



Norwegian University of
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Implementation of Land Use and Land Use Change and its Effects on Biodiversity in Life Cycle Assessment

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Problem Description

Over the last decades, Life Cycle Assessment (LCA) has been developed as a prominent tool to assess the environmental impacts of products and services. There are, however, large differences in how well-developed different impact categories are and to what extent there exists an agreed upon methodology for assessments. Land use and land use change (LULUC) is an impact category that still stands out as immature and is rarely included in LCAs.

Land use impact could be of particular importance when assessing products originating from land extensive activities, e.g., forestry and agricultural activities for food, biomass and bioenergy production.

LULUC has a wide range of possible impacts: climate change, eutrophication, acidification, toxicity, and land competition, to mention a few. Here, the focus is on the impact on biodiversity. Loss of biodiversity as a consequence of land use and land use change is by many regarded as one of the largest environmental problems and it is advocated that this should be included in LCA as well.

The objective of this thesis is to perform a literature survey of existing methods for the implementation of LULUC in LCA, with a particular focus on the effects on biodiversity. The thesis should discuss the different methodologies chosen to assess biodiversity and how this is related to a functional unit. The thesis should discuss the usefulness and applicability of the suggested methods, and discuss the effects on the overall assessment that LULUC are included in the identified studies and give recommendations on a direction for further integration of LULUC and its effects on biodiversity in LCA.

The following questions should be considered in the project work:

1. How is LULUC treated in LCA literature?
2. How is biodiversity assessed in the identified studies?
3. How is impact on biodiversity and LULUC related to the functional unit in the identified studies?
4. To what degree does the inclusion of LULUC affect general conclusions and to what degree are the applied methodologies based on scientific sound assumptions?
5. Is it possible to identify any trends? Are the number of studies increasing or decreasing? Are the methodologies showing any signals of convergence etc.?

Assignment given: 01. February 2010

Supervisor: Anders Hammer Strømman, EPT

Foreword

It has long been an axiom of mine that the little things are infinitely the most important - Arthur Conan Doyle, Sr.

This thesis serves to fulfil the requirements for the award of Master of Science degree in Industrial Ecology (Environmental Systems Analysis option) at the Norwegian University of Science and Technology (NTNU), Trondheim. This was done within the Industrial Ecology Program of the Department of Energy and Process Engineering.

My sincere appreciation goes to my academic supervisor, Prof. Anders Hammer Strømman, and my research advisor, Dr Ottar Michelsen, for their help and guidance during the course of writing this thesis. Also, I want to thank all the staff and my fellow students at the Industrial Ecology program for the time shared together.

Finally, I appreciate my family for their love and care particularly my darling wife and my lovely daughter for their understanding and support throughout the course of my academic pursuits.

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Summary

Land use refers to the use of land for intensive human activities aiming at exclusive use of land for certain purposes and adapting the properties of land areas in view of these purposes. Environmental problems are, however, generated as a result of these human activities which modify the shape and properties of large land areas according to the requirements of human activities and thereby excluding wild animals and plants from coexisting on such land areas and in their neighbourhoods. Land use also leads to the degradation of the natural environment.

Life Cycle Assessment (LCA) methodology is used for evaluating the environmental burdens associated with products or processes while taking their whole life cycle into consideration. LCA is a comprehensive assessment method which considers all aspects of natural environment, human health, and resources. Land use is regarded as an impact category in Life Cycle Assessment and is treated as such. However, the environmental impacts associated with land use and land use change are not being adequately considered in LCA, if considered at all.

Life Cycle Impact Assessment is a part of LCA and is aimed at understanding and evaluating the magnitude and significance of the potential environmental impacts of products or processes and this involves developing characterization factors which link an environmental impact to a category indicator. In the assessment of land use impacts, characterization factors are developed so as to weigh the magnitude of environmental interventions such as land occupation and land transformation on the potentially affected attributes of ecosystem quality such as biodiversity, ecological functions and natural resources.

The goal of this study is to review the progress of the implementation of land use and land use change as an impact category in LCA with a particular focus on biodiversity, recognize limitations, and indicate future prospects for the development of land use impact assessment methodologies and subsequent integration into LCA. Land use impacts are not being widely integrated into LCA because they are dependent on the regional or local situation which is not well known in LCA and land use as an environmental intervention is very complex. However, the importance of land use cannot be overemphasized when assessing products or processes which make use of raw materials that originate from land extensive activities. Despite this importance, there have been diverse arguments on how to include land use impacts, for example,

on biodiversity in LCA so as to provide a common and acceptable methodology for this assessment.

This study focuses on how land use impacts can be included in LCA. With a particular focus on land use impacts on biodiversity, the result of this review shows that only a few studies have been carried out. The problem of non-convergence of the methodology for the assessment of land use in LCA still persists because most of the proposed methodologies deal with different aspects of land use impacts and are therefore conflicting.

Most of the studies reviewed stress the importance of biodiversity measured in terms of vascular plant species diversity. However, there are other methodologies which consider other impact pathways such as life support functions. The number of studies thereby correlates with an increase in the interest in the research area. However, it is difficult to identify any trend of convergence. Different methods are being proposed which do not actually agree with one another. Some of these methods are not “closely” related to the use of land in the normal usage sense. Most of the methods being proposed are exemplified in different regions and these have not been found to be applicable to global cases. This could be a limiting factor for the applicability of the proposed methodologies in LCA. In order to overcome these shortcomings, more research work would be needed before these methodologies could be incorporated into LCA which is presumed to be a global assessment methodology. This will enhance the credibility of the results provided by an LCA and the subsequent acceptability of the LCA methodology.

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1 Introduction

Natural environment can be considered to include both the functional values, represented by the life support functions, and intrinsic values, represented by biodiversity and natural landscapes, of nonhuman life and its environment (Lindeijer, et al., 2002). All aspects of the natural environment are to be considered in environmental life cycle assessment tool used for the analysis of impacts associated with a product or a product system (ISO, 2006a) having interaction with the nature (Wagendorp, et al., 2006) and thereby causing its subsequent degradation (Lindeijer, et al., 2002).

Land is considered as part of natural environment used for the purpose of meeting human requirements such as food and shelter and this has greatly been impacted upon, thereby leading to a reduction in biodiversity, change in the natural landscapes, and disturbances in natural systems (Lindeijer, et al., 2002). Dumanski and Pieri (2000) define land *'as an area of the earth's surface, including all elements of the physical and biological environment that influence land use.'* Land resource does not only refer to soil but to the combined resources of terrain, water, soil and biotic resources that provide the basis for land use while the requirements of land use which include agricultural production, forestry, conservation, and environmental management are usually considered when defining land quality which refers to the condition of land relative to the requirements (Dumanski & Pieri, 2000).

Land use change is expected to have the greatest impacts on biological diversity by year 2100 and it is also expected to have major importance in the tropics (Chapin III, et al., 2000; Sala, et al., 2000). Physical changes in land use resulting from agricultural activities, installation of dams, urbanization and biological resource extraction practices are the main causes of the alarming declines in biological systems worldwide (Schenck, 2001). Land use change may give new species the possibility to establish themselves to the detriment of existing species in an area; examples include transmission corridors, impoundments, peat mining and depositories (Kyläkorpi, et al., 2005). Impact from, for example agriculture, forestry, forest bio-fuel outtake and cooling-water releases, occurs when a continuous land use has negative effects on the existing species while removal results when a particular area is converted to another kind which eliminates the existing species; examples include buildings, roads, foundations and draw-down zone (Kyläkorpi, et al., 2005).

Life cycle assessment (LCA) is an '*internationally standardized environmental assessment method*' (Kloepffer, 2008) used for the assessment of the entire life cycle of a product or a system; that is, beginning from resource extraction and up to the final stage of waste disposal (Rebitzer, et al., 2004). LCA is often regarded as a comprehensive methodology for determining the environmental profile of a product or a production system (ISO, 2006b).

A particular phase of an LCA that has received a great attention is the life cycle impact assessment (Milà i Canals, et al., 2007b; Reap, et al., 2008). Goal and scope description, life cycle inventory, life cycle impact assessment, and interpretation are the main phases considered in the LCA methodology (Rebitzer, et al., 2004). The LCIA phase is regarded as the most challenging of all other three phases because it involves the translation of burdens (emissions or stressors) into environmental impacts (Reap, et al., 2008). An element of an LCIA involves the selection of impact categories (ISO, 2000). Land use is regarded as an impact category in LCA because production or economic systems often require the use of land and this use would possibly have negative consequences on the natural ecosystems (Koellner & Scholz, 2007; Udo de Haes, et al., 1999). The general approach of LCA studies is usually spatially and temporally independent of the environmental impacts derived from a product or production system (ISO, 2006a, 2006b). It is therefore necessary to adjust the LCA methodology to take land use impact assessment into consideration because production systems that interact with the nature, for example agricultural systems, are closely related to local and temporal aspects (Nunez, et al., 2010; Schmidt, 2008b). The introduction of new elementary flows that appropriately reflect and quantify land use and development of appropriate characterization models in order to calculate indicator results from the new elementary flows are normally the two main additions to the conventional LCA methodology in order to assess land use impacts (Geyer, et al., 2010a). Natural environment and the natural resource (that is, biotic production potential) are usually regarded as the most important impact pathways requiring adequate protection in LCIA (Milà i Canals, et al., 2007).

The intrinsic value of biodiversity is regarded as an important aspect of natural environment (Mila i Canals, et al., 2007). The major reason for the decreasing diversity of habitats and wildlife species is the conversion, fragmentation, or degradation of natural and semi-natural ecosystems such as forests, grasslands, and wetlands for human purposes (UNEP, 2000 cited in

Brentrup et al., 2002). Soil degradation and erosion, shifts in ground water availability, loss of biodiversity, and enrichment of environment with toxic chemicals are some of the irreversible consequences of economic activities on the natural ecosystems (Koellner & Scholz, 2007). Ecosystem services that benefit humans may be affected by altered ecosystem processes which result from changes in biodiversity (Chapin III, et al., 2000; Sala, et al., 2000). According to Kylakorpi et al. (2005), change, impact, and removal of ecosystems are the three ecological consequences associated with land use. Human-well being, as shown in Figure 1, is central to the various benefits provided by land and ecosystems. In order to maintain these benefits, ecosystems services must be protected and the protection of biodiversity and life-support functions is central to this (Milà i Canals, et al., 2007b).

Despite the importance of the impact of land use activities on biodiversity, there has been no standard and acceptable methodology for its inclusion in life cycle assessment (Mila i Canals, et al., 2007) and no agreed-upon parameters to consider which could be due to the lack of available data (Nunez, et al., 2010).

1.1 Motivations and Objectives

It is important to include the environmental impacts resulting from land use and land use change in the life cycle assessment of product or production systems (Köllner, 2000; Lindeijer, 2000b; Weidema & Lindeijer, 2001). The assessment of land use impact is important in the life cycle assessment (LCA) studies of products with a major part of their life cycle in biological production, for example agriculture and forestry (Wagendorp, et al., 2006). This necessitates the importance of land use assessment when considering renewable energy options like first and second generation biofuels (Fthenakis & Kim, 2009; Phalan, 2009). In Europe, impact on biodiversity resulting from the cultivation of rapeseed for the purpose of biodiesel for example, has generated a lot of controversy (cf. Schmidt, 2008). There is need to maintain biological systems in a reduced entropy state, which is with less chaos, and this is usually achieved by a high dissipation of energy gradient induced by the sun (Wagendorp, et al., 2006). According to Wagendorp et al. (2006), life and its various forms of diversity have this ability and this must be maintained.

This study focuses on how land use impacts can be included in LCA. The objective of this study is to review the existing literature on land use and compare the methodological approaches used

for assessing land use impacts in life cycle impact assessment with a primary focus on the effects on biodiversity. In this study, future needs and development of a methodology for land use impact assessment are considered. Improvements and adjustments of the LCA methodology to incorporate the effects of land use are particularly discussed. The specific focus on the effects of land use on biodiversity is motivated by the fact that biodiversity plays an important role in determining the state of the environment (Schmidt, 2008a) and these effects are not often dealt with in LCA because of the lack of consensus on how land use should be incorporated into the LCIA phase of the methodology (Lindeijer, et al., 2002; Mila i Canals, et al., 2007). Another motivating factor for this is that land use impact on biodiversity is substantial (Mila i Canals, et al., 2007) and the general acknowledgement of the need to maintain and protect biodiversity (UNEP, 1992) when the extraction of raw materials is considered in activities such as mining, agriculture, and forestry (Michelsen, 2008; Mila i Canals, et al., 2007).

This review covers the most recent publications in the field and has a broader scope. The review is done with a particular reference to the framework proposed within the UNEP/SETAC life cycle initiative on LCIA and land use (Mila i Canals, et al., 2007).

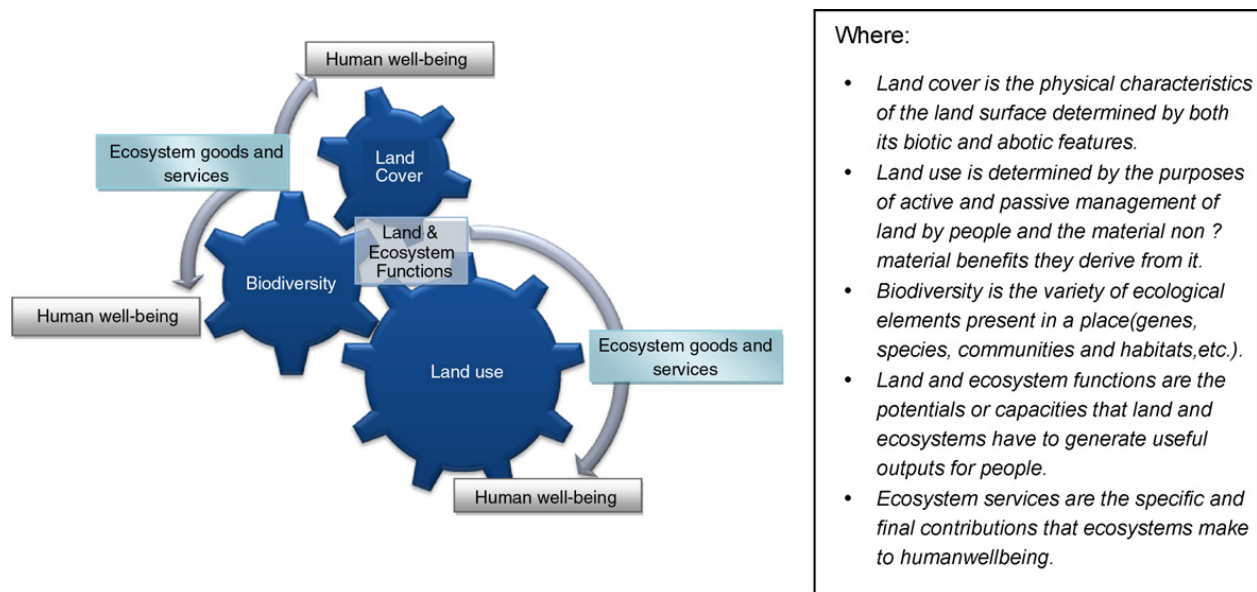


Figure 1: The Relationship between Land Use and Biodiversity (Haines-Young, 2009)

Figure 1 shows the important links between biodiversity, ecological processes and human well-being. The figure shows how human well-being depends so much on the ecosystem goods and services provided by ecosystems and their relations to land and biodiversity.

1.2 Structure of Thesis

This thesis is divided into seven sections:

1. Introduction: This chapter introduces the research problem and its importance. In this section, the motivation and the structure of the thesis are also discussed.
2. Background: This section introduces the LCA methodology framework, land use and land use change concept, biodiversity, the effects of land use impacts on biodiversity and the needs to conserve biodiversity.
3. Methodology: This section discusses the methods employed for the review. The goal and scope of the review are also highlighted.
4. Results: The results of the review are given in tabular forms. These are further discussed to give an insight into the main aspects of the literature.
5. Discussion: This gives an in-depth discussion of the key findings and the limitations of each methodology reviewed are recognized.
6. Conclusion: This chapter brings the review into conclusion by summarizing the objective of the study and the results. This chapter also discusses if the goals of the review have been achieved or not.
7. Recommendations and perspectives: In this chapter, a summary of the work done, an evaluation of the study and some possible areas for future work are presented.

2 Background

2.1 Life Cycle Assessment Methodology

Life cycle assessment can be defined as a tool which provides a useful framework for estimating and assessing the environmental impacts attributable to the life cycle of a product (Rebitzer, et al., 2004). The potential contributions to a wide range of global scale environmental concerns that result from various production systems are identified within this framework (ISO, 1997). The physical inputs, production materials, energy requirements along with the resulting emissions (to air, land, fresh water and oceans) associated with each stage of each production chain (from extraction of materials through production, transport, use and disposal) are assessed (ISO, 1997).

Four methodological components are usually considered within the LCA framework. These are: goal and scope definition, life cycle inventory analysis, life cycle impact assessment, and life cycle interpretation (ISO, 1997). These components and some applications of LCA are shown in Figure 2 below.

The goal and scope definition of an LCA provides a description of the product system in terms of the system boundaries and a functional unit (Rebitzer, et al., 2004). The aspects usually considered in this part include *allocation procedures, the system boundaries, the function of the product system, the functional units, data requirements, type of impact assessment methodology and interpretation to be performed, assumptions and limitations, data quality requirements, type of critical review, if any, and type and format of the report required for the study* (Dantes, 2010). The impact categories to be considered are also specified in line with the goal of the study. The functional unit is the important basis that enables alternative goods, or services, to be compared and analysed (Hertwich, et al., 2002). The choices and assumptions made during system modeling, especially with respect to the system boundaries and what processes to include within these boundaries, are often decisive for the result of an LCA study (Rebitzer, et al., 2004).

The life cycle inventory (LCI) phase involves the compilation, tabulation, and preliminary analysis of all environmental exchanges, for example emissions and resource consumptions (Hertwich, et al., 2002). The aim of the LCI is to calculate the quantities of different resources required and emissions and waste generated per functional unit (Rebitzer, et al., 2004).

The life cycle impact assessment (LCIA) phase involves the calculation as well as the interpretation of the indicators of the potential impacts associated with the exchanges, determined in the LCI phase, with the natural environment (Rebitzer, et al., 2004). The determination of the total environmental impacts caused by environmental interventions resulting from human activities is the focus of LCIA (Mila i Canals, et al., 2007). The result of the LCIA is an evaluation of a product life cycle, on a functional unit basis, in terms of several impacts categories which include climate change, stratospheric ozone depletion, tropospheric ozone creation, eutrophication, acidification, toxicological stress on human health and ecosystems, resources depletion, water use, land use, and noise depending on the goal and scope of the assessment (Rebitzer, et al., 2004).

Life cycle interpretation occurs at every stage in an LCA and the results of the LCI and LCIA stages are used to reach some conclusions (Hertwich, et al., 2002). Based on these conclusions, recommendations are made based on the scope and goal of the LCA study (Hertwich, et al., 2002). An interpretation purely based on the LCI could be conclusive if in the case of comparison of two product alternatives and one alternative shows higher consumption of each material and of each resource (Rebitzer, et al., 2004).

Based on the way a product system should be modeled, two very distinct categories of LCA goals exist (Rebitzer, et al., 2004; Weidema, 1993):

- Attributional (retrospective) LCA which describes a product system and its environmental exchanges, and;
- Consequential (prospective) LCA which describes how the environmental exchanges of the system can be expected to change as a result of actions taken in the system.

The purpose of prospective LCA is for decision support while that of retrospective LCA is for detailed assessment of an existing product or product system (Lemming, et al., 2010).

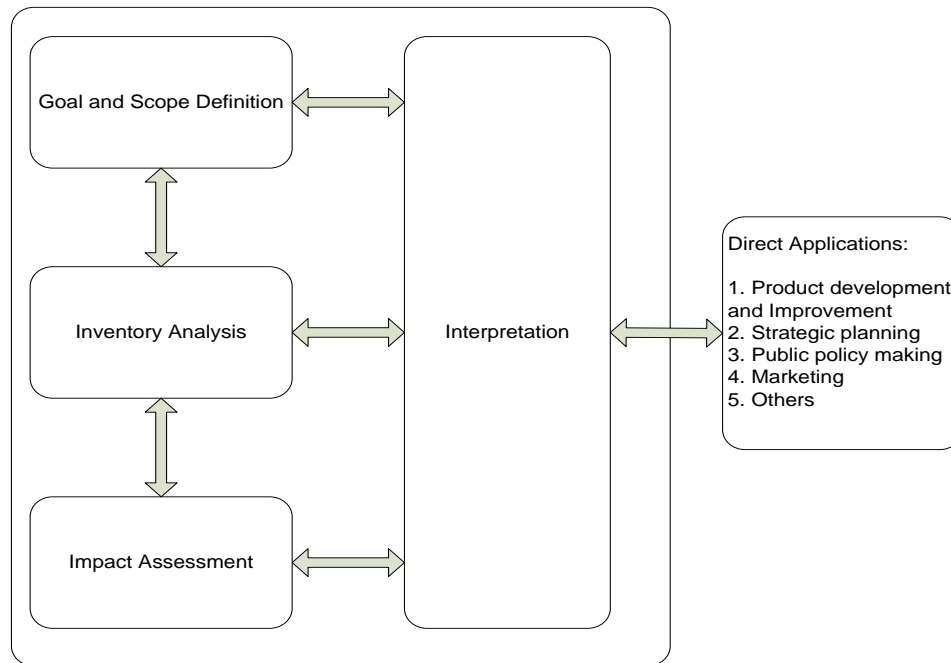


Figure 2: An Overview of LCA Framework and Applications (based on ISO 14040, 1997)

2.2 Land Use and Land Use Change

The use of land for human activities normally induces disturbances and these generally affect the natural environment and its resources (Milà i Canals, et al., 2007b). These human activities transform or maintain the environmental state of the land cover and the economic process of land-use are classified into abiotic resource extraction (mining), biotic resource production (agriculture and forestry) and surface use (housing, industrial plants, leisure parks and traffic infrastructure, and also water reservoirs) (Köllner, 2000). Extraction of abiotic resources or activities leading to the degradation of the ecosystems are example of human activities that often lead to a decrease in the exergy level of natural systems (Wagendorp, et al., 2006).

It is generally acknowledged that land use impact assessment should be included in life cycle assessment (Nunez, et al., 2010). In addition to the area of land occupied by human activities, land use quality is also to be assessed (Mattsson, et al., 2000; Mila i Canals, et al., 2007). Land use impacts should be included in LCIA because ecologically fragile areas are often used to produce raw materials for industrial and economic activities (Mila i Canals, et al., 2007).

In the life cycle assessment of land use, three important impact pathways- the impact on biodiversity, the impact on biotic production, and the impact on the regulating functions of the natural environment-must be included (Mila i Canals, et al., 2007).

Wagendorp, et al. (2006) define land use as a “*human-induced disturbance influencing the exergy level and exergy dissipation rate of an ecosystem which often lead to a temporary or permanent decrease of ecosystem exergy level, indicated by a decrease of biomass and/or canopy cover, simplification, loss of species and a subsequent loss of ecosystem functionality indicated by, for example, biotic deterioration.*” Land use is defined as an environmental intervention which is usually measured as the dimension of land area occupied multiplied by the duration of use, if occupation of the area for a certain purpose is to be expressed. However, transformation of land is measured as the dimension of the area only (Lindeijer, et al., 2002). Land use change is regarded as one of the most important causes of alteration in biodiversity and this can be regulated by changes in policy (Chapin III, et al., 2000). If land use change is defined as a physical entity, then land use types used as entries in the LCI phase should only contain physical activities while chemical emissions should be treated separately (Lindeijer, et al., 2002).

2.2.1 Aspects of Land Use

The physical impacts of land use can be described in two terms. As proposed by the Society for Environmental Toxicology and Chemistry (SETAC) Working Group on Impact Assessment, these are (Lindeijer, 2000a):

- Land transformation
- Land Occupation

According to Köllner (2000), another phase does exist and this is with respect to the temporal phase of land use activities. This is known as *land abandonment* and it is a relevant phase for land use activities with a defined end of use phase (for example, mining) (Köllner, 2000).

Land use change and transformations are regarded as the key drivers of biodiversity which serves as a constraint to how a typical land area may be used (Haines-Young, 2009). Land use impact assessment methods need to account for both land transformation and land occupation (Mila i Canals, et al., 2007). Land use impacts due to conversion or transformation are usually considered more relevant than those due to land occupation (Sala, et al., 2000). Environmental

impacts of land use should be taken into account in LCA because of its effects on the natural ecosystems which are regarded as a safeguard subjects in LCA (Brentrup, et al., 2002). Different types of land use lead to different impacts on the environments (Mueller-Wenk, 1998) and these need to be incorporated in the procedure for determining impacts of land use. Impact assessment of land use is usually related to ecologically homogenous land units, for example, the impact of land use in the Atlantic region of Europe is different from that in the boreal region (Brentrup, et al., 2002).

Occupation impacts have units of a quality factor multiplied by area multiplied by time while transformation impacts have units of a quality factor multiplied by area and are usually represented by the difference between the quality before transformation of a certain area and the quality after renaturalisation of the area has taken place (Schmidt, 2008a). In a simple form, impacts from land use can be represented as follows (Lindeijer, 2000b):

- Land occupation impacts = area of land A * duration of use t * Quality Q
- Land transformation impacts = area of land A * Quality Q

Typical frameworks for transformation and occupation impacts are shown in Figure 3 and Figure 4 respectively. Occupation and transformation impacts are the main focus of several identified methods for land use impact assessment (Schmidt, 2008a). While occupation of a certain area of land results in the reduction in environmental quality normally represented as a delay or postponement of renaturalisation processes, land transformation leads to reduction in environmental quality when an initial land cover is changed into a new type prior to its occupation (Schmidt, 2008a). Another form of impacts which are often considered in the assessment of land use in LCIA is permanent impact and this is defined as '*the permanent loss in quality due to momentary conversion of land use by land transformation or gradual degradation caused by land occupation*' (Schmidt, 2008a) which are assessed qualitatively (Mila i Canals, et al., 2007) or estimated using some uncertain data (Weidema & Lindeijer, 2001). As shown in Figure 3, it is possible for restoration, which could be natural or induced by humans, not to return a particularly transformed and occupied piece of land back to its natural state. It is possible though for a different land cover type to result with the same final quality as the initial untransformed one after a very long restoration time (Koellner, 2001). A permanent change of land cover results when permanent changes in quality occur as a result of the new steady state

reached within natural relaxation, after a severe case of transformation processes, not being equivalent to the reference within the assessment time frame (Mila i Canals, et al., 2007).

In the assessment of already degraded land, Schmidt (2008) argues that the transformation impact experienced when transforming and occupying the land is smaller compared to that of “fresh” land and that the occupation impact remains the same in the two scenarios.

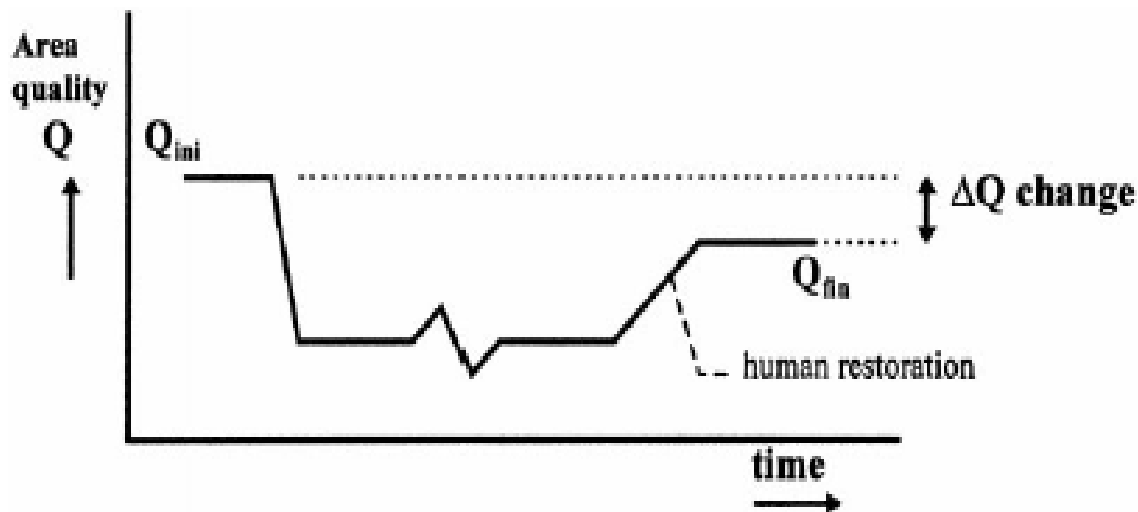


Figure 3: The land use change aspect of land use (Lindeijer, 2000a)

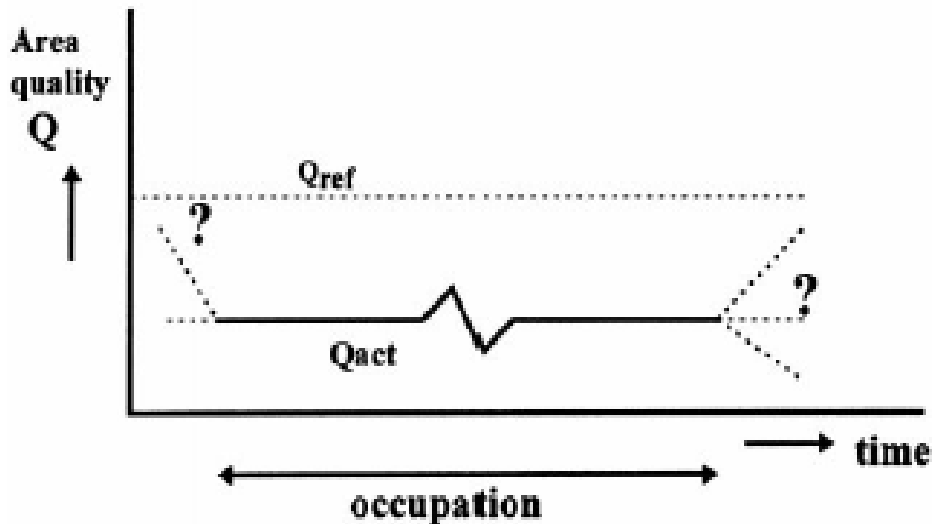


Figure 4: The land occupation aspect of land use (Lindeijer, 2000a)

Chapter 2

In Figures 3 and 4 above (Lindeijer, 2000a):

Q_{ini} = initial quality;

Q_{fin} = final quality;

Q_{act} = actual quality during occupation;

Q_{ref} = reference quality

Depending on the impact pathway considered (for example, biodiversity, biotic production potential or ecological soil quality), the area quality (or land quality) as shown in the figures above may be represented by different parameters measured in different units (Mila i Canals, et al., 2007).

In the inventory phase of LCA methodology, the entries used for land use interventions are occupation and transformation interventions while in the LCIA stage, the results obtained in the LCI stage are characterized and are referred to as occupation and transformation impacts (Schmidt, 2008a). In the easiest form, the intensity of the land use or the original quality of the land is not considered (Wagendorp, et al., 2006). In this form, land use impact is expressed in m^2yr occupied land per functional unit of product, in which case only the area of occupied land and the duration of occupation are considered (Wagendorp, et al., 2006). It is necessary, however, to include the reduction of land quality in the assessment of land use impact because environmental impact is associated with land use and not with its reduced availability (Wagendorp, et al., 2006). A permanent change of land cover results when permanent changes in quality occur as a result of the new steady state reached within natural relaxation, after a severe case of transformation processes, not being equivalent to the reference within the assessment time frame (Mila i Canals, et al., 2007). Allocation of impacts is very key in the assessment of land use change; whether to the first use or to all other following uses (Wagendorp, et al., 2006). In the determination of the biodiversity impact of any land use types, the absolute value calculated from the specific indicator value does not count but the difference from some reference situation (Lindeijer, et al., 2002). Depending on the scope of the LCA study of a system, the reference land use used to determine the size of impact compared to the studied system, may be the non-use of the same piece of land, that is, natural relaxation (referred to as dynamic reference situation in Mila i Canals, et al., (2007)) as in the case of an attributional LCA or an alternative land uses in the case of a consequential LCA (Mila i Canals, et al., 2007).

2.2.2 Framework for Land Use Impacts in Life Cycle Assessment

Biodiversity (existence value), biotic production potential (including soil fertility and use value of biodiversity) and ecological soil quality (including life support functions of soil other than biotic production potential) are the main damages recommended by Mila i Canals and colleagues to be considered in the assessment of land use in LCIA (Mila i Canals, et al., 2007). These represent the impacts that are traditionally not covered by LCIA impact categories. Some of the existing land use impacts assessment methods in LCA so far proposed have two shortcomings, among which are non-differentiation between different land use types and lack of global spatial coverage (Schmidt, 2008a). These shortcomings may be attributed to the nature of LCA methodology which uses non-geospatial information (Geyer, et al., 2010a) and does not conventionally differentiate between different land use types.

These differences must be taken care of and this necessitates the need for a common framework. According to Mila i Canals et al. (2007), the key elements to be considered in the life cycle impact assessment of land use include:

1. Reference for occupation impacts;
2. The impact pathways to be considered;
3. The units of measure in the impact mechanism;
4. Future impacts consideration; and
5. Biogeographical differentiation (Land use interventions may have different consequences on the environment depending on the sensitivity and inherent land quality of the environment).

The framework for land use impact assessment normally has three coordinates, where the y-axis represents the evolution of land quality which depends on the impact pathways being considered; the x-axis represents the time frame of the assessment; and the z-axis denotes the area of the used land (Mila i Canals, et al., 2007). It is possible for the impacts caused by a studied system to be or not to be fully reversed during the time frame of the assessment, depending on the impact pathway being considered (Mila i Canals, et al., 2007). It is necessary to consider the location of where land use occurs in order to assess the impacts on biodiversity properly because species assemblages are dynamic in time and are determined by past land use changes (Geyer, et al.,

2010a). This is important to be considered because impacts due to land use are proportional to the area of used land (Mila i Canals, et al., 2007).

2.3 Biodiversity

Biological diversity is simply referred to as biodiversity (Martens, et al., 2003) and can be defined as ‘ *the variability among living organisms from all sources including, inter alia, terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are part; this includes diversity within species, between species and of ecosystems*’ (UNEP, 1992).

The protection of biodiversity is of international and national concern because of its intrinsic, non-tangible values (for example, value of threatened species) and its instrumental value for human society (e.g. source of pharmaceutical and agricultural products, sustaining of ecosystem functions, recreational use) (Köllner, 2000). Loss of biodiversity does not relate to the “killing” of various species only but the dearth of the resources upon which they depend as a result of human land use activities (Michelsen, 2008).

Biodiversity is usually considered to be subdivided into a genetic, species and ecological diversity (Köllner, 2000; Schmidt, 2008a). The global biological diversity consists of diversity at all organizational levels. This ranges from genetic diversity within populations to the diversity of ecosystems in landscapes. Species diversity normally receives much focus because of the good understanding and availability of data on causes, patterns and consequences of changes in diversity at the species level (Chapin III, et al., 2000).

On the species level, species richness and diversity are distinguished. Species richness indicates the number of species per area or volume while diversity considers the distribution of individuals between species (that is, the number of individuals per species (Köllner, 2000).

Functional consequences of species diversity include the mediation of energy and material fluxes; the alteration of abiotic conditions that regulate process rate; and the influence of the resilience and resistance of ecosystems to environmental change (Chapin III, et al., 2000).

Biodiversity relates to three main functions and these include (Clergue, et al., 2009):

- patrimonial functions which concerns conservation of the aesthetic values of landscape and threatened species;
- agronomical functions which relate to agricultural activities and the production of cultivated ecosystems; and

- ecological functions which define the existence (use value) of habitats and their particular species.

Quality and quantity of biodiversity are two important aspects usually considered while dealing with biodiversity issues (Martens, et al., 2003). The quantity is '*expressed in terms of the size of the population, the abundance of different species, as well as the surface area and number of ecosystems in an area*' whereas quality (or integrity) relates to '*the genetic diversity and the vitality or resilience of a species, ecosystem or natural area*' (Martens, et al., 2003).

The number and kinds of species present determine the species traits that influence ecosystem processes. The number of species present (species richness), their relative abundances (species evenness), the particular species present (species composition), the interactions among species (non-additive effects), and the temporal and spatial variation in these properties are major components of species diversity that determine the expression of species traits (Chapin III, et al., 2000).

2.3.1 Reasons for Protecting Biodiversity

The loss of biodiversity is regarded as a central global problem (Eppink, et al., 2004; Loreau, et al., 2001; Martens, et al., 2003). Habitat destruction and alteration of abiotic conditions in natural areas are regarded as important causes of biodiversity losses (Eppink, et al., 2004). It could then be inferred that biodiversity can better be protected by protecting habitats and conserving abiotic conditions. For over three decades, there have been attempts and several considerations on the need to conserve and protect the environment and its natural inhabitants. These attempts have resulted in several conventions and policy instruments among which include Ramsar Convention (1971), Bern Convention (1979), Bonn Convention (1979), European Commission (EC) birds directive (1979), EC habitats directive (1992), and Convention on Biological Diversity (1992). The most widely accepted of these is the Convention on Biological Diversity because it is holistic in scope and the objectives expressly consider sustainability and these are easy to understand. These objectives as specified in Article 1 are of threefold and include (UNEP, 1992):

- the conservation of biological diversity;
- the sustainable use of its components;

- the fair and equitable sharing of the benefits arising out of the utilisation of genetic resources.

The degradation of biodiversity occurs not only as a result of the changes in the relevant characteristics of land from different land use types but also as a result of the maintenance of the characteristics of the land at a low-quality level by land use types (Lindeijer, et al., 2002).

The cultural, intellectual, aesthetic and spiritual values of biodiversity are important to society and these must be protected and any changes in biodiversity that lead to alteration in ecosystem functioning as a result of human activities must be reduced (Chapin III, et al., 2000). Species serve as sources of marketable commodities in form of food, medicine, industrial values and recreational values (Kunin & Lawton, 1996). Other valuable functions provided by species include environmental modulation, ecosystem functions, ecological roles, knowledge, existence values and aesthetic values (Kunin & Lawton, 1996).

The productivity and stability of ecosystems can be enhanced by conserving species diversity because biodiversity plays an important role in determining the health of ecosystems (Martens, et al., 2003). Good land management of the land use activity can lead to an increase in land quality, for example, improvement in biodiversity (Mila i Canals, et al., 2007).

2.3.2 Effects of Land Use and Land Use Change on Biodiversity

Human activities involving land use cause damages to the natural environment and resources (Koellner & Scholz, 2007; Mila i Canals, et al., 2007; Potschin, 2009). The effects caused by the conversion of ecosystems through human activities include effects on biodiversity, and ecosystem functions among which are regulation of climate, air and water quality, soil formation, and the regulation of flooding and other natural hazards (Millennium Ecosystem Assessment, 2005). Alterations in soil functions have indirect effects on biodiversity (Mila i Canals, et al., 2007). The intrinsic nature value of a region is expressed by the local biodiversity (Lindeijer, 2000b) while life support expresses the dynamic nature value of an area (Lindeijer, 2000a). Biodiversity is usually considered to be a relative measure in reference to a particular reference system, for example, highest present value in a region which infers that species-rich areas should not get a higher value than a less species-rich areas while life support is measured in absolute term based on its contribution to ecosystem development (Lindeijer, 2000b). Change, impact and removal of ecosystems are the ecological consequences that result from different types of land

use (Kyläkorpi, et al., 2005). Desiccation, land fragmentation, landscape degradation, soil degradation, loss of nature development space, loss of biodiversity, impacts on the life support function are some of the effects of land use (Swan, 1998 cited in Kyläkorpi, et al., 2005). Impact on biodiversity, impact on biotic production, and impact on the regulating functions of the natural environment are the three impact pathways which are required to be assessed in the LCA of land use as proposed within the framework of the UNEP/SETAC life cycle initiative on LCIA and land use (Mila i Canals, et al., 2007). Both the effects of land occupation and land transformation are to be included in land use impact assessment (Mila i Canals, et al., 2007). Land transformation leads to changes and reduction in species composition (Mila i Canals, et al., 2007). Land occupation gives rise to a new kind of species composition existing in place of the original composition without the studied system occupying the land (Lindeijer, et al., 2002; Mueller-Wenk, 1998). Impacts due to transformation are considered to be more relevant than occupation impacts (Sala, et al., 2000) because transformation leads to the depletion or loss of scarce ‘*nature*’ (Vogtlander, et al., 2004) and transformation has a significantly high climatic impact resulting from a higher transfer of fossil-combustion-equivalent carbon compared to a corresponding land occupation during one year (Mueller-Wenk & Brandao, 2010).

The types of environments existing in a particular area before any land use change determine the level of impacts that will result, apart from those of land use itself (Kyläkorpi, et al., 2005). The “safeguard subjects” normally considered in the land use impact assessment are biodiversity, life support functions, and the competition of land and cultural values including landscape impacts (Lindeijer, et al., 2002). In order to assess the effects of land use impact on biodiversity, a quality measure, the affected area, and the duration of the impact are the three aspects to be considered (cf. Mila i Canals, et al., 2007 and Michelsen, 2008). Tropical forests (wet and dry), temperate forests, boreal forests, tropical grasslands, and temperate grasslands have been identified as the biomes that suffer most as a result of human land use activities (Mueller-Wenk & Brandao, 2010).

2.3.3 Biodiversity Indicators in Life Cycle Impact Assessment

Developing a good list of indicators is a pre-requisite to assessing and quantifying land use impacts associated with any human land use activities (Lindeijer, 2000b). Indicators are elements that fulfill the three basic functions of simplification, quantification, and communication and

hence, biodiversity indicators can be seen as those elements that are able to quantify the state of the environment and serve as communication tool for the state and trend of biodiversity and for the causal relationships for changes in the state, and the trend (Delbaere, 2002). The indicators developed for land use assessment must be applicable worldwide and these need to be accepted by the scientific community and widely used, and the data for the assessment must be readily available worldwide (Nunez, et al., 2010; Schmidt, 2008a). Wagendorp, *et al.* (2006), asserts that suitable indicators for measuring and monitoring land use impacts should be sensitive for changes in ecosystem state and functionality which coincide with the ‘natural resources’ and ‘natural environment’ areas of protection respectively. The same indicators used to measure biodiversity as a quality must be relevant for both transformation impacts and occupation impacts (Weidema & Lindeijer, 2001).

In order to monitor, interpret, and report accurately the trends and performance of land quality indicators, land quality should be assessed for specific types of land use and management and for specific agroecological zones conditions in a country (Dumanski & Pieri, 2000). Though the assessment of quality within these agroecological zones will be enhanced because the finer the scales used, the better the quality assessment, it must however be ensured that the basis of classification used is consistent (Michelsen, 2008).

There are different proposed indicators for measuring species diversity and some of these take their starting point from species richness (Lindeijer, et al., 2002):

- Number/Percentage of vascular plant species (Köllner 2000, Vogtländer *et al.* 2004)
- Number /Percentage of threatened vascular plant species (Goedkoop & Spriensma 1999, Mueller-Wenk 1998)
- Species accumulation rate (Köllner 2000, Lindeijer 2000)

The most commonly used biodiversity indicator is the species richness measured as vascular plants (Köllner, 2000; Lindeijer, et al., 2002; Schmidt, 2008a). Potentially disappearing fraction of species (PDF) or potentially affected fraction of species (PAF) may serve as indicators for changes in biodiversity (Udo de Haes, 2006). Indicator species are defined as species which occur only in one, two, or a maximum of three ecosystems (Vogtlander, et al., 2004). According to Gaston (1996) as cited in Michelsen (2008), the number of species is widely used as an indicator for biodiversity because of its ability to capture the total essence of biodiversity; the

availability of data on the species level; the easy measurability of the parameter; and its wide understandability. The number of species found in an area is found to increase with an increase in the area surveyed as can be shown by a given species-area curve (Schmidt, 2008a). This number of species is dependent on the land use types, the area size, and the biogeographical conditions of the environment (Lindeijer, et al., 2002). The nonlinear nature of the relationship between species and area must be taken into consideration in order to develop reliable characterization factors (Koellner & Scholz, 2008). The number of species S , for example of vascular plants, is related to the area A by the equation (Köllner, 2000; Lindeijer, et al., 2002; Weidema & Lindeijer, 2001):

$$S = cA^z$$

where c is the parameter for species richness, and z is a parameter for species accumulation rate which differs for land use types. Data of number of species collected on different scales usually result in different species-area curves and this should be taken into account while estimating species richness of a standardized area from these relationships (Schmidt, 2008a). The above equation, when transformed ($\ln S = \ln c + z \ln A$), can be used to adjust the species-area relationships for land use types to derive a standardized species number with a single species-area relationships for all land use types (Koellner & Scholz, 2008).

As an example, the impact of the loss of biodiversity in an area is given by the equation below (Lindeijer, et al., 2002):

$$[(Species_1 - Species_2) / Species_1] * Area * Time_{occupation}$$

where $species_{1 \text{ or } 2}$ represent species composition of two different land use types in which $species_1$ is the reference situation.

Good land use indicators must be meaningful for the biodiversity in any location and accurately reflect the complex, spatially dependent and nonlinear manner in which land use impacts biodiversity (Geyer, et al., 2010a).

The challenges encountered in assessing impacts on biodiversity include (Geyer, et al., 2010a):

- The quantification of biodiversity is complex and this makes it difficult to summarize in a single indicator just like any other environmental interventions such as global warming.

Biological diversity does not include only diversity within species but also between species, and diversity between and within ecosystems (UNEP, 1992). In order to assess impacts of land use on biodiversity therefore, there is need to universal impact indicators that can handle different levels of diversity. An indicator based on species richness has been developed by Lindeijer (2000) and Köllner (2000). Land use indicators based on ecosystem characteristics such as naturalness or wildness (Brentrup, et al., 2002) and exergy (Wagendorp, et al., 2006) have also been developed.

- Another challenge is the spatial aspects of biodiversity. Varying climatic conditions, initial quality of habitats, location within the landscape and the dynamic nature of species often lead to different of species and different species richness in similar biomes of the world (Geyer, et al., 2010a). To deal with this challenge requires that impacts on biodiversity are assessed in the location where the land use occurs.
- The nonlinearity of the relationship between land use and species viability is another challenge. Depending on the absolute level of production, additional increases in fuel crop cultivation may cause either minor or dramatic impacts on biodiversity.

2.3.4 Relating Land Use Impacts on Biodiversity to Functional Units

Life cycle assessment considers a systems perspective of a product and it is being applied for the assessment of systems that provide a function, or that deliver a service (Hauschild, 2005; Lemming, et al., 2010). Life cycle assessment methodology compares the environmental impacts associated with providing a functional unit, which is a core feature of the methodology along with ‘cradle-to-grave’ analysis (Kloepffer, 2008). This functional unit serves as a basis for comparison of services provided by compared products or technologies (Lemming, et al., 2010). A functional unit could be explicitly defined in a study or it could be derived implicitly. For a comparison purpose though, it is usually defined (Kloepffer, 2008). In the life cycle inventory modeling phase of an LCA, occupation and transformation of land are normally related to the functional unit in order to make the assessment of land use impacts feasible (Schmidt, 2008a). The modeling structure of an LCA requires that it should be related to a functional unit and this requires a flow character, either in or out of the considered product system (Udo de Haes, 2006). In land use impact assessment, allocation of land use interventions to functional units is considered to be a challenge (Kloepffer, 2008) especially in the case of transformation of land

(Lindeijer, et al., 2002; Schmidt, 2008a). Allocation of interventions due to occupation of land is relatively easy because occupation of a particular area of land and the functional units provided by the human activities occupying the land are considered to be linearly proportional to the duration of use (Lindeijer, et al., 2002). Occupation impacts are therefore allocated based on the total functional units produced. In the allocation of transformation interventions to functional units however, it is difficult to establish the relationship between the produced functional units and the preceding transformation impacts (Lindeijer, et al., 2002) because the number of functional units that can be supported by the transformation of a certain area of land is not easily determinable (Schmidt, 2008a). Allocation of transformation impacts to functional units is usually dealt with at two different levels: the level at which transformation is followed by successive and different types of occupation; and the level of a single occupation in the successive chain of occupation (Lindeijer, et al., 2002). Lindeijer et al. (2002) propose that at the level of chain of occupation, transformation impacts should be fully allocated to the preceding occupation since the transformation is purposely made for the first occupation and not the subsequent ones in the chain of occupation. Also proposed by Lindeijer et al. (2002) is the splitting of transformation impacts between the functional units in the case of a single occupation type in the successive chain. The determination of the total number of functional units produced during the whole duration of a single occupation is particularly problematic (Lindeijer, et al., 2002) and Lindeijer et al. (2002) propose that the attribution of transformation impacts to functional units in this difficult category should be handled based on attribution by economic depreciation and attribution of trends.

3 Methodology: Review

The objective of this study is to review progress up to date, recognize limitations, and indicate future prospects for the inclusion of land use impact assessment in LCA with a particular focus on the effects on biodiversity. This review takes its starting point from recent publications which are considered relevant to the LCA methodology. With this in mind, it becomes essential to identify a framework used in the assessment of land use impacts as an impact category in LCA.

Lindeijer (2000b) provides a review of methodologies for land use impact assessment. This review covers the development of various approaches for assessing land use impacts within the previous five years. Most of the approaches reviewed take their starting point from species richness measured as vascular plant species diversity (see Lindeijer (2000b)). The author also finds that land management practices have a major influence in the long-term changes experienced during land use. This review is assumed to have covered all earlier proposed methodologies and the main points of this review are believed to be the starting points for the framework proposed by Lindeijer et al. (2002) within the UNEP/SETAC Life Cycle Initiative.

Due to the fact that the methodological framework proposed by Mila i Canals et al. (2007) is a further development and refinement of the one developed by Lindeijer et al. (2002), for example, by the inclusion of a dynamic reference situation, it becomes imperative to include some of the earlier works taking their point of departure from the Lindeijer's framework in this present review, for example the work of Köllner (2000) which was eventually adopted in the Eco-indicator 99 Model (Goedkoop & Spriensma, 2001).

3.1 Goal and Scope of the Review

The goal of this review is to assess how various methodologies for implementing land use and land use change in LCA with a particular focus on impacts on biodiversity are being developed. This review takes its starting point from the work of Mila i Canals, et al. (2007) and it considers methodologies that are proposed, citing this particular framework and those that are particularly being used presently in industries, that is, the Biotope Method proposed by Kyläkorpi et al. (2005), and the proposed methodology of Köllner (2000). This present review considers the studies that are applicable to LCA, and with a particular focus on biodiversity.

The impacts on biodiversity resulting from the utilisation of water as a resource have been included in this review because water flow modification and water pollution have led to far

greater declines in biodiversity than in most terrestrial ecosystems (Dudgeon, et al., 2006). Another motivation for the inclusion of impacts due to water utilisation is that, both land and water utilisation can be jointly regarded as land use (Kyläkorpi, et al., 2005).

The reviewed studies listed in Table 1 in Chapter 4 are divided into two groups based on the way biodiversity is assessed in the studies. These are direct and indirect assessment. Studies based on indirect assessment method make use of structures, functions, and processes possessed by the ecosystem in assessing an ecosystem while those based on direct assessment method make use of dimensions of an ecosystem such as species, populations, communities, ecosystems, and ecosystems at a landscape scale.

4 Results

The results of this review show that only a few studies have been done which considered land use as an impact category in life cycle assessment. The reviewed literature covers articles published during the last three years with the exception of the works of Köllner (2000) and Kyläkorpi (2005) which were published earlier. These were included because of the reasons mentioned earlier in the methodology section. The proposed framework by Milà i Canals (2007) based on that of Lindeijer et al. (2002) has been applied using some good land indicators among which are species diversity proposed by Köllner and Scholz (2007 and 2008), and soil organic matter proposed by Milà i Canals et al (2007). Another proposal is the indirect assessment of biodiversity by means of some key factors proposed by Michelsen (2008). An overview of the reviewed literature with the main findings is given in Table 1 below.

4.1 Land Use and Land Use Change Assessment

There have been several discussions and proposals on how to include impacts due to human land use activities in LCA (Koellner & Scholz, 2007, 2008; Lindeijer, 2000a; Lindeijer, et al., 2002; Milà i Canals, et al., 2007; Milà i Canals, et al., 2007b; Vogtlander, et al., 2004; Weidema & Lindeijer, 2001). Land use is being considered as an impact category because human land use activities have various environmental consequences (Koellner & Scholz, 2008). Land use change is regarded as one of the most important causes of alteration in biodiversity (Chapin III, et al., 2000). One of the challenges faced by the inclusion of land use impacts on biodiversity in LCA is posed by the complexity surrounding the definition of biodiversity (Koellner & Scholz, 2008). Among the studies considered in this review, Köllner (2000) was the first to attempt to quantify the land use impacts on species diversity on both local and regional scales. The major land cover types (also referred to as land use types in his studies) of Europe were used for developing the characterization factors though the focus of the study is on land use activities which serve as entries for the life cycle inventory (LCI). All land use types are grouped into either low-intensity or high-intensity in order to derive the effects of the land use types on the regional scale. Land occupation and land transformation are considered as basic impact types in which land occupation is considered as a continuous intervention.

Table 1: Overview of reviewed literature, showing some key elements of the framework considered

	Reference	Biodiversity Assessment	Indicator(s)	Land Use Aspect(s) considered	Coverage Area	Reference System
1	Schmidt (2008a)	Direct	Vascular plant species	Occupation and transformation	Global	Renaturalisation state
2	Michelsen (2008)	Indirect	Ecosystem scarcity and ecosystem vulnerability	Occupation	Boreal forests but may be extended to other regions	Original quality level immediately before the studied human land use activities
3	Koellner & Scholz (2007) and Koellner & Scholz (2008)	Direct	α -diversity of plants (vascular and threatened), moss and mollusks	Occupation and transformation	Global	For transformation, neutral land use or land use before transformation and the current regional status of ecosystem quality for occupation impact
4	Nunez et al. (2010)	Indirect	Aridity, erosion, aquifer over-exploitation and fire risk	Occupation and transformation	Arid, semi-arid and dry humid regions	No comparison with any reference ecosystem region
5	Mueller-Wenk et al. (2010)	Indirect	Carbon mobilized from soil and vegetation	Occupation and transformation	All climatic zones of the globe	carbon content of land covered with potential natural vegetation is used as baseline
6	Maes et al. (2009)	Indirect	Green (evaporative) water flow	Occupation and transformation	Site-dependent	Evapotranspiration of water by a potential natural vegetation
7	Mila i Canals et al. (2009)	Indirect	Water stress indicator	Occupation and transformation	Main global river basins	Average precipitation in a potential natural vegetation
8	Kløverpris et al. (2008), (2010) and Kløverpris (2009)	Indirect	Biomes (potential natural vegetation)	Occupation and transformation	Global	Dynamic reference situation
9	Kyläkorpi et al. (2005)	Direct	Red-listed species; key features	Occupation and transformation	Local	Comparison of "Before" and "After" situations
10	Geyer et al. (2010a) and (2010b)	Direct	Habitat-type areas	Occupation and transformation	Global	The sum of the habitat areas in the total study area (Current land use)
11	Koellner (2000)	Direct	Vascular plant species	Occupation and transformation	Mid-Europe	Regional species pool (average present species diversity)
12	Mila i Canals et al. (2007)	Not assessed	Soil organic matter	Occupation and transformation	Local	Alternative system (which may or may not be natural relaxation)

Schmidt (2008a) treats land use by considering land use types (for example, grasslands, forest) as entries upon which the LCI should be established. Only occupation and transformation impacts are quantified. Allocation of transformation of land to functional units and the distinction between consequential and attributional LCA are considered. The way to handle the transformation of already degraded land, for example, is outlined. Renaturalisation time and the number of species affected are used to derive the characterization factors which capture differences between cultivation practices.

Kyläkorpi et al. (2005) develops a method for the quantification of impacts on biodiversity caused by changes in land use. It is based on the comparison of the utilised area's biotopes (in terms of distribution and quality) before and after the change. In this method, it is assumed that the gains and losses of biotopes, caused by a change in land use, are indicators of the resulting changes in biodiversity. These gains and losses are quantified by calculating the acreage of the various biotopes. The method is able to differentiate between different impact types and different land use types.

Köllner and Scholz (2007) and (2008) propose a methodology based on ecosystem damage quality measured in terms of species diversity. Ecosystem Damaging Potential (EDP) is used as the characterization function which expresses the ecosystem damage for a specific land use type. The proposed method is not geographically referenced because of the assumption that the exact location of land use is not known in many LCA applications. This is in contrary to Mila i Canals et al. (2007) who proposed inventories and impact assessment that are geographically dependent. Land transformation is assessed based on a factual or virtual restoration time, meaning that the damage of land transformation is largest for land use types which are difficult to restore and need extremely long time to develop. For transformation, the land use before transformation can be used as baselines if the LCIA approach stresses dynamics of damages over time or if absolute damages are more important in the specific LCIA approach, the neutral land use (land use with no effect in the region with $EDP=0$) should be used as baseline. Current regional status of ecosystem quality is used for calculating occupation impacts, that is, static reference situation as opposed to the dynamic reference situation proposed by Mila i Canals et al. (2007).

Nunez et al. (2010) propose an indirect assessment method which is based on ecoregions (terrestrial natural vegetations/regions). The proposed methodology deals with the assessment of desertification (one of the causes of irreversible soil degradation in arid areas) environmental

impact related to land use in which spatial and temporal aspects of land use activities (environmental impacts) are considered. Site-specific environmental effects are measured with the aid of Geographic Information Systems (GIS) and LCA thereby giving rise to site-dependent characterization factors. Combining these ecoregion-specific characterization factors with the area of the process and the area of the ecoregion, the impact is determined.

Mueller-Wenk et al. (2010) focus on the influence of land use (in the narrow sense of quality change in soil and vegetation) on the transfers of CO₂ between atmosphere and land, that is, the climatic impact of CO₂. Types of land transformation that cause a substantial change of carbon storage in vegetation or soil are considered. The net carbon change attributable to the occupation land management method is treated as an additional land transformation because the impact of carbon stock change during occupation does not stop at the end of this occupation. For occupation impact, the reference situation is the carbon content of the specific potential natural state vegetation associated with each of the geographical locations of the world (historic, if this natural vegetation has been absent locally for a long time). The proposed methodology has the ability to differentiate between different climate zones and all types of land transformations and land occupations.

In Michelsen (2008), land use impacts on biodiversity due to forestry operations in a boreal forest are assessed based on key factors that maintain biodiversity in the ecoregion. In this assessment, biodiversity is assumed to be linked to ecosystems in terms of structural components, processes, features of the biological system, for instance, forest in this case. . Three different aspects (quality measure, duration of impact, and affected area) are used to quantify the land use impact on biodiversity. In determining the changes in land quality and total impact due to land use changes, the quality level immediately before the studied human activity is used as a reference situation. Relaxation time is assumed to be equal to the rotation time in the forest. In the application of the proposed methodology to an LCA of forestry operations in the boreal forest, only the occupation impact is considered and this represents the postponement of the natural processes that restore the forest to its natural state and quality. The proposed methodology is able to differentiate among similar activities in different ecoregions and different management practices within one ecoregion.

In Kløverpris et al. (2008), (2010) and Kløverpris (2009), the focus is on the identification of the land actually affected in the systems, meaning land use changes related to marginal crop production. The proposed methodological framework is used for identifying and quantifying the long-term land use consequences of changes in crop demand and their geographical locations. Economic modeling is used to identify land use consequences of crop consumption and the regions in which these occur. The proposed methodology has the ability to differentiate between land use types and geographical locations, that is, different land types (cultivable and grazable) in different regions and can also identify the affected biomes within a given region. The expansion of croplands as a mechanism for increasing crop production is considered as a special case of transformation. Expansion can also lead to delayed release of croplands and be considered an occupation process relative to the ongoing trend in cropland area. Dynamic reference situation is used for measuring transformation impacts. In the methodology, biomes (potential natural vegetations) are ascribed to the areas affected by agricultural expansion in order to provide a basis for assessing the environmental impacts from land use in LCIA.

4.2 Biodiversity Assessment

An ecological system can be described and analyzed in two major ways. These include indirect assessment which involves assessing an ecosystem based on abstract systemic dimensions using, for example, structures, functions, and processes possessed by the ecosystem; and direct assessment which makes use of organismic dimensions of an ecosystem such as species, populations, communities, ecosystems, and ecosystems at a landscape scale (Köllner, 2000). Among the reviewed studies, only five actually assessed biodiversity directly. Köllner (2000) was the first to assess biodiversity by using vascular plant species as proxies for biological diversity. Other studies which also assessed biodiversity directly include Schmidt (2008a), Köllner and Scholz (2007 and 2008), Geyer et al. (2010a and 2010b) and Kyläkorpi et al. (2005). The remaining studies use the indirect assessment method to assess land use impact on biodiversity. Table 2 below shows the reviewed literature which assessed biodiversity directly and measured species richness as absolute or relative. The difference between the relative and absolute approach is the way in which values are being attached to species in species-poor regions and species-rich regions (Schmidt, 2008a).

4.2.1 Direct Assessment

In the direct assessment method, most studies take their starting point from species richness measured as vascular plant species diversity. Vascular plant species is used as a proxy for species richness for a land use type because it constitutes terrestrial ecosystems and there is a correlation between it and other species groups' diversity (Duelli and Obrist, 1998 cited in Köllner & Scholz, 2008).

Köllner (2000) developed the characterization factor species-pool effect potential (SPEP) based on vascular plant diversity as an indicator for biodiversity. Köllner (2000) uses the natural logarithm of the normalized species richness $\ln(S_j/S_{ref})$ to express the characterization factors for biodiversity impact assessment in order to show that as the species richness increases, the marginal utility of ecosystem services diminishes (that is, the redundant species hypothesis (Walker, 1992)). The linear approach is often used when biodiversity and not ecosystem service is the object of the assessment (cf. Geyer et al. 2010b).

The Biotope Method developed by Kyläkorpi et al. (2005) has been widely applied to quantify the impacts caused by changes in land use on biodiversity as a result of electricity generation in Sweden. Red-listed species and key features such as structures in the landscape are used as an indicator to divide the studied area into different kinds of biotopes based on the habitats' compositions. Red-listed species are those threatened species considered to be at risk of disappearing in the near future and cannot, therefore, be true representative of the biodiversity of various ecosystems but do reflect the degree of anthropogenic disturbance of the ecosystems and the deterioration of resident biodiversity (Cederberg et al., 1997 cited in Kyläkorpi et al., 2005).

Schmidt (2008a) develops characterization factors based on species diversity only. In this methodology, weighting between species is not considered, that is, scarce and threatened species are given the same weight regarding species richness as invasive unwanted species. The method for measuring the species richness is based on the absolute approach in which ecosystems in species rich regions are weighted higher than those in species poorer regions, and that each species is given the same value. The species richness is measured for a standardized area of 100m² for all land use types and all regions.

Geyer et al. (2010b) propose four characterization models for biodiversity assessment based on hemeroby, habitat evenness, potential species richness and abundance in which rarity of species

is taken into consideration and the authors find that the weighting of each species based on its rarity increases the characterization factors for each of the habitats suitable for the species. The characterization models derived based on hemeroby, potential species richness, habitat and evenness are absolute qualities while the one based on potential species abundance makes use of a reference state for species abundance.

Köllner & Scholz (2008) estimate the number of threatened species in addition to the average species number as an indicator because of the underestimation of the ecological value of an ecosystem type associated with species number which accounts for a few species of any threat status. According to Köllner & Scholz (2008), if other animal species (in addition to plant species diversity, threatened plant species, moss and mollusks) groups with their specific habitat preferences are integrated into the characterization factors, the values that result would be different for each land use type.

Table 2: Overview of Studies which assess Biodiversity directly

	Reference	Biodiversity Assessment	Measuring Species Richness
1	Schmidt (2008a)	Direct	Absolute
2	Koellner & Scholz (2007) & (2008)	Direct	Relative
3	Kyläkorpi et al. (2005)	Direct	Absolute
4	Geyer et al. (2010a) and (2010b)	Direct/Indirect	Absolute/Relative
5	Koellner (2000)	Direct	Relative

4.2.2 Indirect Assessment

Ecological diversity is included indirectly in the assessment of species diversity because in the cause-effect chain, factors leading to the reduction of habitats directly contribute to the reduction of species diversity (Koellner & Scholz, 2007).

Only two studies (Michelsen (2008) and the works by Kløverpris) are related to land use impacts on biodiversity. Kløverpris develops a methodological framework for identifying and quantifying the long-term land use consequences of changes in crop demand and their geographical

locations. This methodology involves the analysis of consequences caused by a given change or decision, that is, consequential LCA in which economic modeling is used to identify land use consequences of crop consumption and the regions in which these occur. In this method, biomes (potential natural vegetations) are ascribed to the areas affected by agricultural expansion in order to provide a basis for assessing the environmental impacts from land use in LCIA. Michelsen (2008) uses some key factors such as ecosystem scarcity, ecosystem vulnerability, and condition for maintained biodiversity as indirect factors in the assessment of biodiversity. The ecosystem quality (in terms of biodiversity) is then assessed as the product of the three factors. Other studies (Nunez et al. (2010), Mueller-Wenk et al. (2010), Maes et al. (2009), and Mila i Canals (2009)) consider other impacts as related to water use, carbon impacts, and desertification impacts. These are not directly related to biodiversity. For instance, Nunez et al. (2010) use some factors such as aridity, erosion, aquifer overexploitation and fire risk to evaluate the desertification potential of land use activities.

4.3 Relating Land Use Impacts to Functional Units

Functional unit is a very key aspect in the comparison of different options in LCA (Lemming, et al., 2010). Table 3 reveals that some studies actually applied the methodology proposed to an LCA. However, in some studies like Schmidt (2008a), the functional units could be derived from the occupational impact but this is not done in the paper. Michelsen (2008) relates the occupation impact to the production of 1 m³ round wood logs under bark delivered at the gate of a factory. The total impact of land use on biodiversity could be expressed as: quality difference*ha*yr per m³ round wood. Kyläkorpi et al. (2005) argues that the land use impacts related to the production of electricity could be linked to the unit of the electrical energy produced using the Biotope Method in which case the biotope's area information is related to the electricity generated by the activity as, for instance, m²/KWh of electricity. In Kløverpris et al. (2010), transformation and occupation impacts are presented per tonne of increased wheat production in the households in each of the countries (regions) considered.

Table 3: Overview of Reviewed Studies with Land Management Regimes, and LCA Applicability

	Reference	Differentiation between Different Land Management Practices	Differentiation between Land in Different Regions	Renaturalisation Time	Functional Unit	Applied to LCA?
1	Schmidt (2008a)	x	x	x	N/A	N/A
2	Michelsen (2008)	x	x	x	x	x
3	Koellner & Scholz (2007) & (2008)	x	N/A	x	N/A	N/A
4	Nunez et al. (2010)	x	x	N/A	N/A	N/A
5	Mueller-Wenk et al. (2010)	x	x	x	N/A	N/A
6	Maes et al. (2009)	N/A	x	N/A	N/A	N/A
7	Mila i Canals et al. (2009)	N/A	x	N/A	N/A	N/A
8	Kløverpris et al. (2008), (2010) & Kløverpris (2009)	x	x	N/A	x	x
9	Kyläkorpi et al. (2005)	x	x	N/A	x	N/A
10	Geyer et al. (2010a) & (2010b)	x	x	N/A	x	x
11	Koellner (2000)	x	x	N/A	N/A	N/A
12	Mila i Canals et al. (2007)	x	x	x	x	N/A

Nunez et al. (2009) consider the methodological aspects of the assessment of potential desertification environmental impacts in LCA. Desertification is a great concern in the arid areas worldwide and it is considered in life cycle assessment because it leads to irreversible soil degradation (Nunez, et al., 2010). Indicators for this assessment are selected based on the work of DESERTLINKS 2004, cf. Nunez et al., 2009 which recommended that the availability of data sets for the area over a sufficient length of time, and the possibility of obtaining information using remote-sensing technologies are the two main factors necessary for obtaining good scale to derive an indicator.

Maes et al. (2009) propose a methodology to include water quantity-related impact of a land use occupation and its change following land use transformation. This assessment is based on the ecosystem services that water flows provide and these aquatic ecosystem services are maximal when there are no human impacts on a river basin (Maes, et al., 2009). Evaporative flow is used as an indicator for the impact assessment method developed in this study. According to Maes et

al. (2009), minimal impact occurs when a particular land use activity evapotranspires the same amount of water as that of the potential natural vegetation which is the stage with the highest habitat diversity, resistance, and resilience of all succession phases. A land use activity that provides water has minimal impact while the one that consumes more water disturbs the ecosystem functioning until the aquatic ecosystem is irreversibly damaged (Maes, et al., 2009). According to Dumanski and Pieri (2000), Land quality should be assessed for specific types of land use and management and for specific agroecological zones conditions in a given region. From Table 3, several of the proposed methodologies in the studies are able to distinguish not only between different land use types but also land use management practices.

5 Discussion

Land, as a resource, must be considered not just as inputs to the economic systems in LCI but the impacts associated with its use must be assessed in LCIA as well (Kløverpris, et al., 2008). For example, in the case of agricultural product LCA, it is necessary to identify the areas affected by agricultural expansion in order to characterize the areas for a full land use LCIA, implying that the actual occupied area in terms of square metres or hectares may not be enough for a proper consequential LCA (Kloverpris, 2009). This means that land use impact has gone beyond the initial assessment in form of the actual “space” occupied but the impacts resulting from the occupation and possibly initial transformation to make the land fit and usable for the purpose at hand must be assessed. Establishing the boundaries between technical or economic system and the environment can be used to distinguish between activities captured in inventory analysis, and those modeled in impact assessment (Finnveden, et al., 2009).

Why the use of species diversity?

The focus of this review does not only revolve around the diversity of species. However, species diversity is considered by several studies due to the availability of data in this regard and that the focus of land use impact assessment methodologies developed so far have been on species diversity (Köllner, 2000; Schmidt, 2008a). However, Species richness does not give a good indication of completeness and rarity of species because the number of species present in a particular region does not give the value (that is, the importance) of the species (Vogtlander, et al., 2004). In order to identify the biodiversity indicator level for current land use, the use of species richness may not reflect the threat to biodiversity because a particular land use may favour some species more than others (Weidema & Lindeijer, 2001). According to Weidema & Lindeijer (2001), the current biodiversity indicator level should reflect the number of species that are not affected negatively by the current land use. However, with species diversity, it is possible to combine the ecological effects of certain impact categories such as acidification, eutrophication, ecotoxicology with land use for weighting purpose in LCIA if the same endpoints are considered (Köllner, 2000).

Species abundance and evenness are only meaningful in the context of an ecologically relevant spatial area, typically an ecoregion and the geographical scope of LCI modeling with spatially explicit land use should therefore be entire ecoregions (Geyer, et al., 2010a).

Measuring Species Richness

Species richness refers to the number of species per area (Weidema & Lindeijer, 2001). As a biodiversity indicator, species richness could be based exclusively on vascular plant species richness (Goedkoop & Spriensma, 2001; Köllner, 2000; Lindeijer, 2000a). Species richness as an indicator could be measured as the loss of vascular plant species richness in relative terms, through dividing by a local reference state, or as absolute scores for species diversity (Schmidt, 2008a).

Among the reviewed studies that assessed biodiversity directly, Schmidt (2008a) and Kyläkorpi et al. (2005) use the absolute scores for species diversity while Koellner and Scholz (2007) and (2008) and Koellner (2000) choose the relative measure, see Table 2. As stated in Weidema & Lindeijer (2001), the Convention on Biological Diversity supports the relative scores for species diversity.

Michelsen (2008) assesses biodiversity indirectly by considering ecosystem vulnerability and scarcity, which follows the work done by Weidema and Lindeijer (2001). Weidema and Lindeijer (2001) modifies species richness of an ecosystem by two factors at ecosystem level: inherent ecosystem scarcity, which is “*expressed as the inverse of the potential area that could be occupied by the ecosystem if left undisturbed by human activities*”; and ecosystem vulnerability, which “*indicates the relative number of species affected by a change in the ecosystem area, as expressed by the species-area relationship*”. Inherent ecosystem scarcity can only be applied at the biome level with the maximum potential ecosystem area being boreal forests because data for the potential ecosystem area are globally available only at the biome level (Weidema & Lindeijer, 2001).

Weighting between species does affect the ranging of land use types in terms of species richness with rare species having high priority and invasive and common species having low-priority (Schmidt, 2008a). Schmidt (2008a), however, gives the same weight to both scarce and threatened species and invasive unwanted species.

The standard area used for species richness affects the relative difference between species richness of different land use types because different land use types have different accumulation rates (Schmidt, 2008a).

5.1 How the Inclusion of Land Use and Land Use Change affect General Conclusions

The methodology developed by Schmidt (2008a) focuses only on a limited part of biodiversity problems, that is, vascular plant species. This model also does not consider weighting or normalization among different impact categories. It is, therefore, difficult to make any judgement or general conclusion based on this limited aspect. Schmidt (2008a) shows that depending on the renaturalisation times considered, the transformation impacts from the transformation of 1 ha of land could be significantly higher than the occupation impacts from the occupation of 1 ha in 1 year. Species richness and ecosystem vulnerability are two factors that tend to affect the magnitude of the occupation impacts (Schmidt, 2008a). A region with a low species richness could be balanced with high ecosystem vulnerability for occupation impacts, and long renaturalisation time for transformation impacts, and vice versa (Schmidt, 2008a). It could be assumed that the inclusion of LULUC effects on biodiversity in LCA using the proposed methodology would affect the result significantly.

Michelsen (2008) does not suggest any weighting factors for comparing land use to other impact categories. However, the author used the potentially disappeared fraction (PDF) of species as a weighting factor to compare the impact due to acidification with that due to land use by multiplying the space and time needed for logging 1 m³ of timber with the PDF. This leads to a higher land use impact on biodiversity compared to impact due to acidification. However, it is somewhat difficult to infer how this affects the general conclusion because the calculated impact would depend on the annual yield, in this case, of trees, and the ecosystem quality, cf. Michelsen (2008).

The works done by Kløverpris and colleagues (Kløverpris, et al., 2008; Kløverpris, 2009; Kløverpris, et al., 2010) only consider net expansion of agricultural area (in terms of square meters) as a result of marginal crop consumption in a given region. It is difficult to deduce how the effects of land use and land use affect the general conclusions in the studies because land qualities resulting from different forms of transformation cannot be distinguished. However, Kløverpris (2009) takes the works further by ascribing natural potential vegetations (biomes) to the affected areas as a result of the agricultural expansion but these biomes are not characterised for their land quality description.

From the studies done by Geyer et al. (2010a) and (2010b), indicators based on different aspects of biodiversity can result into fundamentally different results. Based on the different production scenarios considered by these authors, different biodiversity indicator results in different biodiversity impacts with some of the indicators coming to very different conclusions. One of the reasons for this could be as a result of the habitats transformed and/or occupied in order to produce a certain output.

5.2 Correlations of the Proposed Methodologies with Scientific Assumptions

According to Lindeijer (2000b), endpoints or safeguard subjects must be considered in connection with the developed indicators so as to enhance decision making by those who are normally non-environmental experts. In the reviewed studies, the impact pathway considered is biodiversity and this is one of the safeguard subjects that need adequate protection to ensure sustainability. Mila i Canals et al. (2007) propose that the implementation of indicators for measuring the effects of land use on biodiversity must be checked with a consistent framework. Among the reviewed studies, Schmidt (2008a) is the first to propose a methodology in connection with the framework.

The development of characterization factors that are unique on a global scale is a requirement for the assessment of any impact category in LCA (Nunez, et al., 2010). Schmidt (2008a) develops a methodology that is able to differentiate between different land use types in different parts of the world. This spatial coverage is a good starting point for the inclusion of land use as an impact category in LCA. The methodology proposed by Michelsen (2008) focuses on the boreal forest but according to the author, this could be extended to cover ecoregions in any part of the world. This will require a good refinement though. The method proposed by Kløverpris et al. (2008) and applied in Kløverpris et al. (2010) and Kløverpris (2009) is also global in the sense that it makes use of a global database which can assess land use impacts in any part of the world. Kyläkorpi et al. (2005) proposes a method which has been applied to Swedish cases but this method can be extended to any region of the world in as much the affected biotopes are recognized. The species-pool effect potentials (SPEP) method proposed by Koellner (2000) is based on mid-Europe but this has been modified and implemented in the Eco-indicator LCIA

method (Goedkoop & Spriensma, 2001). The framework proposed by Geyer et al. (2010) analyses biodiversity at the level of the species composition and abundance in an ecoregion and this does not involve collection of comprehensive field observations of all species or modeling individual species responses to changing conditions.

The elements of Table 3 given in the Results section are further described below in order to understand their implications:

Relaxation Time

The renaturalisation time is *'the duration between end of occupation and the point in time where a land cover type has developed with the same damage potential as the initial one and this does not imply that the same ecosystem type is restored, but one of equal quality and value'* (Koellner & Scholz, 2007). Relaxation (or renaturalisation) time can also be defined as the time required to reach maximum potential land quality after a complete system removal assuming there is no degradation in the relaxation potential of the occupied area (Weidema & Lindeijer, 2001). Land use impact due to transformation tends to increase with increasing restoration time (Köllner & Scholz, 2007). The estimation of renaturalisation times affects the results significantly because most land use impacts are related to transformation of land (Schmidt, 2008a). In Michelsen (2008), rotation time in the forest is assumed to be the relaxation time, however, this may be difficult to determine in forestry where selective felling is carried out thereby making it difficult to identify the rotation periods. Apart from this difficulty, the temporal impacts could be significantly underestimated because of the assumption that the relaxation time could be derived from the rotation time.

The Choice of Reference Situation

The choice of reference situation is often found to affect the result thereby leading to inconsistencies (Schmidt, 2008a). Mueller-Wenk and Brandao (2010) confirm different results for land occupation when the reference situation, potential natural vegetation (PNV), is either grassland or forest. The authors argue that the climatic impact due to the occupation of 1 hectare of cropland for one year is higher when the PNV is forest compared to when the PNV is grassland because of the lower carbon transfer per hectare during transformation of grassland biomes. Köllner (2000) chose average actual or historical levels of species diversity as reference states, while Lindeijer *et al.* (2002) use the maximum actual species diversity. The method

proposed by Koellner & Scholz (2007) and (2008) is not geographically referenced because of the assumption that the exact location of land use is not known in many LCA applications. This is in contrary to Mila i Canals et al. (2007a) who propose inventories and impact assessment that are geographically dependent. In Koellner & Scholz (2007) and (2008), the relative species numbers are calculated by choosing the regional average species richness as a reference for assessing species richness of local plots. In Schmidt (2008a), the reference situation represents the potentially affected number of species on the occupied piece of land.

Different Land Management Practices and Land Use in Different Regions

It is important to develop methodologies that can distinguish between different land use types and management practices because ecosystems are not ecologically homogenous on a larger geographical scale and the protection of an area in a region does not necessarily compensate for the intensive use of another area in a different region (Brentrup, et al., 2002).

The methodology proposed by Michelsen (2008) can differentiate among similar activities in different ecoregions and different management practices within one ecoregion and the indicators can be used at different levels (biome, landscape, vegetation type, etc) provided the necessary data are available and that it fulfills the purpose of the study. However, this may be difficult to ascertain because the differentiation which often results as a reduction in impacts in one region compared to another may result from the increase in the proportion of the areas set aside. As a result of increase in demand for land use (see Kløverpris et al. (2008) and (2010)), it is difficult to expect that areas set aside would increase. The proposal by Schmidt (2008a) is spatially differentiated while taking into account different land management regimes. Kløverpris et al. (2008) proposes a methodology that has the ability to differentiate between different land use types in different regions and identify the affected biomes within a given region. The Biotope method proposed by Kyläkorpi et al. (2005) can also differentiate between different impact types and land uses.

LCA Usefulness and Applicability

It is not just enough to develop a methodology to assess land use impact on biodiversity but the proposed methodology should be viable enough for proper integration into the LCA methodology. Among all the studies reviewed here, only three actually used the proposed

methodologies for an LCA application. These are the studies by Michelsen (2008), Kløverpris et al. (2008), and Geyer et al. (2010) which are applied to forestry, crop consumption, and ethanol production respectively. The methodologies proposed by Schmidt (2008a) and Kyläkorpi et al. (2005) are not used for an LCA application in the literature but these stated the functional units which could serve as basis for LCA applications. The Biotope method developed by Kyläkorpi et al. (2005) has, however, been applied for quantifying impacts on biodiversity caused by changes in land use as a result of electricity generation in Sweden within the Vattenfall Group. All the reviewed studies have some degree of application to LCA and this possibly reflects their usefulness.

5.3 General Trends of Land Use and Land Use Change in LCA

The importance of land use in LCA has generated a lot of interest because of the need to conserve the natural environment and to enhance the applicability of the LCA methodology. This has led to an increase in the number of studies on land use in LCA. The review by Lindeijer (2000b) shows the importance of land use methods in LCA. Most of the studies reviewed by Lindeijer (2000b) stress the importance of biodiversity measured in terms of vascular plant species diversity. However, there have been other methodologies which consider other impact pathways, for example, Milà i Canals et al. (2007b) proposes a method to assess the impacts of land use on life support functions. The number of studies thereby correlates with an increase in the interest in the research area. However, it is difficult to identify any trend of convergence.

Different methods are being proposed which do not actually agree with one another. Some of these methods are not “closely” related to the use of land in the normal usage sense, for example, Mueller-Wenk et al. (2010) focuses on the influence of land use (in the narrow sense of quality change in soil and vegetation) on the transfers of CO₂ between atmosphere and land, that is, the climatic impact of CO₂ and Maes et al. (2009) which develop a land use impact assessment method that is related to water quantity based on the ecosystem services green and blue water flows deliver.

6 Conclusions

Life cycle assessment is a methodology used for the comprehensive assessment of the environmental impacts associated with a product or service and the perspective normally considered is the total life cycle. It is therefore necessary to include land use and its associated impacts in the assessment of products or service involving agriculture, forestry, mining, and transportation activities. Natural environment, natural resources and man-made environment, and human health are the areas of protection directly and indirectly affected by land use. Areas of protection or “safeguard objects” are fundamental objectives in life cycle impact assessment.

Biodiversity is one of the attributes of ecosystem quality and this attribute needs to be protected if ecosystems and their associated ecological services are to be conserved. Ecosystem is an essential “good” and there is no finite compensation for its complete elimination. There is also an established relationship between biodiversity and ecosystem functioning. The loss of biodiversity has become a concern to humans because ecosystems help in the regulation of the Earth as a whole. Loss of biodiversity is one of the impacts associated with land use and biodiversity can be assessed by species richness which has positive effects on ecosystem functions. Though species, genetic, and ecological levels are the three subdivisions of biodiversity yet species diversity is usually used for the assessment of biodiversity because of good data availability which serves as the basis for its use in most developed methodologies for the assessment of biodiversity. The inclusion of loss of biodiversity in LCA is adjudged to be problematic because it does not have a clear flow character in and out of the product system and it often has a local focus (Udo de Haes, 2006). The proposal of indicators for biodiversity which are globally acceptable as a true reflection of biodiversity is a good starting point.

The purpose of this study is to review progress in the development of methodologies that incorporate land use in LCA. In order to accomplish this task, recent publications on land use and land use change and its implementation in LCA were reviewed. The primary focus was on the effects of land use and land use change on biodiversity and the starting point was the framework developed within UNEP/SETAC Life Cycle Initiative on LCIA and proposed by Mila i Canals and colleagues; see Mila i Canals et al. (2007). Using this framework, it is easy to identify a common ground and to see whether the methodologies being proposed are possibly converging or not. Limitations and possible future research areas are also identified.

In this review, the implementation of land use and its effects on biodiversity in LCA has been considered, taking into consideration several studies which have proposed methodologies to cater for this aspect. The number of studies in this area has been on the increase though. However, it is difficult to identify if there is a trend because most of the proposed methodologies are conflicting except for a few which assess biodiversity using species richness measured in terms of vascular plant species. This shows the importance of the research problem and its subsequent consideration in LCA. Moreover, there have been several arguments against the use of vascular plant species but this is still being regarded as the best available option for now. For the integration of land use as an impact category in LCA to be made possible, the non-spatial aspects of the LCA methodology would need adequate refinement in order to incorporate the spatial nature of land use activity and land use impacts as a function of place. In order to overcome the shortcomings identified in this study, more research work would be needed.

7 Recommendations and Perspectives

Species richness alone may not be able to reflect the actual ecosystem functioning (Chapin III, et al., 2000) because the presence of a species in an ecosystem does not really reflect its importance especially if the abundance is below a certain level (Michelsen, 2008). This problem could be tackled by the Shannon-Wiener index because both species richness and evenness are considered (Geyer, et al., 2010b). Another way to tackle this problem is the incorporation of the free net primary productivity (fNPP) into biodiversity assessment methodologies because the global biodiversity of plants often depends on ecosystem productivity (Irigoien, et al., 2004). As suggested by Weidema & Lindeijer (2001), free net primary productivity (fNPP) may serve as an indicator for impacts on biodiversity. The fNPP is defined as the net carbon uptake of an ecosystem less the amount of carbon sequestered for human use (Weidema & Lindeijer, 2001). Also, global biodiversity is assumed to grow with global biomass (Rothman, 2001). Properly-derived indicators of biodiversity could also serve as proxies for land occupation or transformation impacts on life-support functions of a given ecosystem (Lindeijer, et al., 2002).

If the motivation for protecting biodiversity is to conserve ecosystem processes, then species richness may not be a good indicator of biodiversity because the differences in ecosystem processes experienced from one region to another cannot be explained by the differences in species richness but these differences are mostly driven by climate, resource availability, and disturbance (Millennium Ecosystem Assessment, 2005). According to Tietenberg (2006), due to the interdependence of species within ecological communities, any particular species may have a value to the community far beyond its intrinsic value. Certain species contribute balance and stability to their ecological communities by providing food sources or holding the population of the species in check (Tietenberg, 2006).

Freshwater ecosystems along with their biodiversity provide several ecosystem functions such as buffering against droughts and floods, and biodegradation of organic waste (Dudgeon, et al., 2006; Maes, et al., 2009). This, therefore, makes case for the inclusion of the assessment of freshwater ecosystems in life cycle impact assessment. Flow modification and water pollution have also led to greater declines in freshwater biodiversity in freshwater ecosystems compared to terrestrial ecosystems (Dudgeon, et al., 2006). However, the inclusion of water use in land use impact assessment may serve as a source of confusion. One of the motivations for the critics of the LCA methodology is that there are lots of inconsistencies; see Finnveden et al. (2009). It will

be noteworthy to mention here that water and land are different resources and should not be mixed together when assessing impacts of human activities. Since they are both separate resources, it is then necessary to treat them as such. Land competition could be handled as an impact category. The assessment of impacts from water use should be separated from that of impacts from land use (Finnveden, et al., 2009).

The use of geographical information systems (GIS) to obtain information in deriving indicators with wider scale and coverage (Geyer, et al., 2010a; Nunez, et al., 2010) is a good development to reduce the use of marginal values in LCA. This would normally result in a more accurate LCA (cited in Nunez et al. 2010 (Bengtsson, et al., 1998)) because GIS methods are able to give site-specific information on the environmental effects of a product or production system by defining site-dependent characterization factors (Nunez, et al., 2010). This will help to fine-tune the LCA methodology because the spatial independence of the methodology is reduced and the weakness of LCA with no impact categories related to land use is eliminated (Nunez, et al., 2010). With the GIS, LCA can be adapted to account for the effects of land use impacts on biodiversity (Nunez, et al., 2010).

In order to make scientific information of much greater value, it must be presented in more appropriate ways by developing techniques for translating scientific understanding into policy-relevant information for decision-makers. Nunez et al. (2009) recommend that in the assessment of land use impacts, it is necessary to develop indicators with a multiple approach which considers the three aspects of sustainability: biophysical, social, and economic. This is a good perspective to consider if an assessment methodology is to be totally acceptable by all stakeholders. Data on the nature and distribution of biological diversity and relevant knowledge must be bridged in order to develop a good framework for biodiversity assessment that is understandable and acceptable for policy and decision making (Busby, 2002). This will also enhance the acceptability of the LCA methodology.

Since the protection of an area in a region does not necessarily compensate for the intensive use of another area in a different region because ecosystems are not ecologically homogenous on a larger geographical scale (Brentrup, et al., 2002), it then becomes important to develop a land use impact assessment methodology that can differentiate both between different land use types and different regions so as to ensure global and wider spectrum of LCA applicability. This, as

pointed out by Schmidt (2008), must not be “*too coarse-grained regarding the differentiation between different land use types or too narrow regarding spatial coverage*”.

The methodology proposed by Michelsen (2008) depends so much on the selection and scaling of key factors that can be used to assess biodiversity indirectly. The number of key factors selected for different regions will be different and this may affect the homogeneity of the conditions used for the assessment, thereby affecting the credibility of the results and subsequently the integrity of the LCA methodology. Moreover, this selection and scaling process may prove to be cumbersome and requiring so great a deal of data and data manipulations thereby affecting its possible for selecting and assessing the key factors. More so, the assessment also depends on the geographical range used. Using scientific and provable arguments and knowledge may reduce the subjectivity associated with expert knowledge which is often seen as a good approximation.

In measuring species richness, the approach selected, be it relative or absolute, should reflect the purpose of the study. If the motivation is to protect biodiversity, then it does make more sense to give a higher value to each species in species poorer regions than in species richer regions thereby necessitating the relative approach in which the species affected is compared to the regional average species richness as proposed by Koellner (2000). Schmidt (2008a) which adopts the absolute approach however asserts that weighting between species enhances the LCIA method and that rare and threatened species should be given higher weight while invasive and common species is given lower weight which implies low priority.

Biodiversity indicator should reflect the number of indigenous species (as opposed to non-indigenous/neophytes, intentionally introduced by man) and endemic species instead of the overall species (Weidema & Lindeijer, 2001). Though the use of species richness of vascular plants is being used as a biodiversity indicator, it is worth a research effort to focus on biodiversity indicator that can incorporate all indigenous and endemic species occurring in a particular region, if data availability permits.

In order to enhance the credibility of the results provided by an LCA and the subsequent acceptability of the LCA methodology, it is necessary to have an agreed-upon method on how to incorporate land use impacts in LCA. This is very important because of the existing link between the demand or production of goods and service and land use, especially agricultural or extractive

products. It is also important to develop an LCIA method that has a wide geographical coverage because production of goods and services is becoming increasingly global.

8 References

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