraction (PDF) found in IMPACT2002+, ReCiPe and Eco-Indicator 99 and calculates values for the ratio of species lost during a certain time per area, which explains the damage to the ecosystem diversity. An additional species based characterization factor based on rare species is normalised extinction of species (NEX), generated by EPS2000, which calculates the contribution to the extinction of species within one year expressed as dimensionless ratio, combined with a monetary value for the cost of conservations area needed.

The second category of characterization factors rest on ecosystem indicators that capture key factors for maintaining biodiversity and informs about the present conditions of the biodiversity in the area. There are suggested key factors for maintaining biodiversity in the literature (Franc et al. 2000) but they can also be delivered by an ecological expert. The choice of indicators depends on the type of ecosystem, its inherent character of scale in combination with structural, compositional and functional conditions.

Table 4 Overview of analyzed methods for integrating biodiversity in life cycle impact assessment

Reference	**Biodiversity aspect and target taxonomic group (if relevant)**	**Biodiversity indicator and underlying statistical model**	**Reference state****	**Spatial scale**	**Production system or land use type investigated**	**Type of impact** **<u>O</u>cc.** **<u>T</u>ransf.** **<u>P</u>erm.**	**CF (per m^2 for <u>T</u> and per m^2 and year for <u>O</u>)**	**Signification of CF**
Jeanneret et al. (2014)	11 indicator-species groups (ISGs): grassland flora, crop flora, birds, mammals, amphibians, snails, spiders, carabid beetles, butterflies, wild bees, grasshoppers	Impoverishment or promotion of diversity within the 11 ISGs rated 1-5 based on expert knowledge	*Intensive hay production and intensive integrated winter wheat production*	Local to regional	Grassland (hay) and winter wheat	O	Score	Rated and weighted impact on indicator-species groups
Lindner et al. (2014)	Prerequisites for biodiversity	Ecological indicators	Hypothetical highest possible in the ecoregion	Ecoregion	Coppice forestry (Lindqvist et al. 2014)	O	ΔQ	Change in quality of biodiversity as expressed by ecological indicators
Coelho and Michelsen (2014)	Ecological scarcity (ES), ecological vulnerability (EV) and hemeroby	ES, EV and hemeroby	Natural state (=ESxEV)	Local to global	Kiwifruit Forestry plantation (Michelsen et al. 2014)	O	ΔQ	Change in quality of biodiversity in terms of ES, EV and hemeroby
de Baan et al. (2013a)	Species richness of plants, mammals, birds, amphibians, reptiles	α-diversity, SAR, (Koh and Ghazoul 2010)	Natural habitat	Ecoregion and global	Agriculture, pasture, managed forests, urban areas and natural habitats Forestry plantation (Michelsen et al. 2014)	O, T, P	rBDP	Regional biodiversity depletion potential based on extinction of non-endemic and endemic species

Reference	**Biodiversity aspect and target taxonomic group (if relevant)**	**Biodiversity indicator and underlying statistical model**	**Reference state****	**Spatial scale**	**Production system or land use type investigated**	**Type of impact** **<u>O</u>cc.** **<u>T</u>ransf.** **<u>P</u>erm.**	**CF (per m^2 for <u>T</u> and per m^2 and year for <u>O</u>)**	**Signification of CF**
de Baan et al. (2013b)	Species richness of plants (vascular and moss), arthropods, other invertebrates and vertebrates (birds and other)	α-diversity Fisher's α Shannon's entropy *H* MSA Sørensen's S_s	Current late "semi-natural" succession habitat stages	Biome and global, Ecoregion (Michelsen et al. 2014)	Forest, used and not used, agroforestry, annual and permanent crops, pasture, secondary vegetation, artificial areas Forestry plantation (Michelsen et al. 2014)	O	BDP	Biodiversity damage potential based on relative changes in species richness
Maia de Souza et al. (2013)	Functional diversity linked to species richness of mammals, birds and plants	α-diversity, functional diversity based on functional trait values	Late succession stage in actual land cover (representing PNV)	Ecoregion	Agriculture (6 classes of), forests (5 classes), field-margins, grassland, pasture, shrubland,	O	$-ln(FD_N)$ $-ln(SR_N)$	Relative loss of functional diversity and species richness

					wetlands, artificial rivers and lakes, infrastructure			
Geyer et al. (2010)	Hemeroby and species richness, abundance and evenness of terrestrial vertebrates	Hemeroby, species richness (ln S/Sref), richness + abundance, and richness + evenness (Simpson index); the 3 latter based on a species-habitat suitability matrix	Not applicable for 3 of 4 CFs. For sp. richness: the maximum number of known vertebrates in the area	Regional	Corn and sugar beet for ethanol production	O	4 different: see column 3	Various aspects of vertebrate diversity
Ecological* Scarcity (2009)	Species richness of vascular plants	α-diversity, SAR (Hurlbert 1971)	Regional average (Köllner, 2001)	Regional	*NA*	O, T	EDP	Ecosystem damage potential based on relative changes in species richness
Reference	**Biodiversity aspect and target taxonomic group (if relevant)**	**Biodiversity indicator and underlying statistical model**	**Reference state****	**Spatial scale**	**Production system or land use type investigated**	**Type of impact** **Occ.** **Transf.** **Perm.**	**CF (per m^2 for T and per m^2 and year for O)**	**Signification of CF**
ReCiPe* (2008)	Species richness of vascular plants and lower organisms	α-diversity, SAR (Hurlbert 1971)	PNV	Regional	*NA*	O, T	PDF	Potentially disappeared fraction of species
Michelsen (2008)	Ecological scarcity (ES), ecological vulnerability (EV) and	ES, EV and conditions for	Natural state (=ESxEV)	Ecoregion	Spruce forestry	O	ΔQ	Change in quality of biodiversity in terms of ES, EV and CMB

	key factors for biodiversity	maintained biodiversity (CMB)						
Schmidt (2008)	Species richness of vascular plants, ecological vulnerability (EV)	α-diversity, SAR (Arrhenius 1921) EV	Current renaturalisation potential	Regional	Arable (cereals and grass), agroforestry, forestry (managed and nature), nature (heath and scrub, grass, and bog), and sealed land	O, T	wS100	Weighted (by use of EV) species richness on a standardized 100 m^2
LIME2* (2005)	Species richness of endangered vascular plants	α-diversity, SAR (Matsuda 2003)	*Maximum population that can be maintained stably in the habitat*	Global	*NA*	O	EINES	Expected increase in number of extinct species
IMPACT 2002+* (2003)	Species richness of vascular plants	α-diversity, SAR (Hurlbert 1971)	Average species number in the region	Regional	*NA*	O	PDF	Potentially disappeared fraction of species
Weidema and Lindeijer (2001)	Species richness of vascular plants, ecosystem scarcity (ES) and ecosystem vulnerability (EV)	α-diversity, SAR (Arrhenius 1921), ES, EV	Maximum actual species diversity based on ES and EV	Global	Rape seed production in Europe and soy bean production in Brazil	O	ΔQ	Change in quality of biodiversity in terms of vascular plant species richness, ES and EV
Reference	**Biodiversity aspect and target taxonomic group (if relevant)**	**Biodiversity indicator and underlying statistical model**	**Reference state****	**Spatial scale**	**Production system or land use type investigated**	**Type of impact** **Occ.** **Transf.** **Perm.**	**CF (per m^2 for T and per m^2 and year for O)**	**Signification of CF**

Lindeijer (2000)	Species richness of vascular plants	α-diversity	Maximum actual species (α-) diversity in the region	Region, global	Sand extraction, aluminum mining, landfill household waste, hydropower, road traffic, forestry	O, T	ΔQ	Species diversity
Eco-Indicator 99* (2000)	Species richness of vascular plants	α-diversity, SAR (Hurlbert 1971)	Average species number in the region	Regional	*NA*	O, T	PDF	Potentially disappeared fraction of species
Köllner (2000)	Species richness of vascular plants, threatened and common species	α-diversity, SAR (Hurlbert 1971)	Average species number in the region	Local and regional	Urban (4 types), industrial (2), rail, mining fallow, arable (3), meadow (3), broad-leaf forest	O, T	SPEP	Species pool effect potentials; species loss at local and regional level
EPS2000* (1999)	Species richness, large mammals and birds	Number of lost species	*Present state*	Regional	*NA*	O	NEX	Normalized extinction of species

Abbreviations (see text for details):
SAR = Species Area Relationship
MSA = Mean Species Abundance of original species
PNV = Potential Natural Vegetation
NEX = Number of Extinctions

4.2 Case studies

Two case studies were performed and the results from Case study I, applied on a local level, are presented in Table 5 and the results from Case study II, applied on regional level, are presented in Table 6.

4.2.1 Case study I

The purpose with Case study I was to perform a local impact assessment on biodiversity in LCA by the use of the LCIA method on biodiversity developed by Lindner et al. (2014), which is based on the World Wildlife Fund (WWF) ecoregions. The occupational land use investigated was a managed meadow with sparsely distributed hardwood trees of various species located in south part of Sweden. The regional specific prerequisites for biodiversity identified by the ecological expert are presented in Table 5. Table 5 also presents the intermediate results of the biodiversity potential functions and the generated characterization factor, which are described in the following text.

Table 5 Intermediate calculation steps in Case study I, showing the identified parameters important for biodiversity and for each parameter the following are presented, parameter value, normalized value, biodiversity contribution and weighting value. Based on these values the total biodiversity contribution is calculated according to the method developed by Lindner et al. (2014) as a weighted arithmetic mean.

Regional specific prerequisites	**Value**	**Normalized value**	**Contribution**	**Weight***	**Total contribution**
Biomass removal (NPP/ha)	1	0,04	99%	30%	30%
Fertilizer input (kg surplus/ha)	0	0,00	100%	30%	30%
Pesticide (CTUe/ha)	0	0,00	100%	15%	15%
Amount of dead wood (m^3/ha)	7,4	0,02	38%	25%**	6%
Age of dead wood (years)	30	0,30	73%		
Total biodiversity contribution					81%*
CF					**100%*** - 81%** = 19%

*= Weighted arithmetic mean
**= The parameters amount of dead wood and age of dead wood are dependent on each other and are together multiplied with the same weight
*** = Reference situation representing a hypothetical maximum quality biodiversity in the ecoregion according to expert knowledge and in agreement with existing policy documents

As can be seen in Table 5, the total biodiversity contribution of the identified prerequisites is a mean value of 81 percent. Table 5 also shows that no values on fertilizers and pesticides were included. The results from the case study show the proposed management of the forest in which no input of fertilizer is tolerated. It is assumed that all surplus nitrogen (N) is absorbed. Figure 8 illustrates that the area has a tolerance of approximately 20 kg N and the following quick drop reflects that the area is sensitive to N. If pesticides would have been included, the shape of the curve would be very similar to the fertilizer curve. The tolerance for pesticides is smaller and the curve would drop immediately since pesticides are harmful to biodiversity. The results also show that the two identified parameters of biomass removal (NPP/ha) and fertilizer input (kg surplus/ha), were given the greatest weight and thus by applying an arithmetic mean have the greatest influence on the total biodiversity contribution. Table 5 shows the calculation of the characterization factor value of 19 percent, which was obtained by subtracting the total biodiversity contribution value from that of the reference situation. This implies that the quality of biodiversity on the land occupied is reduced with 19 percent compared with the reference situation.

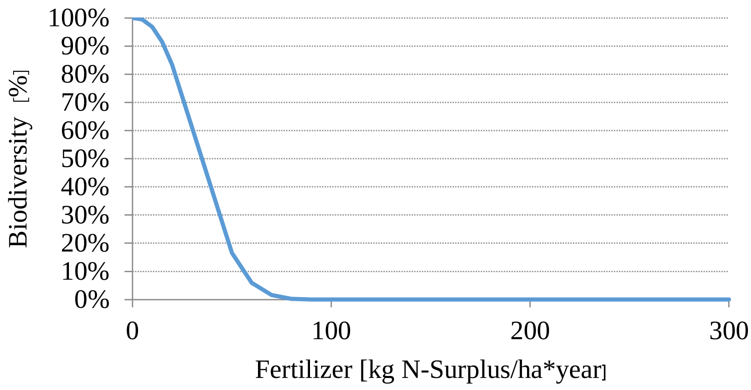


Figure 8 Diagram showing the tolerance for input of N on the studied area.

4.2.2 Case study II

In case study II, two different biodiversity assessment methods in LCA are applied on commercially managed forest in south of Sweden. The method developed by Lindner et al. (2014), which was formerly applied in case study I was selected to be tested also in case study II. The purpose of case study was to compare two methods pertaining their applicability on a regional level. Investigated was the choice of reference situation and different times for when in the production cycle biodiversity assessment is made, choice of biodiversity indicator and feasibility in terms of data availability. A compilation of the results can be seen in Table 6.

Table 6 Compilation of the result of calculated characterization factors for each method tested

Method	CF After felling; ref, spruce	CF After felling; ref, mixed hard wood forest	CF Before felling; ref, spruce	CF Before felling; ref, mixed hard wood forest	CF Whole production cycle; ref, hypothetical description of best quality on biodiversity
Lindner et al. (2014)	-	-	-	-	0.27
de Baan et al. (2013)	0.04	0.29	-0.24	0.08	-

The compilation of the results in Table 6 shows that both methods are applicable for biodiversity assessment on regional level and that the results vary from negative to positive impact on biodiversity. The result generated by the method developed by Lindner et al. (2014), which accounted for the whole production cycle, showed a negative impact. Also the method by de Baan (2013b) generated results indicting negative impact (i.e. positive characterisation factors) except in the case when the assessment was done before felling and old spruce forest was used as a reference. The most negative negative impacts on biodiversity (i.e. the higest postive characterisation factor) was obtained, when the mixed hard wood forest was used as reference and the assessment was done after felling.

For the method developed by de Baan et al. (2013b) not enough data were available to make the assessment for the selected type of forestry in one ecoregion only. Instead, the assessment had to be based on data from two adjacent ecoregions, Scania and Blekinge. The data used was taken provided by SNFI (2010). The method developed by Lindner et al. (2014) relies on the knowledge of expert opinion on the ecoregions in the study. However, finding the right expertise was crucial for the study and so was the development of the biodiversity contribution curves, which was an iterative process between the experts and authors of the study.

All methodological requirements, such as species data and expert opinion proved however to be available and applicable for Scania and Blekinge. Data collection was less time consuming for the species based method than the method based on ecosystem indicators.

4.2.2.1 *Intermediate results generated for the method developed by Lindner et al. (2014)*

With the method developed by Lindner et al. (2014) in focus, the four identified parameters important for biodiversity in commercially managed forests in south of Sweden were soil pH, age of trees (years) and amount of dead wood (m^3/ha) and spruce dominance (%), as shown in Table 7. Table 7 also shows that soil pH, age of trees (years) and amount of dead wood (m^3/ha) contribute more to biodiversity than the parameter of spruce dominance (%). However, soil pH is the parameter that contributes to biodiversity most. Further, all four parameters were given the same weight when an average was calculated.

Table 7 Intermediate calculation steps in Case study II, showing the identified parameters important for biodiversity and for each parameter the following are presented, parameter value, normalized value, biodiversity contribution and weighting value. Based on these values the total biodiversity contribution is calculated according to the method developed by (Lindner et al. 2014) as a weighted arithmetic mean.

Parameter	**Value**	**Normalized value**	**Contribu tion**	**Weight***	**Total contribut ion**
Soil pH	5.5	0.50	99%	0.25	25%
Age of trees (years)	150	0.50	79%	0.25	20%
Amount of dead wood (m^3/ha)	75	0.75	95%	0.25	24%
Norwegian spruce *(Picea abies)* dominance (% biomass)	90	0.90	19%	0.25	5%
Total biodiversity					73%
Age of trees (years)	150	0.50	79%	0.25	20%

*= weighted arithmetic mean

5 Discussion

This section discusses the findings pertaining the results in the literature overview and the application of the methods in case studies.

5.1 Biodiversity indicators

The results generated from the literature overview showed that a dominating part of the LCIA methods use species richness as indicator biodiversity. The earliest methods developed are species based and specifically based on vascular plants. Vascular plants has been used as a proxy for biodiversity for practical reasons, since it is feasible to survey vascular plants and hence data on them are available in in terms of existing data bases. Later developments also include other types of indicators, those of ecosystem indicators and combinations of ecosystem indicators and indicator based on e.g. species richness. In this way biodiversity is captured more holistically, since it is well known that biodiversity made up by far more than one type of species.

Two types of biodiversity indicators were used in the case studies, those based on species and those based on ecosystem indicators. The method developed by Lindner et al. (2014), attempts to capture biodiversity as a whole by identification of prerequisites for biodiversity as captured by ecosystem indicators. No data bases exist for such data, but instead the identification of prerequisites for biodiversity was dependent on expert judgement.

Development potentials for the method developed by Lindner et al. (2014) include the use of two or more experts. The repeatability of the method could be tested if in which the same case study would be tested but by the use of different ecological experts. For instance, it would be useful to test whether a saturation stage can be reached when no additional prerequisites for biodiversity are identified by additional experts, in a way that changes the final CF value. Also amount and intensity of impacts on the same prerequisites as judged by different experts could be tested.

The species based method developed by de Baan et al. (2013b) uses diversity of vascular plants as a proxy for biodiversity. Several additional methods are also based on vascular plants (de Baan et al. 2013b; De Schryver et al. 2010; Müller-Wenk 1998; J. H. Schmidt 2008b; Weidema P Bo and Lindeijer Erwin 2001). However, the use of vascular plants as a single indicator is debatable since the biodiversity of one taxonomic group only partly represent the concept of biodiversity (CBD 1992). There are studies showing that there is not necessarily a clear correlation between species richness in one taxonomic group with that of other taxonomic groups (Prendergast et al. 1997). In order to assess impact on biodiversity more holistically, an understanding of the ecological processes in an ecosystem is required. Later development has suggested ways to get to grips with this problem.
de Souza et al. (2013) has recommended functional diversity to be used recommended to as a compliment to species richness when developing CFs.

5.1.1 Species estimation methods

The literature overview also show that a majority of methods use species estimation methods. However, the metrics of species estimation do not meet important quality criteria such as being unbiased, precise and efficient in terms of focusing on few taxonomic groups and requirement on large number of plots. (Gotelli et al 2011). Neither species variation nor distribution is taken into account. Also, different species estimation methods have different requirements on the input data. They can for example rely in data on rare species, such as unique or duplicate individuals in the plots, or be designed to handle data from equally or differently sized sample plots. Further, the plot size and the numbers of plots monitored on the land occupied are decisive for the spatial resolution of the assessment. The transformation of raw data on species is one source of uncertainty in our case studies. For better understanding of the role method for of species estimation, we suggest future studies in wich different species estimation metrics are applied in the same biodiversity assessment method.

5.2 Temporal resolution

Productions of different crops have different rotation cycles, implying different time scale within the occupational land use phase with varying impact on biodiversity. Examples include e.g. short rotation crops such as wheat, sunflower and soy beans, crops with rotation period of a couple of years such as salix and sugar cane and long rotation crops such as conventional forestry. In the framework developed by Köllner et al. (2013a) it is assumed that the quality on biodiversity remains the same during the whole occupational land use phase. However, that is not true since impact on biodiversity varies over the production cycle, and in particular this becomes evident for long rotation crops such as wood.

In case study II, the assessment of biodiversity was done at different times in the production cycle, which generated different CFs. This is what could be expected since removing the dense canopy during harvest increases sun light, which is favourable for plant biodiversity To receive a CF representing the whole production cycle, I suggest assessing biodiversity at several phases in the production cycle, for calculation of a mean CF value.

5.3 Reference situations

5.3.1 Definitions on reference situation in currently existing LCIA methods on biodiversity

The results of the literature study showed that the reference state is most often described as "natural vegetation", although under different names such as potential natural vegetation, semi-natural vegetation or late succession vegetation. Sometimes the reference state is represented by an actual landscape where biodiversity assessments have been made, but sometimes it's a hypothetical "highest possible biodiversity". A completely different approach, applied in a few methods, is to use the regional average number of species as a reference state.

A majority of methods which suggest the use of regional reference situation rely on data sets that are species average on the Swiss lowlands. All the ready-made LCIA packages such as EcoIndicator99 and Impact 2002+ rely on the data on numbers of species in Switzerland. Few methods, those of Michelsen (2008), de Souza et al. (2013), LIME (2005) use a larger geographical scale as reference situation. The variety of reference situation used indicates that the recommendations from the guideline are unclear.

In the UNEP-SETCs guidelines proposals on three reference situations are given, 1. PNV, 2. quasi natural land cover in e.g. each ecoregion or biome and 3. current mix of land uses. Based on the proposed descriptions in the guidelines it is difficult to interpret which type of reference situation to use. The suggested reference situation which is constituted by PNV is based on the definition made by Chiarucci et al. (2010). However, according to Chiarucci et al. (2010) it is problematic to define PNV and the authors suggest that the concept is abandoned.

The majority of LCIA methods on biodiversity suggest the use of regional reference situations but does not define the reference situation further. Interestingly only one LCIA method, LIME, does not include a reference situation. This method instead relies on the expected extinction time of vascular plants.

5.3.2 Reference situations used in case studies I and II

The two methods used in the case studies used two different types of reference situations; a hypothetical description of the maximum quality on biodiversity in the region based on the opinion of an expert and PNV.

The method developed by de Baan et al. (2013b) proposes the use of semi natural reference situation, which is represented by current late succession habitat stages often used as targets for restoration ecology. In order to apply semi natural reference situation to the case study an interpretation of the definition semi natural in Scania and Blekinge had to be made. However, although it is well known that mature mixed hard wood forests peak biodiversity at an age between 200-300 years, data were not available for such forests, why we had to refrain to younger forests.

The definition on semi natural leaves great room for interpretations as it could be represented by different types of forests and vary between regions. The dominating land use in Scania and Blekinge is intensive agriculture and limited amount of forests of all types exists. In particular this is true for old forests. The search for forests which are targets for restoration ecology resulted in a selection of forests, which were not representative as references situations, according to experts. Reasons for this were that this type of forest may be carefully managed.

The chosen reference situation included when applying the method developed by Lindner et al. (2014) is a hypothetical maximum quality biodiversity in the ecoregion according to expert knowledge and in agreement with existing policy documents. The description resulted in a reference situation constituted by a regional landscape with best quality on biodiversity in accordance with national strategy documents. The description made by the expert of such landscape was very difficult to express and resulted in a very diffuse answer.

An analysis of the biodiversity contribution expressed by the expert shows that the parameter values with the highest quality on biodiversity combined, peaking at 100%, represent the highest quality on biodiversity and could presumably represent an optimal reference situation. However, these values combined do not account for land use competition e.g. food production or infrastructure, which may reduce biodiversity. Neither is management taken into account, which has the possibilities to improve the quality of biodiversity.

This hypothetical reference situation used in this study is close to that of PNV, which is also difficult to define due to the fact that ecosystems are dynamic but PNV is not. Additional difficulties to define PNV are related to major drivers that have impact on ecosystems e.g. large mammals, forest management, wild and cultural fires, soil and invasive species. The impacts on ecosystems caused by major drivers are difficult to predict, which makes it a challenge to define PNV, a future stable ecosystem, based on current mature vegetation Chiarucci et al. (2010). Because the hypothetical reference situation used in this study is very similar to PNV, this could be an explanation why the description was very diffuse and extremely difficult to predict for the expert.

5.4 Data availability

5.4.1 Expert opinion

Data were available to apply both methods on a regional level. However, the feasibility in terms of applying them to the case studies differed. Data sets on different formats were adjusted for implementation in the selected methods. The use of the method developed by Lindner et al. (2014) relied on ecosystem indicators identified by judgement from an expert. Contact was taken with the University of Alnarp in order to find the right expertise. Finding the right expert, who has regional specific knowledge, was crucial for further assessment of the method. The interview methodology according to Lindner et al. (2014) constitutes of four questions. It was clear that management play an important role for biodiversity in spruce forest. Since management proved to be difficult to be expressed as a biodiversity contribution curve, the closest description was taken as the percentage of alien tree species, which in this case was spruce. Another challenge was the description of the reference situation which was extremely difficult for the expert to answer. The use of the method included a very iterative process between the method developer, the ecological expert and the author of the study in order generate the following biodiversity contribution curves, leaving room for bias and coincidence. Since the interview questions allow for interpretations a second opinion was included in case study II to increase the validity and reliability of the results.

5.4.2 Species richness

Data sets with suitable format were available for the species based methods developed by de Baan et al. (2013b). However, data set on vascular plants in Scania was not sufficiently large to be used in the species estimation method. The number of inventoried plots in the chosen type of forest with the chosen stand age was not enough. Because of this the geographic scale had to be extended to include data sets from Blekinge. The choice of including data sets from Blekinge led to the inclusion of two ecoregions instead of one. An option could have been to include data sets from spruce forest from other countries, but within the same ecoregion.

A combination of standardized biodiversity survey methodology and coordinated databases across ecoregions could be one way to facilitate the development of comparable LCIA results on biodiversity. Many biodiversity surveys are based on different sampling methodology. Also, coordination between databases with information on biodiversity within and across the borders of nations is one way to generate data with biological relevance. For Europe the European Environment Agency (EEA) is assigned to coordinate independent information on the environment and in which standardized information on biodiversity could be compiled.

6 Conclusions

This thesis presents a literature overview on LCIA methods on biodiversity and two case studies. Selected methods were applied to forestry in Southern Sweden and the methods represents two different methodological approaches for assessment of biodiversity in LCA. The discussion in this thesis has highlighted a number of issues that have led to the following conclusions.

6.1 Biodiversity indicators

- Quantification of biodiversity using vascular plants as a proxy indicator limites the perspective on biodiversity.
- The use of ecosystem indicators allows for a holistic approach on biodiversity but includes large uncertainties as the description on ecosystem indicators are dependent on individual expert judgments. However, the exstent of the uncertainties is difficult to tell.
- There is a potential for development of combinations of species based indicators, including fuctional diversity with ecosystem indicators.
- Methods including expert judgement could be improved by testing the robustness of different judgments.

6.2 Temporal resolution

- The CFs obtained varied dependent on the period of time when the data was collected.
- It is evident that the assumption made in the UNEP-SETAC framework, that of the quality of biodiversity is constant during the occupational land use phase does not hold.
- I suggest development of characterisation factors that are valid for whole production cycles for long rotation crops such as wood.

6.3 Reference situations

- The definitions on reference situations needs to be clearer in order to increase the feasibility to operationalize the LCIA methods on biodiversity.
- The definition on PNV is problematic to define as proposed by Chiarucci et al. (2010).
- With PNV in focus, the case studies showed that it was problematic to describe future types of forests in Scania and Blekinge.
- The data availability was limited pertaining old forests in Scania and Blekinge.
- The hypothetic reference situation used in the method developed by Linder et al. (2014) was very difficult for the experts interviewed to describe.

6.4 Data availability

- Data was available for applying the selected methods on a regional level. However, the data sets had to be extended to include two ecoregions in order to receive sufficient amount of data.

- It was difficult to find a reference situation providing data sets with suitable data format.
- The data availibility for the method based on ecosystem indicators is dependent on available experts willing to express a judgement.

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