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NATIONAL COUNCIL FOR AIR AND STREAM IMPROVEMENT

METHODS FOR CHARACTERIZING FOREST-RELATED LAND USE IMPACTS IN LIFE CYCLE ASSESSMENT

SPECIAL REPORT NO. 15-04

APRIL 2015

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PRESIDENT'S NOTE

Forests provide many important functions, goods, and services such as support for biological diversity, clean water, carbon storage, recreational opportunities, and the raw materials required to manufacture products that society needs and demands. Land uses, however, including the management that provides products and services from forests, have long been accompanied by dialogue about sustainability. Recently, that dialogue has expanded to include discussion about the environmental aspects of all stages in the production of goods and services and life cycle assessment (LCA) has emerged as a tool for organizing and considering relevant scientific information. As a result, there is growing recognition of the need to integrate consideration for land use impacts such as those related to forestry into LCA. However, few LCAs have addressed the environmental aspects of land use, and there is ongoing debate about approaches for doing so.

This special report discusses proposals for evaluating land use impacts in LCA, including a general framework proposed by the United Nations Environment Programme (UNEP) and the Society for Environmental Toxicology and Chemistry (SETAC). The report explains impact assessment in LCA, how evaluation of land use impacts fits in that context, the general framework proposed by UNEP-SETAC, different proposals for biodiversity and ecosystem services impact indicators for use in LCA, and challenges related to addressing biodiversity and ecosystem services through LCA. The report concludes that LCA is not currently, and may never be, suited to providing reliable site-specific assessment results concerning the complexities of biodiversity and ecosystem services associated with land use, including forest management, largely because of the complexities of biodiversity and the global and comprehensive nature of LCA. Nonetheless, there is a need to integrate consideration for land use such as forestry within life cycle approaches, potentially through the use of complementary site-specific and/or territorial assessment approaches.

Dirk J. Krouskop

April 2015

NOTE DU PRÉSIDENT

Les forêts fournissent de nombreuses fonctions importantes, des biens et des services tels que le soutien à la diversité biologique, l'eau propre, le stockage du carbone, les possibilités de loisirs et les matières premières nécessaires pour fabriquer des produits dont la société a besoin et exige. L'utilisation des terres, cependant, y compris la gestion de celles-ci pour fournir des produits et services provenant des forêts, est généralement accompagnée d'un dialogue sur la durabilité. Récemment, ce dialogue a été élargi pour inclure une discussion des aspects environnementaux de toutes les étapes de la production de biens et services et l'analyse du cycle de vie (ACV) a émergé comme un outil pour la considération et l'organisation de l'information scientifique pertinente dans ce contexte. En conséquence, il y a une reconnaissance croissante de la nécessité d'intégrer les considérations des impacts de l'utilisation des terres, tels que ceux liés à la foresterie dans les études d'ACV. Cependant, peu d'études d'ACV ont abordé les aspects environnementaux de l'utilisation des terres et les approches pour le faire sont toujours débattues.

Ce rapport spécial examine les propositions pour l'évaluation des impacts de l'utilisation des terres en ACV, y compris un cadre général proposé par le Programme des Nations Unies pour l'Environnement (PNUE) et la "Society for Environmental Toxicology and Chemistry" (SETAC). Le rapport explique l'évaluation des l'impacts en ACV, comment l'évaluation des impacts de l'utilisation des terres s'inscrit dans ce contexte, le cadre général proposé par le PNUE-SETAC, différentes propositions d'indicateurs d'impact sur la biodiversité et les services écosystémiques à utiliser en ACV ainsi que les défis liés l'utilisation de ces indicateurs en ACV. Le rapport conclut que l'ACV n'est pas, et ne sera peut-être jamais, adaptée à fournir des résultats d'évaluation des impacts reliés à l'utilisation des terres qui sont spécifiques au site et fiables étant donné la nature holistique et globale de l'ACV et la complexité même de la biodiversité et des services écosystémiques, surtout en ce qui concerne la gestion des forêts. Néanmoins, il y a tout de même un besoin d'intégrer la considération des impacts potentiels de l'utilisation des terres, telle que la foresterie, dans les approches du cycle de vie, éventuellement par l'utilisation de méthodes d'évaluation complémentaires propres au site et/ou territoriales.

Dirk Krouskop

Avril 2015

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ABSTRACT

This report discusses proposals for evaluating land use impacts in LCA, including a general framework proposed by the United Nations Environment Programme (UNEP) and the Society for Environmental Toxicology and Chemistry (SETAC). The report explains impact assessment in life cycle assessment (LCA), how evaluation of land use impacts fits in that context, the general framework proposed by UNEP-SETAC, different proposals for biodiversity and ecosystem services impact indicators for use in LCA, and challenges related to addressing biodiversity and ecosystem services through LCA. The report concludes that LCA is not currently, and may never be, suited to providing reliable site-specific assessment results concerning the complexities of biodiversity and ecosystem services associated with land use, including forest management, largely because of the complexities of biodiversity and the global and comprehensive nature of LCA. Nonetheless, there is a need to integrate consideration for land use such as forestry within life cycle approaches, potentially through the use of complementary site-specific and/or territorial assessment approaches.

KEYWORDS

biodiversity, forest products, land use, life cycle assessment

RELATED NCASI PUBLICATIONS

None.

MÉTHODES POUR LA CARACTÉRISATION DES IMPACTS DE L'UTILISATION DES TERRES RELIÉS À LA FORESTERIE EN ANALYSE DU CYCLE DE VIE

RAPPORT SPÉCIAL N[O] 15-04
AVRIL 2015

RÉSUMÉ

Ce rapport discute des propositions pour évaluer les impacts de l'utilisation des terres en analyse du cycle de vie (ACV), y compris un cadre général proposé par le Programme des Nations Unies pour l'Environnement (PNUE) et la "Society for Environmental Toxicology and Chemistry" (SETAC). Le rapport explique l'évaluation des l'impacts en ACV, comment l'évaluation des impacts de l'utilisation des terres s'inscrit dans ce contexte, le cadre général proposé par le PNUE-SETAC, différentes propositions d'indicateurs d'impact sur la biodiversité et les services ecosystémiques à utiliser en ACV ainsi que les défis liés l'utilisation de ces indicateurs en ACV. Le rapport conclut que l'ACV n'est pas, et ne sera peut-être jamais, adaptée à fournir des résultats d'évaluation des impacts reliés à l'utilisation des terres qui sont spécifiques au site et fiables étant donné la nature holistique et globale de l'ACV et la complexité même de la biodiversité et des services écosystémiques, surtout en ce qui concerne la gestion des forêts. Néanmoins, il y a tout de même un besoin d'intégrer la considération des impacts potentiels de l'utilisation des terres, telle que la foresterie, dans les approches du cycle de vie, éventuellement par l'utilisation de méthodes d'évaluation complémentaires propres au site et/ou territoriales.

MOTS-CLÉS

Utilisation des terres, analyse du cycle de vie, biodiversité, produits forestiers

AUTRES PUBLICATIONS DE NCASI

Aucune

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METHODS FOR CHARACTERIZING FOREST-RELATED LAND USE IMPACTS IN LIFE CYCLE ASSESSMENT

1.0 INTRODUCTION

Life cycle assessment (LCA) is one of several environmental evaluation tools for characterizing a comprehensive set of environmental aspects and impacts[1], positive or negative, related to products and services over their full life cycle (i.e., from raw material acquisition to final disposal). LCA can be used for many different applications, for instance to identify opportunities to improve the environmental performance of a product or a service or to compare the environmental performance of competing products.

One key feature of LCA is the requirement to consider a comprehensive set of environmental aspects when assessing product and services. This can include global environmental impacts (e.g., global warming), regional impacts (e.g., smog) and local impacts (e.g., human health effects). Impacts at the global and regional levels are typically well covered in LCA. This is not the case, however, for local impacts such as those related to land use, the reasons for which are discussed in this report. There is growing recognition that the use of land by agriculture, mining, urban development, forestry and other anthropogenic activities can lead to significant impacts on biodiversity and on ecosystem services (e.g., Milà i Canals et al. 2007a). As a result, there is broad agreement that there is a need for better assessment and understanding of land use impacts (Chapin et al. 2000; EEA 1995; FAO 1976; Oldemann, Hakkeling, and Sombroek 1991; Pimentel et al. 1995), especially in the context of the growing number of studies on the environmental aspects related to biofuels. The interest in land use-related aspects is highlighted in several international conventions such as the Convention on Biological Diversity (http://www.cbd.int/), the United Nations (UN) Convention to Combat Desertification (http://www.unccd.int/), the Convention on the Conservation of Migratory Species (http://www.cms.int/), and the Ramsar Convention on Wetlands (www.ramsar.org).

While the interest in land use impacts is clear, characterizing the environmental aspects of land use associated with goods and services can be challenging. For example, in many ecosystems, disturbance by factors such as fire, wind, and storms is a natural process (Lorimer 1977) and land management may influence ecosystems in ways that are similar to those impacts. In some cases, historical natural disturbances such as fire have been suppressed by humans, and land management may actually be substituting for natural disturbance. Thus, discerning the environmentally significant aspects related to land management undertaken to produce goods and services can be challenging. Furthermore, assessments of environmental impacts depend upon the indicators chosen and the temporal and spatial scales of analysis. As there is no general agreement on land use effect indicators, a common practice has been to report land use in surface-time (e.g., ha*year) units and land use change (transformation) in units of surface area, based on the assumption that less affected area means less negative impact. Note, however, that this practice does not allow for incorporation of perspective on impact pathways related to biodiversity and ecosystem services, and does not allow differentiation of the effects related to various occupation intensity levels.

This report discusses early and more recent proposals for evaluating land use impacts, including the general framework proposed by UNEP-SETAC[2] as well as specific indicators of biodiversity and ecosystem services that have been developed outside that process. First, the "impact assessment" step in LCA is explained, along with the evaluation of land use impacts within that step. Then, the general

[11] In this report, unless otherwise indicated, the term "impact" should be interpreted to reflect effects that may be positive or negative.
[2] In 2002, the United Nations Environment Programme (UNEP) and the Society for Environmental Toxicology and Chemistry (SETAC) launched an International Life Cycle Partnership, known as the Life Cycle Initiative (LCI), to enable users around the world to put life cycle thinking into effective practice. This initiative is known as the UNEP-SETAC Life Cycle Initiative.

framework proposed by UNEP-SETAC, and different proposals for biodiversity and ecosystem services impact indicators and related challenges are discussed.

2.0 AN OVERVIEW OF LIFE CYCLE IMPACT ASSESSMENT

The International Organization for Standardization (ISO) 14044 Standard describes LCA as a methodology for the "compilation and evaluation of the inputs, outputs and the potential environmental impacts of a product system throughout its life cycle" (ISO 2006b, p. 2). The life cycle of a product system covers extraction of raw materials required for production (tree growth and harvest, in the case of forest products) through final disposal of the product (i.e., from "cradle to grave").

LCA studies evaluate a variety of potential environmental impacts (e.g., global warming, acidification, smog, toxicity, etc.). LCA can serve multiple purposes such as educating customers and stakeholders about the environmental aspect of the industry's products and/or providing a basis for documenting improvements in these attributes over time.

LCA consists of a 4-phase methodology as illustrated in Figure 2.1 as well as the relevant ISO standards and reports (ISO 2006a, 2006b, 2012a, 2012b). The goal and scope definition is the phase of LCA where the objectives of the study, the specification of the studied system (including boundary and function), and the methods to be used in the next phases (including impact assessment methods) are defined. Two primary application approaches for LCA have been recognized: one that aims at describing the environmental impacts associated with a given system ("attributional" LCA or ALCA) and one that aims at describing how environmental loads are changed when a given system is modified ("consequential" LCA or CLCA). More details regarding ALCA and CLCA can be found in Appendix A.

Data are collected and relevant input and output flows are quantified during the life cycle inventory (LCI) phase. The life cycle impact assessment (LCIA) phase involves transforming flow information collected in the inventory phase into impact indicator values, with the objective of understanding and evaluating the magnitude of potential environmental impacts. As it is important to have a minimum understanding of how LCIA is generally performed in LCA to understand the step within LCA in which land use impacts would be characterized, LCIA principles are presented in further detail in this report. Finally, the life cycle interpretation phase aims at providing conclusions and recommendations based on the findings of other phases of the LCA.

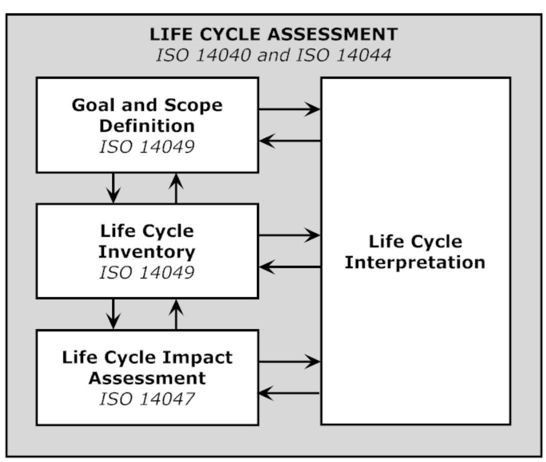

Figure 2.1 Phases of an LCA
[Adapted from ISO 14040 (ISO 2006a)]

LCA methodology has been standardized by a series of standards and supporting documents from the International Organization for Standardization (ISO), often referred to as the ISO 14040 series. ISO 14040 outlines the general principles of LCA (ISO 2006a). ISO 14044 details the requirements for its undertaking (ISO 2006b). ISO 14049 provides illustrative examples on how to define the goal and scope of an LCA and perform the inventory analysis (ISO 2012b). ISO 14048 provides the requirements and a structure for a data documentation format, to be used for transparent and unambiguous documentation and exchange of LCA and LCI data (ISO 2002). ISO 14047 provides examples to illustrate current practice in carrying out a life cycle impact assessment (ISO 2012a).

ISO 14040 proposes seven fundamental principles that should guide decisions relating to both the planning and execution of an LCA. These principles are also useful in evaluating the quality and appropriateness of existing LCA studies. The seven principles are listed below.

1) LCA should consider the **full life cycle** of a product[3] to ensure that any shifting of a potential environmental impact between life cycle stages is captured.
2) LCA addresses only the **environmental aspects** of a product.
3) LCA is a relative approach based on a **functional unit**[4]**.**
4) LCA is an **iterative** methodology in the sense that the individual phases of an LCA use results from all the other phases, and as new information becomes available for any one phase, all other phases may be revisited.
5) **Transparency** is a key to ensuring adequate interpretation of the results of an LCA.
6) LCA should consider **all pertinent environmental aspects** related to the studied product to enable identification of potential trade-offs, in particular related to a cross-media perspective (comprehensiveness).
7) Where possible, decisions to be performed in undertaking an LCA should be based on natural **science**. When it is not possible, other scientific approaches should be used. Decisions based on value choices (preferences), should be used only in cases where neither a scientific basis exists nor a justification based on other scientific approaches or international conventions is possible.

2.1 Life Cycle Impact Assessment (LCIA)

The overarching purpose of life cycle impact assessment (LCIA) is to provide additional information to help in assessing the environmental significance of a product system's input and output flows. In the life cycle inventory (LCI) phase of an LCA, inputs (i.e., resources extracted from the environment and area of land used and transformed) and outputs (i.e., releases to air, water, and soil) are quantified for the studied system throughout its life cycle. LCIA is used to evaluate the significance of these environmental interventions[5]. The primary purpose of LCIA is therefore to determine the relative significance of each environmental intervention in the context of a given environmental impact category and to aggregate these interventions into a manageable set of indicators (Owens 1999). Baumann and Tillman (2004) also suggested the following goals for LCIA:

- to make the LCI results more relevant, comprehensible, and easier to communicate (e.g., it is easier for the public to relate to the environmental impact (e.g., acidification) than to the emissions themselves (e.g., emissions of sulfur dioxide);

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[3] Products also include services.
[4] The functional unit is the "quantified performance of a product system for use as a reference unit" (ISO 2006b, p. 4).
[5] In LCA, the term "environmental intervention" is used to describe the physical interaction between a system (being studied) and the environment. This physical interaction is defined in terms of the extraction of resources; emissions to air, water, or land; space occupied by waste; or structures or area of disturbance.

- to streamline the analysis of the LCI results (the number of substances modeled in LCI is large, and the subsequent application of LCIA significantly reduces the number of parameter to evaluate); and
- to facilitate the analysis of trade-offs between substances.

When interpreting LCIA results it is of paramount importance to keep in mind that LCIA does not predict absolute or precise environmental impacts. This is because potential environmental impacts are expressed relative to a reference unit, environmental data are integrated over space and time, there is inherent uncertainty in modeling environmental impacts, and some possible environmental impacts may occur in the future.

2.1.1 LCIA Elements

The mandatory and optional elements of life cycle impact assessment (LCIA) methodology, as specified in ISO 14040, are presented in Figure 2.2. The selection of impact categories and methods, classification, and characterization are mandatory elements of LCIA. During the life cycle inventory phase of an LCA, data on resources used (including land occupied and transformed), as well as on releases to air, water, and soil (referred to as “elementary flows” in LCA) are compiled for the entire life cycle of the studied product. These data are the main input to LCIA. The first step of LCIA consists of selecting impact categories, category indicators, and characterization models that will be used to convert the LCI results into information that can be used to better understand their environmental significance. This is normally achieved through the choice of one or more existing impact assessment methods, for instance TRACI (Bare et al. 2003) from the United States Environmental Protection Agency, IMPACT 2002+ (Jolliet et al. 2003), etc. Then, each LCI result is assigned to one or more impact categories (e.g., CO_2 is assigned to global warming and CH_4 is assigned to global warming and to smog). This is referred to as classification. Once LCI results have been classified, a characterization factor is used to convert the results for each substance in each impact category to the common unit that has been defined for the impact category (e.g., for the global warming impact category, the common unit is kg CO_2 equivalents, the characterization factor for carbon dioxide is 1.00 kg CO_2 equivalent (eq.)/kg CO_2, and the characterization factor for methane is 25 kg CO_2 eq./kg of methane). Classification and characterization are usually performed using LCA software in which the impact assessment method is selected and applied. The core of LCIA is the development of the characterization factors; this is discussed in Section 2.1.5.

Optional elements of LCIA include normalization, grouping, and weighting. Normalization consists of calculating the magnitude of category indicator results relative to reference information. Examples of reference information include the total impact in a given region or the impacts of a base case scenario. Grouping consists of assigning the different impact categories into predefined sets (e.g., global, regional and local impacts; impacts of high, medium, and low importance). Finally, weighting is the process of assigning numeral factors, based on value choices, to the various impact categories that represent the relative importance of these categories. These optional elements of LCIA will not be further discussed in this document.

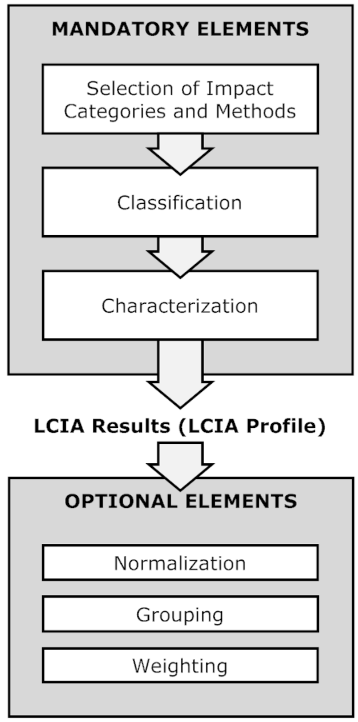

Figure 2.2 Elements of the LCIA Methodology
[Adapted from ISO 14040 (ISO 2006a)]

2.1.2 Important LCIA Definitions

To better understand how LCIA works in practice, it is necessary to understand several important concepts. These are summarized in Table 2.1 and further illustrated using the environmental impact category of "climate change" in Figure 2.3 below.

Table 2.1 LCIA Important Concepts

Concept	**Definition**
Impact category	Class representing environmental issues of concern to which life cycle inventory analysis results may be assigned (ISO 2006b, p. 5)
Category endpoint	Attribute or aspect of natural environment, human health, or resources, identifying an environmental issue giving cause for concern (ISO 2006b, p. 5)
Category indicator	Quantifiable representation of an impact category (ISO 2006b, p. 6)
Environmental mechanism	System of physical, chemical and biological processes for a given impact category, linking the life cycle inventory analysis results (often referred to as the environmental intervention) to category indicators and to category endpoints (ISO 2006b, p. 5)
Characterization model	Science-based model used to determine the characterization factor (adapted from Baumann and Tillman 2004)
Characterization factor	Factor derived from a characterization model which is applied to convert an assigned life cycle inventory analysis result to the common unit of the category indicator (ISO 2006b, p. 5)
Indicator results	Result for a specific impact category

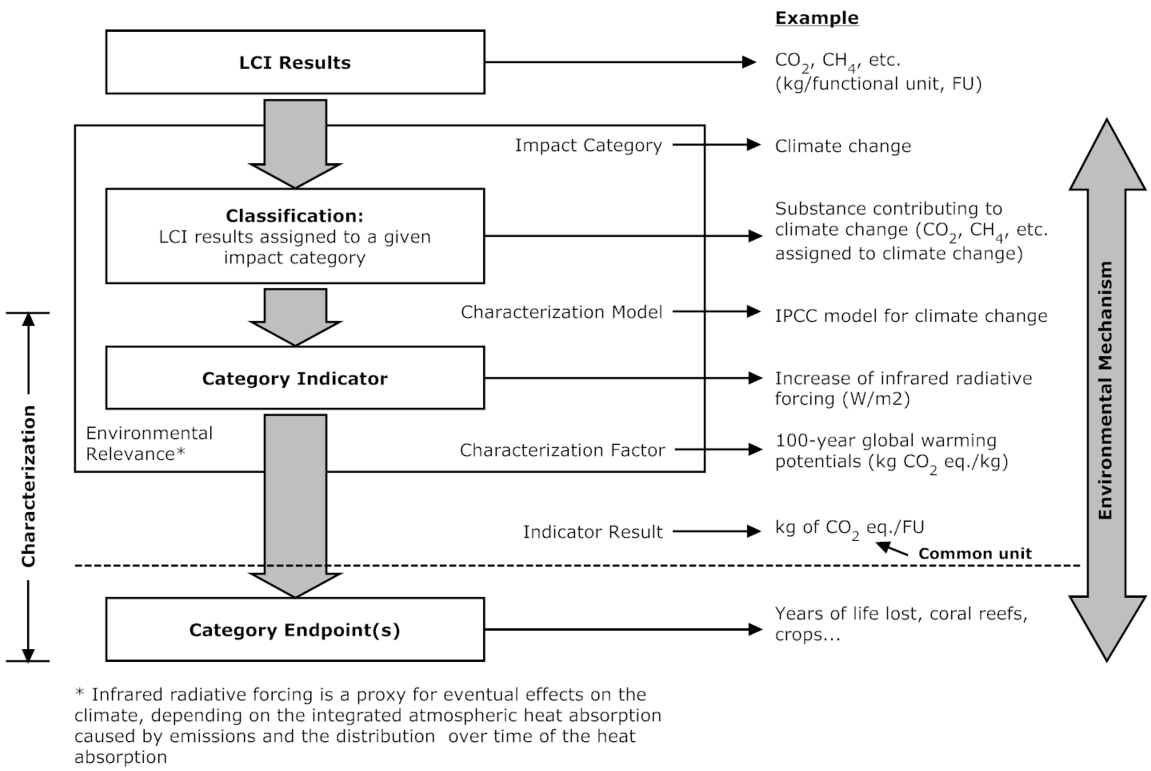

Figure 2.3 Concept of Category Indicator – Example for the Climate Change Impact Category [Adapted from ISO (2006b)]

2.1.3 Selection of Impact Categories

The selection of impact categories is most often based on an existing LCIA method. The indicator for an impact category can be chosen anywhere along the environmental mechanism between the environmental intervention and the category endpoint. LCIA methods can be classified into two groups, depending on the point in the environmental mechanism at which category indicators are defined. Midpoint methods, also called problem-oriented methods, define the category indicator somewhere along (but before the end of) the environmental mechanism (Jolliet et al. 2004), sometimes relatively close to the environmental intervention (Guinée et al. 2002a). Endpoint methods, also called damage-oriented methods, define the category indicators at the level of the category endpoints. Four endpoint categories of recognized value to society are defined in the literature. These categories are often referred as *Areas of Protection* (AoPs) or as "safeguard subjects" and include human health, biotic and abiotic natural environment, biotic and abiotic natural resources, and biotic and abiotic man-made environment (Jolliet et al. 2004).

The most common approach is for models to be applied at the midpoint level (Jolliet et al. 2004). Midpoint impact categories group together substances that have similar type of effects on the environment. For instance, it is possible to group together substances that can potentially affect the global climate into a "climate change" impact category and to group together potentially carcinogenic substances into a "human health cancer" impact category. The ISO 14047 Technical Report (ISO 2012a, p. 11) proposes a list of commonly used midpoint impact categories that are divided into "output related" impact categories and "input related" impact categories. The impact categories described as "commonly used" in ISO 14047 are listed in Table 2.2. ISO 14047 recognizes that the list may not be complete and provides some more examples, also listed in Table 2.2, for which it specifies that there are no widely accepted characterization methods.

Some categories have more than one endpoint. A conceptual framework was proposed to link midpoint and endpoint impact categories through the four areas of protection listed above (e.g., Jolliet et al. 2004). Each impact category at the midpoint level can be associated with one or more of area of protection, as characterized by one or more endpoints. The relationships between midpoint and endpoint impact categories are depicted in Figure 2.4. In this figure, the dashed lines indicate that a relationship is more uncertain.

Table 2.2 Example of Impact Categories Listed in ISO 14047

Output Related	**Input Related**
Described as "Commonly used"	
Climate change (global warming), stratospheric ozone depletion, photo-oxidant formation (smog), acidification, nutrification (eutrophication), human toxicity, ecotoxicity	Depletion of abiotic resources (e.g., fossil fuels, minerals), depletion of biotic resources (e.g., wood, fish)
Listed as other examples	
Radiation, noise and odor, working environment impacts	Land use impacts

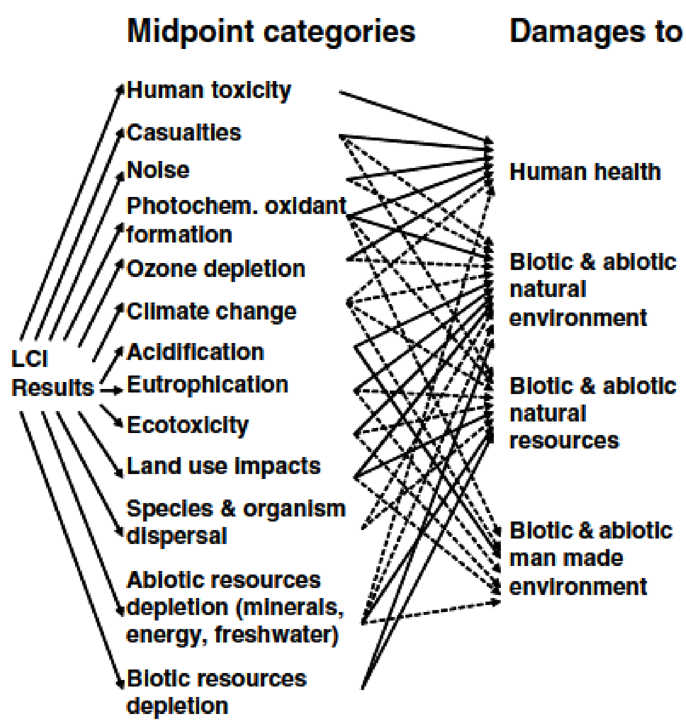

Figure 2.4 Relation between Midpoint and Endpoint Impact Categories
[From Jolliet et al. 2004]

The ISO 14040 series does not express any explicit preference for midpoint or endpoint indicators. Indeed, the ISO 14044 Standard specifies that "the category indicator can be chosen anywhere along the environmental mechanism between the LCI results and the category endpoint(s)" (ISO 2006b, p. 18) and its ISO 14047 accompanying technical report mentions that "often indicators are chosen at an intermediate level somewhere along that mechanism, sometimes they are chosen at endpoint level" (ISO 2012a, p. 4). However, a set of principles can be derived from the ISO 14040 series that are useful for performing life cycle impact assessment:

- **Comprehensiveness:** "the selection of impact categories shall reflect a comprehensive set of environmental issues related to the product system being studied" (ISO 2006b, p. 17);
- **Environmental relevance:** "the category indicators should be environmentally relevant"[6] (ISO 2006b, p. 19);
- **Transparency:** "due to the inherent complexity in LCA, **transparency** is an important guiding principle in executing LCAs, in order to ensure a proper interpretation of the results" (ISO 2006a, p. 7);
- **Priority of scientific approach:** "decisions within an LCA are preferably based on natural science" (ISO 2006a, p. 7) and "value-choices and assumptions made during the selection of impact categories, category indicators and characterization models should be minimized" (ISO 2006b, p. 19);
- **Usefulness for decision-making:** "the selection of impact categories [...] shall be [...]consistent with the goal and scope of the LCA" (ISO 2006b, p. 17); and
- **International acceptance:** "the impact categories, category indicators and characterization models should be internationally accepted, i.e., based on an international agreement or approved by a competent international body" (ISO 2006b, p. 19).

In 2000, an international workshop held under the umbrella of UNEP provided a forum for LCA experts to discuss midpoint vs. endpoint modeling. They discussed the strengths and weaknesses of the two approaches in the context of the principles listed above as well as other considerations. Their findings are summarized in Table 2.3.

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[6] Environmental relevance is defined by ISO 14044 as "the degree of linkage between category indicator result and category endpoints" (ISO 2006b, p. 18). For this reason, it is often argued that endpoint methods are more environmentally relevant as acknowledged in the ISO 14047 Technical Report that states that "typically, the environmental relevance is higher for indicators chosen later in the environmental mechanism" (ISO 2012a, p. 4).

Table 2.3 Strengths and Weaknesses of Midpoint and Endpoint Indicators

<table>
<tr><th>ISO Principles/Other Considerations</th><th>Midpoint Indicators</th><th>Endpoint Indicators</th></tr>
<tr><td>Comprehensiveness</td><td>In general, reflect a more comprehensive range of impacts, though not necessarily specified or known</td><td>In general, endpoint indicators focus on a smaller number of environmental mechanisms</td></tr>
<tr><td>Environmental relevance</td><td>Less relevant from a decision-making perspective[a]</td><td>More relevant from a decision-making perspective</td></tr>
<tr><td>Transparency</td><td>Intrinsically more transparent[b]</td><td>Intrinsically less transparent[b]</td></tr>
<tr><td rowspan="2">Usefulness for decision-support</td><td>May be preferred for scientific communication purposes
Easier to communicate</td><td>May lead to more understandable results
More valuable in those cases where aggregation is desired (i.e., where it is desirable to determine the relative importance of an impact category compared to another)</td></tr>
<tr><td colspan="2">The international workshop concluded that that both midpoint and endpoint methodologies provide useful information to the decision maker, prompting the call for tools that include both in a consistent framework.</td></tr>
<tr><td>Uncertainty</td><td>Lower model and parameter uncertainty[c]</td><td>Higher model and parameter uncertainty[c]</td></tr>
<tr><td>International acceptance</td><td colspan="2">Although not discussed in the workshop, there are probably more impact categories defined at a midpoint level for which an international agreement exists (e.g., global warming and ozone depletion) than it is the case for impact categories defined at the endpoint level.</td></tr>
</table>

SOURCE: The information in the table is based on Bare et al. (2000).
[a]Some midpoint indicators, such as global warming potential, can also be of high environmental relevance.
[b]The more complex the model, the harder it is to maintain transparency and the greater the level of required documentation.
[c]Model uncertainty reflects the accuracy of the model. Parameter uncertainty is the uncertainty associated with the accuracy of the input data.

2.1.4 Classification

Classification consists of assigning each substance extracted from, or released to, the environment to one or more impact categories. In practice, this is almost always performed through the application of an existing LCIA method. However, determining which substances contribute to specific impact categories is not always straightforward because some substances can have multiple effects according to two distinct impact pathways:

- a given substance may have **simultaneous parallel** effects, for instance sulfur dioxide is a cause of acidification but is also toxic when inhaled; and
- a given substance may have an effect that in turns generates other effects (**in series**), for instance sulfur dioxide is a cause of acidification, and acidification can cause certain metals in the soil to mobilize, increasing their toxicity.

2.1.5 Characterization

Once each substance extracted from, or released to, the environment has been classified into one or more impact categories, one needs to estimate the contribution of each substance to each impact category. This is known as characterization and is performed using characterization models. Given the complexity of the environmental mechanisms involved, different characterization models have been developed for this purpose. Several different LCIA methods have evolved [e.g., TRACI (Bare et al. 2003)], each of which uses a unique characterization model for each impact category, some of which may be the same from one method to another [e.g., most LCIA methods use International Panel on Climate Change (IPCC) models for climate change]. The evolution of different LCIA methods has, in part, reflected the need to incorporate geographic specificity. LCIA methods differ in the development of the characterization factors (derived from characterization models) for each substance in each impact category but are generally based on a similar conceptual approach, which is described in this section.

The impact score for a given midpoint impact category is calculated by multiplying, for each substance assigned to that category, the quantity of the substance and its characterization factor (Equation 2.1):

$$MI_j = \sum_{i=1}^{n} Q_i \times CF_{i,j} \quad \text{[Equation 2.1]}$$

where $\boldsymbol{MI_j}$ is the impact score for midpoint impact category j, $\boldsymbol{Q_i}$ is the quantity of substance i in the inventory results and $\boldsymbol{CF_{i,j}}$, is the characterization factor for substance i under impact category j.

Characterization factors are derived from science-based models that aim at describing, in a quantitative manner, a given potential environmental impact by reflecting the environmental mechanism that characterizes it. An environmental mechanism can be seen as the cause-to-effect chain between the inventoried substance and the category endpoint. An example for climate change is illustrated in Figure 2.3. The environmental mechanism for climate change can be further defined, for instance:

Emissions of GHGs → Increase of infrared radiative forcing → Increase of Earth's temperature → Increase of sea levels → Effects on human health and ecosystems

To establish a characterization factor for a given impact category, it is first necessary to select one of the elements in the environmental mechanism to use as a reference (i.e., a category indicator). For instance, for the climate change impact category, the increase of infrared radiative forcing (expressed as W/m^2) is often selected. Then, each substance contributing to the impact category is compared to a reference substance (i.e., a common unit) in the context of the category indicator. For the climate change impact category, the common unit is usually 1 kilogram of carbon dioxide equivalents (global warming potential, GWP, in kg CO_2 eq./kg). The potential effect of other substances classified under climate change is evaluated compared to carbon dioxide. For instance, using the International Panel on Climate Change (IPCC) 100-year characterization model for climate change (IPCC 2006b), methane (CH_4) is considered to cause an increase of infrared radiative forcing that is 25 times higher than for carbon dioxide. The 100-year global warming potential of methane is 25 kg CO_2 eq./kg CH_4. Other elements in the environmental mechanism, however, can also be used to develop a characterization factor for the global warming indicator. For instance, in its more recent assessment report, IPCC calculates a second indicator, global temperature change potential (GTP) that characterizes the impact one step further along in the environmental mechanism than GWP (IPCC 2013).

Characterization can also be applied the same way for endpoint impact categories, but is very often derived from midpoint results (Equation 2.2):

$$EI_k = \sum_{j=1}^{n} MI_j \times CF_{j,k} \quad \text{[Equation 2.2]}$$

where $\boldsymbol{EI_k}$ is the impact score for endpoint impact category k, $\boldsymbol{MI_j}$ is the impact score for midpoint impact category j and $\boldsymbol{CF_{j,k}}$, is the characterization factor for midpoint impact category j under endpoint impact category k.

Although there is currently no generally accepted method (Baumann and Tillman 2004), characterization factors for resource extraction are usually defined differently than those for output-related impact categories. For instance, they can be determined by estimating the quantity of energy that will be required in the future to extract more resources, taking into account the increased difficulty to do so.

2.2 Main Challenges in LCIA

Three main challenges concerning life cycle impact assessment are identified in the literature (Finnveden 2000; Reap et al. 2008):

- selection of appropriate impact categories;
- consideration of spatial variability; and
- consideration of the dynamics of the environmental mechanisms and the decisions on temporal horizons.

2.2.1 Selection of Impact Categories

The type of impact categories to include in an LCA is one important choice to be made. ISO 14040 and 14044 recommend that all pertinent environmental aspects related to the studied product be considered. In trying to make this recommendation operational, different groups of LCA researchers and practitioners have proposed lists of environmental impact categories to include in LCA, resulting in a lack of harmonization despite ongoing efforts at standardization. The main differences are due, in large part, to the use of midpoint versus endpoint indicators, debates on whether certain environmental impacts should be considered on their own or included within broader impact categories (e.g., should soil salinity and erosion be their own category or included within a broader land use impacts category), and debates on whether LCA should be limited to environmental impacts or should it also include impacts that are more of a socio-economic nature (e.g., human health, resource depletion) (Klinglmair, Sala, and Brandão 2014; Reap et al. 2008; Weidema, Finnveden, and Stewart 2005). The lack of standardization has resulted in the disuse of some impact categories that may be relevant to the studied product system (Finnveden 2000). The lack of inclusion of potentially relevant impact categories within LCA can also be caused by insufficient data to support a comprehensive assessment of that category, the belief that a given impact category is not relevant to the studied product, and/or the absence of that category in the LCIA method selected or LCA software tool used. Traditionally, some impact categories such as land use, toxicity and ecotoxicity, aquatic eutrophication, and photo-oxidant formation, have suffered from significant data gaps (Finnveden 2000). Of these, the toxicity and ecotoxicity impact categories are expected to be the most difficult to improve over time, despite efforts at consensus building, due to the large number of chemicals used in society and their potential synergistic effects (Finnveden 2000; Pennington 2001, Rosenbaum et al. 2008).

2.2.2 Spatial Variability and Temporal Boundary

LCA has been designed as a spatially and temporally independent tool. Indeed, the typical assumption made in LCA is that all extraction/emissions in the life cycle of the studied product occur at the same place and time. In reality, the life cycle of a product may involve unit processes that operate in different locations and it may occur over years or even decades. Unlike global impacts such as global warming, those affecting local, regional, or continental scales require spatial information to accurately portray the impacts in the receiving environments. Regionalization of impact assessment is currently an important field for LCA-related research. Environmental impacts are a function of several temporally related factors including the timing of emissions, the rate of their release and the dynamics of the environment. In addition, LCA typically involves arbitrary decisions regarding the time horizons considered in evaluating the environmental impacts.

3.0 GENERAL FRAMEWORK FOR LAND USE IMPACT ASSESSMENT

3.1 Land Use Impacts in the Context of LCA

The ISO 14044 Standard recommends that a comprehensive set of environmental issues be considered when performing LCIA. The ISO 14047 Technical Report (ISO 2012a) mentions land use impacts as a possible consideration in LCA, although the report notes that they are not commonly included due to lack of generally accepted methods. Land use impacts relate to the occupation, reshaping, and management of land for human purposes (Brentrup et al. 2002). Natural ecosystems (natural biotic environment) are considered as an area of protection (AoP) in LCA (see Section 2.1.3 above). In that context, it is important to understand that pollutant releases that arise from land use are considered in other indicators of LCIA and that “land use impacts” include only the environmental consequences associated with the land use itself, for instance through the reduction of landscape elements (e.g., by removing forests, hedges, ponds, bushes), the planting of agricultural crops or artificial vegetation (e.g., gardens), or the sealing of surfaces (e.g., for buildings or roads) (Brentrup et al. 2002).

There is a general recognition that three impact pathways should be considered when addressing land use in LCA: biodiversity, biotic production potential, and life support functions (i.e., ecosystem services mainly related to soil quality). Biodiversity impacts refer to the species composition of the land and how that is affected by land occupation and transformation. The biotic production potential of the land is of interest primarily in the context of the use of land to meet human needs. Land has a capacity to provide life support functions, for instance cycling of nutrients and water, buffering, and filtering capacities, etc. These mechanisms are mainly related to soil quality.

Land use related impacts are one of the most debated topics in the LCA community (Baumann and Tillman 2004). One reason for this is that there is limited knowledge and data on the influence of land use on the environment (Baumann and Tillman 2004). Although numerous indicators have been proposed for the three impact pathways associated with land use impacts, these indicators have rarely been developed within a consistent framework (Milà i Canals et al. 2007a). In general, the lack of consensus is due to a lack of understanding that the approach for characterizing land use impacts may vary depending on the goal of the LCA study and due to the failure to recognize that value judgments are necessary for any land use assessment methodology (Milà i Canals et al. 2007a). These value judgments are related to, for instance, the function of the land studied; the perception of ownership; indicators used to describe the land quality; assumptions regarding future or alternative usages of the land; the time horizon, including the definition of a reference situation; and the perception of the land’s recovery capacity (Milà i Canals et al. 2007a).

An additional challenge is that LCA has been developed to characterize the potential environmental impacts related to flows to and from the environment independent of spatial and temporal considerations. Land occupation and transformation are not “flows” and land use impacts are site- and time-dependent.

3.2 Conceptual Model

Land use impacts relate to both transformation processes (often referred to as "land use change" or LUC) and occupation processes (often referred to as "land use" or LU) of the land and the potential effects of these processes on biodiversity and ecosystem services[7].

Transformation occurs when land goes from one type of occupation, including no occupation, to a different type of occupation and generally results in a change in ecosystem "quality" (positive or negative)[8]. Transformation generally occurs over a relatively short period of time and thus, the temporal dimension is generally neglected.

Occupation is the anthropogenic use of the land for a specific purpose (e.g., agriculture, waste disposal, harvesting). In the early 2000s, occupation impacts were defined by LCA researchers and practitioners as "the maintenance of the [ecosystem quality] in a state different from that steady state which can be reached after the relaxation period" (Weidema and Lindeijer 2001, p. 13) or, in other words, "the prevention of potential renaturalisation" (Lindeijer et al. 2002, p. 41). Under this definition, occupation impacts are not related to the difference in quality between the beginning and the end of the occupation but rather to the difference in quality between the land as occupied and a potential quality in the absence of occupation. In fact, it is generally assumed that occupation maintains the ecosystem quality at a certain level, neglecting any difference in land quality between the start and end of the occupation process. Under that definition, one could have no impact at all on biodiversity, for instance, or one could ostensibly improve biodiversity and still be assigned a positive occupation impact (i.e., a deterioration in quality). In other words, occupation impacts are only related to the time the land is maintained at a certain quality level and the difference between this level and a reference level.

Ecosystem quality (Q) has been defined by UNEP-SETAC (Milà i Canals et al. 2007a, p. 1190) as *"the capacity of an ecosystem [...] to sustain biodiversity and to deliver services to the human society."* Ecosystem quality can be measured using various indicators discussed later in this report. Under the UNEP-SETAC approach, ecosystem quality is affected by land occupation to the extent that it is maintained at a different level than would naturally/otherwise be present, and ecosystem quality is affected by land transformation to the extent that the characteristics of ecosystems are deliberately altered. Impacts of land occupation and transformation on ecosystem quality are depicted schematically in Figure 3.1 with the severity of the impacts represented by the shaded areas.

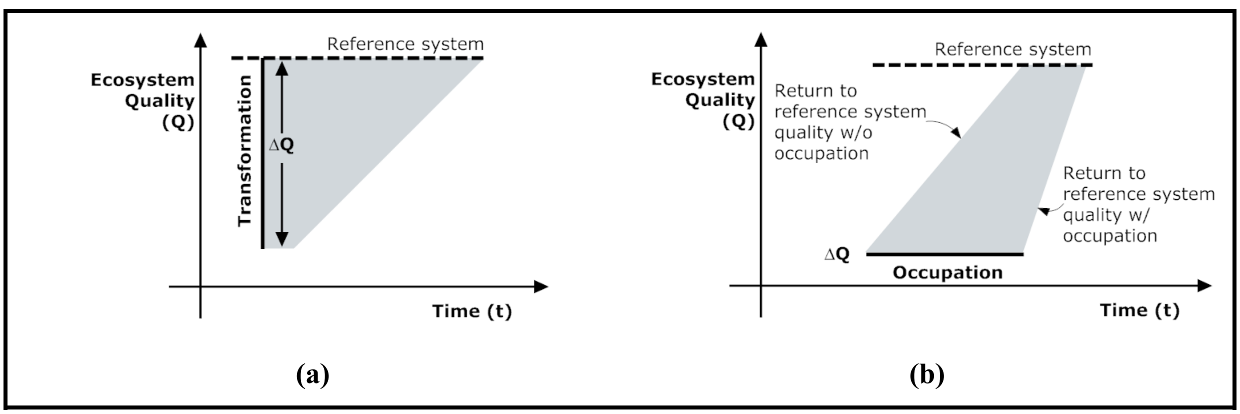


Figure 3.1 Characterization of Land Use Impacts Due to a) Transformation and b) Occupation [Adapted from Milà i Canals et al. 2007a]

[7] Other impacts (e.g., climate change impacts) can be caused by land transformation but those are generally characterized using other impact indicators.
[8] Note that characterizations of "quality" is influenced by value judgments, as discussed later in this document.

Figure 3.2 presents and explains, in a simplified way, the conceptual model for quantification of land use and land use change introduced by Lindeijer et al. (2002) and adopted by UNEP-SETAC in 2007 (Milà i Canals et al. 2007a). More complex examples can be found in Milà i Canals, Rigarlsford, and Sim (2013). As illustrated in Figure 3.1a, in cases where a transformation process would not be followed by any land occupation processes, the change of land quality (ΔQ) would be followed by a gradual natural reversal of the quality (regeneration) until it becomes roughly equivalent to the initial quality (Milà i Canals et al. 2007a; Milà i Canals, Rigarlsford, and Sim 2013). As the temporal dynamics of ecosystems are generally not well known, a linear trajectory is assumed. The regeneration time (t_{Reg}) depends on the severity of the transformation. The transformation impacts are represented by the integral of the change in quality over time or, in other words, by the shaded area between the dotted line (reference situation) and the solid vertical line. For the example illustrated in Figure 3.2, the impact on ecosystem quality from the transformation process (TI) can be calculated using Equation 3.1:

$$TI = 0.5 \times \Delta Q \times t_{Reg} \times A = 0.5\big(Q_{Ref} - Q_1\big)(t_2 - t_1)A \quad \text{[Equation 3.1]}$$

where $\boldsymbol{0.5 \times \Delta Q \times t_{Reg}}$ is the characterization factor and $\boldsymbol{A}$, the inventory parameter.

One could have elected not to consider regeneration times in computing transformation impacts. For instance, as depicted by Equation 3.2, in its Product Standard, the GHG Protocol (WRI and WBCSD 2011) defines land use change impacts on climate change as the difference in carbon stocks between two steady states, the one before transformation and the one after transformation, irrespective of the regeneration time.

$$TI_{Carbon} = \Delta CS \times A = (CS_{SS2} - CS_{SS1})A \quad \text{[Equation 3.2]}$$

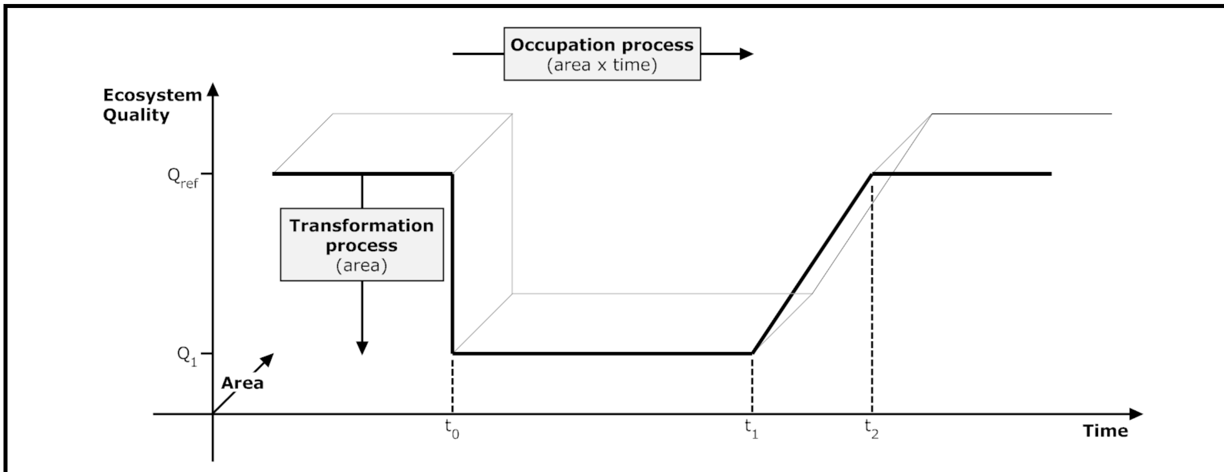
where TI_{Carbon} is the impact of land use change on climate change; *CS* the carbon stocks; *SS2* the steady state after transformation; and *SS1*, the steady state before transformation. The introduction of the regeneration time into transformation impacts allows these impacts to become additive with occupation impacts, giving more weight to transformations that take more time to return to natural conditions. The main difficulty with such an approach is that regeneration times are very uncertain and sometimes unknown (Goedkoop et al. 2013).

Occupation impacts are not due to a change in the ecosystem quality but to the fact that, as illustrated in Figure 3.1b, occupation prevents the ecosystem quality from returning to that which would occur if the land was not occupied. In other words, the natural return of ecosystem quality to a reference condition is postponed because of the occupation. The land occupation impact is calculated by subtracting the integral over time of ΔQ without occupation from the integral over time of ΔQ with occupation, and is illustrated in gray in Figure 3.1b. For the example illustrated in Figure 3.2, the impact on ecosystem quality from the occupation process (OI) can be calculated using Equation 3.3:

$$OI = \Delta Q \times t_{Occ} \times A = \big(Q_{Ref} - Q_1\big)(t_2 - t_1) \quad \text{[Equation 3.3]}$$

where $\boldsymbol{t_{OCC}}$ is the time the land is being occupied to fulfill the functional unit of the studied system, $\boldsymbol{\Delta Q}$, the characterization factor and $\boldsymbol{t_{Occ} \times A}$, the inventory parameter. For both the transformation and occupation impact indicators (TI and OI), a positive result means a deterioration of the ecosystem quality and a negative result, an improvement.

The example described below in Figure 3.2 assumes that effects on ecosystem quality from land use are fully reversible, meaning the ecosystem quality of a given reference situation can be reestablished. In practice, ecosystems modified by human activities may never return to their exact pre-activity condition, but they will probably eventually return to something essentially equivalent. For this reason, UNEP-SETAC has suggested that land use impacts generally be considered reversible with the exception of situations where regeneration times are known to exceed temporal horizons modeled in the LCA study. In these situations, impacts are considered to be permanent. UNEP-SETAC proposes to consider permanent impacts separately from the reversible ones.



Time stage	Anthropogenic intervention	Potential effect on land quality (Q)
Before t_0	No human intervention: a certain land cover is in place (e.g., natural forest) in a steady state	There is a slow natural evolution of the quality (depicted as a static state for simplicity, although natural fluctuations in Q are likely to occur)
At t_0 (transformation)	Transformation process: the land is transformed by humans to make it suitable for a new use (e.g., natural forest is harvested for the first time and converted into managed forest)	Land quality changes abruptly from Q_{ref} to Q_1 (a decrease is shown in the figure but it could be an increase depending on the actual intervention and the quality parameter)
Between t_0 et t_1 (occupation)	Occupation process : the land is used for the new use (e.g., ongoing harvesting)	Land quality gradually evolves under the new land use from Q_0 to Q_{fin} (in the figure, the quality is shown as stable but could also be increasing or decreasing depending on the actual intervention and the quality parameter)
Between t_1 and t_2 (relaxation)	No human intervention: the land quality varies naturally	Land quality changes from Q_1 to Q_2 (this can be accelerated through human restoration; the figure shows a relaxation time shorter than the occupation but it can be the opposite)
After t_2	A new land use is in place (e.g., unused land)	If land is left undisturbed, land quality probably reaches a new steady state; otherwise, quality evolution depends on the new use

Figure 3.2 Example of the UNEP-SETAC Conceptual Model for Land Use Impact Assessment [Adapted from Lindeijer et al. 2002, Milà i Canals et al. 2007a]

4.0 SCOPE AND INVENTORY ASPECTS

When developing an LCA that considers land use, it is necessary to define the reference situation, compute the quantity and quality of land being used and transformed, and allocate land transformation between multiple subsequent usages, etc. These aspects are discussed in the next paragraphs.

4.1 Land Use Elementary Flows

While land transformation causes a change in ecosystem quality, land occupation delays its recovery (e.g., Koellner et al. 2013b). Koellner et al. (2013b) provides an example where they describe the conversion of tropical forest into cropland causing a drop in biodiversity while the continuous use of such cropland prevents the regrowth of the original tropical forest. Also according to the authors, the minimum information required to perform land use impact assessment is the type of land use, its location, and timing of its use. More specifically, in the life cycle inventory of an LCA, the following information needs to be collected for land occupation:

- area and time occupied (m^2yr);
- land use type; and
- region in which the land occupation takes place.

For land transformation, the following information needs to be collected:

- area of land transformed (m^2);
- land use type before transformation;
- land use type after transformation; and
- region in which land transformation takes place.

4.2 Land Use and Cover Typology

Land use impact assessment requires a classification system or "typology" for land use and cover types. Several different land classification systems for land use impact assessment are available in the literature, with some of the more widely recognized ones summarized in Table 4.1. Also, because land use impacts can be very region-specific, for land use impact assessment to be meaningful, a regional approach is required (Koellner et al. 2013b). Several approaches are available to regionalize land use in LCA, for instance:

- the ***Holdridge Life Zone System*** that classifies world's vegetation types based on mean annual precipitation, mean annual biotemperature (which is measured as the mean of all temperatures above freezing, with all temperatures below freezing adjusted to 0 °C), and ratio of evapotranspiration to rainfall (Holdridge 1947);
- the ***Terrestrial Ecoregions of the World*** in which ecoregions are defined based on biodiversity, environmental properties, climatic conditions, and habitat diversity (Olson and Dinerstein 1998; Olson et al. 2001);
- the ***Marine Ecoregions of the World*** *(MEOW)* in which ecoregions are defined based on species composition, determined by the predominance of a small number of ecosystems and/or a distinct suite of oceanographic or topographic features (Spalding et al. 2007); and
- the ***Global Ecoregions*** developed by the World Wildlife Fund, which defines an ecoregion as a "large unit of land or water containing a geographically distinct assemblage of species, natural communities, and environmental conditions" based on work from Olson et al. 2001.

Building on existing classification schemes and regionalization approaches, the UNEP-SETAC Life Cycle Initiative has proposed a multilevel regionalized global land use and land cover classification system for LCA (hereinafter UNEP-SETAC LULC classification) summarized in Table 4.1. This multilevel classification allows for varying level of detail depending on the goal and scope of the LCA. However, UNEP-SETAC does not provide guidelines in terms of which levels to choose under which conditions.

Table 4.1 Land Classification Schemes Potentially Useful for LCA

Author(s)	**Classification Scheme**	**Usage/Features**
Outside LCA		
FAO (Di Gregorio and Jansen 2005)	Land Cover Classification System (LCCS)	Created for mapping exercises Provides a flexible framework for the description of land cover types, globally
JRC (2000)	Global Land Cover 2000 (GLC 2000)	Based on LCCS Provides, for the year 2000, a harmonized land cover database over the entire globe Divides world surface into 18 regions
European Space Agency (e.g., Arino et al. 2007)	GlobCover	Based on LCCS More recent and higher resolution than GLC 2000
Bossard, Feranec, and Otahel (2000)	CORINE	Provides a detailed classification of land in Europe as well as several types of environmental information related to the land
Alkemade et al. (2009)	GLOBIO3	Global biodiversity model to assess the impacts of land use change on terrestrial biodiversity Land cover classification based on GLC 2000, aggregated into seven broad classes
Within LCA		
Koellner et al. (Koellner and Scholz 2008a, 2008b; Köllner 2003)	CORINE Plus	Proposed for use in LCI and LCIA Builds on CORINE system but has been adapted to the context of LCA Takes into account the distinct methods applied in cropland and pasture
ecoinvent version 2.0 (Althaus et al. 2007)	Streamlined CORINE Plus	Land classification is based on CORINE Plus but with fewer land use classes
ReCiPe (Goedkoop et al. 2009)	ReCiPe	Developed their own classification scheme that includes 18 categories
UNEP-SETAC (Koellner et al. 2013b)	UNEP-SETAC LULC Classification	Multilevel classification system to be used in LCA (see more details in Table 4.2)
ecoinvent version 3.0 (Weidema et al. 2013)	Based upon UNEP-SETAC LULC Classification	See above

SOURCE: The information in the table is based on Koellner et al. (2013b).

Table 4.2 Overview of the UNEP-SETAC LULC Classification

<table>
<tr><th colspan="2">Land Use and Land Cover</th><th colspan="2">Regionalization</th></tr>
<tr><th>Level</th><th>Description</th><th>Level</th><th>Description</th></tr>
<tr><td>1</td><td>General land use and land cover classes from GLC 2000</td><td>1</td><td>Biomes</td></tr>
<tr><td>2</td><td>Refinement of level one based on ecoinvent v2.0 and GLOBIO3</td><td>2</td><td>Climatic regions</td></tr>
<tr><td rowspan="2">3</td><td rowspan="2">More information on land management (e.g., irrigated vs. non-irrigated)</td><td>3</td><td>Terrestrial and water biomes (based on Olson et al. 2001 and Spalding at al. 2007)</td></tr>
<tr><td>4</td><td>Terrestrial and marine ecoregions (based on Olson et al. 2001 and Spalding at al. 2007)</td></tr>
<tr><td>4</td><td>Specification of the intensity of land uses (extensive vs. intensive)</td><td>5</td><td>Exact geo-referenced information on land use in grid cells of 1.23 km^2 or less</td></tr>
</table>

SOURCE: The information in the table is based on Koellner et al. (2013b).

4.3 Reference Situation

As shown in Figure 3.1, land use impacts cannot be evaluated without defining a reference situation. Indeed, land use impacts are generally defined in LCA as being proportional to the difference in ecosystem quality between the studied system and a reference system. In the case of transformation processes, for attributional and consequential LCAs, the reference situation is straightforward, at least in concept, and is defined as the original steady state of the land before transformation (Lindeijer et al. 2002).

While the reference situation for occupation processes in the case of consequential LCAs is generally agreed to be the state of the land in the alternative scenario, there is much more debate pertaining to occupation impacts in attributional LCA.

Although occupation impacts were originally defined by the LCA community as “the prevention of potential renaturalisation” (Lindeijer et al. 2002, p. 41), subsequent proposals concerning the reference situation to be used for occupation processes for attributional LCAs have not always been aligned with this definition. The most commonly considered reference conditions for attributional LCA are

- the natural state of the land prior to any human intervention;
- the state of the land before the start of the occupation process; and
- some sort of regeneration potential.

It has been suggested by Lindeijer et al. (2002) that the choice of a reference state should enable the distinction between occupation and transformation processes. Occupation impacts should be defined in a way that avoids overlap with transformation impacts and which fully express the impacts not captured by transformation impacts. The three proposals above are discussed below, in this context.

4.3.1 Natural State of the Land Prior to Human Intervention

One proposal is to use the natural state of the land prior to any human intervention (e.g., Brentrup et al. 2002) as the reference situation for quantifying land use occupation impacts. The rationale for this approach is the assumption that land in its natural state is better for the environment. Using this approach, each human activity happening on the land is assigned a portion of any permanent ecosystem impact from the first activity on the land in proportion to its duration. It has been argued, however, that while the natural state of the land is relevant as a reference for the transformation impacts associated with the first activity on the land, this state is not related to occupation impacts of subsequent activities and completely ignores the potential “renaturalisation” potential from these activities (Lindeijer et al. 2002; Weidema and Lindeijer 2001).

4.3.2 Situation before the Start of the Occupation Process

Another option is to use the state of the land just before the start of the occupation process (Baitz, Kreissig, and Wolf 2000; Blonk, Lindeijer, and Broers 1997). Blonk, Lindeijer, and Broers (1997) stated that that this is the only option that characterizes changes in ecosystem quality due to the occupation process, but on the other hand, it requires potentially costly data on ecosystem attributes before and after the occupation process (i.e., not very practical), and has difficulty differentiating natural effects from anthropogenic ones. Weidema and Lindeijer (2001, p. 17) argue that this approach would

- “imply that until all human activity on the area ends, the first human activity on the area [...]would continue to be ascribed occupation impacts[...] as if it had not terminated, or these impacts would not be ascribed to any activity, while any subsequent activity would be ascribed occupation”;
- “eliminate the distinction between permanent ecosystem impacts and occupation impacts”; and
- imply that “continuation of land use as it is [...]not be ascribed any impact [, thus] ignoring the impacts due to this land occupation and prevention of relaxation.”

4.3.3 Regeneration Potential

Others have proposed a reference condition based on the regeneration potential of the land. Several variations of that concept are available in the literature.

Blonk, Lindeijer, and Broers (1997) suggest that an option to be used as the reference for occupation impacts is the situation without human intervention (i.e., the “would-be” natural situation). It is not clear whether this relates to the natural state of the land prior to any human intervention or the relaxation potential as described above. One reason for the lack of clarity might be that the authors assume that these two situations are equivalent. The authors argue that this option results in unbiased results, as the situation without human intervention is not necessarily a preferred situation, but they acknowledge that a drawback is that an activity is held responsible for degradation that happened in the past. While the concern is understandable, this aspect of the Blonk et al. approach is conceptually consistent with how occupation impacts are often characterized in that they usually hold an activity responsible for postponing the return to natural conditions. Blonk, Lindeijer, and Broers (1997) highlight that this reference situation could be difficult to implement in the absence of good knowledge of the state of the land just prior to starting an activity.

Weidema and Lindeijer (2001) propose a reference condition equal to the current relaxation potential, representing the final steady state condition following the current land occupation, if the current occupation were to be the last. They point out that if the reference condition is chosen to be a level less impacted than the current relaxation potential, the impact measured would include a part of the permanent ecosystem impact (i.e., it would overlap with transformation impacts) and if the reference condition was chosen to be a level more impacted than the current relaxation potential, the impact measured would exclude part of the occupation impact of the current land use. Relaxation/ regeneration times are discussed in Section 4.4. Weidema and Lindeijer note that the relaxation potential can be affected by human interventions during the relaxation phase and suggest that this effect be considered if it can be foreseen.

Lindeijer et al. (2002) use the term “renaturalisation potential” to describe something very similar to what was originally proposed by Weidema and Lindeijer (2001). However, they emphasize that there is ongoing discussion on how this renaturalisation potential should be defined. They give the example of biodiversity for which the renaturalisation potential has been alternatively defined as “the maximum biodiversity of the region” and the “maximum biodiversity in recent history.”

Milà i Canals et al. (2007a) argue that the relaxation or renaturalisation potential does not account for dynamics of land evolution and raises the problem of allocation between successive uses of the land. As an alternative, they propose using the natural relaxation (no human intervention) evaluated in a more dynamic way. Milà i Canals et al. (2007a) also recognize that expert judgment and modeling expertise is required to describe the evolution of land quality under natural relaxation. They suggest that the relaxation quality could be interpolated from land areas in the same region being in natural or quasi-natural state.

4.3.4 UNEP-SETAC Framework

As illustrated above in Figure 3.2, the UNEP-SETAC Conceptual Model for Land Use Impact Assessment (Koellner et al. 2013a) proposes a reference situation defined in a general enough way to be equally applicable to occupation and transformation impacts, and to both attributional and consequential LCAs. Each type of land use is judged against the relevant reference situation. If the comparison of interest is between two different land occupation processes, the reference situation becomes irrelevant because both processes are judged against the same reference situation. Transformation process impacts are computed as the difference in land quality before and after the transformation.

Koellner et al. (2013a) identify three main proposals for defining the reference state for occupation impacts:

1. the potential natural vegetation (PNV);
2. the quasi-natural land cover in the ecoregion or biome; and
3. the current mix of land uses in a given region (Koellner and Scholz 2007)

The potential natural vegetation (PNV) is the expected state of mature vegetation in the absence of human intervention. Chiarucci et al. (2010) argue that the PNV concept is not practical and overlooks vegetation dynamics. The “quasi-natural land cover” is usually used to suggest a “natural” (untouched by humans) ecosystem. In practice, “quasi-natural land cover” likely means that the reference land has some, but not all, of the features or characteristics of “natural” (untouched) land. Both the PNV and the quasi-natural land cover proposals characterize impacts in terms of regeneration potential. The third proposal, that of using a current mix of land use in a given region, instead characterizes occupation impacts in terms of how they differ from those associated with the current average land use. Koellner et al. (2013a) argue that, in the context of LCA, options 1 and 2 are probably close enough to be considered the same, although option 2 may be more practical because there are more data on potential natural land cover than data on the expected state of mature vegetation. They also argue that option 3 involves a reference situation that evolves in time, thus being impractical. For these reasons, they recommend using quasi-natural land cover, without providing guidance on what “quasi-natural” actually means. However, they recognize that in cases where the objective is to protect the current environmental conditions, the idealistic quasi-natural conditions may be of no interest.

In summary, at present, the literature suggests there is general agreement within the LCA community that reference situations for occupation impacts in attributional LCA should reflect regeneration potential. However, there is a lack of agreement on how this regeneration potential should be defined.

Also, as discussed in Appendix A, the reference state to be used in attributional LCA should be consistent with the chosen definition of attributional LCA.

4.4 Regeneration Time

Computation of transformation impacts requires that the regeneration time be estimated. Assuming that the impact is reversible, the regeneration time is that required for anthropogenic impacts to vegetation and soil to essentially disappear (Koellner et al. 2013a).The regeneration time depends on a variety of factors, including

- the impact pathway (e.g., on a given land type, or plot, it may take more time to regenerate biodiversity than to regenerate the biotic production capacity);
- the type of land transformation (e.g., it takes more time to regenerate a forest converted into urban land than converted into cropland); and
- the biogeographical conditions of the location (i.e., regeneration times are typically shorter in a warm and humid climate than in a cold or dry climate).

Regeneration times are highly variable and uncertain. Regeneration times proposed in the literature range from a few years to tens of thousands years depending on the land type and impact pathway (see for instance Table 5.3). However, different data sources give regeneration times that can vary by a factor of ten for the same land use type (Schmidt 2008). For that reason, several proposed methods for land use impacts do not include characterization factors for transformation impacts.

Also, the calculated transformation impact is very sensitive to the assumed regeneration time. For this reason, the UNEP-SETAC approach recommends that LCIA method developers provide uncertainty estimates and that sensitivity analyses be performed when applying characterization factors for transformation impacts (e.g., by using low and high estimates of regeneration times). In that context, Schmidt (2008) underlines the importance of research for better definition of regeneration times.

4.5 Allocation of Impacts for Transformation Processes

When land is transformed from one type of occupation to another, it can subsequently be used multiple times for the same purpose. For instance, a natural forest can be converted to a corn field that will produce corn for many years. In such cases, one must decide how to allocate the transformation impacts to the subsequent uses of the land. There are three main options for doing so:

- fully allocate the transformation impacts to use of the land responsible for the transformation;
- allocate the transformation impacts over a fixed amortization period, for instance 20 years, by calculating the production output over that amortization period; or
- neglect transformation impacts because they become insignificant when spread over multiple uses of the land after conversion.

IPCC recommends using an amortization period of 20 years (IPCC 2006a) for soil organic carbon emissions. The Product Standard of the GHG Protocol (WRI and WBCSD 2011; Draucker, pers. comm.) recommends various approaches for allocating changes in carbon stocks caused by land transformation, depending on the harvest period.

- For products from annually harvested crops, $1/20^{th}$ of the change in carbon stocks is attributed to the products produced from each yearly harvest for 20 years following the transformation.
- For products with a harvest period less than 20 years but greater than one year, the change in carbon stocks is distributed equally across the products made from the land

over 20 years, including those produced from wood harvested during the transformation, on a production basis.
- For products with a harvest period greater than 20 years but no greater than 50 years, the change in carbon stocks is distributed across the products made from the harvest of the land causing the transformation and products made from the first subsequent rotation period, on a production basis.
- For products with a harvest period greater than 50 years, the change in carbon stocks is fully attributed to the product that generated the land transformation.

Using amortization approaches that vary depending on the land use type has been recommended and implemented in the ecoinvent database, a public database for life cycle inventories (Frischknecht et al. 2007, Table 5.5). Koellner et al. (2013a) have also suggested a linear depreciation along the regeneration period as an option.

Koellner et al. (2013a) state that the drawback of a short allocation period is quickly losing sight of the transformation impacts and that of a long allocation period is the quasi-elimination of these impacts. They argue that there is no clear scientific amortization period and hence, recommend using 20 years. They also recommend sensitivity analyses.

The developers of the ReCiPe method (Goedkoop et al. 2013) argue that when it can be demonstrated than an infinite quantity of product can be made from the transformed land, then transformation impacts can be neglected. However, according to these authors, there are a few clear cases where it is possible to allocate transformation impacts without arbitrariness. They give the example of a mining operation where it is possible to determine a link between the production of a kilogram of ore and the area or volume of the mine. With each tonne, a number of square metres of area are converted from the existing land-use type to a mining area. Another example is landfilling, for which each additional tonne landfilled will occupy an additional area.

4.6 Modeling Transformation Impacts

Koellner et al. (2013a, p. 1197) argue that modeling transformation impacts requires the definition of a modeling period that they define as the "time over which the impacts caused by land transformation are integrated." Milà i Canals et al. (2007a) suggest that the impact on ecosystem quality should be assessed at the point where a new steady state is reached (regeneration). This would be consistent with some carbon-related standards, for instance the Product Standard of the GHG Protocol (WRI and WBCSD 2011) that provides a guideline on how to perform product carbon footprints, in which the change in carbon stocks is expressed as the difference between two steady states. The main caveat to this is that the time it takes to reach the new steady state is not a consideration. According to Koellner et al. (2013a), using the regeneration time as the modeling period for transformation impacts would introduce inconsistencies for different transformation processes and would be difficult to implement. For this reason, they recommend using a modeling period of 500 years, one of the three time frames for which IPCC calculates global warming potentials and the only one of the three long enough to accommodate longer regeneration times such as those related to temperate or boreal forest biomes.

4.7 Simplifying Assumptions

Recognizing the complexity of land use impacts characterization, Koellner et al. (2013a) recommend some simplifying assumptions:

- discrete land cover types;
- constant ecosystem quality during occupation;
- zero transformation time;
- time substitutable for space;

- ecological impacts that vary linearly with the type of intervention;
- independence of biodiversity and ecosystem services;
- no interaction between drivers of ecosystem services and biodiversity;
- regeneration being linear and independent of land history; and
- no active restoration.

5.0 EVALUATION OF IMPACTS OF LAND USE ON ECOSYSTEM QUALITY

Several aspects of ecosystem quality need to be considered when evaluating the effects of land use. Based on international treaties and declarations, Koellner (2000) first suggested the following attributes be considered as they have intrinsic or instrumental value to stakeholders: biodiversity, ecosystem functions, and natural resources. UNEP-SETAC (Koellner et al. 2013a) propose using the typology of ecosystem services of the Millennium Ecosystem Assessment (Millennium Ecosystem Assessment 2005) to structure the LCIA related to land use. Therefore, they distinguish between only two main impact pathways for which characterization factors need to be defined—biodiversity and ecosystem services—given that natural resources are now considered in LCA as an area of protection in itself, i.e., distinct from ecosystem quality. Given this context, "natural resources" are thus not discussed in this report.

The impact pathway associated with biodiversity can be considered at different organizational levels, for instance genetic diversity (genes), population diversity (species), and ecological diversity (ecosystem diversity). Methods to characterize land use impacts on biodiversity in LCA have mainly used population diversity indicators, primarily species richness (Curran et al. 2011). Some studies have included ecosystem level indicators directly. Genetic diversity is generally not used in LCA because there is a lack of information on the potential effect of land use on the genetic diversity of populations and species (Köllner 2000). UNEP-SETAC recommends characterizing biodiversity from two perspectives: the protection of global species diversity, and the functional diversity of species in ecosystems.

The second pathway relates to ecosystem services and structure consistent with the classification suggested by the Millennium Ecosystem Assessment in that it includes the impacts on biotic production potential, freshwater regulation potential, water purification potential, erosion regulation potential, and climate regulation potential. Figure 5.1 summarizes the attributes of ecosystem quality to be considered for land use impact assessment under the UNEP-SETAC approach.

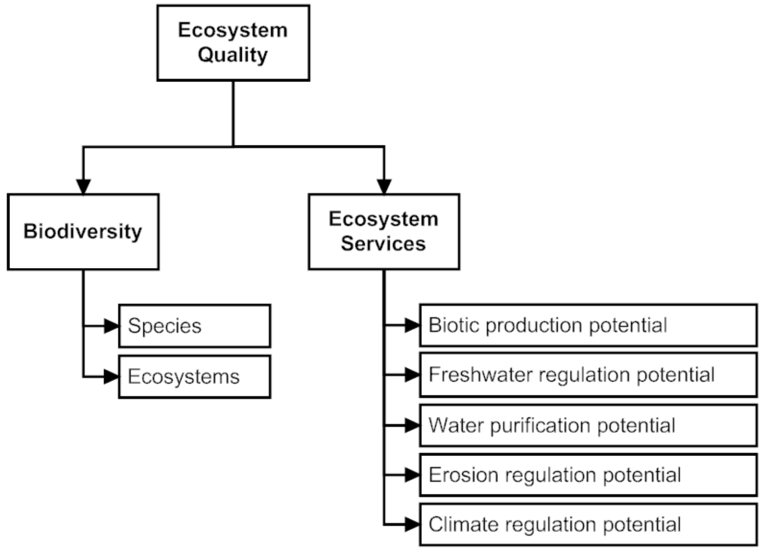


Figure 5.1 Attributes of Ecosystem Quality
[Adapted from Koellner (2000) using information from Koellner et al. (2013a)]

Table 5.1 lists various indicators that have been proposed to address the two impact pathways of land use. Some of the indicators have been proposed under the UNEP-SETAC guidelines (in gray) and some have been proposed by others (in white). The table also lists examples of indicators developed specifically in the context of forestry (in italic). These indicators are further discussed in Sections 5.1 through 5.3 and the main data and assumptions for publicly available characterization factors are subsequently summarized in Section 5.4.

Table 5.1 Indicators for Land Use Impact Assessment

<table>
<tr><th colspan="2">Proposed Impact Categories</th><th colspan="2">Description/Indicator*</th><th>Example References</th></tr>
<tr><td colspan="5">BIODIVERSITY: Impacts on global species diversity and functional species diversity</td></tr>
<tr><td rowspan="6"></td><td rowspan="6">Species diversity</td><td rowspan="6">Capacity of ecosystems to support global species diversity</td><td>Plant species richness (number of vascular plant species)</td><td>Goedkoop et al. (2013); Koellner (2000); Koellner and Scholz (2008a); Lindeijer (2000); Lindeijer et al. (1998); Schmidt (2008); Vogtlander et al. (2004)</td></tr>
<tr><td>Potential disappeared fraction (of species) (PDF.m^{2}yr)</td><td>Goedkoop and Spriensma (2001); Jolliet et al. (2003)</td></tr>
<tr><td>Biodiversity damage potential i.e., relative change in observed species richness (%)</td><td>de Baan et al. (2013); Frischknecht and Büsser Knöpel (2013)</td></tr>
<tr><td>Number of threatened species</td><td>Müller-Wenk (1998)</td></tr>
<tr><td>Species richness and abundance</td><td>Geyer et al. (2010)</td></tr>
<tr><td>Ecosystem damage potential (EDP)</td><td>Frischknecht et al. (2009)</td></tr>
<tr><td rowspan="5"></td><td rowspan="5">Ecosystem diversity</td><td colspan="2">Quality of biodiversity as an aggregated indicator of species richness, ecosystem scarcity and ecosystem vulnerability</td><td>Weidema and Lindeijer (2001)</td></tr>
<tr><td colspan="2">Biodiversity quality in terms of ecosystem scarcity, ecosystem vulnerability and conditions for maintained biodiversity</td><td>Michelsen (2008)</td></tr>
<tr><td rowspan="2">Fragmentation</td><td>Integrity index using USGS 30-meter resolution land cover adjacencies</td><td>Schenck (2006)</td></tr>
<tr><td>Land disturbance using a parameterized measure of fragmentation that relies on “edge effects”</td><td>Jordaan et al. (2009)</td></tr>
<tr><td>Capacity of ecosystems to support functional diversity</td><td>Functional diversity lost per area for a specific land cover relative to a reference land cover (%)</td><td>de Souza et al. (2013)</td></tr>
</table>

(Continued on next page. See notes at end of table).

Table 5.1 Continued

Proposed Impact Categories	Description/Indicator*		Example References
ECOSYSTEM SERVICES: Impact on global ecosystem services			
Biotic production potential (BPP)	Capacity of ecosystems to produce biomass	Back up technology (see below)	Stewart and Weidema (2005)
		Deficit in soil organic matter (SOM, Mg SOM.yr)	Baitz et al. (2000); Brandão and Milà i Canals (2013); Brandão et al. (2011); Milà i Canals et al. (2007b)
Freshwater regulation potential (FWRP)	Capacity of ecosystems to regulate peak and base flows of surface water	Water regulation capacity	N/A
	Capacity of ecosystems to recharge ground water	Ground water recharge rate (mm/year)	Baitz et al. (2000); Milà i Canals et al. (2009); Saad et al. (2013)
Water purification potential (WPP)	Chemical, physical and mechanical capacity of ecosystems to clean a polluted suspension of water	Mechanical filtration capacity measured based on soil permeability (k_f, cm/d)	Beck et al. (2010); Saad et al. (2013)
		Cation exchange capacity (mol C/kg soil)	Baitz et al. (2000); Saad et al. (2013)
Erosion regulation potential (ERP)	Capacity of ecosystems to stabilise soil and to prevent sediment accumulation downstream	Erosion resistance (ton/ha year)	Baitz et al. (2000); Beck et al. (2010); Saad et al. (2013)
Climate regulation potential (CRP)	Capacity of ecosystem to uptake carbon from air	Carbon flows (t C/ m^2yr)	Müller-Wenk and Brandão (2010)
OTHERS			
Competition	Temporary unavailability of the land because of its occupation	Quantity of land occupied (m^2yr)	Goedkoop et al. (2013); Guinée et al. (2002b)
Hemoroby	Measure of human influence on ecosystems	Naturalness degradation potential (NDP)	Brentrup et al. (2002)
Net primary productivity (NPP)	Total energy (or nutrients) accumulated by an ecological unit of interest (such as an organism, a population, or an entire community)		Lindeijer (2000); Weidema and Lindeijer (2001)
Sustainability factor	Factor to assess the sustainability of forest management practices that integrates qualitative and quantitative information on conservation of biological diversity (species, ecosystem, and genetic), maintenance and enhancement of forest ecosystem condition and productivity, and conservation of soil and water resources.		Axel Springer Verlag AG et al. (1998)

SOURCE: The table was adapted and augmented from Milà i Canals, Rigarlsford, and Sim (2013).
NOTE: In this table, UNEP-SETAC recommendations (Koellner et al. 2013a) are highlighted in light gray and the one forestry-specific indicator in italics.
*Names of the indicators are those used by the authors.

5.1 Biodiversity Indicators

A commonly cited definition of biodiversity is that of the Convention of Biological Diversity (UNEP 1992, p. 3):

> "Biological diversity means 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."

The challenge with this definition is that it makes biodiversity very difficult to quantify directly and, as a result, indirect indicators are often used, especially in LCA. Thus, the focus is often on conditions important for biodiversity. Hansson (2000) suggests that many features of ecosystems can be used as the basis for biodiversity indicators such as structural components, i.e., processes or features of the system that ensure the maintenance or restoration of diversity. The next sections of the report summarize the various proposals that have been made to compute the impact of land use on biodiversity in LCA.

5.1.1 Species Diversity

5.1.1.1 Description of Proposed Indicators

Several species diversity indicators have been proposed for use in LCA, but species richness is probably the most commonly applied (Vogtlander et al. 2004). There are a few potential reasons for this. It is often thought that species richness captures much of the essence of biodiversity and many authors have used species richness and biodiversity as synonyms. Species richness is generally well understood and measurable. Data for species richness of particular taxonomic groups (e.g., birds, mammals, and vascular plants) often exist in most regions, although data for others groups (e.g., invertebrates and microbes) are often lacking.

Several authors have suggested that, in the context of LCA, vascular plant species richness could be used as a surrogate for total species richness (e.g., Köllner 2000; Koellner and Scholz 2008a; Lindeijer 2000; Lindeijer et al. 1998; Vogtlander et al. 2004). Although plants represent only a minor part of species richness, Koellner (2000) identifies three reasons for this choice: 1) it is possible to develop a reasonable estimate of the number of plant species; 2) some research in Switzerland has shown a correlation between plant species richness and the total number of plant, insect, and spider species; and 3) it is the taxonomic group for which there are the most data. However, many studies have shown no correlation between species richness in one taxonomic group and the others (e.g., Chapin et al. 2000; Dobson et al. 1997; Lawton et al. 1998; Prendergast et al. 1993). It also has long been recognized that the total number of species, and that of vascular plant species in particular, in an area is influenced by its size (e.g., MacArthur and Wilson 1967; Preston 1962; Rosenzweig 1995). This concept is known as the species-area relationship (SAR)[9].

One assumption behind LCA is that once an area is occupied, it is prevented from returning to a reference state. Occupation is considered to be "damage" if the number of species in the occupied area is lower than the number of species in the reference situation. If the occupied area has a higher number of species, it is considered to cause a negative damage (i.e., a benefit). ReCiPe (Goedkoop et al. 2013), a commercial LCIA method, uses a *species-area relationship* (SAR) to develop a species diversity indicator (species*yr). Ecosystem response to land occupation is estimated using the

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[9] The SAR describes the rising number of species (S) present due to a rising area size (A) using the equation $S = cA^z$, where c is the species richness factor and z, the species accumulation factor (i.e., rate at which species are encountered in a system).

difference in species richness between the reference land type and the occupied area, and the occupation time—the reference land being Europe-average undisturbed woodlands. The impact of transformation activities is estimated using the difference in species richness of the area before and after the transformation, along with the assumed restoration time.

Eco-Indicator 99 (Goedkoop and Spriensma 2001) and Impact 2002+ (Jolliet et al. 2003), two other commercial LCIA methods, use the potential disappeared fraction (PDF) to characterize the potential impacts on ecosystem quality. This approach is also based on species diversity. Although the complete and irreversible extinction of species is probably the most fundamental damage to ecosystems, this would be very difficult to model in LCA because no single product life cycle could be deemed responsible for the extinction. For this reason, the PDF approach assumes that the damage caused by a given life cycle is a temporary stress on ecosystems and expresses the damage to ecosystems as the decrease of the number of species (in terms of fraction) over a certain area and time. The PDF approach can be interpreted as the fraction of species that has a high probability of extinction due to unfavorable conditions (Goedkoop and Spriensma 2001). It is based on empirical observations of the number of vascular plant species per area of land cover type. As an example, Goedkoop and Spriensma (2001) estimated the occupation impacts of extensive forestry (see definitions in Table 5.2) on a per area basis as 13.1% of that of a dump site or 9.6% of that of a permanent crop or clearcutting forestry[10]. They also estimated that managing a natural forest would have no ecosystem quality impact compared to the reference but that intensive management using clearcuts would have an ecosystem quality impact equivalent to conversion of the forest to arable land or industrial area. This would seem to indicate a lack of sensitivity of the models employed. In fact, it is interesting to note that other non-LCA studies have shown that measures of biodiversity in even-aged forests (i.e. more intensively managed forests), particularly when considered at the landscape scale, can be comparable to those of extensively managed forests (e.g., Miller, Wigley, and Miller 2009).

De Baan, Alkemade, and Koellner (2013) proposed a biodiversity damage potential (BDP) that compares the species richness of different land use types to a quasi-natural regional reference situation to calculate relative changes in species richness. Their approach includes several species groups (e.g., vascular plants, moss, and mollusks). The authors used the comparison of species richness as the basis for their approach to other indicators of biodiversity including Fisher's α[11], Shannon's entropy H[12], Sørensen's S_s[13], and mean species abundance of original species (MSA)[14]. They found that the choice of indicator strongly influenced results, and that species richness was less sensitive to land use than are indicators that consider similarity of species between the reference and the land use situations.

According to Vogtlander et al. (2004), the main advantage of approaches based on species richness is that they are relatively easy to apply. Total species richness, however, has limitations as an indicator of biological diversity. Species richness fails to account for the compositional, structural, and functional aspects of biodiversity. For example, it fails to account for taxonomy or for whether the species involved are of conservation significance, and it underestimates the conservation value of important but naturally species-poor habitats (Ferris and Humphrey 1999; Smith et al. 2008; Vogtlander et al. 2004). Müller-Wenk (1998) proposed using the ***number of threatened species*** as a

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[10] Goedkoop and Spriensma (2001) do not provide precise definitions of the different forest management regimes.

[11] Corrects for incomplete sampling by estimating "true" species richness from a sample, fitting the observed values of species richness and total number of individuals to a theoretical (empirically derived) relationship between "true" species richness and "true" number of individuals.

[12] Combines information on species abundance and richness into one number and reaches a maximum when all species occurring in a sample are equally abundant.

[13] Computes how many reference-habitat species occur in a given land use type.

[14] Assesses changes in abundance of each reference-habitat species and thus reports changes in species composition earlier than Sørensen's S_s, which only indicates a complete absence of a species from a site.

potential indicator of species diversity instead of the number of total species. Also, Geyer et al. (2010) argued that biodiversity assessment in LCA needs to capture additional aspects of biodiversity, such as abundance and evenness, and proposed weighting each species according to its rarity, i.e., lack of abundance.

Ecological Scarcity 2006 (Frischknecht, Steiner, and Jungbluth 2009), a former commercial LCIA method, has been used to assess the various types of land cover according to their plant biodiversity (Koellner and Scholz 2007, 2008a; Köllner 2003). The method used Ecosystem Damage Potential (EDP) factors, which were developed primarily from the Swiss Biodiversity Programme, based on the predicted number of species and the actual number of species found on a specific land type compared to the regional average. Positive EDP factors for land use imply that there has been ecosystem "damage" because plant biodiversity is below average and negative factors indicate "improvement" because plant diversity is above average. Land types were defined using the CORINE inventory (Commission of the European Communities 1991a, b). EDP factors were a logarithmic function of relative species richness, i.e., number of vascular plant species on occupied land relative to an average standardized number of species in the region calculated by correlating the size and species number for all local and regional plots of Switzerland. Parameters in the logarithmic function were based on results of an expert survey on the expected functional form of the relationship between biodiversity and ecosystem function (Schläpfer 1999). The LCA practitioner specified time of occupation and surface occupied. The Ecological Scarcity 2006 methodology did not include characterization factors for land transformation. In 2013, the Ecological Scarcity method was updated (Frischknecht and Büsser Knöpel 2013) and replaced with the biodiversity method proposed by de Baan, Alkemade, and Koellner (2013).

5.1.1.2 Comparison of Proposed Indicators in the Context of Forestry

In this section, the different species diversity indicators are compared in terms of their application to various forest management regimes. While some authors do not differentiate between the different forest management regimes in characterizing land uses impacts, others do. Table 5.2 presents various definitions of forest management regimes that can be found in the LCA literature.

Table 5.2 Definition of Different Forest Management Regimes

Author	Forest Management Regime Nomenclature Used in LCA Methods[a]					
	Forest, Extensive[b]	**Forest, Managed**	**Forest, Intensive (Normal)[b]**	**Forest, Intensive, Short-Cycle[b]**	**Forest, Intensive, Clear-Cutting[b]**	**Used Forest**
Brentrup et al. (2002, p. 341)	Forestry that includes only "little removal of timber, trees of different age at the same site, [old growth forest], introduction of site-atypical species possible"	Listed but not defined, described as having moderate human influence	Listed but not defined, described as having moderate to strong human influence	N/A	N/A	N/A
Eco-Indicator 99 (Goedkoop and Spriensma 2001)	Forestry which "changes land sooner or later into natural or near-to-natural state or maintains it in this state"[c]	N/A	Forestry which "changes land sooner or later into natural or near-to-natural state or maintains it in this state"[c]	Forestry which "changes land sooner or later into a non-natural state or maintains it in this state"[c]		N/A
ReCiPe (Goedkoop et al. 2013)	Monoculture broadleaf, mixed forest and woodland	N/A	Broad-leafed plantation	Mixed plantations	Mixed plantations	N/A
de Baan et al. (2013)	N/A	N/A	N/A	N/A	N/A	Forest used by humans[d]
Ecological Scarcity (Frischknecht and Büsser Knöpel 2013)	Semi-natural broad-leafed, coniferous or mixed forest	N/A	Broad-leafed, coniferous, or mixed forest	Broad-leafed, plantations	Broad-leafed, plantations	N/A

[a]In some cases, authors of LCIA assessment methods did not provide a direct definition for a given forest management regime and it was necessary to infer it from the characterization factor proposed for this regime in the SimaPro LCA software (Pré Consultants 2011). [b]Ecoinvent v2.2 (ecoinvent Centre 2010) nomenclature. Several methods adopted this nomenclature, for consistency purposes. However, they had to "force-fit" their land use category into that of ecoinvent. [c]Definition of land use intensity is based on Müller-Wenk (1998, p. 22) definitions of land use intensities. [d]Definition based on Koellner et al. (2013b).

Figure 5.2 compares the biodiversity impact of various forest land occupation processes, for which the definitions are provided in Table 5.2, relative to permanent crop for the different methods for which characterization factors[15] are readily available. Several observations can be made from this figure.

- The Eco-Indicator 99 and ReCiPe methods do not distinguish between forest management regimes in terms of biodiversity, with the exception of clearcutting, for which the biodiversity is deemed equivalent to that of a permanent crop. This lack of distinction is because the difference in observed species richness between management regimes was not significant when compared to the reference. All land occupation types are considered to result in "positive" biodiversity impacts, meaning a reduction in observed species compared to the reference.
- De Baan, Alkemade, and Koellner (2013) provide characterization factors for one type of forests only, which they identify as "used forest." However, they provide first and third quartiles for the proposed characterization factors. It can be seen from the figure that using world average characterization factors would result in high uncertainty. Even in a specific biome, the uncertainty is high.
- The Ecological Scarcity method gives a negative biodiversity impact for what is referred to as "extensive" forestry, meaning an improvement in quality. This is because extensive forestry was shown using this method to have higher species richness than average (i.e., including all land use types) in Switzerland. On the other hand, using this method, short-rotation (cycle) plantations (classified either under "clear-cutting" or "forest, intensive, short-cycle"[16] under the ecoinvent nomenclature; see Table 5.2) was the land use type for which there was the most observed impact on species richness. The results for plantation forest, however, fail to account for issues related to species composition or spatial and temporal scale. For example, species richness does not provide information about which species occur, no longer occur, or change in abundance. Species richness also may decline initially following clear-cut harvest and planting, but increase subsequently as the regenerating plantation forest matures (e.g., Dickson, Conner, and Williamson 1984). Or, plantation forest within a landscape composed predominantly of mature naturally regenerated forest may increase species richness across the entire landscape by expanding habitat available for species associated with young seral stages. Landscapes consisting of a mix of plantation forests of different age and structural conditions, and areas of lightly managed or unmanaged forest, have been documented to have high levels of bird species richness (e.g., Mitchell et al. 2006). Young forests also are important habitat for some mature forest-associated species during certain times of year. For example, during the post-fledging period, birds associated with mature forest during other times of the year can be abundant in young forests (e.g., Porneluzi et al. 2014).
- Most importantly in the context of LCA, different LCIA impact assessment methods give different ranking of land occupation types. The observed differences are mainly due to fundamental differences in methodologies applied. That said, another potential explanation for the differences observed is the different definitions used for the various forest management regimes.

[15] Characterization factors are always provided relative to a reference. Eco-Indicator 99 and de Baan, Alkemade, and Koellner use some sort of natural state, ReCiPe uses undisturbed woodlands, Ecological Scarcity uses a regional average.
[16] Not shown in Figure 5.2.

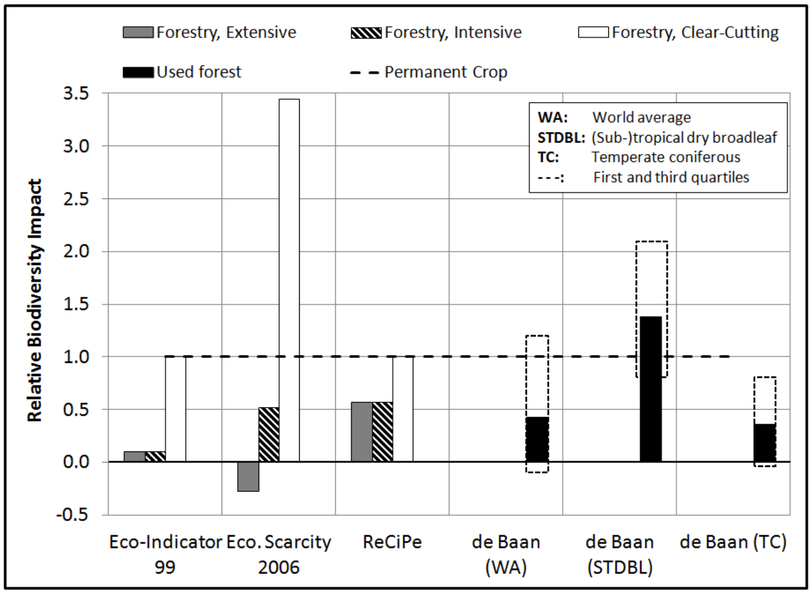

Figure 5.2 Comparison of Biodiversity Impacts from Different Land Occupation Types Obtained Using Various Proposed Impact Assessment Methods [Definition of the forest management regimes employed by each method is presented in Table 5.2. For instance, under the Ecological Scarcity method, "clearcutting" actually means "short-rotation plantation."]

As illustrated in Figure 5.3, an interesting feature of the method proposed by de Baan, Alkemade, and Koellner (2013) is that it provides characterization factors for different biomes. It can be seen from that figure that, when using this method, the greatest biodiversity impact from forestry occurs in temperate broadleaf forest, while the lowest impact is from forestry in (sub-)tropical (meaning sub-tropical and tropical) grassland and savannah. However, de Baan, Alkemade, and Koellner (2013) note that for many combinations of land use types and biomes, with the exception of (sub-)tropical moist broadleaf forests, very few data are available. Also, some LCIA method developers consider LCA as quite primitive with regards to discerning the effects of different forest management regimes from a land use impacts perspective (e.g., Goedkoop, pers. comm.).

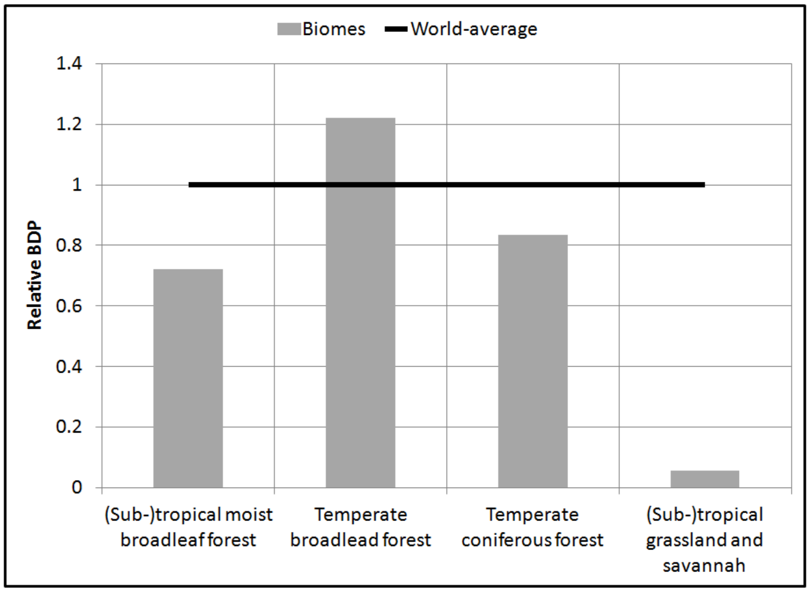

Figure 5.3 Comparison of Biodiversity Damage Potentials Related to the Occupation of Land for Forestry in Different Biomes
[Data from de Baan, Alkemade, and Koellner (2013)]

5.1.2 Ecosystem Diversity

Müller-Wenk (1998) proposed using the geographical surface of "intact scarce habitats" as an indicator of ecosystem health based on the assumptions that 1) the total surface of scarce habitats[17] in a given region, combined with an indication of the quality status of habitats, is the key element in controlling species diversity; and 2) monitoring the area in scarce habitats is easier than monitoring species.

Vogtlander et al. (2004) suggested that species richness is a weak indicator of the botanical "value" of land[18], defined, because they argue it assumes that the presence of many species indicates the probable presence of species of high conservation priority. They proposed using a combination of species richness and "rare ecosystems" but concluded that species richness is a good proxy for biodiversity in most cases.

Weidema and Lindeijer (2001) proposed assessing land use impacts on biodiversity at the ecosystem level by quantifying the biodiversity value of reference habitats of different biomes based on vascular plant species richness, ecosystem scarcity, and ecosystem vulnerability.

Michelsen (2008) suggested that existing methods for assessing biodiversity impacts based on species richness are difficult to apply, especially in the context of forestry, and because species richness is not an appropriate measure of ecosystem functioning. More specifically, Michelsen listed three challenges related to the use of species richness indicators. First, existing methods are too coarse to distinguish between different management regimes. Second, they are often based on invalid assumptions. For instance, several proposals use vascular plant diversity as the indicator of biodiversity, while numerous studies have shown no correlation between species richness in one taxonomic group and the others (e.g., Chapin et al. 2000; Dobson et al. 1997; Lawton et al. 1998; Prendergast et al. 1993). Third, not reflected in a species richness indicator is the fact that it is not

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[17] Scarce habitats are defined by Müller-Wenk as habitats whose total surface has shown a negative development during the past decades.
[18] Defined as a combination of completeness in terms of "indicator species" for that ecosystem type and rarity of the ecosystems.

only important to account for what species are present, but to maintain conditions that allow species to persist on the landscape. For these reasons, Michelsen proposed an indicator of biodiversity quality (Q) based on various aspects of biodiversity, notably ecosystem scarcity (ES), ecosystem vulnerability (EV), and the conditions for maintaining biodiversity (QMB). More details can be found in Appendix C.

The Institute for Environmental Research and Education (IERE) first proposed and tested fragmentation as an indicator of biodiversity in LCA (Schenck 2006). A rationale for using fragmentation as an indicator of biodiversity is that changes in the spatial organization of ecosystems influence the functions and services they provide (Milà i Canals et al. 2006a). Fragmentation can be defined as a process in which *"a large expanse of habitat is transformed into a number of smaller patches of smaller total area, isolated from each other by a matrix of habitats unlike the original"* (Wilcove 1987). However, several different definitions of fragmentation can be found in the literature contributing to confusion and debate regarding the effects of fragmentation (NCASI 2008).

Recognizing that the focus of proposed indicators for biodiversity in LCA has been mainly on taxonomic measures, de Souza et al. (2013) proposed an indicator of the functional diversity of ecosystems. They argued that using indicators based on species richness attributes equal weight to each species, regardless of their functional characteristics. They gave examples of how carbon storage, nutrient cycling, and biotic productivity are influenced by activities of species. Thus, functional diversity can be based on aspects of species such as feeding behavior, quantity of resources consumed, phosphorus uptake, etc. The use of functional diversity indicators reflects the fact that the loss of species can be compensated for by other species playing similar functional roles in an ecosystem. The approach proposed by de Souza et al. is based on a regional meta-analysis of richness and functional diversity of mammal, bird, and plant species associated with different land use types in the Americas (Flynn et al. 2009).

5.2 Ecosystem Services Indicators

UNEP-SETAC (Koellner et al. 2013a) recommend using the classification suggested by the Millennium Ecosystem Assessment (Millennium Ecosystem Assessment 2005) for ecosystem services. They recommend the following impact categories: biotic production potential (i.e., potential of the ecosystem to produce biomass), freshwater regulation potential (i.e., water quantity), water purification potential (i.e., water quality), erosion regulation potential (i.e., soil quality and quantity), and climate regulation potential (i.e., effect on climate by influencing the carbon sequestration in top soil and land cover).

5.2.1 Biotic Production Potential (BPP)

Although land use can have an effect, positive or negative, on the land's biological production capacity, it has been debated whether the effect of land use on biotic production potential should be considered as an environmental endpoint. For instance, on one hand, Müller-Wenk (1998, pp. 12-13) proposed it should not for the following reasons.

- He doubts that society really considers the status of the ecosystems is improved if the production of biomass is high (for instance oligotrophic conditions in lakes and rivers are favored by public voices and aimed at by laws and farmers are often encouraged to increase fallow land surfaces and reduce fertilizer application).
- He notes that a reduction of the actual biomass production on a piece of land does not necessarily mean that the future potential for biomass production on this land is lowered.

On the other hand, Milà i Canals et al. (2007a) argued that any ecological function of land not covered through economic assessments, including biotic production potential, should be covered in land use impact assessment. They suggest that biotic production is the main ecosystem service directly used by humans, and thus can be considered as the natural resource aspect of soil, ecosystem services mainly being related to soil quality. Brandão and Milà i Canals (2013) suggest that biotic production potential can be seen as referring to the conditions responsible for biological/biomass/ ecosystem productivity.

A few soil quality indicators have been proposed for ecosystem services, for instance, erosion (Cowell and Clift 2000). These generally do not refer explicitly to BPP and because there are many aspects of soil quality, an integrated indicator may be required (Brandão and Milà i Canals 2013).

Stewart and Weidema (2005) suggested the use of a functional approach to all resource-related impacts including land use. The proposed approach considers the functional values of natural resources as opposed to their intrinsic values. It is based on the concept of the quality state of resource inputs and outputs to and from a production system. A production system can 1) render a resource unavailable through use and disposal (i.e., resource is dissipated), 2) reduce its functionality, or 3) maintain or increase its functionality. In this approach, two parameters need to be defined for each resource: the ultimate quality limit and the back-up technology. The ultimate quality limit is level of quality differentiating a resource that has no possible future use from that which has a sufficient functionality to ensure that there is a potential (future) use for the resource, if retained. Stewart and Weidema (2005) proposed that this limit is determined theoretically based on thermodynamics. The alternative technology that would be applied to compensate in the future for a loss of functionality, either by extracting more of the same resource at a higher price or by regenerating the loss in quality of already extracted resources, is referred to as the back-up technology. Stewart and Weidema illustrate the back-up technology with the example of high quality mineral ores that are depleted and as a recourse, ores of lower quality must be used, requiring more effort (energy) and perhaps more land, water, and auxiliary chemicals and materials. According to the authors, using the notions of resource functionality, ultimate quality, and back-up technology makes it possible to quantify the impacts of using any resource, including land.

The most cited indicator of BPP is Soil Organic Matter (SOM) primarily due to its role in influencing soil quality and biotic production in general (Baitz, Kreissig, and Wolf 2000; Brandão and Milà i Canals 2013; Brandão, Milà i Canals , and Clift 2011; Milà i Canals, Romanyà, and Cowell 2007b). According to Milà i Canals, Romanyà, and Cowell (2007b), SOM has a role in biotic production through physical and biological fertility, in climate regulation, and in the maintenance of substance cycles; hence, some argument could be made for its use as a single indicator of ecosystem services, although some aspects of soil quality are not covered through SOM (e.g., erosion).

5.2.2 Freshwater Regulation Potential (FRP)

There is a need for freshwater indicators in LCA that reflect how freshwater availability for aquatic ecosystems is reduced due to competition with human uses, potentially leading to impacts on ecosystem quality (Milà i Canals et al. 2009). In that context, Koellner et al. (2013a) proposed using two indicators of freshwater regulation potential: water regulation capacity and ground water recharge rate. Water regulation capacity is proposed as a description of the capacity of ecosystems to regulate peak and base flows of surface water. No further information is provided in the paper. Groundwater recharge rate was proposed by Baitz, Kreissig, and Wolf (2000); Milà i Canals et al. (2009); and Saad, Koellner, and Margni (2013). It is measured in millimeters of water recharged annually and represents the soil's ability to recharge groundwater in order to regulate peak flow through the magnitude of runoff and aquifer recharge (Saad, Koellner, and Margni 2013).

5.2.3 Water Purification Potential (WPP)

Water purification potential (WPP) is the chemical, physical, and mechanical capacity of ecosystems to clean polluted water. Physicochemical filtration, or cation exchange capacity, are potential impact indicators for WPP (Baitz, Kreissig, and Wolf 2000; Saad, Koellner, and Margni 2013). Cation exchange capacity represents the soil's ability to act as a sorption matrix and to adsorb dissolved substances, and is measured in moles of cation fixed per kilogram of soil. Beck et al. (2010) and Saad, Koellner, and Margni (2013) also proposed mechanical filtration as another potential impact indicator for WPP. Mechanical filtration represents the soil's capacity to mechanically clarify a suspension through soil infiltration to ensure groundwater protection. It is measured as the rate of water passage over a given amount of time.

5.2.4 Erosion Regulation Potential (ERP)

Erosion regulation potential (ERP) is the capacity of an ecosystem to stabilize soil and prevent sediment accumulation downstream. Erosion resistance has been proposed as an indicator of ERP (Baitz, Kreissig, and Wolf 2000; Beck et al. 2010; Saad, Koellner, and Margni 2013). It represents the ability of a terrestrial ecosystem to withstand soil loss through erosion and is measured in tonnes of soil eroded per hectare per year (i.e., lower values mean higher erosion resistance).

5.2.5 Climate Regulation Potential

UNEP-SETAC (Koellner et al. 2013a) recommends the capacity of an ecosystem to remove carbon from air as an indicator of climate regulation potential, to characterize these impacts of land use in LCA. This recommendation is based on the work from Müller-Wenk and Brandão (2010).

Müller-Wenk and Brandão (2010, p. 172) argue that "compared with the potential natural vegetation [...], areas getting transformed by man (land transformations) as well as areas forced to maintain their current non-natural state (land occupations) may store reduced amounts of carbon in soil and vegetation, whereby the mobilized carbon is essentially transferred to the atmosphere in form of CO_2, contributing to global warming." They propose determining the magnitude of this climate impact by calculating the amount of carbon transferred to the air per hectare, as well as by the duration of the carbon's presence in air.

Increasingly, climate impact from land use change (i.e., transformation) is addressed within the "global warming" indicator of LCA (e.g., ISO 2013; WRI and WBCSD 2011)[19]. However, because the Millennium Ecosystem Assessment described climate regulation as an aspect of ecosystem services, it has been proposed by UNEP-SETAC (Koellner et al. 2013a) that it should be included in assessing land use impacts. Including change in carbon stocks from land use change both in the global warming indicator and under land use impacts would be double-counting. The main difference between including carbon stock change in the global warming indicator versus including it under land use impacts is that, in the case of land use impacts, the flows of carbon are weighted with the theoretical regeneration time of the land, which is not the case when carbon stocks are considered within climate change. Another difference is that, internationally accepted carbon footprint standards (e.g., ISO 2013; WRI and WBCSD 2011) typically do not consider climate impacts from land occupation, at least not explicitly. The indicators discussed here do not address other climate-related impacts of land use, such as impacts on albedo (the reflectivity of the land) and evapotranspiration.

[19] There is ongoing research to develop methodologies and models, and to generate data for the inclusion of soil carbon change in greenhouse gas reporting (ISO 2013). Hence, change in soil carbon is often neglected.

5.3 Other Land Use Impact Indicators

In this section, we describe other land use impact indicators that cannot be directly described under biodiversity or ecosystem services mainly because they have been proposed as a surrogate for land use impacts in general or as aggregate indicators of various aspects of land use.

5.3.1 Competition

The easiest way to compute land occupation impacts is to use the area of land affected by the activity. This approach assumes that all types of land use are equivalent (Heijungs, Guinée, and Huppes 1997). This has been described as a form of competition for land, meaning that when land is used for a given purpose, it is temporarily lost for other purposes (Goedkoop et al. 2009; Guinée et al. 2002b). The main problems with this approach are that it neglects to consider that uses of different land types have different impacts on the environment (Müller-Wenk 1998), and that not all land uses are mutually exclusive.

5.3.2 Hemoroby

Hemoroby is a simple concept that attempts to relate land use to habitats, and habitats to biodiversity. Hemoroby has been defined as a measure of human influence on ecosystems or, in other words, the deviation from naturalness. Using this concept, a given land use is assigned a hemoroby score (the naturalness degradation potential, NDP) that varies from 0 to 1, with 0 for land with no human influence and 1 for dedicated human use of the land. For instance, the NDP of an “intensively” managed forest is 0.40, while that of an “extensively” managed forest is 0.20 (Brentrup et al. 2002). That of a natural forest would be 0. Definitions of forest management regimes according to Brentrup can be found in Table 5.2.

5.3.3 Net Primary Productivity (NPP)

Net Primary Productivity (NPP) is the rate at which plants incorporate atmospheric carbon into biomass through photosynthesis. NPP has been proposed as a surrogate for several aspects of land use impact.

Weidema and Lindeijer (2001) argue that net primary productivity (NPP), defined as the net carbon uptake of the ecosystem (i.e., fixation through photosynthesis minus losses through respiration), “appears to be a reasonable mid-point indicator for the impact of altered species composition and population volumes on biotic resources, potential for agriculture, and life-support functions of natural systems” (Weidema and Lindeijer 2001, p. 21). Lindeijer (2000) has suggested using free net primary productivity (fNPP), defined as NPP minus the amount of carbon sequestered for human use, as an indicator for ecosystem services. According to Weidema and Lindeijer (2001, p. 22) the fNPP form of the indicator relates more to biodiversity and ecosystem services because it is a measure of “the amount of biomass nature can apply freely for its own development.” However, the nature of the relationship between productivity measures, such as NPP, and biodiversity is still debated, and depends heavily on spatial scale, species life history traits, and the measure of productivity used in the analysis (Evans, Greenwood, and Gaston 2005; Phillips, Hansen, and Flather 2008; Storch, Evans, and Gaston 2005). Weidema and Lindeijer (2001) proposed using the overall NPP as an indicator of ecosystem services. NPP is an interesting indicator in that man-made ecosystems may have a higher NPP than natural ecosystems, due for instance to fertilization, irrigation, and other management practices. The implied assumption is that “managed ecosystems can have a higher ‘nature value’ than natural systems” (Weidema and Lindeijer 2001, p. 22). The reason for that is that the “nature value” these authors try to capture with the NPP indicator is primarily related to the effect of vegetation on climate and substance flows. According to Weidema and Lindeijer, one limitation of NPP is that it may not reflect the effect of harvesting appropriately because an ecosystem that is repeatedly harvested by removing the majority of above-ground biomass may still have high NPP, while important ecosystem services, for instance, the influence of vegetation on wind speed, interception of precipitation, evapotranspiration, and albedo, may be affected immediately after harvest and until an adequate plant cover is re-established. Furthermore, ecosystems with low productivity can be species rich and can support rare species, and thus can be of high conservation priority.

Net Primary Productivity (NPP) and free NPP have also been proposed as indicators of soil quality and its biotic production potential (Lindeijer 2000; Weidema and Lindeijer 2001). However, the main shortcomings of using NPP as an indicator of soil quality are that it is affected by several factors other than the soil itself (Burger and Kelting 1999) and that low-quality soils may have a high productivity due the type of management applied (Bouma 2002).

5.3.4 Sustainability Factor

Axel Springer Verlag AG, Stora, and Canfor (1998) proposed an approach to evaluate land use impacts in LCA that builds on the assumption that the more sustainable forest management practices are, the less damage to the ecosystem is caused. The approach includes criteria and indicators from forest certification programs [e.g., that from the Forest Stewardship Council (FSC), or the Canadian Standards Association – Sustainable Forest Management Standard (CSA)], excluding social and economic aspects. More specifically, the approach includes the following three criteria: conservation of biological diversity (species, ecosystem, and genes); maintenance and enhancement of forest ecosystem condition and productivity; and conservation of soil and water resources.

In this context, the proposed approach, although published earlier than the recommendations of UNEP-SETAC discussed above, is generally consistent with them. One exception is that it actively excludes any climate regulation aspects of ecosystems because they are already included in the climate change indicator.

Axel Springer Verlag AG, Stora, and Canfor (1998) used an indirect approach to the qualification of the three criteria listed above because a direct approach would require counting species, populations, trophic levels, or ecosystem types; and measuring genetic variability, soil compaction, or nutrient balances of specific sites. These measurements are difficult to make. Instead, the indirect approach attempts to measure factors that are known to affect habitats and species, including reduction of habitat area, fragmentation and habitat degradation. The proposed approach recognizes that ecosystem quality can only be properly measured at the landscape level. Also, instead of quantifying habitat metrics within the landscape, it attempts to evaluate ongoing management plans based on their potential effects on ecosystem quality. The authors proposed this approach because they note that an adequate direct quantification of habitat is difficult and tends to reflect past land history rather than present forest management.

Based on that, the authors defined the sustainability factor (SF) as follows:

$$SF = QLSF \times QTSF \times AF \quad \text{[Equation 5.1]}$$

where *QLSF* is the qualitative sustainability factor; *QTSF*, the quantitative sustainability factor and AF, the area factor.

Qualitative sustainability aspects cover three indicators—conservation of biological diversity (species, ecosystem, and genetic), maintenance of forest ecosystem conditions and soil productivity, and conservation of water resources—through 13 indicators. It is based on responses to questionnaires. The qualitative assessment is based on a scale from good to bad (A to E), where A is for forestry best practices, B for better than industry average but not best practice, C for current industrial average forestry, D for obsolete or almost obsolete forestry, and E is reserved for entirely unsuitable practices leading to forest destruction. The authors proposed that impacts of forestry on these indicators vary exponentially with a decreasing level of sustainable management. The actual score attributed to each letter would have to be assigned by forestry experts. For instance, a score of A could be assigned a numerical value of 1, B a value of 2, C a value of 3, and so on. The quantitative sustainability aspects cover the rate at which wood is harvested and the rate at which it re-grows. The area factor considers how much of the original forest has been lost to the creation of roads, landings etc.

5.4 Summary of Data Used in Developing Ecosystem Quality Characterization Factors

Table 5.3 summarizes the publicly available data, assumptions, and references used to develop the main characterization factors for ecosystem quality. In theory, characterization factors provided in the methods listed below could be further regionalized by collecting equivalent data specific to the geographical context of the system studied in the LCA. In practice, however, this can be difficult to achieve, especially in the case of forest products, as wood used in a given product can come from several different locations for which site-specific information may not be available.

Table 5.3 Data Used, Assumptions, and References Used in Developing Publicly Available Characterization Factors for Ecosystem Quality

Method/Indicator	Data and Main Assumptions Used in Developing the Characterization factors	Reference Situation	Assumed Restoration Times: Land Type	Assumed Restoration Times: Years
BIODIVERSITY				
ReCiPe (Goedkoop et al. 2013), species richness based on SAR	Three important sources of information have been used to compute the SAR parameter values[a]. Crawley and Harral (2001) provide an in-depth analysis of the variability of species accumulation curves at different land area sizes in the UK and provide data for both for the z values and the c values, but the number of land use types is rather limited. The UK's Countryside Survey 2000 (Haines-Young et al. 2000) gives species counts for land use types in the UK, for different area sizes, with a good separation of main land use types, but does not provide values for z and c. Koellner (2001) gives c values for a wide range of land use types in Switzerland, but assumes a uniform value for z, irrespective of the land use type. ReCiPe offers three sets of characterization factors based on three cultural values perspective: individualist (I)[b], hierarchist (H)[c], and egalitarian (E)[d]. The main assumptions for these three perspectives are - **I:** regional effect assumption B and mean restoration times with a maximum of 100 years; - **H:** regional effect assumption A, assuming that fragmentation is an overall problem that requires attention, and mean restoration times; and - **E:** regional effect assumption A and maximum restoration times.	Extensive Broadleaf, mixed and yew LOW woodland	Vegetation of arable land, pioneer vegetation	2.2-2.2-5[e]
			Species-poor meadows and tall-herb communities, mature pioneer vegetation	11-11-25[e]
			Species-poor immature hedgerows and shrubs, oligotroph vegetation of areas silting up, relatively species-rich marshland with sedges, meadows, dry meadows and heath land	35-35-50[e]
			Forests quite rich in species, shrubs and hedgerows	100-100-200[e]
			Low and medium (immature) peat bogs, old dry meadows and heath land	100-450-1,000[e]
			High (mature) peat bogs, old growth forest	100-3,300-10,000[e]

(Continued on next page. See notes at end of table).

Table 5.3 Continued

Method/Indicator	Data and Main Assumptions Used in Developing the Characterization Factors	Reference Situation	Assumed Restoration Times	
			Land Type	**Years**
BIODIVERSITY				
Eco-Indicator 99 (Goedkoop and Spriensma 2001) and Impact 2002+ (Jolliet et al. 2003), Potential disappeared fraction(PDF)	The PDF is used to express the effects on vascular plant populations from land use. Vascular plants are considered representative of the total ecosystem. PDFs are based on empirical data such as observation of species numbers for different types of land cover. All vascular plants are considered to be of the same value. The PDF fraction of species is expressed as the relative difference between the number of species under the reference conditions and under the conditions created by land conversion or maintained by land occupation. The species richness factor of the SAR from Köllner (2000) is used as a proxy for species numbers in different land use types. Local and regional effects are considered.	Conversion: land before conversion Occupation: Swiss lowlands as a proxy for the natural state	From agricultural to urban areas and vice-versa	5
			From natural areas to urban or agricultural	30, unless obvious it is longer
Biodiversity damage potential (de Baan et al. 2013; Frisch-knecht and Büsser Knöpel 2013), relative species richness	Ecosystem is expressed as relative species richness. Two data sources are used to quantify biodiversity of different land use types and reference situations for different world regions: the GLOBIO3 database (Alkemade et al. 2009) and the national biodiversity monitoring data of Switzerland (BDM 2004). The relative presence or absence of species within a sampling area is assessed and an equal weight is given to all species recorded in a sample, no matter how abundant or biologically distinct they are.	Site-specific (semi)-natural habitat (defined as the current, late-succession habitat stages)	None. Transformation impacts not included.	
Functional diversity (de Souza et al. 2013)	The method proposed is based on species richness and functional diversity data across land use intensification gradients compiled by Flynn et al. (2009) and Gibson et al. (2011). Three taxonomic groups are included: mammals, birds, and plants. The functional diversity index is computed using presence-absence information for each species (Petchey and Gaston 2002).	Most natural or close to natural states are present in their dataset (i.e., land use types that represent a stable climax community representative of the PNV of the study region)	None. Transformation impacts not included.	

(Continued on next page. See notes at end of table).

Table 5.3 Continued

Method/Indicator	Data and Main Assumptions Used in Developing the Characterization Factors	Reference Situation	Assumed Restoration Times	
			Land Type	**Years**
ECOSYSTEM SERVICES				
Biotic production potential (Brandão and Milà i Canals 2013), deficit in soil organic matter (SOM)	To determine the average reference SOM in the different biomes or climate regions, a weighted average is applied to the values associated with the different soil types proposed by IPCC (IPCC 2006a) within each climate region, which reflects the share of those soil types in each climate region. The estimation of the change in SOM due to land use is also based on IPCC (2003, 2006a).	(Quasi-) natural land cover predominant in global biomes and ecoregions	All	20
Freshwater regulation potential (Saad et al. 2013), ground water recharge rate Water purification potential (Saad et al. 2013), soil's ability to act as a sorption matrix and to adsorb dissolved substances Erosion regulation potential (Saad et al. 2013)	Modeling is based on the LANCA model (Beck et al. 2010), which was calibrated using data from the Harmonized World Soil Database (FAO/IIASA/ISRIC/ ISS-CAS/JRC 2009), Olson et al. (2001) and the U.S. Geological Survey Earth Resources Observation and Science (EROS) Center (2004).	Potential natural vegetation	World Generic	85
			Tropical & Subtropical Moist Broadleaf Forests	55
			Tropical & Subtropical Dry Broadleaf Forests	55
			Tropical & Subtropical Coniferous Forests	78
			Temperate Broadleaf & Mixed Forests	97
			Temperate Conifer Forests	96
			Boreal Forests/Taiga	86
			Tropical & Subtropical Grasslands, Savannas & Shrublands	57
			Temperate Grasslands, Savannas & Shrublands	95
			Flooded Grasslands & Savannas	63
			Montane Grasslands & Shrublands	138
			Tundra	97
			Mediterranean Forests, Woodlands & Scrub	140
			Deserts & Xeric Shrublands	80
			Mangroves	52

(Continued on next page. See notes at end of table).

Table 5.3 Continued

Method/Indicator	Data and Main Assumptions Used in Developing the Characterization Factors	Reference Situation	Assumed Restoration Times	
			Land Type	Years
ECOSYSTEM SERVICES				
Climate regulation potential (Müller-Wenk and Brandão 2010)	The estimates of terrestrial carbon stocks (globally aggregated values by biome in megagram or tonne of carbon) are from IPCC (IPCC 2001, Table 3.2). Other data sources are used to estimate relaxation times and imputable mean carbon stay in air for main types of transformation in main biomes (e.g., IPCC 2000).	Potential natural vegetation	Relaxation after tropical forest to cropland	62
			Relaxation after tropical forest to pastureland	65
			Relaxation after tropical forest to artificial	62
			Relaxation after temperate forest to cropland	74
			Relaxation after tropical forest to pastureland	74
			Relaxation after tropical forest to artificial	74
			Relaxation after boreal forest to cropland	238
			Relaxation after boreal forest to pastureland	133
			Relaxation after boreal forest to artificial	238
			Relaxation after tropical grassland to cropland	97
			Relaxation after tropical grassland to artificial	97
			Relaxation after temperate grassland to cropland	110
			Relaxation after temperate grassland to artificial	110

[a]The SAR describes the rising number of species (S) present due to a rising area size (A) using the equation $S = cA^z$ where c is the species richness factor and z, the species accumulation factor (i.e., rate at which species are encountered in a system).
[b]The individualist perspective is based on short-term interest, impact types that are undisputed, and technological optimism as regards human adaptation.
[c]The hierarchist perspective is based on the most common policy principles with regards to time frame and other issues.
[d]The egalitarian perspective is the most precautionary in that it takes into account the longest time frame, and impact types that are not yet fully established but for which some indication is available, etc.
[e]For individualist, hierarchist, and egalitarian, respectively.

6.0 LAND USE IMPACT ASSESSMENT IN LCA: THE CHALLENGES

6.1 Challenges Related to the LCA Framework

The methodological framework of LCA poses particular difficulties for integrating biodiversity considerations. LCA aims to cover the entire life cycle of a product or service, but information on where and when environmental interventions occurred is often partially or entirely missing. Impact characterization is therefore typically generic in space and is summed across a time horizon. Moreover, impacts from emissions and resource consumption are linked to a functional unit[20] and this contrasts with other methods developed to assess the potential impacts of a specific project or chemical localized in space and time. Thus, other tools may be more useful than LCA for some types of evaluations (Milà i Canals et al. 2007a).

Existing land use LCIA methods were mainly developed for one specific region (often Europe) and most frequently use species richness of vascular plants as an indicator (Curran et al. 2011; de Baan, Alkemade, and Koellner 2013). Plants are an important component of terrestrial ecosystems but they only make up an estimated 2% of all species (Heywood, Watson, and UN Environment Programme 1995) and the effects of land use on plant communities is not necessarily representative of the potential impacts on other species groups. Although de Baan, Alkemade, and Koellner (2013) made a significant step forward from what had been proposed up until then by addressing these challenges, they still conclude that “the presented characterization factors for BDP [Biodiversity Damage Potential] can approximate land use impacts on biodiversity in LCA studies that are not intended to directly support decision-making on land management practices. For such studies, more detailed and site-dependent assessments are required.”

In this context, recent efforts and initiatives such as the project for land use impacts on biodiversity and ecosystem services in LCIA within the UNEP/SETAC Life Cycle Initiative (LULCIA, www.pes.uni-bayreuth.de/en/research/projects/LULCIA) and the LC_IMPACT project[21] demonstrate the growing interest and research activity around geographically differentiated LCIA. It is therefore expected that the next generation of LCIA methodologies, such as IMPACT World+ (http://www.impactworldplus.org/en/), will systematically address spatial differentiation and include uncertainty information that encompasses both spatial variability and model uncertainty. This will allow application of more environmentally relevant characterization factors by addressing regional assessment of geo-referenced emissions, although the resolution of these characterization models will remain too coarse to perform site-specific assessment.

6.2 Challenges Related to the Intrinsic Complexity of Biodiversity

Curran et al. (2011) found serious conceptual shortcomings in the way models of biodiversity change are constructed: (1) scale considerations are largely absent, (2) there is a disproportionate focus on indicators that reflect changes in compositional aspects of biodiversity (usually as changes in species richness), (3) functional and structural attributes of biodiversity are largely neglected, (4) taxonomic and geographic coverage is problematic because the majority of models are restricted to only one or a few taxonomic groups and geographic regions, and (5) only three of the five drivers of biodiversity loss identified by the Millennium Ecosystem Assessment are included in current impact categories (habitat change, climate change, and pollution), while two are not (invasive species and overexploitation).

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[20] Quantified performance of a product that is used as a reference unit in LCA.
[21] LC_IMPACT project www.lc-impact.eu/about-lc-impact; supported by the European Commission’s 7th Framework Programme for Research; involves more than a dozen organizations (including research centres and industry).

In addition to the shortcomings listed by Curran et al. (2011), there are other significant weaknesses associated with current biodiversity metrics used in LCA because of assumptions around (1) species richness, (2) reference land, (3) the directionality of biodiversity response to land use, (4) interpretation of fragmentation metrics, and (5) use of single biodiversity metrics.

Total species richness provides no information on which species are present, and biodiversity is not a collection of similar species that responds to habitat changes in a linear manner (Failing and Gregory 2003). Assuming linearity can lead to oversimplification of biodiversity metrics used to assess the impact of a specific land use action on areas of protection (AoPs).

Occupied/transformed land is compared to reference land using biodiversity response metrics in most of the proposed approaches. However, natural disturbances such as fire, windstorms, ice storms, alluvial processes, and landslides are important processes in ecosystems (Pickett and White 1985; White 1979) and usually lead to forest landscapes that are a dynamic mosaic of forest ages and conditions. Each patch in the mosaic can be characterized by a somewhat unique but potentially overlapping assemblage of fauna and flora, and this makes it difficult to determine what constitutes appropriate reference land. LCIA method developers and LCA practitioners therefore need to take great care when choosing reference land, and it may be necessary to consider a range of landscape conditions rather than a single ecosystem state as a reference in LCIA methodologies for biodiversity.

Many biodiversity assessment methodologies used in LCA have a uni-directional focus on loss, damage, and extinction. However, the interconnection between landscape components and biodiversity is highly complex, and in many cases can be multi-directional; this can make indicators difficult to interpret within an LCA context. For example, forest disturbance can have negative or positive effects on indicators related to species groups, landscape heterogeneity, and species diversity (Huston 1999; McWethy, Hansen, and Verschuyl 2010; Robbins et al. 2006) and there can be a wide variation in the temporal dynamics of effects (Grime 1973; Huston 1999). Other considerations are that many species are adapted to or rely upon forest conditions that develop following disturbance (e.g., Kimmins 2003; Litvaitis 2001), and estimation of biodiversity measures can vary depending on the spatial and temporal scale of analysis.

Responses of species to common fragmentation metrics (e.g., edge density) can vary by edge type, landscape context, disturbance intensity, community structure, productivity, and species' life history traits (Halpern 1988; Harper et al. 2005; McWethy, Hansen, and Verschuyl 2009). Mature forest patches are often surrounded by lands that are structurally less complex, and thus a negative species response to fragmentation is assumed (Murcia 1995). However, land use adjacent to forest patches can vary substantially (e.g., regenerating forests, agriculture, pasture, and urban or sub-urban environments). Species responses to edges, in turn, are driven by direct biological effects (temperature, light intensity, solar radiation, vapor pressure deficit, etc.) and indirectly through vegetation response to those abiotic factors, all of which vary significantly with the type of edge. Fragmentation metrics are thus problematic because of multi-directional issues and the difficulty of interpreting responses to highly variable disturbances within the landscape.

Finally, many indicators can be used to describe ecosystem quality and in particular to measure and quantify change in biodiversity (Duelli and Obrist 2003; Milà i Canals et al. 2007a). The use of multiple measures will likely be required, therefore, to fully capture the complexity of biodiversity, which is a multi-dimensional concept, and to provide the information necessary to understand the implications of trade-offs. Biodiversity indicators can be (1) "direct" indicators that are biological or taxon-based (e.g., indicator species, richness of functional groups or guilds), or (2) "indirect" or vegetation structure-based indicators that reflect local or landscape-level habitat conditions (e.g., forest stand structural complexity, measures of landscape structure) (Lindenmayer, Margules, and Botkin 2000; McElhinny et al. 2005; Rossi 2011). The usefulness of taxon-based indicators depends upon the taxonomic resolution and taxonomic groups considered, and richness among different taxonomic groups is sometimes not strongly correlated. Furthermore, estimation of direct measures of diversity may require expensive and time-consuming studies. Habitat-based indicators may have a narrow scope and be related only to certain taxa or influenced by the values of LCA practitioners (Duelli and Obrist 2003; Rossi 2011). Failing and Gregory (2003) encourage users of biodiversity indicators to clarify the value-oriented basis for biodiversity indicator selection and to design indicators that are concise, relevant, and meaningful to decision makers.

6.3 Challenges Specific to Effects on Ecosystem Services

The characterization of ecosystem services impacts shares several of the same challenges as the biodiversity impact assessment, especially in that biodiversity is often related to, and may influence, those services (e.g., water and air quality, development, etc.). The potential effects of land transformation on biotic production and climate regulation are susceptible to being double-counted, as these are typically included in the climate change indicator. The main difference lies in consideration of time in the land use impact evaluation framework proposed by the UNEP-SETAC. In the UNEP-SETAC land use impact framework, the change in carbon is "weighted" with the time it takes to regenerate the level of carbon in the soil or on the land. When accounted for in the climate change indicators, however, only the difference in carbon stocks between the two types of land use (normalized to the functional unit) is typically computed. As a result, when these changes in carbon stocks are considered under the UNEP-SETAC framework for land use impacts, rather than included within the climate change indicator, uncertainty is introduced through the assumptions about regeneration time.

7.0 DISCUSSION OF IMPLICATIONS FOR LCAs OF FOREST PRODUCTS

Characterizing the environmental aspects of the management practices used to produce a forest product or service can be complex. For example, in some regions manufacturers may acquire wood from a large number of landowners who collectively employ a range of silvicultural practices. Thus, it may not be possible to attribute the wood used to manufacture any particular product to a single location that was harvested or to a single silvicultural regime. However, a large body of research has examined the environmental aspects of sustainable forestry and provides information useful in the context of LCAs. This research has confirmed that managed forest systems can provide high levels of biodiversity, water quality, carbon storage, and other goods and services. It is also important to recognize that sustainable forestry in North America is conducted within a well-established framework of environmental laws, regulations, and guidelines that ensure high levels of environmental protection and consideration. Examples include laws such as the Clean Water Act and Endangered Species Act in the US, and the Species at Risk Act in Canada, along with their state or provincial forestry best management practices and regulations for protecting rare and threatened species and water quality.

The production of bioenergy feedstocks in general, and forest biomass in particular, illustrate the challenges of incorporating the environmental aspects of land management practices into LCAs. The cellulosic feedstocks required to meet future bioenergy demand will be derived from a variety of settings including agricultural, grassland, forest, urban, and aquatic ecosystems, and feedstock production systems in these ecosystems vary widely. In forest ecosystems alone, biomass for bioenergy is currently being derived from at least four production systems: thinnings, removal of harvest residues, intercropping of herbaceous vegetation, and short-rotation woody crops (Riffell et al. 2011a, 2011b, 2012; Verschuyl et al. 2011). Biodiversity response to biomass harvesting will vary based on biomass production system, site productivity, context of the surrounding landscape, scope of land use change, frequency and intensity of biomass harvest, structure of the wildlife communities present, individual species life history traits, and the potential to maintain elements of habitat structure.

Biomass energy feedstocks in agricultural systems are derived from annual grain crops, perennial grasses, woody perennials, specialty crops, and crop stovers (leaves and tops). Annual crop plants include corn, soybeans, sorghum, sugar beets, wheat, and barley, and examples of perennials include miscanthus (*Miscanthus* spp.), hybrid poplar (*Populus* spp.), and sugar cane (*Saccharum* spp.). Production systems may involve tillage, multiple annual applications of fertilizers and pesticides, supplemental irrigation, and removal of crop stover. Biodiversity response to these production systems will vary with tillage method, crop species, timing of harvest, amount of retained grain and stover, character of retained field borders, landscape context, and other factors. These practices contrast with those most often used in forests (e.g., thinning, removal of harvest residues) and hence lead to contrasting biodiversity responses. Identifying a single meaningful metric of biodiversity response that can be applied consistently in LCA across these and other production systems is therefore a significant challenge.

Another barrier to incorporating biodiversity response into LCAs that involve biomass production systems is lack of knowledge; few field studies have investigated this question, and Campbell and Doswald (2009, p.27) note in their review of the topic for liquid biofuels that "more research is needed, especially at the local level since much of the current literature reviewed focuses on global overviews." Recent meta-analyses of manipulative and observational field studies provide insight into potential biodiversity responses to practices associated with intensive biomass production systems in North American forests (Riffell et al. 2011a, 2011b, 2012; Verschuyl et al. 2011). Biodiversity responses varied among taxa and production systems reviewed. Most taxa responded positively to thinning treatments (Verschuyl et al. 2011). Diversity and abundance of birds were substantially and consistently lower in treatments with lower amounts of downed coarse woody debris (CWD) and/or standing snags, as was biomass of invertebrates (Riffell et al. 2011a). Other taxa did not respond strongly to reduced downed CWD and/or snags, but these conclusions were based on fewer studies. Little is currently known about biodiversity response to harvest of fine woody debris. Riffell et al. (2011a) concluded that if reductions in coarse woody debris from actual harvests are less than the 70-95% used in experimental studies, then overall biodiversity responses may be minimal.

Diversity and abundance of bird and mammal guilds are often lower on short-rotation plantations compared with reference woodlands, but abundance of individual species varies (Riffell et al. 2011b). Shrub-associated birds are often more abundant in short-rotation woody crops than in reference forests, but species associated with mature forest and cavity nesters are often less abundant. Differences between bird communities in short-rotation woody crops and reference forests diminish as short-rotation woody crops mature. However, a wide variety of reference forests have been used.

Several recent studies have evaluated biodiversity response to different levels of harvest residue retention following biomass harvests. Fritts et al. (2014) tested a retention area-based Biomass Harvesting Guideline (BHG) strategy for maintaining desired volumes of down woody debris (DWD) and concluded that the treatments resulted in DWD retention levels that approximated those prescribed. Thus, they suggested that BHGs can be implemented successfully in an operational setting. Prescribed retention levels were 0%, 15%, 30%, and 100% of pre-harvest DWD levels with the DWD in the15% and 30% retention levels spatially dispersed or scattered. The authors did not detect consistent differences in shrew relative abundance among these treatments, but relative abundance of all shrew species increased over time as vegetation became established. They concluded that shrews in their study area were associated more with vegetation characteristics than DWD and that removal of harvest residues may have little influence on shrew abundances in the southeastern United States Coastal Plain.

Based on a review of the literature, Berger et al. (2013) compared the degree of departure of energy-wood harvesting from less intensive conventional and whole-tree harvesting in the north central and northeastern United States. They discussed differences in forest structure, forest carbon, nutrient retention, ground-layer plants, saproxylic organisms, amphibians, reptiles, and small mammals. The authors report finding that residual structural and functional conditions resulting from forest harvesting differed from those resulting from natural disturbance. Differences generally were least with conventional harvesting, greatest with energy-wood harvesting, and intermediate with whole-tree harvesting.

Also based on a review of the literature, Otto, Kroll, and McKenny (2013) reported that although studies involving terrestrial salamanders often stressed the importance of retaining DWD in harvested forests, empirical support for this conclusion is uncertain due to study- and species-specific variation in responses. Lack of a DWD effect on amphibians has often been attributed in published papers to downed wood that was not well decayed or was too small for amphibian use. However, the authors concluded that their review provides evidence that retention can ameliorate initial effects of timber harvest on amphibian communities.

Results from studies of biodiversity response to intercropping of native, warm season grasses in commercial forests also are now emerging. Marshall et al. (2012) recently reported initial effects of removing woody biomass after clear-cutting and intercropping switchgrass on rodents for two years post-treatment in regenerating pine plantations in North Carolina. Species richness and diversity of rodents did not change due to switchgrass intercropping or biomass removal. However, abundance of the two species differed between the treatments. *Peromyscus leucopus* was more abundant and had the greatest survival in treatments without switchgrass, while the invasive *Mus musculus* was most abundant in treatments with switchgrass. Four-year results for small mammals were similar to these and the authors concluded that natural succession exerted greater effects on rodent species and the rodent community than did biofuel production regimes (Homyack et al. 2014). Briones et al. (2013) found that diet and trophic position of *P. leucopus* was not influenced by treatments on these sites. Based on a study on these same study sites, Homyack et al. (2013) found that neither intercropping switchgrass with pine nor removal of harvest residuals caused herpetofauna diversity or abundance of common species to differ from traditional plantation management during the first two years following treatment establishment. Riffell et al. (2012) noted that research with grasses in row crop agriculture suggests that effects of intercropping on biodiversity will likely vary with habitat needs of individual species and communities and that intercropping regimes favouring mixed native warm-season grasses over switchgrass only, spring harvests over fall, and that rotational harvests producing mosaics of grass heights would likely benefit biodiversity.

The complexity of these findings demonstrates the challenges of incorporating biodiversity considerations in LCA involving land uses, and forest management in particular. Characterizations of biodiversity response to forest management must consider the temporal and spatial scales of analysis, the forest management system(s) to be assessed, landscape context, and many other factors.

8.0 CONCLUSIONS

A framework and several proposed indicators for land use impact assessment in LCA are available in the literature. The proposed framework considers the effects from both land transformation (or conversion) and occupation. Transformation effects are generally described as the difference in ecosystem quality between pre- and post-transformation. On the other hand, the effects of land occupation are generally described as the difference between the ecosystem quality in the occupation conditions and that of some sort of "ideal" conditions in the absence of the occupation processes weighted by the time the occupation process occurs, rather than as the difference in ecosystem quality between the start and the end of the occupation process. The change in ecosystem quality between the start and the end of the occupation process is generally considered negligible for the purpose of LCA.

Integrating biodiversity and ecosystem services considerations into the methodological framework of LCA for forest products systems poses particular challenges. Many proposed approaches rely on a single biodiversity indicator. Biodiversity, however, is a multi-dimensional concept that can never be fully represented by a single number. Reliance on a single metric over-simplifies "biodiversity" and will undoubtedly lead to inappropriate conclusions in LCA, thereby failing to support decision-making related to local land management practices. That said, using several indicators to characterize biodiversity may lead to unfair comparisons in the context of other LCA single-indicator metrics such as climate change. In addition, the interconnections between landscape components and biodiversity can often be multi-directional and yet many of the current methodologies for biodiversity assessment within LCA are unable to incorporate positive effects because of a uni-directional focus on loss, damage, and extinction. Finally, the empirical basis for addressing site-specific biodiversity in LCA is limited because of the lack of field research investigating responses of biological diversity to actual biomass production practices.

LCA is not currently suited to providing reliable site-specific assessment results concerning biodiversity and ecosystem services due to the complexities discussed above, and probably never will be because of the inherent global and comprehensive nature of LCA. Nonetheless, land use is a key aspect of forest products manufacture that should somehow be incorporated within life cycle approaches to reduce the risk of environmental burden shifting across impact categories or across life cycle stages. Site-specific and/or territorial assessment approaches are thus an essential complementary tool when LCA is applied in the context of land use impacts and can be used to guard against inaccurate conclusions. This type of paired assessment allows for acknowledgement of the relevance of potential biodiversity-related impacts within the context of LCA, while recognizing that effective examination of the complexities of biodiversity responses requires significant additional site-specific analysis.

Aside from selecting indicators appropriate for characterizing land use impacts, several key decisions still need to be made by LCA practitioners in analyzing land use impacts that can have enormous implications to the results, including the choice of a reference state and the estimation of regeneration times. Given this significant uncertainty, there is a need to distinguish the types of decisions that require information on the impacts of land use and to identify those that need and can be supported by LCA. For instance, a group of LCA experts highlighted that "LCA seems appropriate to bring a life cycle perspective to support complex decisions where the scope of other tools (e.g., Environmental Impact Assessment, Environmental Risk Assessment) is too limited or inappropriate" but "may not be adequate to aid in concrete land management decisions where other tools may be more appropriate" (Milà i Canals et al. 2006b, p. 324).

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APPENDIX A

ATTRIBUTIONAL AND CONSEQUENTIAL LCA

Difference between Attributional and Consequential LCA

According to the ISO 14040 Standard (ISO 2006, p. 19), "it is necessary to consider the decision-making context when defining the scope of an LCA; i.e. the product systems studied should adequately address the products and processes affected by the intended application" and "two possible different approaches to LCA have developed during the recent years. These are

- one which assigns elementary flows and potential environmental impacts to a specific product system typically as an account of the history of the product, and
- one which studies the environmental consequences of possible (future) changes between alternative product systems."

These two approaches are often referred to as attributional and consequential LCA (see for instance Curran, Mann, and Norris 2002; Ekvall and Weidema 2004, Finnveden et al. 2009, Tillman et al. 1994).

Attributional LCA (ALCA) aims at describing the environmental properties of a life cycle and its subsystems. Its main characteristics are that it generally includes the full life cycle of the studied system, uses average data, and uses allocation to deal with multifunctional systems (e.g., economic value, mass). ALCA has been described as an "average [representation] of a static system irrespective of economic or policy context" (Plevin, Delucchi, and Creutzig 2014). Consequential LCA (CLCA) aims at describing the effects of changes within a life cycle. Its main characteristics are that it generally includes only processes that are affected by the change, uses data that reflect expected effects of changes (e.g., marginal data), and avoids allocation through system expansion. The appropriate uses of ALCA and CLCA are still debated in the LCA literature (Finnveden et al. 2009). However, it has been argued that while CLCA can be used to estimate the implications of a given decision, ALCA cannot (Curran, Mann, and Norris 2002; Earles and Halog 2011; Ekvall and Andrae 2006; Reinhard and Zah 2011; Whitefoot et al. 2011).

Within ALCA, two distinct definitions have emerged with significant implications for LCAs of forest products, of particular importance to land use impacts (including climate change-related). Curran, Mann, and Norris. (2002, 2005) have defined ALCA as "[serving] to allocate or attribute, to each product being produced in the economy at a given point in time, portions of the total pollution (and resource consumption flows) occurring from the economy as it is at a given point of time." With this definition, the land use impacts of occupying the land should be measured as the difference between land quality at the end and at the beginning of the defined temporal horizon. Helin et al. (2013, pp. 478 and 484) argue the environmental relevance of such an approach can be disputed and rather suggest using natural relaxation to assess any impacts related to land use in ALCA because "ignoring the 'no use' reference land-use situation in ALCA might result in conclusions that do not reflect the environmental impacts of the system studied." The ILCD Handbook (EC-JRC-IES 2010, p. 238) requires that, for ALCA, "only the net interventions related to human land management activities shall be inventoried" and that the "reference system under attributional modelling shall be the independent behaviour of the site, starting from the status of the land at that moment when the area of the analysed system is prepared for the modelled system." Helin et al. (2013), however, recognize that the application of a virtual reference situation describing something that did not take place might be controversial.

Implication of the Definition of Attributional LCA for the Selection of a Reference Situation for Land Use Impacts Calculation

Attributional LCA aims at "describing the environmentally relevant physical flows to and from a life cycle and its subsystems" (Ekvall and Weidema 2004, p. 161), but disagreement remains as to what this means when it comes to land occupation. Two distinct interpretations can be found in the LCA literature.

According to the first interpretation, an attributional LCA attempts "to allocate or attribute, to each product being produced in the economy **at a given point in time** [emphasis added], portions of the total pollution (and resource consumption flows) occurring from the economy as it is at a given point of time" or, in other words, to quantify the flows of substance to and from a specific product system boundary within a temporal horizon defined by the product's life cycle (Curran, Mann, and Norris 2005, p. 856). Clearly, using "quasi-natural" conditions as the reference for occupation impacts, as recommended by UNEP-SETAC, does not achieve this objective of measuring the "actual" environmental impact attributable to a specific product at a given point in time. Instead, the UNEP-SETAC approach attempts to quantify the distance from some sort of ideal condition. Using the conditions of the land before the start of the occupation process as the reference is more aligned with the objective of measuring the actual environmental impact attributable to a specific product, as it is the only approach that enables the characterization of the actual degradation or improvement in ecosystem quality due to the occupation process. However, this approach can require very site-specific and potentially costly data on ecosystem quality before and after the occupation process. In contrast, not only does the UNEP-SETAC approach recommend using an ideal reference condition, it also recommends neglecting differences in ecosystem quality between the start and end of the occupation process.

According to the second interpretation of attributional LCA, "only the net interventions related to human land management activities shall be inventoried in LCI. Interventions that would occur also if the site was unused shall not be inventoried" (EC-JRC-IES 2010, p. 238). This approach requires the definition of a no-use system that reflects a state of "relaxation" of the land. In specific, according to the EC-JRC-IES (2010, p. 238), the no-use system reflects the "independent behaviour of the site, starting from the status of the land at that moment when the area of the analysed system is prepared for the modelled system." This interpretation of attributional LCA is generally aligned with the use of reference systems based on quasi-natural conditions, as recommended by UNEP-SETAC. However, modeling how the land would evolve in the absence of human activities is highly uncertain and there is no general agreement on what "quasi-natural" means.

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APPENDIX B

UNEP-SETAC LULC CLASSIFICATION

Table B.1 UNEP-SETAC Land Use and Land Cover Classification

LEVEL 1			LEVEL 2			LEVEL 3			LEVEL 4		
	Land Use and Land Cover Classes	**Description**		**Refinement of Land Use and Land Cover Classes**	**Description**		**Land Management**	**Description**		**Land Use Intensity**	**Description**
0	Unspecified	Land use and cover not known	1	Used	Human land use and resulting land cover not known						
			2	Natural*	Natural land cover not known						
1	Forest	Areas with tree cover >15%	1	Natural*	Forest not used by humans	1	Primary	Forests minimally disturbed by human impact, where flora and fauna species abundance is near pristine			
						2	Secondary	Areas originally covered with forest or woodlands, where vegetation has been removed, forest is re-growing and is no longer in use			
			2	Used	Forest used by humans				1	Extensive	Forests with extractive use and associated disturbance like hunting, and selective logging, where timber extraction is followed by re-growth including at least three naturally occurring tree species
									2	Intensive	Forests with extractive use, with either even-aged stands and clear-cut patches, or less than three naturally occurring species at planting/seeding

(Continued on next page. See notes at end of table.)

Table B.1 Continued

LEVEL 1			LEVEL 2			LEVEL 3			LEVEL 4		
	Land Use and Land Cover Classes	**Description**		**Refinement of Land Use and Land Cover Classes**	**Description**		**Land Management**	**Description**		**Land Use Intensity**	**Description**
2	Wetlands	Areas regularly flooded, eventually with tree cover, closed to open (>15%)	1	Coastal*	Areas tidally, seasonally, or permanently waterlogged with brackish or saline water. Includes costal marshland, mangroves, and salt marshes. Excludes coastal land with infrastructure or agriculture						
			2	Inland*	Areas partially, seasonally, or permanently waterlogged. The water may be stagnant or circulating. Includes inland marshland, swamp forests, and peat bogs						
3	Shrub land*	Areas with shrub-dominated sclerophyllous vegetation									
4	Grassland	Herbaceous cover, closed to open (>15%) with scattered shrubs or trees	1	Grassland	Naturally grassland dominated vegetation	1	Natural*	Grassland-dominated vegetation, fauna, and flora near pristine (e.g., steppe, tundra, savannah)			
						2	For livestock grazing	Grasslands where wildlife is replaced by grazing livestock			
			2	Pasture/meadow	Areas that have been converted to grasslands for livestock grazing or fodder production				1	Extensive	Pasture with low number of livestock or meadows mechanically harvested 2 or 3 times per year, reduced input of fertilizer
									2	Intensive	Pasture with high number of livestock or meadows mechanically harvested 3 times or more per year, fertilizer applied
5	Agriculture	Areas used for crop production	1	Arable	Cultivated areas regularly ploughed and generally under a rotation system. Cereals, legumes, fodder crops, and root crops. Includes flower and tree (nurseries) cultivation and vegetables as well as aromatic, medicinal and culinary plants. Excludes permanent pastures	1	Fallow	Cropland temporarily not used (<2 years)			

(Continued on next page. See notes at end of table.)

Table B.1 Continued

LEVEL 1			LEVEL 2			LEVEL 3			LEVEL 4		
Land Use and Land Cover Classes		Description	Refinement of Land Use and Land Cover Classes		Description	Land Management		Description	Land Use Intensity		Description
5	Agriculture	Areas used for crop production	1	Arable	Cultivated areas regularly ploughed and generally under a rotation system. Cereals, legumes, fodder crops, and root crops. Includes flower and tree (nurseries) cultivation and vegetables as well as aromatic, medicinal and culinary plants. Excludes permanent pastures	1	Fallow	Cropland temporarily not used (<2 years)			
						2	Non-irrigated	Annual crop production based on natural precipitation (rainfed agriculture)	1	Extensive	+ Use of chemical–synthetic and organic fertilizer as well as pesticides is reduced
									2	Intensive	+ Chemical–synthetic and organic fertilizer as well as pesticides are applied
						3	Irrigated	Annual crops irrigated permanently or periodically, using a permanent infrastructure (irrigation channels, drainage network). Most of these crops like rice could not be cultivated without an artificial water supply. Does not include sporadically irrigated land	1	Extensive	+ Use of chemical–synthetic and organic fertilizer as well as pesticides is reduced
									2	Intensive	+ Chemical–synthetic and organic fertilizer as well as pesticides are applied
						4	Flooded crops	Areas developed for rice cultivation. Flat surfaces with irrigation channels. Surfaces regularly flooded			
						5	Greenhouse	Crop production under plastic or glass			
						6	Field margins/ hedgerows	Areas between fields with natural vegetation			

(Continued on next page. See notes at end of table.)

Table B.1 Continued

LEVEL 1			LEVEL 2			LEVEL 3			LEVEL 4		
	Land Use and Land Cover Classes	**Description**		**Refinement of Land Use and Land Cover Classes**	**Description**		**Land Management**	**Description**		**Land Use Intensity**	**Description**
5	Agriculture	See above	2	Permanent crops	Perennial crops not under a rotation system which provide repeated harvests and occupy the land for a long period before it is ploughed and replanted: mainly plantations of woody crops	1	Non-irrigated	Perennial crops production based on natural precipitation (rainfed agriculture)	1	Extensive	+ Use of chemical–synthetic and organic fertilizer as well as pesticides is reduced
									2	Intensive	+ Chemical–synthetic and organic fertilizer as well as pesticides are applied
						2	Irrigated	Perennial crops with artificial input of water	1	Extensive	+ Use of chemical–synthetic and organic fertilizer as well as pesticides is reduced
									2	Intensive	+ Chemical–synthetic and organic fertilizer as well as pesticides are applied
6	Agriculture, mosaic	Heterogeneous, agricultural production intercropped with (native) trees. Trees or shrubs are kept for shade or as wind shelter; or use of timber or non-timber products (e.g., agro-forestry)									
7	Artificial areas	Artificial surfaces and associated area(s)	1	Urban	Areas with infrastructure for living and businesses	1	Industrial fallow	Areas with remains of industrial buildings; deposits of rubble, gravel, sand and industrial waste. Can be vegetated			
						2	Continuously built	Buildings cover most of the land. Roads and artificially surfaced area cover almost all the ground. Non-linear areas of vegetation and bare soil are exceptional. At least 80% of the total area is sealed			

(Continued on next page. See notes at end of table.)

Table B.1 Continued

LEVEL 1			LEVEL 2			LEVEL 3			LEVEL 4	
	Land Use and Land Cover Classes	**Description**		**Refinement of Land Use and Land Cover Classes**	**Description**		**Land Management**	**Description**	**Land Use Intensity**	**Description**
7	Artificial areas	See above	1	Urban	See above	3	Discontinuously built	Most of the land is covered by structures. Buildings, roads, and artificially surfaced areas associated with areas with vegetation and bare soil, which occupy discontinuous but significant surfaces. Less than 80% of the total area is sealed		
						4	Green areas	Areas with vegetation within urban fabric. Includes parks with vegetation		
			2	Industrial area	Artificially surfaced areas (with concrete, asphalt, or stabilized, e.g., beaten earth) devoid of vegetation occupy most of the area in question, which also contains buildings and/or areas with vegetation					
			3	Mineral extraction site	Areas with open-pit extraction of industrial minerals (sandpits, quarries) or other minerals (opencast mines). Includes flooded gravel pits, except for riverbed extraction					
			4	Dump site	Landfill or mine dump sites, industrial or public					
			5	Construction site	Areas under construction development, soil or bedrock excavations, earthworks					
			6	Traffic area	Areas used for traffic infrastructure	1	Road network	Motorways, including associated installations (gas stations)		
						2	Rail network	Railways, including associated installations (stations, platforms)		
						3	Rail/road embankment	Vegetated area along motorways and railways		
8	Bare area*	Areas permanently without vegetation (e.g., deserts, high alpine areas)								
9	Snow and ice*	Areas permanently covered with snow or ice considered undisturbed								
10	Water bodies	Areas covered permanently with water	1	Rivers	Areas covered with watercourses	1	Natural*	Rivers nearly undisturbed by human use		
						2	Artificial	Artificial watercourses serving as water drainage channels. Includes canals		
						3	Used	Riverbed heavily influenced by human use, e.g. due to straightening or infrastructure.		
			2	Lakes	Body of slow-moving or standing water that occupies an inland basin	1	Natural*	Lakebed, nearly undisturbed by human use		

(Continued on next page. See notes at end of table.)

Table B.1 Continued

LEVEL 1			LEVEL 2			LEVEL 3			LEVEL 4		
Land Use and Land Cover Classes		**Description**	**Refinement of Land Use and Land Cover Classes**		**Description**	**Land Management**		**Description**	**Land Use Intensity**		**Description**
10	Water bodies	See above	2	Lakes	See above	2	Artificial	Reservoir in a valley because of damming up a river			
						3	Used	Lakebed disturbed by human use, e.g., by infrastructure			
			3	Seabed	Areas covered permanently with salt water	1	Natural*	Natural seabed, nearly undisturbed by human use			
						2	Used	Seabed influenced by human use	1	Fisheries for dredging	Seabed disturbed by dredging due to fisheries
									2	Sediment dumping	Seabed disturbed due to dumping of sediments
									3	Marine infrastructure	Seabed disturbed due to infrastructure like harbors or platforms
									4	Oil drilling	Sea bed disturbed due to oil drilling
									5	Mining	Sea bed disturbed due to mining

SOURCE: The table is derived from Koellner et al. (2013b).
* Refers to land covers that serve as a natural reference (Koellner et al. 2013a, 2013b).

Table B.2 UNEP-SETAC Regionalization Approach

LEVEL 1 **Biome**		**LEVEL 2** **Climatic Region**		**LEVEL 3** **Terrestrial and Water Biomes**	
1	Terrestrial biomes	1	Tropical and subtropical	1	Moist broadleaf forests
				2	Dry broadleaf forests
				3	Coniferous forest
				4	Savannas and shrublands
				5	Flooded grasslands and savannas
				6	Mangroves
				7	Deserts and xeric shrublands
		2	Temperate	1	Broadleaf and mixed forests
				2	Coniferous forests
				3	Grasslands, savannas and shrublands
				4	Mediterranean forests, woodlands and scrubs
				5	Deserts and xeric shrublands
		3	Boreal	1	Forests/Taiga
				2	Tundra
				3	Montane grasslands and shrublands
				4	Deserts and xeric shrublands
		4	Polar	1	Rock and ice
				2	Deserts and xeric shrublands
2	Freshwater biomes	1	Tropical and subtropical		
		2	Temperate		
		3	Boreal		
		4	Polar		
3	Coastal water and shelf biomes (shallower than 200 m)	1	Tropical		
		2	Temperate		
		3	Polar		
4	Deep sea biomes (deeper than 200 m)				

SOURCE: The table is derived from Koellner et al. (2013b).
NOTE: Level 4 (ecoregion) and level 5 (exact geo-referenced location) are not shown in this table.

Figures A.1 and A.2 show maps of terrestrial and marine ecoregions.

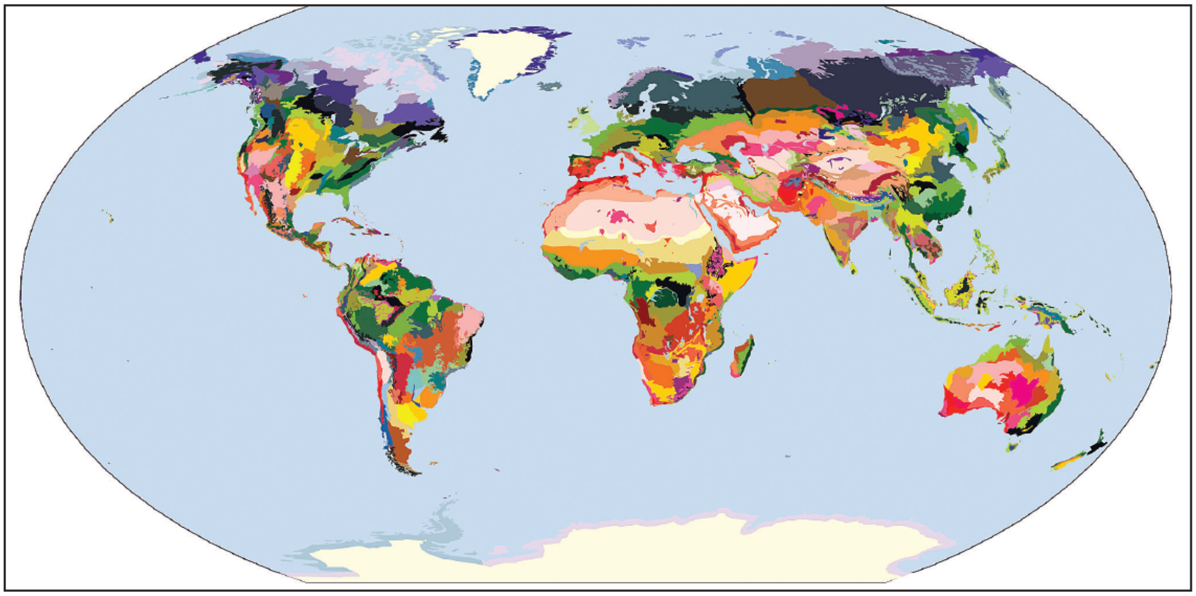

Figure B.1 Map of Terrestrial Ecoregions
[Figure taken from Olson et al. (2001)]

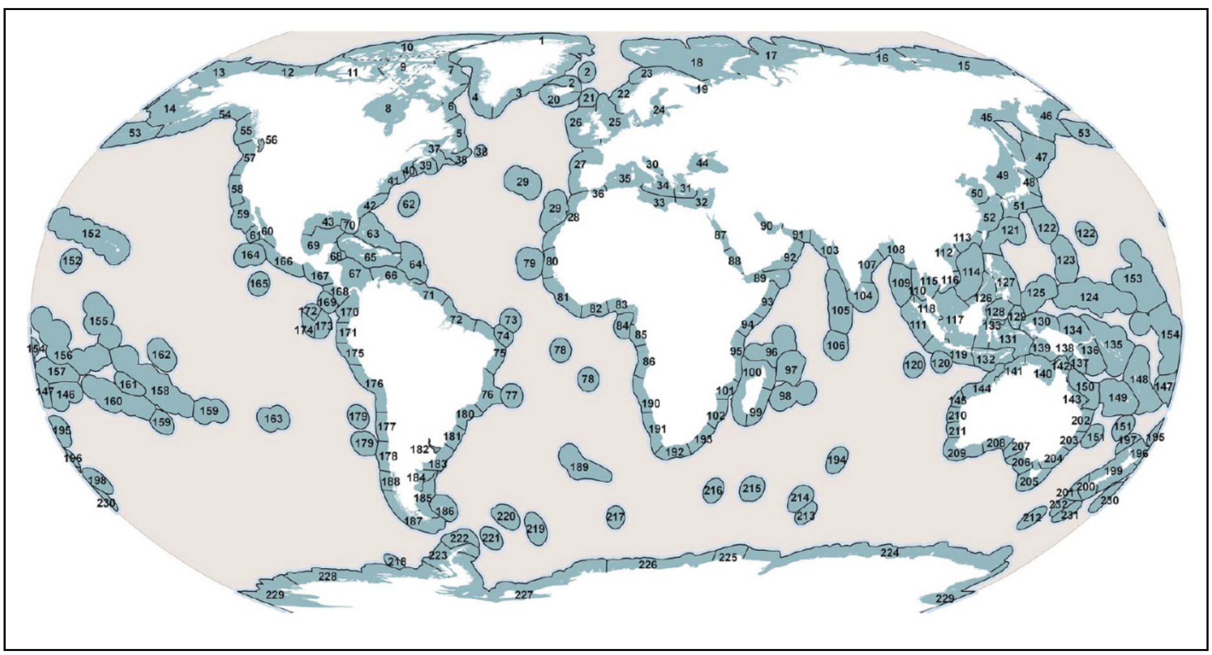

Figure B.2 Map of Marine Ecoregions
[Figure taken from Spalding et al. (2007)]

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APPENDIX C

MICHELSEN'S INDICATOR OF BIODIVERISTY QUALITY

Michelsen (2008) proposed an indicator that makes an attempt to evaluate biodiversity quality (Q) using various aspects of biodiversity, notably, the ecosystem scarcity (ES), the ecosystem vulnerability (EV), and the conditions for a maintained biodiversity (QMB) based on key factors of biodiversity (Larsson 2001).

Biodiversity quality at a given location and time can hence be calculated as the product of ecosystem scarcity, ecosystem vulnerability and conditions for maintained biodiversity index:

$$Q = ES \times EV \times CMB$$

The **ecological scarcity** concept, in the context of LCA, was first introduce by Weidema and Lindeijer (2001). The rationale for its use is that biodiversity linked to scarce ecosystems is assumed to be more vulnerable than that of more widespread ecosystems. It is expressed as the inverse of the potential structure[22] area (A_{pot}) that could be occupied by the ecosystem if left undisturbed by human activities or in its normalized form as

$$ES = 1 - \frac{A_{pot}}{A_{max}}$$

where A_{max} is the potential area of the most widespread structure at the relevant level (e.g., biome, landscape, etc.) . The more scarce an ecosystem, the closer ES is to 1. For this index to be useful for LCA of forest products, information on A_{pot} is required for various forest types.

The objective of the **ecosystem vulnerability** (EV) index is to provide information on the relative number of species affected by a change in the ecosystem area (species-area relationship, see body of the report). The rationale is that the greater the area of ecosystem lost, the more valuable is the remaining area. A few different equations have been proposed to compute EV and data are normally more difficult to find, resulting in proxies being used the majority of the time. Michelsen (2008) suggested that, in the absence of better data, the following semi-quantitative framework is applied: for ecoregions with a "critical" conservation status, EV=1; for ecoregions with a "vulnerable" conservation status, EV=0.5; and for intact ecoregions, EV=0.1. Note that Michelsen provides no definition of "intactness."

While the ecosystem scarcity and vulnerability indices provide information on the intrinsic, or potential, biodiversity value of an area, the objective of the **conditions for maintained biodiversity** (CMB) index is to provide information on the actual condition of the studied area in terms of its biodiversity. CMB is a combined index of the key factors for biodiversity (Larsson 2001) and is ecosystem-specific. CMB is calculated as follows:

$$CMB = 1 - \frac{\sum_{i=1}^{n} KF_i}{\sum_{i=1}^{n} KF_{i,max}}$$

where KF_i is any implemented key factor (i.e., known to be important for biodiversity in the particular structure) for biodiversity (varies depending on the ecosystem type). The key factors can also be weighted to reflect their relative importance. In the absence of impact, CMB = 1.

[22] The term "structure" implies that different levels can be used for the analysis (e.g., biome, landscape, ecosystem, vegetation type, etc.).

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