Linking land use and ecosystem services

Development of a Life Cycle Impact Assessment method to improve evaluation of biobased products

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Preface

"*The question is, whether it is really possible to describe the basic interactions in a manner that produces more than frustration due to the enormous inherent complexity*." (Kandziora et al., 2013)

This quote from a paper on ecosystem service quantification describes how I felt somewhere halfway in the process of writing this thesis. I choose this subject, because it involves two very important global issues, but it is very complex and (almost) impossible to classify and measure. But this was exactly the aim of this research.. So, like my supervisor Jesús wanted, I even dreamed about land use and ecosystem services and tried my best to produce the best master thesis possible.. By following the planned steps of my research, I eventually found the indicators and data that I was looking for and was able to finish my thesis well before the final deadline.

Thus, I would like to thank the developers of the ecological integrity indicators and the 'matrix model', also for answering my questions: Prof. Dr. F. Müller, PD Dr. habil. B. Burkhard, Dr. S. Stoll, and M. Kruse, MSc. In particular, I would like to thank my supervisors, Jesús Rosales Carreon and Elisabeth Keijzer, for helping me when necessary, and leaving me free to fill in the research as I wanted. A discussion with Aat Barendregt and Jerry van Dijk on ecological integrity of organic agriculture was helpful and I appreciate the critical look on my thesis by Mara Hauck and thinking about some complicated LCA issues together. Lastly, I am happy with the positive feedback I got from everyone mentioned, and Suzanne de Vos, Tom Ligthart and Max Rietkerk.

I hope that my work contributes to evaluating land use in LCA and can help with choosing sustainable product alternatives, and thus in the end helps reduce the human ecological footprint. I think my thesis shows how complex ecosystems are and how little we know about their functioning. We do know that all human actions have an impact and therefore that not using anything is better than choosing the 'best alternative'. So, I'm sorry that I wrote so many pages.. when you decide to print it, consider just printing the most interesting pages ☺. I hope you enjoy reading it.

Linda Knoester, Utrecht, 16 April 2015

Abstract

Life Cycle Assessment (LCA) is a widely used standardized method to assess all environmental impacts of products or processes. However, LCA cannot fully assess the impacts of land use, which leads to an incomplete assessment of biobased products. To date, Life Cycle Impact Assessment (LCIA) methods evaluate land use impacts on biodiversity and soil quality, but do not include impacts on ecosystem services. Hence, the objective of this research was to develop an LCIA method to evaluate potential impacts of different land use types on ecosystem services.

Evaluation of scientific literature led to the selection of ecological integrity indicators to represent the potential of land to provide ecosystem services. The use of ecological integrity indicators and the application of the 'matrix model' based on quantitative data in combination with expert judgment is a new approach to assessing land use impacts. The 8 ecological integrity indicators represent the structural and functional aspects of ecosystem quality and hereby enable a more complete assessment of ecosystem quality besides biodiversity. The new land use LCIA method evaluates damage to ecological integrity and is called in short the EID method.

The EID method was applied to a case study on natural fibers from forestry and several agricultural feedstocks. The EID method was compared to the ReCiPe (H) endpoint method, a state of the art LCIA method which evaluates land use impacts on biodiversity. The case study demonstrated that the EID method is able to compare land use impacts of renewable material alternatives, and different land management practices. By differentiating between different agricultural types, the method is particularly useful for assessing the trade-off between the amount of land use and the relative quality of a certain land type. This means that larger land requirements do not necessarily lead to a larger impact when a certain land type can maintain ecosystem health and thus ensure sustainable land use. This more complete assessment of land use impacts improves the comparison of biobased and fossil-based products and can therefore contribute to choosing sustainable product alternatives with lowest environmental impacts.

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List of abbreviations

AoP	Area of Protection
CF	Characterization factor
CF_{occ}	Occupation characterization factor
CF_{trans}	Transformation characterization factor
CICES	Common International Classification for Ecosystem Services
CLC	CORINE Land Cover
EC-JRC	European Commission - Joint Research Centre
EI	Ecological integrity
EID	Ecological integrity damage
ES	Ecosystem services
ILCD	International reference Life Cycle Data System
LCA	Life Cycle Assessment
LCI	Life Cycle Inventory
LCIA	Life Cycle Impact Assessment
LU LCIA	Land use Life Cycle Impact Assessment
MA	Millennium Ecosystem Assessment
NPP	Net primary production
PDF	Potentially Disappeared Fraction of Species
SOM	Soil organic matter
TEEB	The Economics of Ecosystems and Biodiversity
TI	Transformation impact

1. Introduction

1.1 Background

The continuously growing global population and use of materials is putting great pressures on the environment (Krausmann et al., 2009; MA, 2005). Humanity currently uses around 60 billion tons of materials per year, of which 70% consists of non-renewable materials (Krausmann et al., 2013). The extraction and use of non-renewable, or fossil, resources such as fossil fuels and minerals leads to increasing scarcity of these resources, contributes to climate change, releases toxins into the environment, and produces large amounts of non-biodegradable waste (Krausmann et al., 2009; UNEP, 2012). Biomass, on the other hand, is biodegradable, generally non-toxic, and when the regeneration flux exceeds the extraction flux, biomass is carbon-neutral and can be extracted indefinitely (Balat, 2008; Johnson, 2009).

For these reasons, the use of biomaterials from agricultural and forestry feed stocks is being researched as a sustainable alternative to non-renewable materials (Mohanty et al., 2002). The production of renewable resources requires the use of land, in the form of agriculture or forestry, which is accompanied with environmental impacts such as soil degradation, deforestation, loss of biodiversity, and impacts on water quantity and quality (Krausmann et al., 2009). These impacts threaten the ability of ecosystems to continue to provide natural goods and services that are essential for human survival and well-being (MA, 2005). In order to reduce environmental impacts, total material use must decrease, and the use of materials with the least environmental impacts should be encouraged (Behrens et al., 2007). Therefore, it is important to be able to evaluate environmental impacts of different resources.

1.2 Problem description

Life Cycle Assessment (LCA) is a widely used standardized method to assess the environmental impacts of products or processes from 'cradle to grave'. LCA was developed to assess the environmental impacts of fossil-based products made from non-renewable resources, and cannot yet fully assess environmental impacts related to biobased products (Mattsson et al., 2000; Jolliet et al. 2004). So far, it is not possible to include all land use impacts of renewable materials, which leads to an incomplete assessment of biobased products (Sandin et al., 2013). This hampers comparison among biobased alternatives, and comparison between biobased and fossil-based products.

Different Life Cycle Impact Assessment (LCIA) methods exist that can be applied within the framework of LCA, each using different indicators and calculation methods to assess environmental impacts. Several LCIA methods evaluate land use as the area and time that land is used, without differentiating between the quality of different land use types (Milà i Canals et al., 2007a). For example, urban land use has a higher environmental impact than forestry, and organic agriculture might have fewer environmental impacts than conventional agriculture. Several state of the art land use LCIA methods include impacts of different land use types on biodiversity or soil quality (Goedkoop et al., 2013; Milà i Canals et al., 2007b). However, land use has several other impacts on ecosystem quality and functioning, which should also be included for a complete assessment (EC-JRC, 2010a). The UNEP/SETAC Life Cycle Initiative stresses the need for inclusion of ecosystem services in a land use impact assessment method, with a distinction between impacts of different land use types (Koellner et al., 2013b).

1.3 Research aim and research questions

The aim of this research was to develop a Life Cycle Impact Assessment method to evaluate potential impacts of different land use types on ecosystem services. Therefore, knowledge about land use, ecosystem functioning, ecosystem services quantification, and LCIA methods was combined to develop an LCIA method which includes all relevant aspects of potential ecosystem service provisioning by land. The central research question of this research was:

How can a Life Cycle Impact Assessment method be developed to assess land use impacts on ecosystem services?

This research project was divided into three phases: 1) gathering the necessary background information, 2) development of a new impact assessment method, and 3) application and evaluation of the new method in a case study (figure 1). Each phase contained the answering of one research question and its sub-questions:

1. **How are core concepts in this research defined?**
   1.1 How is land use defined?
   1.2 What types of land use exist?
   1.3 What are the environmental impacts of land use?
   1.4 What are ecosystem services?
   1.5 How can ecosystem services be quantified?
   1.6 How does land use relate to ecosystem services?
   1.7 What is Life Cycle Impact Assessment?
   1.8 How is land use currently incorporated in Life Cycle Impact Assessment?
2. **How can ecosystem services be incorporated in Life Cycle Impact Assessment?**
   2.1 Which indicators can be used to quantify potential supply of ecosystem services by land?
   2.2 How can characterization factors be developed from the indicator values?
3. **How does the new method function in a comparison to state of the art land use assessment?**
   3.1 How does the new method function in a comparison among different products?
   3.2 How do the results of an LCA using the new method compare with the results using a conventional method?

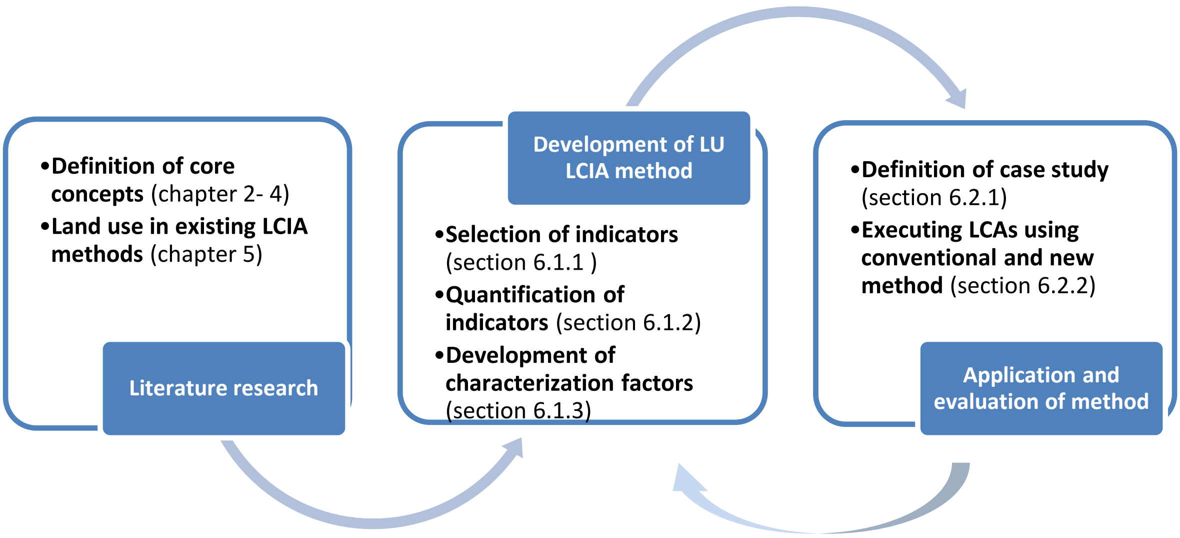


Figure 1. Research framework

1.4 Thesis structure

Chapters 2 and 3 provide the scientific background of land use and ecosystem services and their relation. Chapter 4 gives a description of the methodology of Life Cycle Assessment, in particular Life Cycle Impact Assessment (LCIA). In chapter 5 an overview is given of the state of the art of land use impact assessment in LCA, which led to the knowledge gap that was targeted in this research. Chapter 6 describes how a new LCIA method was developed, and gives a description of the case study to which the method was applied. The results of the case study using the new method and a conventional method are presented in chapter 7. The implication of these results and the choices made in the development of the method are discussed in chapter 8. Finally, a conclusion and suggestions for further research are given in chapter 9.

2. Land use

Land use includes a range of human activities and results in a variety of effects on the natural environment. This complex interaction is explored in this chapter: section 2.1 gives the scientific definition of land use, section 2.2 briefly discusses the socioeconomic drivers of land use, and section 2.3 describes the resulting environmental and ecological impacts. Section 2.4 gives an overview of the different land cover and land use classification schemes.

2.1 Definition

Land use and land cover are terms that are used interchangeably to indicate the characteristics of the observed land surface on a certain geographic location. The difference between the two is that the (bio)physical aspect of land in terms rock, soil, vegetation and water bodies is called land cover, while the human utilization of a piece of land in terms of arrangement, activities and inputs is called land use (MA, 2003; di Gregorio and Jansen, 2005). Land is a basic source of mass and energy throughput in terrestrial ecosystems, and its characteristics determine the type and quality of the ecosystem it sustains (di Gregorio and Jansen, 2005).

Land cover can change due to natural phenomena, such as weathering, glaciers and vegetative succession, and by human activities (USGS, 2013). A land transformation causes a change in quality of the land and the ecosystem it sustains, while a subsequent occupation maintains a certain land quality (Koellner and Scholz, 2007; Weidema et al., 2013). After an occupation has ended, land tends to return to an equilibrium situation. This regeneration may take a very long time, or the land may not reach a similar quality as before the transformation. It these cases, the transformation impact can be seen as irreversible (Koellner and Scholz, 2007). The total impact of land use therefore results from the change in quality due to the transformation integrated over the regeneration time and the impact of the occupation integrated over the duration of the occupation (Koellner and Scholz, 2007).

2.2 Drivers of land use

The ultimate goal of all land use practices is the same, namely the acquisition of natural resources for immediate human needs; in other words acquiring food, fiber, water and shelter (Foley et al., 2005). The drivers of land use depend on multiple natural and socioeconomic variables. Lambin et al. (2001) argue that land use depends on interactions between population density, technical capacity, mode of resource use, use of (economic) opportunities afforded by ecosystems (Lambin et al., 2001). Land that is most suitable for a certain function is initially used, which indicates that land use is dependent on ecosystem quality. Population growth has led to a larger demand for natural resources, which in turn increased the transformation and occupation of land (Ellis, 2011). The organization of people in extensive social networks and global trading systems in the globalized world of today offers the possibility of land use in other places than the place of demand (Vitousek et al., 1997; Smith, 2007). Markets and policies create opportunities and constraints for new land uses; poverty often leads to inappropriate land use and degradation because of restricted options, while concentrated and unchecked wealth can lead to environmental degradation due to unrestricted opportunities for development and resource extraction (Lambin et al., 2001).

2.3 Impacts of land use

The environmental impacts of land use are complex and difficult to measure directly. Land use occurs in local places, while potentially causing ecological degradation across temporal and spatial scales (Foley et al., 2005). Which impacts occur, and to what extent, depends on the land use type, the intensity of use or land management, and local conditions (Asner et al., 2004b). There are clear bioclimatic and soil-related controls over the vulnerability of ecosystems to degradation due to land use: ecosystems that are already vulnerable due to climatic or soil conditions show a stronger response to land use (Asner et al., 2004b). The different impacts of land use interact with each other and are exacerbated by other environmental impacts, such as climate change and pollution. This leads to indirect responses and feedbacks among the various ecosystem responses after the initial direct impacts (Asner et al., 2004b). Because of the complex and dynamic nature of ecosystems, responses to land use are non-linear and can include tipping points (DeFries et al., 2004). Despite of the complex ecosystem responses, there are multiple direct impacts that can be attributed to land use, shown in table 1. In general terms these include the altering of fluxes of water, nutrients, energy, and species (Asner et al., 2004b).

Land transformation directly leads to the loss and fragmentation of habitat, and the loss of natural resources and biodiversity. Land use directly affects soil and water quality, and influences biogeochemical cycles, which leads to a change in availability of water and nutrients. Changing vegetation and soil properties may further influence soil and water quality and quantity, and change biogeochemical cycles. Land use also influences species composition and can cause an increase of invasive species. Furthermore, the change of vegetation, soil properties and water fluxes influences regional climate and therefore weather patterns, while the release of carbon to the atmosphere contributes to global climate change. The biogeochemical and ecological properties of land determine the provisioning of ecosystem services that support human needs, which are therefore directly affected by land use and indirectly through all other impacts. Finally, the degradation of ecosystem services and loss of biodiversity leads to increased vulnerability of ecosystems and its inhabitants for diseases, forest fires and extreme events.

Table 1. Environmental impacts of land use

Environmental impacts	**References**
Habitat loss	Pimm and Raven, 2000
Fragmentation of habitat	Fischer and Lindenmayer, 2007
Loss of natural resources	Ramankutty and Foley, 1999
Biodiversity loss	Pimm and Raven, 2000; Sala et al., 2000
Soil degradation	Asner et al., 2004a
Change of water quality	Bennet et al., 2001
Change of carbon cycle	Houghton et al., 1999; Houghton, 2003
Change of nutrient cycles	Bennet et al., 2001
Change of water cycle	Postel et al., 1996; Vörösmarty et al., 2000
Change of ecosystem services	Ricketts et al., 2004; Vitousek et al., 1997
Spread of invasive species	Mooney and Hobbs, 2000
Increased vulnerability of ecosystem and inhabitants for diseases and disturbances	Patz et al., 2004
Regional climate change	Kalnay and Cai, 2003
Global climate change	Houghton, 2003

2.4 Land classification

Remote sensing has made it possible to construct global maps of the Earth's surface and has enabled the classification of major land cover types (Congalton et al., 2014). Different classification schemes are being used, which classify land types according to their observable characteristics such as soil type, natural vegetation, crops and human structures that cover the land surface (Congalton et al., 2014; Verburg et al., 2009). Classification schemes are hierarchically structured and include natural land cover and human land use. The first steps for hierarchical classification of land use and land cover based on global remote sensing data were made by Anderson et al. (1976), who included 9 major land cover classes (table 2).

The Land Cover Classification System (LCCS) developed by the FAO and UNEP is widely used as a framework for global land mapping. It has based the major classes on presence of vegetation, soil condition (terrestrial or aquatic), and artificiality of cover (Congalton et al., 2014). In the next hierarchical level, the set of classifiers differs for each of the 8 major classes and includes e.g. life form and cover, water seasonality, and leaf type. Additionally, environmental attributes such as climate, altitude and soils, and specific technical attributes such as floristic aspects (species composition) or crop type may be added (di Gregorio and Jansen, 2005). The resulting land cover classes depend on the chosen classifiers and the level of detail.

Table 2. Comparison of different classification schemes

Name	**Anderson et al., 1976**	**CORINE Land Cover (EEA, 2007)**	**LCCS (di Gregorio and Jansen, 2005)**	**Anthromes (Ellis et al., 2011)**
Number of hierarchical levels	3	3	Depends on chosen level of detail, could be up to approximately 14	2
Total number of classes	Over 100	44	Depends on chosen level of detail	19
Major classes	Urban or build-up land	Artificial areas	Artificial Surfaces and Associated Areas	Dense settlements
				Villages
	Agricultural land	Agriculture	Cultivated and Managed Terrestrial Areas	Croplands
		Agriculture, mosaic		
	Rangeland	Grassland	Natural and Semi-Natural Vegetation	Rangeland
	Forest land	Forest		-Seminatural lands -Wildlands
	Tundra	Shrub land		
	Water	Water bodies	-Natural Waterbodies, Snow and Ice -Artificial Waterbodies, Snow and Ice	
	Perennial snow or ice	Snow and ice		
	Wetland	Wetlands	Natural and Semi-Natural Aquatic or Regularly Flooded Vegetation	
			Cultivated Aquatic or Regularly Flooded Areas	
	Barren land	Bare area	Bare Areas	

Within the European Union, the CORINE Land Cover (CLC) classification is widely used, e.g. in the ecoinvent database for LCA (Frishknecht and Jungbluth, 2007). CORINE Land Cover is a project of the European Commission with the objective of assessing the land cover and land use types of the EU member states using satellite image data. Data are available for the years 1990, 2000 and 2006 (Stoll et al., 2015). The standard CLC nomenclature includes 44 land cover classes, grouped into three hierarchical levels. The five main categories are: 1) artificial surfaces, 2) agricultural areas, 3) forests and semi-natural areas, 4) wetlands, and 5) water bodies (European Environment Agency, 2007). The second hierarchical level includes a more specific land use type, e.g. arable land, permanent crops, pastures, and heterogeneous agricultural areas. The third level includes more detail, for example relating to land management: non-irrigated arable land, permanently irrigated land, and rice fields. Koellner and Scholz (2008) have suggested to add a fourth hierarchical level which includes details on intensity of use (e.g. intensive versus extensive agriculture), which would match the level of detail of land interventions in LCA databases such as ecoinvent. The definitions and characteristics of CORINE land use types can be found in the CORINE land cover technical guide (Bossard et al., 2000) and in short in appendix A.

Instead of mapping land cover or land use types, Ellis and Ramunkutty (2008) have defined 18 'anthromes' or antropogenic biomes, that represent the idea that humans have influenced the largest part of the Earth's surface. Currently, about 40% of all ice-free land on Earth is occupied by agriculture, pastures and urban settlements (Ellis et al., 2010; Foley et al., 2005). 37% is not used for this purpose, but embedded within human land use areas. 22% is left as wildlands, of which most is located in cold and dry biomes of the world (Ellis et al., 2010). Fortunately, land that is influenced by human activity can still sustain high levels of biodiversity and provide a number of ecosystem services (Chazdon et al., 2009). Therefore, another way of classifying land is by determining the ecosystems it sustains or the functions of land area (USGS, 2014). After exploring the concept of ecosystem services in chapter 3, and the methodology of LCA in chapter 4 and 5, a method is developed which links ecosystem services with the land types used in LCA (chapter 6).

3. Ecosystem services

This chapter explains the concept of ecosystem services and its relation with land use, in order to find an appropriate way of quantifying this relationship in LCA. Section 3.1 gives an overview of the scientific definition and typology of the concept of ecosystem services, and section 3.2 explores possible indicators for ecosystem services and how they can be quantified.

3.1 Definition and typology

Scientists and economists have discussed the general concepts behind natural capital, ecosystem services, and their value since the 1920s (Costanza et al., 2011). However, the first explicit mention of the term "ecosystem services" in the peer-reviewed scientific literature was by Ehrlich and Mooney (1983). The publication of Costanza et al. (1997) was one of the first studies that attempted to value ecosystem services and since then the concept of ecosystem services has been used to represent the benefits of nature for humans, either for conservation, management, or economic purposes (MA, 2003;2005; van Oudenhoven et al., 2012; de Groot et al., 2012). However, there has been much debate about the definition and classification of ecosystem services (Wallace, 2007; Boyd and Banzhaf, 2007; Fisher et al., 2009; Costanza, 2008; de Groot et al., 2010; Nahlik et al., 2012). Several prominent ecosystem services publications were analyzed in this research in order to build an understanding of concepts related to ecosystem services, and to select the most appropriate definitions for further use in this research. Table B1 in appendix B shows an overview of the definitions used in the analyzed publications.

3.1.1 Definitions

Generally, there is agreement among the analyzed articles on the definitions of biodiversity, ecosystems, and human well-being. In this research these concepts are defined as follows: an ecosystem is a dynamic complex of plant, animal and microorganism communities and the nonliving environment interacting as a functional unit (MA, 2003). These plants, animals, micro-organisms and the non-living environment are the ecosystem elements (Wallace, 2007). The abundance and variety of these elements represents ecosystem composition, and the physical distribution of the elements is called ecosystem structure (McGarigal et al., 1994). Biodiversity is considered as the number, abundance, composition, spatial distribution, and interactions of genotypes, populations, species, functional traits, and landscape units (Díaz et al., 2006). The definition of human well-being of the Millennium Ecosystem Assessment (MA, 2003) is adopted in this research, although the focus is on the capacity of ecosystem services to provide services, and therefore does not consider actual contribution to human well-being (see table 3 for full definition).

Most disagreement among ecosystem services publications is found in the definition of ecosystem function(ing), ecosystem services, and benefits. According to the definitions by Costanza et al. (1997) and the Millennium Ecosystem Assessment (MA, 2003), ecosystem services are the benefits that humans derive from ecosystem(s) (functions). Costanza et al. (1997) and de Groot et al. (2002; 2010) include ecosystem functions as an intermediate between ecosystem properties, and benefits for humans. An ecosystem function is defined as the capacity of an ecosystem to deliver services, independent of whether anyone wants or needs that service. Observed ecosystem functions are reconceptualized as 'ecosystem goods or services' when human values are implied (de Groot et al., 2002). The Millennium Ecosystem Assessment does not use 'ecosystem function' as an intermediate between properties and benefits, therefore, according to their definition, benefits can be derived from ecosystem components or processes directly. Wallace (2007) agrees with the MA and argues that the term ecosystem function is unnecessary when ecosystem processes and ecosystem services are adequately defined.

Wallace (2007), in accordance with Costanza (1997) and the Millennium Ecosystem Assessment (MA, 2003), sees benefits as analogous to ecosystem services. De Groot et al. (2002;2010;2012), however, define benefits as contributions to human well-being that result from the actual use of goods and services. Boyd and Banzhaf (2007) and Fisher et al. (2009) agree with de Groot et al. (2002; 2010; 2012) that ecosystem services are not benefits, and see benefits as the explicit impact on changes in human welfare, which is expressed in e.g. water for irrigation, drinking water or timber. Total benefit then is the quantity of explicit impacts times their value (Boyd and Banzhaf, 2007). Boyd and Banzhaf (2007) and Fisher et al. (2009) therefore take an economic perspective, while de Groot et al. (2002; 2010; 2012) focus on social implications of ecosystem services.

From the various approaches to ecosystem services definitions, in this research the approach of de Groot et al. (2002; 2010) is followed, because it enables the separation of the *potential* of an ecosystem to deliver services (function), the *actual* ecosystem services, and the benefits they provide to humans (which can be expressed in, for example, monetary units). The capacity or potential supply of services is relevant for land use evaluation in Life Cycle Assessment, as LCA is a generic and globally applicable assessment and can only include potential impacts (EC-JRC, 2010a; Koellner et al., 2013b). Actual ecosystem services supply can be assessed by local studies or risk assessments. Thus, in this research, ecosystem services are defined as those natural ecosystem components and processes that provide a benefit to human well-being (de Groot et al., 2002; 2010). Table 3 gives an overview of the definitions used in this research.

Table 3. Definition of concepts related to ecosystem services

Term	Definition	Examples
Ecosystem elements/components	Biotic and abiotic tangible entities described in amount (Wallace, 2007)	-rock formations -soil -water bodies -flora and fauna
Ecosystem composition	Abundance and variety of ecosystem elements/components ("the pieces of the puzzle") (adapted from landscape composition definition from McGarigal et al., 1994)	-amount of biomass and the forms in which it is present (e.g. soil organic matter, vegetation) -number and relative abundance of species -soil depth, type and water content
Ecosystem structure	The physical distribution or spatial character of ecosystem elements/components ("how the pieces are arranged") (adapted from landscape configuration definition from McGarigal et al., 1994)	-the spatial distribution of ecosystem elements/ components -relief -fragmentation
Human well-being	Includes multiple constituents: -Basic material for a good life: secure and adequate livelihoods, enough food, shelter, clothing, access to goods -health, including a healthy environment -good social relations -security, including access to resources, and security from disasters -freedom of choice and action (MA, 2005)	

Ecosystem processes	Biological, chemical and physical interactions between ecosystem components (Boyd and Banzhaf, 2007).	-nutrient cycling -photosynthesis -evapotranspiration -decomposition -predation -reproduction
Ecosystem properties	Characteristics of an ecosystem, which includes composition, structure, processes, size, biodiversity, human influence, and emergent characteristics such as resilience (combination of UK National Ecosystem Assessment, n.d. and Joint Nature Conservation Committee, n.d.)	
Ecosystem functioning	The combination of all ecosystem processes which maintain the integrity of an ecosystem (combination of Costanza et al., 1997 and MA, 2005)	The integrity of a coral reef can be compromised by acidification and thereby degrades/changes the functioning of the coral reef as a functional unit. Relates to resilience and tipping points.
Ecological integrity	The support and preservation of those processes and structures which are essential prerequisites of the ecological ability for self-organization of ecosystems (Barkmann et al., 2001)	
Ecosystem function	The capacity to deliver a service independently of whether anyone wants or needs that service (de Groot et al., 2010; Haines-Young and Potschin, 2010)	
Ecosystem services	The natural ecosystem components and processes that provide a (direct and indirect) benefit to human well-being. It therefore encompasses natural goods and services. (de Groot et al., 2002;2010; Fisher et al., 2009)	-food -drinking water -natural and genetic resources -stable climate -pollination -protection from disease/disturbances -aesthetic landscape
(Human) benefit	Contribution to human well-being or protection from adverse conditions (de Groot et al., 2002;2010;2012; Boyd and Banzhaf, 2007; Fisher et al., 2009)	-nutrition -safety and health -aesthetic, recreational and spiritual experiences

3.1.2 Classification

Several classification schemes exist that organize ecosystem services into three or four categories; provisioning, regulating, cultural, and supporting/habitat services (see table 4). Supporting and habitat services are seen as prerequisites for services in other categories (MA, 2005; Sukhdev, 2014). Regulating processes are under certain conditions supportive for provisioning and cultural services, which depends on the benefit that is considered. Therefore, it is argued to distinguish between intermediate and final services (Costanza, 2009; Boyd and Banzhaf, 2007). For example, purification of water can be of direct interest for humans when this water is used for drinking purposes and can then be called a final service. However, when a healthy fish population is the issue of interest, clean water is supportive of that service and in this case can be considered an intermediate service.

Several authors have noted that not making a distinction between intermediate and final services may lead to double counting when ecosystem services are quantified (Wallace, 2007; Boyd and Banzhaf, 2007). Boyd and Banzhaf (2007) look at ecosystem services with an economic perspective and aim to quantify ecosystem service analogous to GDP. In this case, only the final services should be quantified, as the value of the intermediate services is incorporated in the value of the final services. Therefore, the purpose of a study determines which type of classification should be used and which ecosystem services should be considered (Fisher et al., 2009).

The classification of the Millennium Ecosystem Assessment is most commonly used as illustrative of the different ecosystem services that exist (MA, 2003; 2005). The MA assesses the consequences of ecosystem change for human well-being, and hereby gives a scientific basis for the action needed to conserve ecosystems (MA, n.d.). Thus, the main purpose is raising awareness for, and stimulating action against, the degradation of ecosystem services. It has divided ecosystem services into four categories: provisioning, regulating, supporting and cultural services (table 4).

The Economics of Ecosystems and Biodiversity (TEEB) aims to draw attention to the economic benefits of biodiversity and ecosystems. It follows the MA classification, but has omitted supporting services because they are seen as ecological processes, which do not directly provide a benefit that should be valued economically. Instead, provision of habitat for migratory species and protection of the gene-pool were added in a category called habitat services (TEEB, 2010).

The Common International Classification for Ecosystem Services (CICES) was developed by the European Environmental Agency (EEA) with the goal of developing a common international classification, specifically for quantification and valuation of ecosystem services. Therefore attention was paid to overcoming the problem of double counting, which led to omission of supporting and habitat services. Supporting and habitat services are assumed to be embedded within each of the other categories (Haines-Young and Potschin, 2010). The CICES has a hierarchical structure which allows for adaptation of the classification according to new insights or specific conditions, and the application of the appropriate level of detail. The first three levels are shown in table 4; service section (provisioning, regulation and maintenance, cultural), service division, and service group.

Table 4. Comparison of different ecosystem services classification schemes

	Millennium Ecosystem Assessment (MA, 2005)	TEEB (2010)	CICES (Haines-Young and Potschin, 2013)	
Service category			*Service Group*	*Service Division*
Provisioning	Food	Food	Biomass	Nutrition
	Fresh water	Fresh water	Water	
	Fuelwood	Raw materials	Biomass	Materials
	Fiber			
	Biochemicals			
	Genetic resources	Genetic resources		
		Medicinal resources		
		Ornamental resources		
			Biomass-based energy sources	Energy
			Mechanical energy	
Regulating (and maintenance)[1]	Water regulation	Extreme events (incl. regulation of water flows)	Liquid flows	Mediation of flows
			Gaseous/air flows	
		Air quality regulation	Atmospheric composition and climate regulation	Maintenance of physical, chemical, biological conditions
	Climate regulation	Climate regulation		
		Maintenance of soil fertility	Soil formation and composition	
		Erosion prevention		
	Pollination	Pollination	Lifecycle maintenance, habitat and gene pool protection	
	Disease regulation	Biological control	Pest and disease control	
	Water purification	Waste treatment	Water conditions	
			Mediation by biota	Mediation of waste, toxics and other nuisances
			Mediation by ecosystems	
Cultural	Recreation and ecotourism	Opportunities for recreation and tourism	Physical and experiential interactions	Physical and intellectual interaction with biota, ecosystems and land-/seascapes [environmental settings]
	Educational	Information for cognitive development (education and science)	Intellectual and representative interactions	
	Spiritual and religious	Spiritual experience	Spiritual and/or emblematic	Spiritual, symbolic and other interactions with biota, ecosystems, and land-seascapes [environmental settings]
	Aesthetic	Aesthetic information	Other cultural outputs	
	Cultural heritage			
	Inspirational	Inspiration for culture, art and design		
	Sense of place			
Supporting	Soil formation			
	Nutrient cycling			
	Primary production			
Habitat		Life cycle maintenance (esp. nursery service)		
		Protection of gene pool		

[1] CICES calls the category regulation and maintenance, while the MA and TEEB call it regulating services.

3.1.3 Hierachical structure of ecosystem service concepts

The definitions related to ecosystem services can be structured hierarchically, which is generally referred to as the 'ecosystem services cascade' (de Groot et al., 2010; Kandziora et al., 2013; Haines-Young and Potchin, 2013). The cascade distinguishes a 'natural box' which comprises ecosystem structures, processes and biodiversity, and a 'human box' which comprises benefits, well-being, and values. The natural box represents the 'pure' ecological processes, which exist independently of possible benefits for humans (de Groot et al., 2010). Ecosystem services link the two boxes together by representing those ecosystem structures or processes that provide a benefit for human beings (figure 2).

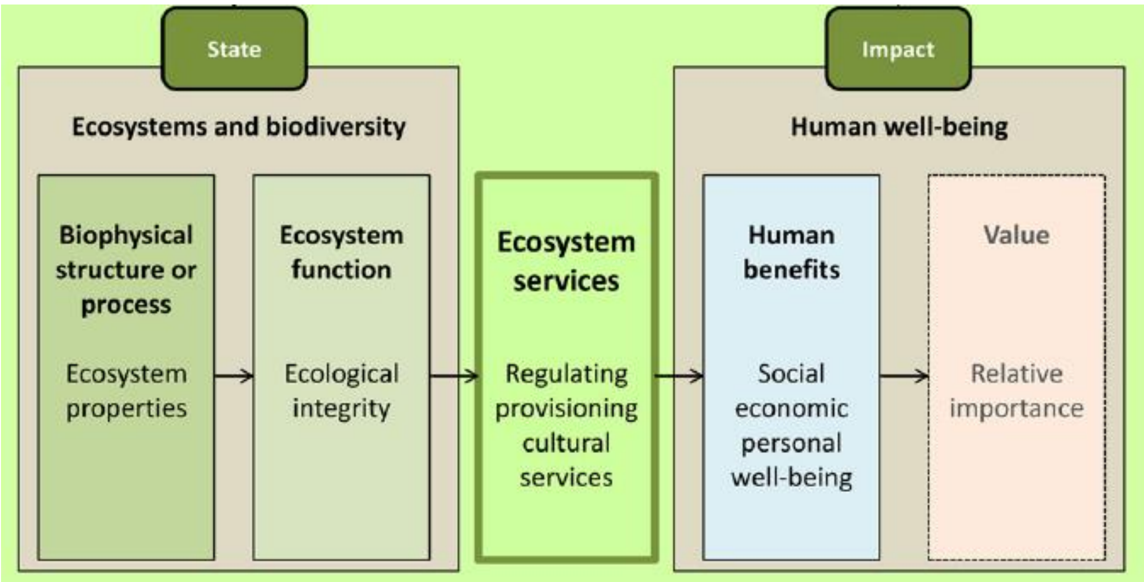


Figure 2. The 'ecosystem services cascade' (adapted from Kandziora et al., 2013)

The cascade starts with biophysical structures and processes (ecosystem properties), which determine the capacity of an ecosystem to deliver services. This capacity to deliver a service is called an ecosystem function (de Groot et al., 2010; Haines-Young and Potschin, 2010). The total of these functions constitute ecosystem functioning, in other words the maintenance of ecological integrity (MA, 2003). Ecological integrity represents the support and preservation of ecological structures and processes that are required for the self-organizing capacity of an ecosystem (Barkmann et al., 2001; Kandziora et al., 2013). These structures and processes are similar to the supporting and habitat services from the Millennium Ecosystem Assessment and the TEEB (MA, 2003; TEEB, 2010) (see section 3.1.2).

3.1.4 The influence of land use on ecosystem services

Land cover is part of the biophysical structure of ecosystems. These (bio)physical aspects of land in terms of rock, soil, vegetation, water bodies determine ecosystem services on the landscape level (see section 2.1). Human activities and man-made structures are not seen as ecosystem components and cannot provide ecosystem services, but can have an impact on ecosystem properties; for example by changing forest to grassland or by the input of fertilizer (see section 2.3).

Van Oudenhoven et al. (2012) adapted the ecosystem services cascade to assess land management impacts on ecosystem services (figure 3). Van Oudenhoven et al. (2012) define land management as the human activities that can affect ecosystem properties and function, as well as the ecosystem services that can be provided, and therefore can be seen as analogous to the definition of land use that is used in this research. Land use can influence different levels of the cascade: 1) by changing ecosystem properties, 2) by influencing ecosystem functioning, and 3) by determining the purpose of the land and thus which ecosystem functions are considered ecosystem services.

For example: 1) changing a diverse rainforest into a single-crop system, 2) increased fertilizer input changes the nutrient cycles and thereby ecosystem functioning, and 3) changing a production forest into a protected forest: in the first case the service that was primarily valued was timber production, while in the latter case, regulating or cultural services are valued most.

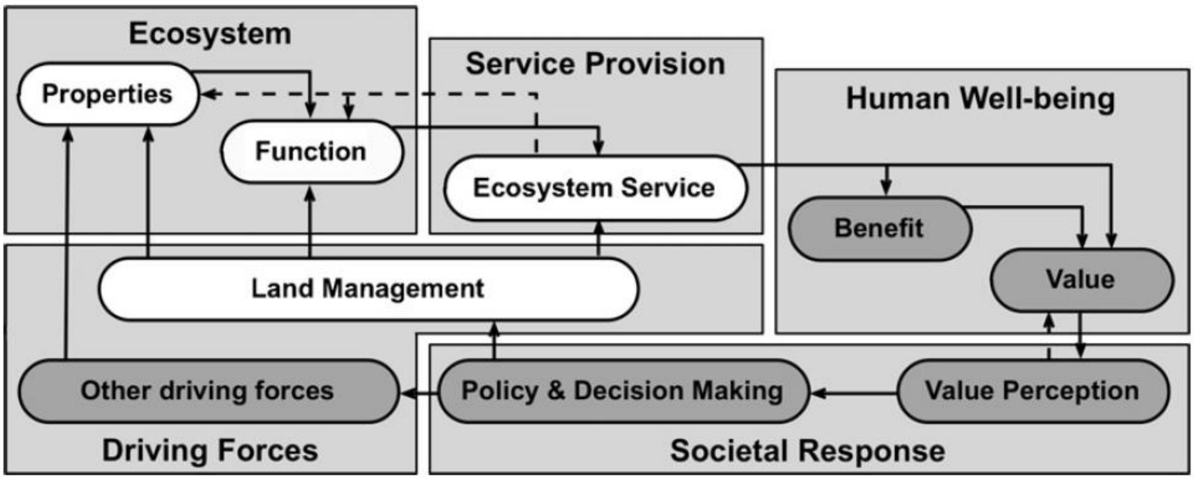


Figure 3. Framework to asses land management impact on ecosystem properties, functions and services (van Oudenhoven et al., 2012).

Generally, landscapes and ecosystems are multi-functional and can provide several services. However, different land cover types and ecosystems contain different components and processes and therefore can provide a different set of services, and the increase in one service often has a negative effect on other services (Haines-Young et al., 2012). The most important trade-off is between agricultural production and regulating services (Maes et al., 2011). The intensity of land use has a large influence on potential ecosystem services provision, which is a result of decreased 'stock' and diversity of ecosystem components (Costanza et al., 1997; Braat and ten Brink, 2008). The natural capital of biophysical structures and processes is necessary for the flow of materials (water and nutrients), energy and information (biodiversity) which constitute ecosystem functioning and integrity (see figure 2 and section 2.3).

Biodiversity is an important component of natural capital. The functional composition of organisms (identity, abundance and range of species traits) is often identified as the most important component of biodiversity for the delivery of ecosystem services, such as primary production and decomposition (Hooper et al., 2005a; Balvanera et al., 2006; Loreau et al., 2001; Naeem et al., 1994). In addition, genetic and species diversity directly provides several services, such as medicines, and the variety of food, fibers and other resources. Furthermore, landscape diversity and species diversity are valued highly for their cultural services (Kienast et al., 2009). Thus, land with high diversity and functional natural processes has a large capacity to deliver ecosystem services, while more intensively used land with low diversity and little natural processes has a low capacity to deliver services (Braat and ten Brink, 2008). A transformation from the former to the latter would constitute a large impact on potential ecosystem services supply.

3.2 Ecosystem services assessment

To evaluate the impact of land use and land management decisions on ecosystem services, ecosystem services provision of different land types need to be quantified. Ecosystem services assessments are done for different purposes and on several spatial scales: for one particular area or the whole world, for conservation or economic purposes, and for one, several or all ecosystem services. For example, if one is interested in climate change mitigation, land that stores large amounts of carbon should be conserved or if possible expanded (e.g. Houghton, 2003). Another application is to estimate the total economic or monetary value of ecosystems, where the aim is to value the natural capital and/or flow of a certain area (e.g. Costanza et al., 1997). In this way, ecosystems can be taken into account in economic analyses. For the application in LCA, a large scale assessment of all ecosystem services for all general land use types is of interest.

3.2.1 Ecosystem service indicators

To assess impacts on ecosystem services, not only a single indicator can be used, but a set of indicators is needed (van Oudenhoven et al., 2012). Because of the difficulties with classification of ecosystem services (as explained in section 3.1.2), and the complex relationship of land use and ecosystem services, there is no agreement yet on a set of indicators for the assessment of (global) land use (de Groot et al., 2010). Nevertheless, several publications include suggestions for ecosystem service indicators for land management (e.g. Burkhard et al., 2014; Maes et al., 2011; van Oudenhoven et al, 2012; de Groot et al., 2010; Kandziora et al., 2013; Tsonkova et al., 2014). These indicators are often classified as either capacity (state, integrity or function) indicators or flow (process or service) indicators. Capacity indicators represent the potential to provide services, while flow indicators represent the actual provision of a service experienced by people (Tsonkova et al., 2014). Most indicator sets include both capacity and flow indicators, as ecosystem services itself either depend on a stock (e.g. carbon storage) or flow (e.g. erosion prevention) (Tsonkova et al., 2014).

Some ecosystem services can be measured directly, which are mainly the provisioning services: e.g. harvested crops, wood or fiber, number of animals, and withdrawal of freshwater (Kandziora et al, 2013). Cultural services depend on human values and can only be estimated indirectly by for example the number of visitors or facilities, preference surveys, and number of protected species or habitats (Kandziora et al., 2013; de Groot et al., 2010). Regulating and supporting services are the most difficult to measure, because they often result from complex interactions of ecosystem properties, or take place on different spatial scales than the observed landscape scale (e.g. climate regulation or genepool protection). A list of potential indicators for regulating services is shown in table 5.

Table 5. Potential indicators for regulating ecosystem services (Kandziora et al., 2013)

Regulating service	**Potential indicators**	**Definition**
Global climate regulation	Source-sink of methane, carbon dioxide and water vapour (t C/ha*a) Amount of stored trace gases in marine systems, vegetation and soils (t C/ha)	Long-term storage of greenhouse gases in ecosystems.
Local climate regulation	Temperature (°C), albedo (%), precipitation (mm), wind (Bft), temperature amplitudes (°C), evapotranspiration (mm), shaded areas (ha, %)	Changes in local climate components like wind, precipitation, temperature, radiation due to ecosystem properties.
Air quality regulation	Leaf area index Air quality amplitudes (ppb) Air quality standards deviation (ppb) Level of pollutants in the air Critical loads (kg/ha*a) Difference between open land and throughfall (kg/ha*a)	Capturing/filtering of dust, chemicals and gases.
Water flow regulation	Groundwater recharge rate (mm/ha*a)	Maintaining of water cycle features (e.g. water storage and buffer, natural drainage, irrigation and drought prevention).
Water purification	Water quality indicators: Sediment load (g/l) Total dissolved solids (mg/l)	The capacity of an ecosystem to purify water, e.g. from sediments, pesticides, disease-causing microbes and pathogens.
Nutrient regulation	Water quality indicators, e.g. N (mg/l), P (mg/l) Leakage of nutrients (kg/ha*a) Electrical conductivity (μS/cm) Total dissolved solids (mg/l) Turnover rates of nutrients, e.g. N, P (y^{-1})	The capacity of an ecosystem to recycle nutrients, e.g. N, P.
Erosion regulation	Vegetation cover (%) Loss of soil particles by water and wind (kg/ha*a) USLE factors for assessment of landslide frequency (n/ha*a)	Soil retention and the capacity to prevent and mitigate soil erosion and landslides.
Natural hazard protection	Number of prevented hazards (n/a) Natural barriers (dunes, mangroves, wetlands, coral reefs) (%, ha)	Protection and mitigation of floods, storms (hurricanes, typhoons …), fires and avalanches.
Pollination	Species numbers and amount of pollinators (n/ha)	Bees, birds, bats, moths, flies, wind, non-flying animals contribute to the dispersal of seeds and the reproduction of lots of plants.
Pest and disease control	Populations of biological disease and pest control agents (n/ha)	The capacity of an ecosystem to control pests and diseases due to genetic variations of plants and animals making them less disease-prone and by actions of predators and parasites.
Regulation of waste	Amount and number of decomposers (n/ha) Decomposition rate (kg/ha*a)	The capacity of an ecosystem to filter and decompose organic material in water and soils.

Supporting services are often neglected in ecosystem services assessment, due to accounting problems (Müller et al., 2010). They are often not seen as delivering a direct benefit but considered as intermediate services and thus are left out of the assessment (see section 3.1.2). Tsonkova et al. (2014) developed an indicator set to represent the relevant ecosystem services by agro-ecosystems and includes several supporting ecosystem services (table 6). Unsustainable agriculture has large impacts on soil, water and biodiversity and therefore affects carbon sequestration, soil fertility, erosion control, water regulation, water quality, and habitat provision. The set of indicators by Tsonkova et al. (2014) is relevant for land use impact assessment, as agriculture comprises a large part of global land use (see section 2.4).

Table 6. Indicator set to assess relevant ecosystem services provided by agro-ecosystems (Tsonkova et al., 2014)

Ecosystem service	**Indicator**	**Definition**	**Area**
Carbon sequestration, Soil fertility	Carbon stock in soil ($t\ ha^{-1}$)	The quantity of organic C contained in the soil (0–30 cm)	Soil
Soil fertility	Nutrient use efficiency ($kg\ kg^{-1}$)	Total harvested biomass in dry matter (DM) produced per unit of nutrient assimilated	
Erosion control, Water quality	Erosion by water ($t\ ha^{-1}\ yr^{-1}$)	The amount of soil lost from a field by water runoff	
Erosion control	Erosion by wind (–)	Risk of soil loss due to wind	
Water regulation	Seepage rate ($mm\ yr^{-1}$)	The amount of water that leaves the rooting zone toward the groundwater table	Water
Water quality, Soil fertility	Nitrate concentration ($mg\ l^{-1}\ yr^{-1}$)	Potential nitrate concentration in seepage water	
Water quality, Soil fertility	Phosphorus loss ($kg\ ha^{-1}\ yr^{-1}$)	Potential amount of particulate phosphorus removed with soil lost by water erosion	
Habitat provision	Diversity of plant community (–)	The variety of plant community expressed as species richness and structural diversity	Biodiversity
Soil fertility, Water quality, Habitat provision	Plant protection products (%)	Application of PPP, e.g., herbicides, insecticides, and plant growth regulators on farm over the total management period	

In conclusion, many indicators for different ecosystem services exists, representing capacity or flow of ecosystem services. However, it is difficult to compose a set of indicators representing all ecosystem services, because of the complex interlinkages between services and therefore the possibility of double counting (see section 3.1.2). Also, which ecosystem components and processes are considered ecosystem services depends on the purpose of the assessment and value judgements.

3.2.2 Ecosystem service quantification

Many site-specific studies have been done which quantify and map the supply of several ecosystem services based on quantitative and spatially explicit data (e.g. Tsonkova et al., 2014; Remme et al., 2014; Kaiser et al., 2013; van Oudenhoven et al., 2012). Also, several regional studies have quantified ecosystem services, often using geo-referenced data which results in spatial representation of ecosystem services supply (e.g. Egoh et al., 2008; Chan et al., 2006). However, a quantitative study on a global or continental scale remains challenging because of the lack of data and difficulties in upscaling results of regional studies (Kienast et al., 2009). Naidoo et al. (2008) were able to map only four ecosystem services at a global scale (carbon sequestration, carbon storage, grassland production of livestock, and water provision). Increasingly, expert knowledge is being applied in addition to quantitative data to fill the gap that exists between general land properties and the ability to provide services (Koschke et al., 2012; Kienast et al., 2009; Maes et al., 2011; Burkhard et al., 2009).

Some studies use binary links to indicate the ability of a land use type to provide certain services: a land characteristic or land use type gets the value 0 when it has a neutral role and a value 1 if it has a supportive role for an ecosystem service (Kienast et al., 2009; Maes et al., 2011). Kienast et al. (2009) applied binary links to the land characteristics of 581 administrative units of Europe which resulted in a spatially explicit landscape function assessment. 15 production, regulation, habitat and information functions were considered. Land characteristics existed mostly of CORINE land cover classes, complemented with several other parameter such as net primary production, heterogeneity of land use classes, and forest patch heterogeneity index. Linking landscape functions with land characteristics was done coarsely and intuitively on a European scale. However, it does assess many different ecosystem services, as opposed to more sophisticated, complex models that can often only include a few provisioning or regulating services. The authors suggest that additional quality data could refine the results, for example willingness to pay information or land prices. A subsequent research used the classification of CICES and a finer spatial scale to assess crop-based production, wildlife products, habitat diversity, and recreation (Haines-Young et al., 2012).

Burkhard et al. (2009) used a scale of 0 to 5 to indicate the relevance of CORINE land cover types for 29 ecosystem services, where 0 indicates no relevant capacity, and 5 indicates high relevant capacity for ecosystem service supply. Definitions and indicators for ecosystem services were formulated and scores were given on the basis of expert evaluations in combination with quantitative data. The normalization to the relative 0-5 scale aims at making different ecosystem services, measured and assessed by various indicators and units, comparable with each other. The normalized scale can also be used as a representative for biogeophysical or currency units, for which the formulated indicators can be used (Burkhard et al., 2009). It is emphasized that land cover is only one aspect of ecosystems, and therefore only a proxy indicator. The integration of more comprehensive quantification methods (models, measurements, statistics, surveys) and geobiophysical, land use intensity and socio-economic date will improve the reliability of the results (Burkhard et al., 2014).

The ecosystem service 'matrix method' of Burkhard et al. (2009) has been applied to quantify and map ecosystem services in several case studies (e.g. Burkhard et al., 2012; Kaiser et al., 2013) and inspired the development of other ecosystem service mapping studies, e.g. by the European Commission (Maes et al., 2011). The European Commission have applied a similar approach as Kienast et al. (2009) and Burhard et al. (2009) and apply a binary score to CORINE land cover classes that are thought to be related to an ecosystem service (Maes et al., 2011). Maes et al. (2011) use only quantitative CORINE data, which is linked to spatially explicit indicators for 13 ecosystem services.

The ecosystem matrix method has been improved in 2012 and 2014 (Burkhard et al., 2012; 2014). In 2015, the values were updated by using data and expert judgment of 28 European sites belonging to the Long-Term Ecological Research network (LTER) and made it possible to indicate the range of values for each entry (Stoll et al., 2015). Their results suggest that the matrix approach to assess ecosystem services principally works on broad spatial scales, and therefore is suitable to be used in a life cycle impact assessment method.

3.2.3 Ecological integrity indicators

It is possible instead of using indicators representing ecosystem services, to use indicators for ecosystem functioning or ecological integrity. Land use strongly influences ecosystem properties and ecosystem functioning: it determines the biophysical stuctures of an ecosystem and influences the flow of materials, energy and information (see section 2.3 and 3.1.3). It therefore makes sense to use an indicator set that represents ecological functioning or integrity when assessing land use impact, as it represent the potential of an ecosystem to deliver services (Müller and Burkhard, 2010) (see section 3.1.3). Moreover, a set of ecological integrity indicators is smaller than a set of ecosystem services indicators.

The term ecological integrity has been used by Leopold (1949) in his definition of 'land ethic': "A thing is right when it tends to preserve the integrity, stability and beauty of the biotic community. It is wrong when it tends otherwise". Ecological integrity has since been used as a synonym to 'naturalness' or ecosystem health, but has evolved to incorporate those aspects of ecosystems that maintain its stability and functionality (Martin-López et al., 2009). Ecological integrity incorporates aspects such as biodiversity, stability, and sustainability which are necessary for functioning of an ecosystem (Martin-López et al., 2009; Barkmann et al., 2001).

Müller (2005) composed a set of indicators that represent five components of ecological integrity; biotic structures, abiotic structures, energy balance, water balance, and matter balance (table 7). This indicator set represents the degree of self-organization in an ecosystem and can be used to depict the generalized potential to provide ecosystem services (Müller, 2005). It is based on the definition of Barkmann et al. (2001): ecological integrity is the "support and preservation of those processes and structures which are essential prerequisites of the ecological ability for self-organization of ecosystems". Self-organization is interpreted as the spontaneous process of gaining order by the emerging arrangement of the ecosystem components and processes, which results in a specific performance and development of the system's functions (Müller, 2005). Ecological integrity therefore has structural and functional aspects that determine the whole ensemble of flows, storages, and regulations. It is mainly based on variables of energy and matter budgets and structural features of whole ecosystems (Burkhard et al., 2009; Müller, 2005). Human influences, such as land use, can constrain this self-organization and therefore affect ecological integrity.

Table 7. Ecological integrity indicators (Müller, 2005; Müller and Burkhard, 2007; Kandziora et al., 2013)

Orientor group	Integrity indicators	Potential key variables (Müller, 2005; Müller and Burkhard, 2007; Kandziora et al., 2013)
Energy balance	Exergy capture	Gross or Net Primary Production (GPP, NPP), leaf area index (LAI)
	Entropy production	C/yr from respiration and evapotranspiration, entropy balance, entropy production after Aoki, entropy production after Svirezhev and Steinborn
	Metabolic efficiency	Respiration/biomass (low energy needed to maintain biomass indicates efficient ecosystem)
Matter balance	Storage capacity	N/C_{org} in soil, N/C in biomass
	Nutrient loss reduction	Leaching of nutrients
Water balance	Biotic water flows (water uptake by plants)	Transpiration/total evapotranspiration
Abiotic structures	Biotope heterogeneity	Abiotic habitat components' heterogeneity indices, nr/area of habitants
Biotic structures	Biodiversity	Number and identity of (selected) species

As ecological integrity can be seen as synonym to supporting services, similar indicators can be used. The indicator set by Müller (2005) shows resemblance to the agro-ecosystems ecosystem services indicator set by Tsonkova et al. (2014) (table 6); they both include indicators for storage, efficiency, nutrient loss, and biodiversity. Also, several ecological integrity indicators are similar to indicators or variables for regulating services; e.g. water and carbon storage, net primary productivity, biodiversity, turnover rates and nutrient loss (see table 5). A matrix interrelation with the four categories of ecosystems services by Müller and Burkhard (2007) confirms that the set of ecological integrity represents all categories of ecosystem services included in the Millennium Ecosystem Assessment (see table 4), especially supporting and regulating services (figure 4). Cultural services are in a lesser degree represented, but because supporting services are a prerequisite for regulating, provisioning and cultural services, ecological integrity is a good way of representing all ecosystem services.

		Ecosystem services			
		Supporting services	Provisioning services	Regulating services	Cultural services
Components of ecological integrity	Exergy capture	O	O	O	
	Exergy dissipation	O		O	
	Biotic waterflows	O	O	O	
	Metabolic efficiency	O		O	
	Nutrient loss	O	O	O	
	Storage capacity	O	O	O	
	Biotic diversity	O	O	O	O
	Organisation	O	O	O	O

Figure 4. Matrix showing the interrelations between ecological integrity indicators and ecosystem services (Müller and Burkhard, 2007)

3.2.4 Ecological integrity quantification

Quantification of ecological integrity indicators faces similar quantification challenges as ecosystem services indicators due to complex interrelations and difficulty of global assessment (see section 3.2.2). However, ecological integrity includes purely ecological aspects, and can therefore be assessed more objectively than ecosystem services (Clerici et al., 2014). Moreover, less data needs to be collected due to the small indicator set, and the ecological integrity components are generally better-studied than ecosystem services (Stoll et al., 2015).

The ecological integrity indicators have been applied to several case studies for which quantitative date was available (e.g. Burkhard et al., 2011; Burkhard and Müller, 2008; Nunneri et al., 2007). On larger spatial scales, the 'matrix approach' of Burkhard et al. (2009) has been used to estimate ecological integrity values for the different land use types (Burkhard et al., 2009;2012;2014). The latest application of the matrix approach by Stoll et al. (2015) estimates all 8 ecological integrity indicators by Müller for 44 CORINE land types on the third hierarchical level. Quantification was based on quantitative data and expert judgment from the European Long-Term Ecological Research (LTER-Europe) network (28 LTER sites from 11 countries). The results indicate that ecological integrity is highest for forest land types and other natural land types, and low for artificial land types and sparsely vegetated areas (Stoll et al., 2015).

Stoll et al. (2015) analysed the expert values for heterogeneity within the different land types. Most artificial land types showed little heterogeneity which indicates that artificial land types are generally valued similarly among Europe. In general it was concluded that heterogeneity of ecosystem service provision and ecological integrity increases with increasing 'naturalness' of the land cover classes. Stoll et al. (2015) suggest that this can be explained by variable exploitation and management options of natural systems, whereas agricultural and artificial systems are often fully exploited. Moreover, less knowledge is available for natural systems. Most ecological integrity indicators showed little heterogeneity of expert judgement values, in constrast to several ecosystem services, except abiotic heterogeneity and biodiversity. Stoll et al. (2015) suggest that this is because ecological integrity components are generally well studied and clearly defined. This suggests that ecological integrity indicators are more suitable for assessments on large spatial scales than ecosystem services indicators.

The European Commission has used the ecological integrity indicators to represent the capacity of land to provide supporting, regulating and provisioning services (Clerici et al., 2014). The matrix model was applied to assess the consequences of land cover change on ecological integrity of riparian zones (Clerici et al., 2014). They recognize that proxy-based methods, such as the matrix model, are generally unsuitable for identifying hotspots or priority areas for multiple services, but mostly appropriate for identifying broad-scale trends of ecosystem services (Eigenbrod et al., 2010). Such methods are especially valuable when deriving trends on a large scale, if the collection of primary data is not feasible or if other techniques would bring an undesired level of complexity (Clerici et al., 2014). For the assessment of land occupation and transformation in LCA therefore a similar approach can be taken.

In conclusion, ecological integrity can be evaluated by a small set of ecological indicators that represents the functioning of an ecosystem and thus its potential to provide ecosystem services (see section 3.1.3). Land use strongly influences ecological integrity by altering energy, matter and information fluxes (see section 2.3). The 'matrix approach' provides estimates of the ecological indicator values for CORINE land types wich can be used for large scale assessment of land use impacts, which fits the scale of LCA (Clerici et al., 2014). The following chapters 4 and 5 explain what LCA is and how land use is incorporated in LCA, in order to learn how ecological integrity can be included in land use assessment methods.

4. Life Cycle Assessment

This chapter briefly explains what LCA is, by whom and for what goals it is being used, and explains the different phases of an LCA. The focus is on the impact assessment phase, as the goal of this research is to develop a new impact assessment method. The information in this chapter is based on the International reference Life Cycle Data System (ILCD) handbooks by the Joint Research Centre - Institute for Environment and Sustainability of the European Commission (EC-JRC, 2010a; 2010b; 2011).

4.1 LCA overview

Life Cycle Assessment (LCA) is a structured, comprehensive and internationally standardized method that is widely applied to assess the potential environmental impacts of the entire life cycle of products (both goods and services); their production, transport, use, and waste management. It quantifies all relevant emissions and consumed resources and calculates the related environmental and health impacts and resource depletion issues that are associated with a product. The ISO 14040 and 14044 standards provide the framework for LCA. Additionally, several guidelines have been developed, such as the ILCD handbooks by the European Commission (EC-JRC 2010a; 2010b, 2011), a guide to the ISO standards by Guinée (2001), and guidelines by the UNEP/SETAC Life Cycle Initiative (e.g. on land use assessment: Milà i Canals et al., 2007; Koellner et al., 2013).

LCA is mainly used for product development, product improvement, product comparisons, decision making and communication - with the ultimate goal of minimizing environmental impacts. An LCA can help to identify opportunities to improve the environmental performance of a process or product by pointing out the most influential life cycle stages and processes. An LCA consists of four stages: 1) goal and scope definition, 2) inventory analysis, 3) impact assessment and 4) interpretation (figure 5). These stages will be subsequently explained, but it should be kept in mind that an LCA study is almost always an iterative process.

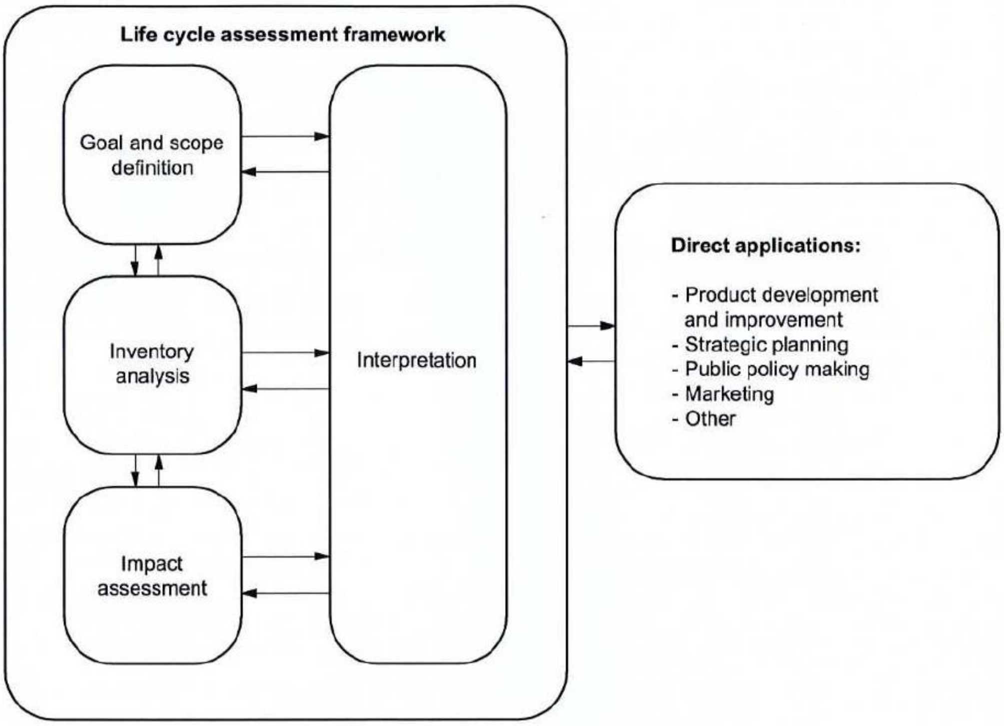


Figure 5. Phases of Life Cycle Assessment (EC-JRC, 2010a)

4.2 Goal and scope definition

The goal and scope definition is decisive for all the other phases of the LCA. The decision-context and intended application(s) of the study are identified and the targeted audience are named. The scope definition defines the product of the study and the system boundaries, and determines the requirements for the LCA methodology; allocation procedures, data requirements, impacts to be evaluated, and the methodology applied for assessing those impacts. A functional unit is clearly defined which represents the qualitative and quantitative aspects of the function of the product. This function could therefore also be fulfilled by an alternative product or process, the definition in the form of a functional unit enables comparison between alternatives. The reference flow represents the specific product under study to which all the input and output flows will be related in the inventory phase. The system boundaries define which parts of the life cycle and which processes belong to the analysed system and therefore for which processes data needs to be collected. The type of LCA that is executed also determines data requirements; attributional life cycle assessment focuses on describing the environmentally relevant physical flows to and from a product or process within the system boundary, while consequential assessment describes how relevant environmental flows will change in response to possible decisions. The system boundaries of a consequential LCA may therefore be beyond the cradle-to-grave system being investigated, depending on the choice made in the goal and scope definition. Consequential LCA includes economic concepts and dynamic models and is therefore more complex and includes more assumptions than attributional modelling.

4.3 Inventory analysis

During the Life Cycle Inventory (LCI) phase the data collection and modelling of the system is done in line with the goal definition and meeting the requirements derived in the scope phase. The LCI results are the input to the subsequent impact assessment phase. The results of the LCI work also provide feedback to the scope phase, as initial scope settings often needs adjustments. The LCI should inform on all input/output data within the system boundaries and throughout the different life cycle stages that are assessed.

4.4 Impact assessment

The ISO 14044 standard defines Life Cycle Impact Assessment (LCIA) as the phase of life cycle assessment aimed at understanding and evaluating the magnitude and significance of the potential environmental impacts of a product system (ISO 14044:2006). In this phase, the inputs and outputs of elementary flows that have been collected and reported in the inventory phase are assigned to translated into impact indicator results, by multiplying them with the characterization factors provided by a chosen LCIA method (figure 6). According to ISO 14044, the indicator of an impact category can be chosen anywhere along the impact pathway, which links inventory data to the 'Areas of Protection' (AoPs). The Areas of Protection that should be considered are 'Human health', 'Natural environment', and 'Natural resources'. Indicators that directly model damage to an AoP are called endpoint indicators, while characterization at midpoint level models envrionmental impact using an indicator located somewhere along the mechanism.

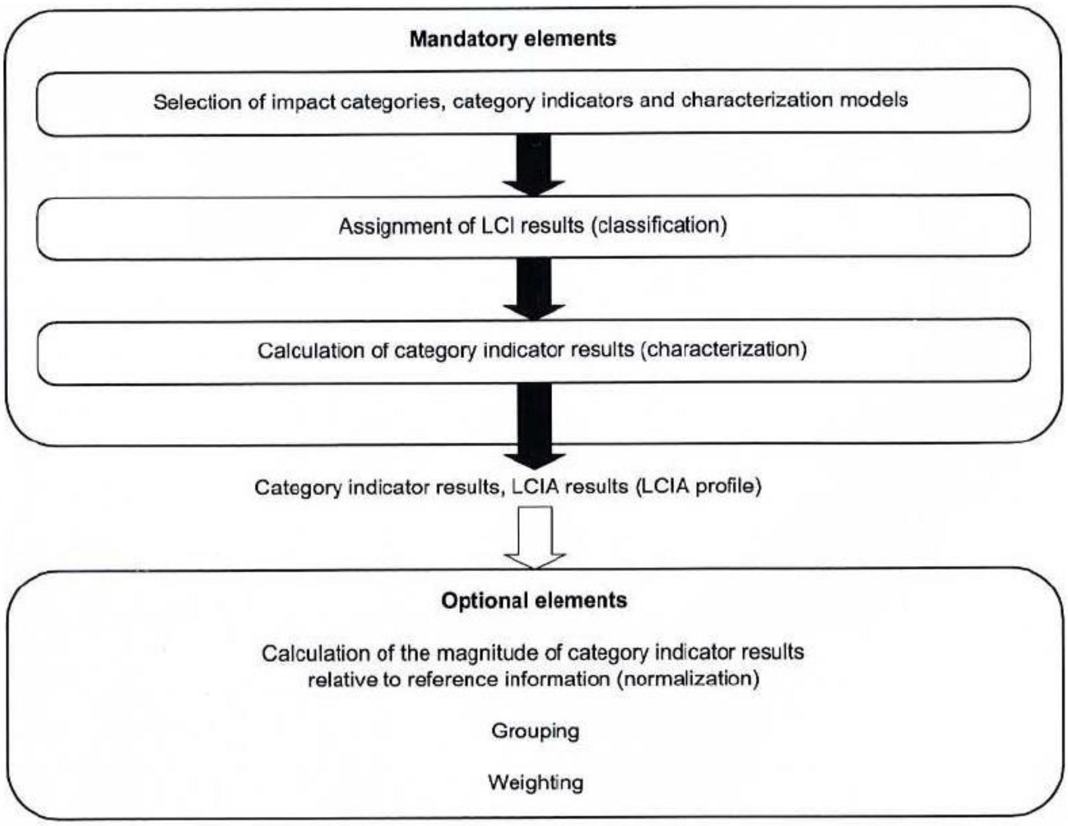


Figure 6. Elements of the LCIA-phase (ISO 14040:2006)

Impact categories at midpoint level are defined at the place where a common mechanism for a variety of substances within that specific impact category exists. The selection of impact categories must be comprehensive in the sense that they cover all relevant environmental issues related to the analyzed system. The ILCD recommends the use of the following impact categories at midpoint level: 'climate change', 'ozone depletion', 'human toxicity', 'respiratory inorganics', 'ionizing radiation', 'photochemical ozone formation', 'acidification', 'eutrophication', 'ecotoxicity', 'land use', and 'resource depletion (minerals, fossil and renewable energy resources, water)'.

Endpoint modeling directly quantifies the damage to the Areas of Protection, for example damage to human health in Disability Adjusted Life Years (DALY), which represents a decrease in quality of life and life expectancy. Damage to the natural environment should reflect changes to the function and structure of ecosystems. The complexity of ecosystems and the different levels of biodiversity make it a challenging task to include all damage to ecosystems. It is recommended by the ILCD to use the Potentially Disappeared Fraction of Species (PDF) concept as an endpoint indicator, because this is an ecological indicator which is sufficiently mature for application in LCIA. The ILCD does not give recommendations for endpoint indicators for the AoP Natural Resources. The AoP Natural Resources has strong links with the AoPs Human Health and Natural Environment, as the extraction and use of resources has impacts on the other AoPs. Often indicators are used that represent the effort needed to safeguard the availability of resources and focus on the use value for humans, thereby excluding its non-use and intrinsic value.

To support interpretation of the indicator results, normalization and weighting may be applied, although it is not required according to ISO standards. LCIA results can be multiplied with normalization factors that represent the overall inventory of a reference (e.g. a whole country or an average citizen), obtaining dimensionless, normalized LCIA results. This way the results of the different impact categories can be compared with each other. In a second step these normalized LCIA results can be multiplied by a set of weighting factors, that indicate the relevance that the different impact categories or AoP may have. These normalized and weighted LCIA results can be summed up to a single score. A weighting set always involves value choices, which may be based on scientific information or on political, economic or cultural considerations. The weighting set must be justified and documented in the scope phase and be in line with the application of the study and its target audience.

Since the early 1990s, numerous LCIA methods have been developed. The UNEP/SETAC Life Cycle Initiative and other organizations such as the United States Environmental Protection Agency (US EPA) and the European Commission have worked on finding consensus and recommendations on best practice and standardizations. In chapter 5, guidelines and state of the art methods for land use impact assessment methods are described.

4.5 Interpretation

An LCA concludes with interpretation of the results, which takes into account uncertainties and the (value) choices that were made in the previous steps. The results of the LCA are analyzed for accuracy, completeness and sensitivity. In light of the goal and scope definition and taking into account the assumptions and limitations, conclusions and recommendations are made.

5. Land use in LCA

Chapter 3 described that land use is an important driver of various environmental impacts, and especially has large effects on biodiversity and ecosystem services. However, not all of these impacts can be accurately measured and evaluated with LCA yet. This chapter explains how land use is included in inventory databases (section 5.1), how land occupation and transformation impact can be calculated (section 5.2), and how land use impacts are being assessed in state of the art LCIA methods (section 5.3). This enabled identification of the remaining gaps in land use impact assessment on which the development of a new method was focused.

5.1 Inventory

In life cycle inventory databases, land occupation and transformation are elementary flows which are noted separately (Koellner et al., 2013a). In the LCI phase, information is collected on the type, area and duration of occupations, and the type of land before and after transformation, and the area that is transformed. The chosen impact assessment method determines the exact data requirements (EC-JRC, 2010b). Different LCI databases exist, such as the European reference Life Cycle Database (ELCD) developed by the EC-JRC, but the most commonly used and most transparent database is ecoinvent.

Ecoinvent uses the CORINE land classification with maximum four levels of detail, based on the recommendations of the UNEP/SETAC Life Cycle Initiative (Weidema et al., 2013). Land occupation is recorded in m^2year of a certain land use type (e.g. annual crop 100 m^2year). Land transformation consists of two factors: a transformation from a certain type in m^2 (e.g. *transformation from forest* in 100 m^2) and a transformation to another type in m^2 (e.g. *transformation to annual crop* in 100 m^2) (Weidema et al., 2013). Sometimes the previous land type is unknown, then a reference land type is used that is determined by the chosen LCIA method. When a land transformation was a long time ago, a decision needs to be made if and how it should be allocated to the current land use (Goedkoop et al., 2013). Another issue is the allocation of transformation to a production system when many products are being produced per year and area, it that case the transformation factor may become close to zero. It is suggested by Koellner et al. (2013b) to allocate transformation impacts over a period of twenty years.

Several other assumptions are made in the inventory phase, e.g. that discrete land cover types can distinguished, and that the transformation from a certain type to another time does not take any time (Koellner et al., 2013b). Because occupation is expressed in m^2year, time and area are substitutable, although in reality it can make a difference for ecological impact if a small area is used for a long time or a large area for a short time, or one single large plot instead of many smaller ones (Koellner and Scholz, 2007; Koellner et al., 2013b).

5.2 Impact Assessment

Different methods exist for the calculation of land use impacts. The ILCD handbook includes a framework and requirements for LCIA models and indicators (EC-JRC, 2010c). Several criteria for the land use impact category need to be taken into account besides the general criteria for impact categories, such as environmental relevance and scientific robustness. A land use impact assessment method should be based on a specific underlying environmental model, consider both land transformation and occupation, and include changes to all relevant aspects of land (see section 5.2.1).

5.2.1 Indicators

Land use impact is considered as a decrease in ecosystem quality compared to a certain natural reference state. Ecosystem quality of a certain land use type can be quantified by an appropriate indicator along a relevant environmental pathway, either at midpoint or endpoint level (ISO 14044:2006). The ILCD sees several impacts as relevant for determining land use impacts, such as changes to habitat and biodiversity, soil, net primary production, and climate change related changes (EC-JRC, 2010c). The EC-JRC and UNEP/SETAC both see the pathways identified by Milà i Canals et al. (2007a) as most relevant: biotic production potential, biodiversity and ecological soil quality. Examples of indicators for these pathways are primary productivity, species loss, and soil organic matter (SOM). In order to incorporate all of the relevant impacts of land use, it is suggested by the UNEP/SETAC to include ecosystem services in a land use impact assessment method (Koellner et al., 2013b).

5.2.2 Calculation of land use impacts

Because land use impacts are complex (see section 2.3), several assumptions need to be made in order to calculate land use impacts. The UNEP/SETAC Life Cycle Initiative makes the following assumptions: the quality of an area under occupation remains constant, ecological impact increases linearly with the area and time of occupation and area transformed, and the drivers of ecosystem quality loss do not interact (e.g. climate change and land use) (Koellner et al., 2013b). Furthermore, regeneration to a natural state after land use ends is linear and independent from the land use history (Koellner et al., 2013b). Often it is assumed that land can recovery towards its original quality, although in reality, the restored land might be of lower quality or regeneration takes a longer time than the modelling period of the LCA. It that case, the UNEP/SETAC advises to use a maximum regeneration time of 500 years, and include possible further impacts as permanent impacts, although permanent transformation impacts are often neglected (e.g. de Baan et al. 2013; de Souza et al., 2013; Milà i Canals et al., 2013). There is much uncertainty and complexity related to regeneration time and it is therefore still under much debate (de Baan et al., 2013; Koellner et al., 2013b). Which regeneration times are used for the different land use types, differs per LCIA method.

5.2.2.1 Land occupation impact

For the calculation of land use impacts the method by Koellner et al. (2013b) is followed in this research. To calculate occupation characterization factors (CFs), information is required on the value of a quality indicator for each land use type and a reference situation. The occupation characterization factors (CF_{occ}) indicate the difference in ecosystem quality (ΔQ) of the land type of interest and a reference land type, and are expressed in $\Delta Q/m^2yr$. The impact of land occupation is calculated by multiplying the characterization factors by the inventory data (in m^2year). This results in the occupation impact expressed as the decrease in the chosen ecosystem quality indicator compared to a reference situation. The UNEP/SETAC recommends to use the predominant (quasi-)natural land cover as a reference when assessing land use impact on a global scale (Koellner et al., 2013b). The potential natural vegetation (PNV) can be seen as a prediction based on the most mature vegetation stage that can currently be observed in a given site (Chiarucci et al., 2010).

5.2.2.2 Land transformation impact

Land transformation impact is calculated from information on ecosystem quality and the duration of regeneration after land use has ended. In ecoinvent, transformation is split into two interventions: 1) *transformation, from land type A* (in m^2), and 2) *transformation, to land type B* (in m^2) (figure 7). They can be seen as *transformation from A to reference* and *transformation from reference to B*. As a reference a natural land type with no impact is commonly used (Koellner et al., 2013b). The impact of each of these two transformation interventions is calculated by the difference in quality between land type A/B and the reference land type, integrated over the regeneration time from land use type A/B to the reference land type, multiplied by the transformed area in m^2.

The *transformation from A to reference* represents the regeneration of land use type A to a natural state, which results in an increase in ecosystem quality and thus is a negative impact. The *transformation from reference to B* represents a decrease in ecosystem quality and thus an environmental impact. To calculate the net transformation impact (TI_{net}), the negative impact of a transformation from A to reference ($TI_{A \to ref}$) and the impact of a transformation from the reference to B ($TI_{ref \to B}$) are added up:

$TI_{net} = TI_{A \to ref} + TI_{ref \to B}$

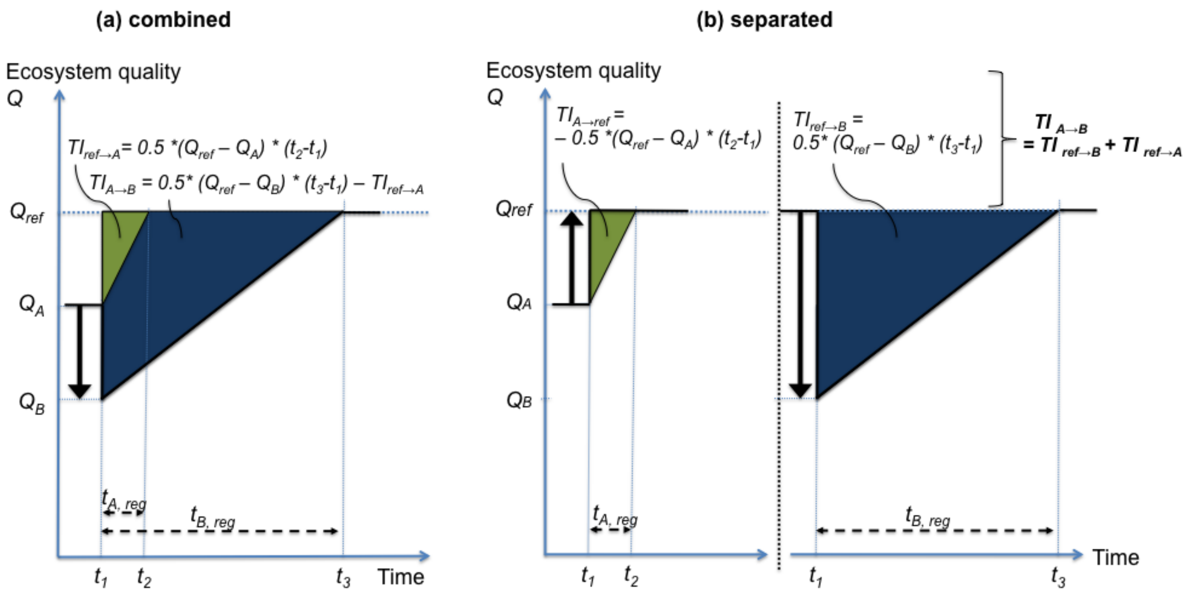


Figure 7. Transformation from land type A to land type B, calculated as (a) a combined impact or (b) two separated impacts (Koellner et al., 2013b)

Figure 7 depicts the calculation of net transformation impact. Transformation from ecosystem quality A to a lower ecosystem quality B is assumed to occur instantaneously at t_1. No occupation takes place, thus regeneration starts immediately after the transformation and it takes some time ($t_{B,reg}$) for the land to reach the quality of the reference situation. Regeneration is assumed to follow a linear trajectory and thus transformation impact of land intervention *transformation from baseline to B* can be calculated by multiplying the surface area of the 'triangle' of ΔQ*regeneration time with the transformed area (the area represents a third dimension, which is not shown in figure 7):

$TI_{ref \to B} = 0.5*(Q_{ref} - Q_{LUB})* t_{LUB,\ reg}*$ area

The green triangle that represents the *transformation from A to baseline* needs to be subtracted from the transformation impact. The transformation characterization factors in $\Delta Q*t_{reg}/m^2$ can be determined by multiplying CF_{occ} with 0.5*regeneration time, where the *transformations from* get a negative value.

5.3 LCIA methods

The EC-JRC reviewed a number of land use assessment methods based on their criteria (EC-JRC, 2011). Some midpoint methods use indicators like soil structure, soil pH or soil organic matter. Endpoint characterization factors mostly refer to the amount of species lost or to the change in Net Primary Productivity (NPP) of the land. The EC-JRC selected three midpoint methods that were reviewed in more detail; ReCiPe v1.05 (not based on a specific underlying model), the method by Milà i Canals et al. (2007b) (based on soil organic matter), and the method by Baitz (Bos and Wittstock (2009) (based on seven quality indicators). Five endpoint methods were reviewed, which were based on species diversity loss (e.g. Eco-Indicator 99 and ReCiPe) and in two cases on production of wood.

The EC-JRC found that in general, land use methods do not score highly against the formulated criteria. The method by Milà i Canals et al. (2007b) is recommended as the most suitable midpoint method by the EC-JRC, because it includes a sophisticated calculation involving a relevant aspect of ecosystem quality, although the scope should widen to include impacts on biodiversity. Also, the inclusion of an operational set of characterization factors would improve applicability for the LCA practicioner, instead of having to calculate the characterization factors her or himself.

All endpoint models are seen as too immature, but ReCiPe is recommended as an interim solution as it includes occupation and transformation effects and distinguishes between several different land use types and intensities. It could be improved by including CFs for more land use types and intensities of use and by including impacts on primary productivity. Furthermore, it would be an improvement if globally applicable characterization factors would be developed, by basing them on global input data and additional land use types, instead of the current European focus of impact assessment methods. The two recommended methods are further discussed in the following sections (5.3.1 and 5.3.2). In section 5.3.3 the recommendations of the UNEP/SETAC are described, as well as their work on the development of characterization factors for several ecosystem services.

5.3.1 Milà i Canals et al. (2007b)

The method by Milà i Canals et al. (2007b) was developed in the context of the UNEP/SETAC Life Cycle Initiative. The UNEP/SETAC developed a framework for assessing land use impacts, as there was no method yet to assess land use impacts on biodiversity, biotic production and soil quality (Milà i Canals et al., 2007a). The land use assessment method that was developed in the same year, described by Milà i Canals et al. (2007b), focused on the impacts of land occupation and transformation on soil quality within the AoP Natural Environment. Soil quality is an important aspect of life support functions, and was therefore selected as the subject of the impact pathway, measured by the indicator soil organic carbon. Life support functions can be seen as analogous to regulating and supporting ecosystem services according to Milà i Canals et al. (2007b), and therefore impacts on biodiversity and other life support functions are lacking from this method. These should be assessed in parallel, although no midpoint methods exist to date that do this.

5.3.2 ReCiPe endpoint method

ReCiPe[2] is a state of the art LCIA method with a wide range of impact categories and the possibility for midpoint and endpoint assessment. Land occupation and transformation are part of the Area of Protection Ecosystems, divided into three midpoint impact categories: agricultural land occupation, urban land occupation and natural land transformation. No further distinction between the quality of different land use types is made at midpoint level.

[2] The acronym ReCiPe represents the initials of the main developers of the method: RIVM, Radboud Universty, CML and PRé

At endpoint level, ReCiPe differentiates between the impact of different land use types according to Potentially Disappeared Fraction of species (PDF). The damage to the AoP Ecosystems is calculated by multiplying the PDF with the LCI parameter and the species density (SD), and in case of transformation, with the regeneration time. Species density is calculated by the species-area relationship, based on number of plant species per area. Transformation is only deemed relevant for natural systems, therefore transformation CFs are only developed for the natural land types (Goedkoop et al., 2013).

ReCiPe includes three different cultural perspectives (individualist, hierarchist and egalitarian) in order to incorporate uncertainties related to assumptions. For example, when a new occupation by agricultural land is located next to existing agricultural land, the species number decreases relative to a reference land type, but the expansion of agricultural area can have positive effect on the local plant species due to the species-area relationship. The hierarchist and egalitarian perspectives do not take into account such possible positive effects of expansion of a certain land type, while the individualist does. The main assumption related to transformation impacts also differs per cultural perspective: the hierarchist and individualist assume regeneration times of 100 years for forests, shrubs and hedgerows, and 3300 for tropical forest. For the egalitarian perspective longer regeneration times are assumed: 200 years for forests, shrubs and hedgerows and 10,000 for tropical forest (Goedkoop et al., 2013).

5.3.3 UNEP/SETAC Life Cycle Initiative

The review of the EC-JRC (2011) indicated several issues with state of the art methods that urgently need improvement and this is confirmed by the UNEP/SETAC Life Cycle Initiative (Koellner et al., 2013b). Firstly, all relevant impact pathways should be included, covering biodiversity and ecosystem services. Second, the method should be easily applicable and therefore include characterization factors that are compatible with LCI databases, instead of requiring the user to make calculations (like is the case with the method of Milà i Canals et al. (2007b)). Furthermore, these characterization factors should preferably be globally applicable, in line with the goal of LCA. In addition, because land use impact depend on local conditions, biogeographical differentiation is recommended to capture part of the variability. Lastly, differentiation between land use types with several levels of intensity of use is needed, in order to evaluate impacts of different land management practices (Koellner et al., 2013a).

There have been several developments, mainly within the project land use in LCIA (LULCIA) of the Life Cycle Initiative, that addressed these four points of improvement. This resulted in characterization factors, differentiated per biome, to evaluate land use impacts on biodiversity, biotic production, erosion regulation potential, freshwater regulation potential and water purification potential (de Baan et al. 2013; Brandão and Milà i Canals 2013; Saad et al. 2013). According to the cause-effect chain for land use impacts from Koellner et al. (2013b), all relevant ecosystem services are covered by the recent developments of the UNEP/SETAC Life Cycle Initiative (figure 8). However, several ecosystem services as defined by CICES are still missing; e.g. disease regulation, pollination, habitat provision, protection against floods and storms, and cultural services (see section 5.3.4). Moreover, by developing characterization factors for each of the ecosystem services separately, interdependencies are neglected and double counting can occur (see section 4.2). Furthermore, only for biotic production (measured by soil organic carbon) a distinction is made between several agricultural types, all other developments include only characterization factors on roughly the second hierarchical level.

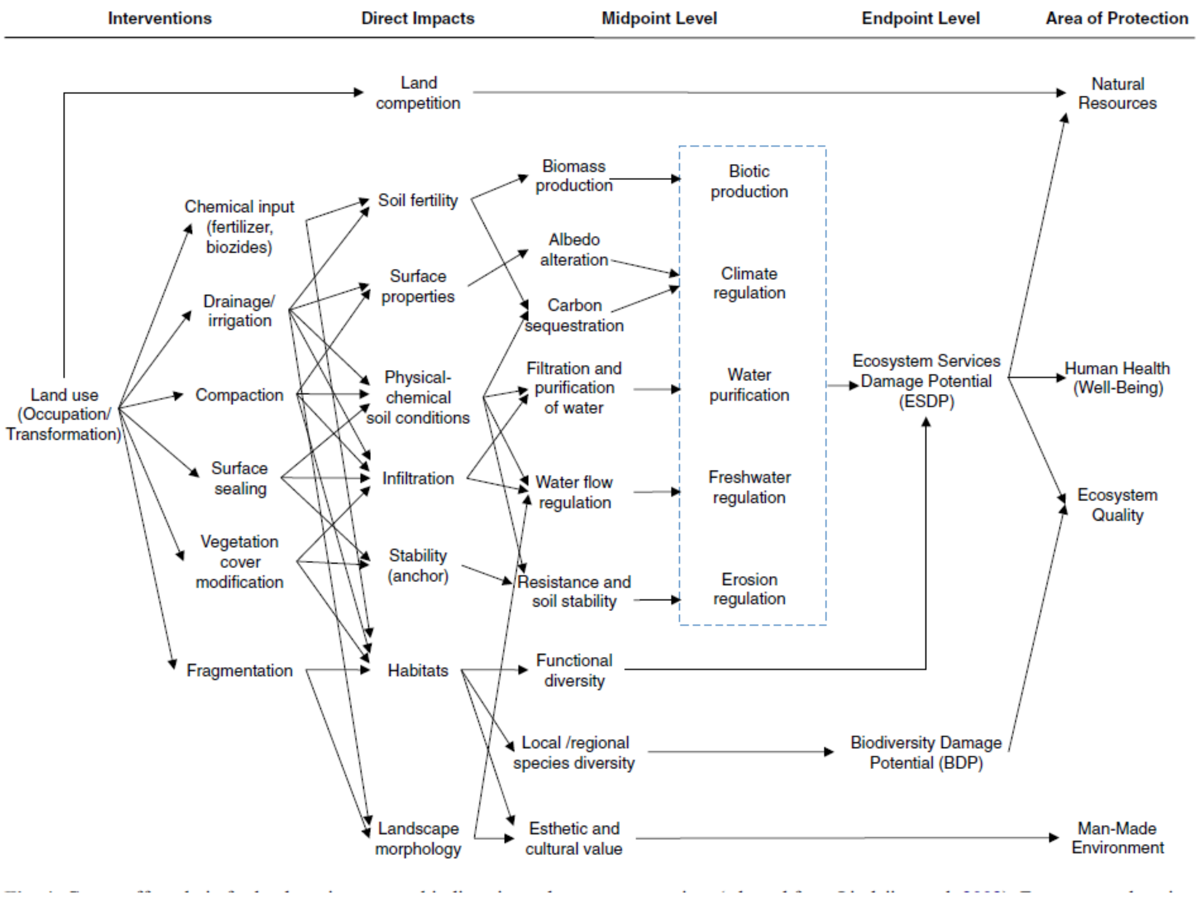


Figure 8. Land use impacts in LCA (Koellner et al., 2013b)

5.3.4 Remaining gaps in land use impact assessment

Section 5.3.3 concluded that despite of recent developments in land use impact assessment, there is still need for a land use impact assessment method that: 1) includes impacts on all ecosystem services and takes into account possible interactions between these services, 2) is easily applicable and compatible with LCI databases and thus includes one set of characterization factors which represent all land use impacts, 3) uses globally applicable characterization factors, or if possible biogeographically differentiated, and 4) differentiates between land use types and intensity of use, especially for different agricultural types.

6. Method

Chapter 2 and 3 have given the background information on land use impacts on ecosystem services and ways of evaluating ecosystem services. However, these impacts cannot be fully assessed yet, because life cycle impact assessment methods do not include all land use impacts and do not differentiate sufficiently between different land use types (as discussed in section 5.3.4). This chapter describes the steps that were followed to develop a new land use impact assessment method that aims to fill this gap (figure 9). Section 6.1 describes how the knowledge from the previous chapters was used to develop a new LU LCIA method, and section 6.2 describes how the method was applied in an LCA on a case study in order to illustrate and evaluate the functioning of the new method.

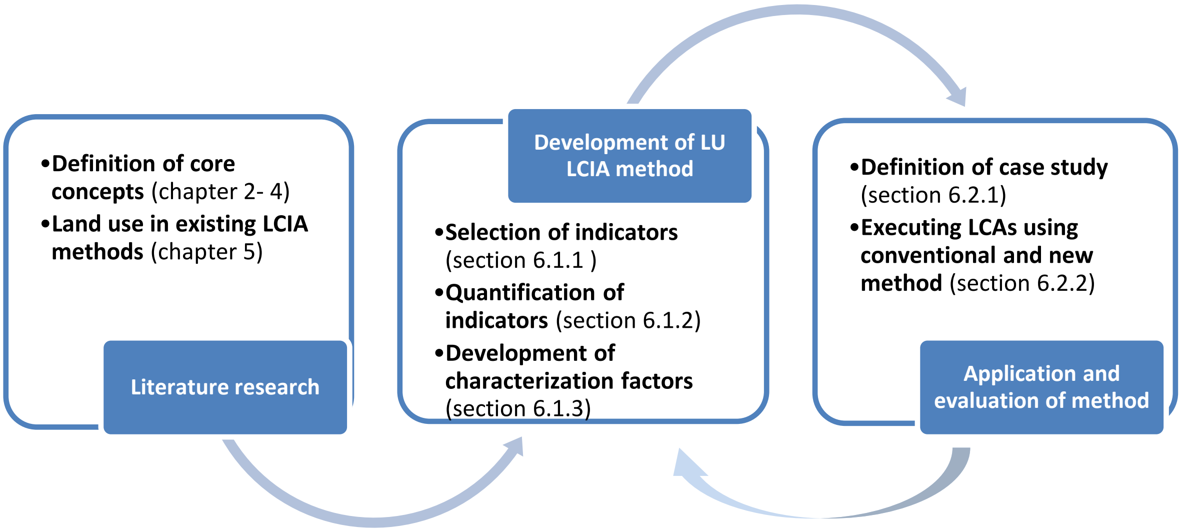


Figure 9. Research framework

6.1 Development of a land use impact assessment method

The development of the new LU LCIA method involved the following steps: 1) selection of appropriate indicators (section 6.1.1), 2) valuing the different land types according to their potential to deliver ecosystem services (section 6.1.2), 3) translating these values into characterization factors expressing damage to ecosystem quality from occupation and transformation (section 6.1.3).

6.1.1 Selection of indicators and data

The extensive review of literature on ecosystem services typology and quantification, especially in relation with land management, has led to the selection of ecological integrity indicators to represent the potential supply of ecosystem services by land (see chapter 3). The ecological integrity indicators developed by Müller (2005) were used: 1) exergy capture, 2) entropy production, 3) metabolic efficiency, 4) storage capacity, 5) nutrient loss reduction, 6) biotic water flows, 7) biotope heterogeneity, and 8) biodiversity (see section 3.2.3).

The ecological integrity (EI) indicator values that were quantified by Stoll et al. (2015) were used in this research. The values range between 0 and 5, where a high number indicates high ecological integrity and therefore large potential to deliver ecosystem services. Stoll et al. (2015) rounded the average EI value to an integer number, which makes comparison between e.g. irrigated and non-irrigated agriculture impossible (both have an average EI value of 3). For use in this research, the average value of the 8 indicator values was therefore rounded to 1 decimal, which enables differentiation between different land use types.

These EI values were then related to the land types used in ecoinvent. Stoll et al. (2015) valued land types on the third hierarchical level, while ecoinvent also includes land interventions on the first, second, and fourth hierarchical levels (see appendix A). In the case that a land type was defined at a lower hierarchical level in ecoinvent than valued by Stoll et al. (2015), the value for the higher hierarchical level was used (e.g. *extensive pasture and meadow* and *intensive pasture and meadow* both got the value determined for *pastures*). In the case that a land type was defined at a higher hierarchical level than valued by Stoll et al. (2015), the average of the specific types was used (e.g. *permanent crops* got the average of *vineyards*, *fruit tree and berry plantations* and *olive groves*). Ecoinvent also includes the land type t*ropical rain forest*, while this type is no longer included in the latest CORINE classification (CLC2006) and was not quantified by Stoll et al. (2015). In this research, *tropical rain forest* was assgined the same value as *mixed forest*. An unknown land use was given an average of all values, taking into account the hierarchical structure of the CORINE classification. An *unspecified, natural* land type was given the average value of all natural types (*forests and semi-natural areas*, *wetlands* and *water bodies*), and an *unspecified, used* land type was given the average of a*rtificial surfaces* and *agricultural areas*.

6.1.2 Indicator quantification for organic agriculture

Stoll et al. (2015) make a distinction between irrigated and non-irrigated arable land, but not between intensive versus extensive agriculture, and conventional versus organic agriculture. Because management practice has a large impact on potential ecosystem service provision (see section 3.4), and there is a need for refinement of impact from different agricultural practices, this effect was important to include in this research. The difference between intensive and extensive agriculture lies in the amount of labor and capital for application of fertilizers, pesticides and machinery (Encyclopaedia Britannica, 2014). There is, however, not a clear boundary between the two and therefore this research did not include this distinction. Organic agriculture, on the other hand, is well-defined and strictly regulated and therefore allows better comparison of the performances of farming systems with and without agrochemical inputs, and with or without the adoption of certain management practices (Gomiero et al., 2011).

Organic agriculture is defined as "a holistic production management system which promotes and enhances agro-ecosystem health, including biodiversity, biological cycles, and soil biological activity. It emphasizes the use of management practices in preference to the use of off-farm inputs, taking into account that regional conditions require locally adapted systems. This is accomplished by using, where possible, agronomic, biological, and mechanical methods, as opposed to using synthetic materials, to fulfill any specific function within the system" (Codex Alimentarius Commission, 2007). For this research, organic agriculture is considered distinct from conventional agriculture by the restraint of pesticide and (chemical) fertilizer use.

Data to quantify the ecological integrity indicators for organic agriculture were not available, therefore scientific literature was reviewed to determine if each indicator value is likely higher, lower, or similar for organic agriculture compared to conventional agriculture. Table 8 shows the explanation of the reasoning behind each estimation, which was discussed with, and confirmed by, soil and landscape ecology experts. The values of Stoll et al. (2015) for all agricultural land use types range between 2.1 (*vineyards*) and 3.9 (*agro-forestry areas*), a difference of 1.8. A full point increase or decrease per indicator value for organic agriculture compared to conventional agriculture would result in the same value as is assigned to agro-forestry. This does not seem appropriate, as agroforestry is closer to a natural forest system than organic agriculture. An increase or decrease of each indicator value with 0.5 was deemed more appropriate, which resulted in a total ecological integrity value of 3.6 for organic arable land. Table C1 in appendix C shows the EI values for all land types in ecoinvent.

Table 8. Estimation of difference in indicator values for organic arable land compared to conventional arable land

Ecological integrity indicators	Assumed difference between conventional and organic arable land	Reasoning	Reference
Exergy capture (NPP)	-	Primary production of intensive agriculture is generally higher	Matson et al., 1997; Mäder et al., 2002
Entropy production (respiration)	+	Due to increased microbial activity, respiration from organic soil is found to be higher than conventional agricultural soils	Wang et al., 2011; López-López et al., 2012
Storage capacity (SOM)	+	Soil loss is reduced under organic practices and therefore SOM is higher	El-Hage Scialabba, 2013; Gomiero et al., 2011
Reduction of nutrient loss	+	Less fertilizers are used and therefore less leaching of nutrients occurs	El-Hage Scialabba, 2013; Gomiero et al., 2011
Biotic waterflows (transpiration/evapotranspiration)	+	Improved soil structure increases soil moisture and therefore water uptake by plants (during a period of low rainfall)	El-Hage Scialabba, 2013; Gomiero et al., 2011
Metabolic efficiency (respiration/biomass)	+	Microbial communities with an increased diversity in organic soils likely transform carbon from organic debris into biomass at lower energy costs, building up a higher microbial biomass. This showed from the results of Mäder et al. (2002) where metabolic quotient (qCO_2) was lower.	Mäder et al., 2002
Abiotic (soil) heterogeneity	+	Spatial heterogeneity is found to be higher in extensively cultivated soils	Li et al., 2010
Biodiversity	+	Reduced use of pesticides and fertilizers, and high soil quality increases below and aboveground biodiversity	Gomiero et al., 2011; Mäder et al., 2002

6.1.3 Characterization factors

Because Life Cycle Assessment evaluates environmental impacts, the ecological integrity values needed to be transformed to be expressed as a damage to ecological integrity. The new impact assessment method therefore represents the difference in the capacity of land to provide ecosystem services compared to a natural reference situation. The ecological integrity damage (EID) was calculated by subtracting the ecological integrity (EI) value from 5. The maximum EI value of 5.0 therefore corresponds with 0 EID, while a minimum EI value of 0.0 corresponds with a maximum EID of 5.0.

The EID values were considered as the damage to ecological integrity per m^2*year (EID/m^2yr) and form the characterization factors (CFs) for land occupation. The CFs have to be multiplied with the land occupation inventory values (in m^2year), and consequently added up, to result in the total land occupation impact for a functional unit (in EID, dimensionless).

In order to calculate CFs for land transformation impact, regeneration times are required (see section 5.2.2). Because there is much uncertainty and complexity related to regeneration time (de Baan et al., 2013; Koellner et al., 2013b), and no data was available for regeneration of ecological integrity, in this research the same regeneration time for all land types was used. This enables transparent evaluation of the method, and can be easily adapted by LCA practitioners when more specific information is available. *Forest* (*broad-leaved, coniferous,* or *mixed*) is the land type with the highest ecological integrity value in the CORINE land classification and reflects the potential natural vegetation (PNV) in Europe (de Baan et al., 2013). The regeneration time thus is based on the regeneration of a forest, which is considered 50-200 years by Bastian and Schreiber (1999). Taking into account the estimations of regeneration time from different land types towards forest determined by Koellner and Scholz (2008), 50 years was deemed appropriate for an average land type to regenerate towards a European (quasi-)natural forest.

Multiplying the occupation CFs with 0.5*regeneration time resulted in the CFs for *transformation, to B* (in EID/m^2). Characterization factors for *transformation, from A* (in EID/m^2) received a negative value, since it should represent the transformation from A to baseline (see section 5.2.2). No permanent transformation impacts are assumed. The total transformation impact (in EID, dimensionless) can be calculated by multiplying the land transformation inventory values (in m^2) with the transformation CFs (in EID/m^2) and consequently adding these up. Because the total damage from land occupation and transformation are both expressed in EID (dimensionless), they can be compared with each other, and added up to get the total land use impact on ecological integrity.

6.2 Application and evaluation of the new method

The new method was applied in a Life Cycle Assessment on a case study, to illustrate the application of the method and evaluate its functioning. The case study is described in section 6.2.1, followed by a specification of the steps followed in the LCA in section 6.2.2.

6.2.1 Case study description

A case study can provide context and detailed information to a general problem (Neale et al., 2006). A case was selected that involves different types of land use, in order to evaluate the results of the new method for different land use types. By choosing a product that can also be produced from fossil resources, the results of the case study can make a contribution to a better comparison of fossil- and biobased alternatives. In section 2.2 it was described that land use serves the ultimate goal of acquiring resources, mainly for food, fiber, water and shelter (Foley et al., 2005). Because fibers can be produced from different renewable feedstocks and thus involve both agricultural and forest land types, a case study on different fibers is appropriate to evaluate the new land use impact assessment method.

Total world fiber production exists of 60% synthetic and 40% natural fibers, which can be used for the production of garments, household textiles or technical textiles (European Commission, 2011). 75% of synthetic fibers are made from polyester, and 85% of natural fibers are cotton fibers (FAO and ICAC, 2013). Cotton has a large environmental impact due to the large scale irrigation and pesticide use (Kooistra and Temorhuizen, 2006). Therefore interest is growing for a more sustainable and organic cultivation of cotton, and the use of alternative natural fibers (Truscott et al., 2013; van der Werf and Turunen, 2008). Natural fibers such as hemp and flax have a limited requirement of pesticides, fertilizers and irrigation (van der Werf and Turunen, 2008). Man-made cellulosic fibers, such as viscose made from wood pulp, were also found to have a lower environmental impact as cotton and polyester (Shen et al., 2010b). LCAs that compare synthetic and natural fibers, conclude that land use has a large influence on the results (e.g. Sandin et al., 2013; Shen et al., 2010b). A sound comparison of land use impact of the different natural fibers could, however, not be done yet due to the incomplete land use LCIA methods (see chapter 5). The new method developed in this research is meant to provide a better comparison and is therefore applied to this case study.

The fibers that were assessed in this study are three out of the four most commonly used natural fibers; cotton, viscose, and flax. Wool (the third most commonly used natural fiber) was omitted, because time constraints did not allow to deal with methodological issues, such as allocation issues

(Henry, 2012). The effect of different land use types on total environmental impacts was evaluated by using different agricultural practices for cotton and flax. For flax, conventional versus organic cultivation was considered, both non-irrigated as water requirements of flax can be covered by rainfall (Le Duigou et al., 2011). For cotton, irrigated versus non-irrigated, and conventional versus organic cultivation were considered. Viscose is made from wood pulp and therefore adds a forest land type to the case study.

6.2.2 Application of LCA to case

All steps of a Life Cycle Assessment (see chapter 4) were taken for the case study, with an emphasis on the impact assessment phase. In the following sections the application of each phase to the case study is described.

6.2.2.1 Goal definition

The goal of the LCA was to illustrate the application of the new method and evaluate of its functioning in a comparison with a conventional LCIA method (see section 1.3). Therefore land use impacts of several natural fibers were assessed using both the new LU LCIA method and a conventional method, of which the results were compared. The conventional method was selected based on the following requirements, it should 1) include land occupation and transformation impacts, 2) evaluate a relevant aspect of ecosystem quality at endpoint level, and 3) make a distinction between different land types, in order to compare results of the new method with the conventional method. The method that best meets these requirements is ReCiPe.

6.2.2.2 Scope definition

The functional unit was *1000 kg of dry staple fibers*, which is a commonly used functional unit for environmental impact assessment of fibers (Sandin et al., 2013; Shen et al., 2010a;2010b). The scope was defined as the 'cradle-to-gate' production of fibers, which includes the cultivation and harvesting of the different fiber crops, and the separation of seeds and fibers. Further production of yarns and garments was outside of the scope of this study, because the first phase of cultivation of natural fibers includes most land use, which is the impact category of interest for this research. Only land use impacts were assessed, as this is the focus of this research. The reference flows are fibers made from: 1) irrigated conventional cotton, 2) non-irrigated conventional cotton, 3) non-irrigated organic cotton, 4) non-irrigated conventional flax, 5) non-irrigated organic flax, and 6) viscose from wood pulp (table 9).

Table 9. Reference flows considered in the LCA

Reference flow	Main data source	Main land use type (ecoinvent)	Adjustments to main data source
Cotton, irrigated, conventional	Cotton fibres, ginned, at farm/CN U (ecoinvent v2.0)	*Arable, irrigated*	
Cotton, non-irrigated, conventional	Cotton fibres, ginned at farm/CN U (ecoinvent v2.0)	*Arable, non-irrigated*	-45% of irrigated yield -irrigation excluded
Cotton, non-irrigated, organic,	Cotton fibres, ginned at farm/CN U (ecoinvent v2.0)	*Arable, organic*	-45% of irrigated yield -irrigation excluded -pesticides excluded -fertilizers excluded
Flax, non-irrigated, conventional	Based on Le Duigou et al. (2011) and Labouze et al. (2007)	*Arable, non-irrigated*	
Flax, non-irrigated, organic	Based on le Duigou et al. (2011) and Labouze et al. (2007)	*Arable, organic*	-56% of conventional yield -pesticides excluded -fertilizers excluded
Viscose	Viscose fibres, at plant/GLO U (ecoinvent v2.0)	*Forest, intensive*	

6.2.2.3 Inventory data

Data from the ecoinvent v2.0 database were used for cotton and viscose fibers. Data for cotton production in China was used, because this is the largest producer of cotton fibers (FAOSTAT, 2012). The record for cotton fiber production in China includes irrigation, and the land occupation *arable land*. The land type for occupation was adjusted to *Occupation, arable, irrigated* to enable evaluation of ecological integrity damage from irrigated arable land. A copy of the record was made, for which the land type was changed to *Occupation, arable, non-irrigated* and the irrigation process and water inputs were excluded. The influence of irrigation on yield depends on the amount of rainfall and therefore shows large variability. However, in general a higher yield is assumed for irrigated cotton cultivation, due to the high water requirements of the cotton plant (Kooistra and Temorshuizen, 2006). A yield of 45 percent from the conventional cultivation was modelled for non-irrigated cotton production, based on Soth et al. (1999).

A review of literature showed that there are no significant differences in yield from conventional or organic cotton cultivation (Blaise, 2006; Mygdakos et al., 2007; Forster et al., 2013; Bilalis et al., 2010). Therefore pesticide and fertilizer use and resulting emissions were set to zero in the organic scenario, while the yield was left the same as for conventional production. Water use and type of irrigation are not explicitly included in the standards for organic production, but the majority of organic cotton systems are not irrigated (Kooistra and Temorshuizen, 2006). Furthermore, the CORINE classification and ecoinvent do not include an irrigated organic agriculture type. Therefore the organic scenario was adapted from non-irrigated cotton cultivation, and irrigated organic agriculture was left out of the analysis.

For conventional flax, a record made by TNO was used, based on full LCI data from Le Duigou et al. (2011) and Labouze et al. (2007). A copy of this record was made for which pesticides and fertilizer use and resulting emissions were set to zero and the land occupation *arable, organic* was used instead of *arable, non-irrigated*. The yield of organic flax was assumed to be 56% of conventional yield, based on a 2007 survey among flax producers in Canada (McGilp et al., 2007). Table 9 gives an overview of the data used for each reference flow.

The records used in this research did not contain transformations in the foreground system (the main records as described in this paragraph). For all agricultural fiber cultivation it was assumed that this has been the land use for a long time and thus no transformations were related to fiber cultivation (see Althaus et al, 2007). Therefore only the transformations related to the background system, e.g. the production of fertilizers, were evaluated.

6.2.2.4 Impact assessment

The new land use impact assessment method represents damage to ecological integrity from land use, in short called the EID method. It includes characterization factors for all land occupation and transformation interventions in ecoinvent, examples are shown in table 10 and 11 and the full list is shown in appendix C. The EID method is an endpoint method, as it evaluates damage to the function and structure of ecosystems, which are the relevant aspects of the AoP Natural Environment (see section 4.4).

Table 10. Examples of characterization factors for *Land occupation,* full table in Appendix C

Intervention	CF	Unit
Occupation, arable	1,8	EID/m^{2}yr
Occupation, arable, irrigated	1,9	EID/m^{2}yr
Occupation, arable, non-irrigated	1,7	EID/m^{2}yr
Occupation, arable, organic	1,4	EID/m^{2}yr

Table 11. Examples of characterization factors for *Land transformation*, full table in Appendix C

Intervention	CF	Unit
Transformation, from arable	-45	EID/m^2
Transformation, from arable, irrigated	-47,5	EID/m^2
Transformation, from arable, non-irrigated	-42,5	EID/m^2
Transformation, from arable, organic	-35	EID/m^2
Transformation, to arable	45	EID/m^2
Transformation, to arable, irrigated	47,5	EID/m^2
Transformation, to arable, non-irrigated	42,5	EID/m^2
Transformation, to arable, organic	35	EID/m^2

The results of the EID method and the ReCiPe v.1.07 endpoint method were compared using SimaPro 7.3.3. Of the three different scenarios that ReCiPe uses (see section 5.3.2), the hierarchist perspective was used, because it is the default version of ReCiPe (Weidema, 2015).

Because ReCiPe includes two separate categories for 'Agricultural land occupation' and 'Urban land occupation', these two were added up to form the category 'Total land occupation' in order to make a clearer comparison to the EID method. Because ReCiPe does not include a characterization factor for *Occupation, arable, irrigated*, the characterization factor of *Occupation, arable* was used to characterize irrigated arable land in ReCiPe. The impacts within an impact category were normalized to 100% of the largest impact in order to compare the results of the two methods with each other, as they use different units to express damage to Natural Environment.

6.2.2.5 Interpretation

The results of the LCAs were analyzed in order to evaluate the EID method. The dominant processes, land types and transformations were determined for the different fibers and agricultural types. The causes of differences between ReCiPe and the EID method were investigated and explained. Several important aspects of the EID method were further investigated by sensitivity analyses: differentiation between forest types, influence of EI value, differentiated regeneration time, and foreground transformation. The most important results of the LCA analysis and interpretation are reported in chapter 7.

7. LCA results

This chapter describes the most important results of the Life Cycle Assessments using the EID method and ReCiPe. Section 7.1 describes the main differences between both methods, illustrated by results from the case study. In section 7.2, the case conclusions from the EID method and ReCiPe are discussed. Four issues were selected for a sensitivity analysis, which are described in section 7.3.

7.1 Comparison EID method and ReCiPe endpoint

The comparison of results from the EID method and ReCiPe indicated several methodological differences. The most important differences are related to: differentiation between agricultural types (section 7.1.1), relative forest damage and differentiation between forest types (section 7.1.2), and non-natural land transformations (section 7.1.3).

7.1.1 Differentiation between agricultural types

The case study included fibers from different agricultural cultivation types, to evaluate the functioning of the EID method to differentiate between land type and intensity of use. As the EID method takes into account 7 indicators in addition to biodiversity, the results from the LCA differ from ReCiPe. In section 7.1.1.1 the evaluation of organic versus conventional land is discussed, and in 7.1.1.2 evaluation of irrigated versus non-irrigated land is discussed.

7.1.1.1 Organic versus conventional arable land

Figure 10 shows that the occupation impact of organic flax is higher than conventional flax using both methods, which is due to the lower yield and thus larger area of land needed to produce the same functional unit. Occupation impact of organic flax is 32% larger than occupation impact of conventional flax using the EID method and 42% using ReCiPe. The smaller difference when using the EID method is because of the lower CF of 18% attributed to organic arable land compared to non-irrigated arable land. In ReCiPe, organic arable land has a 7% lower CF than conventional arable land.

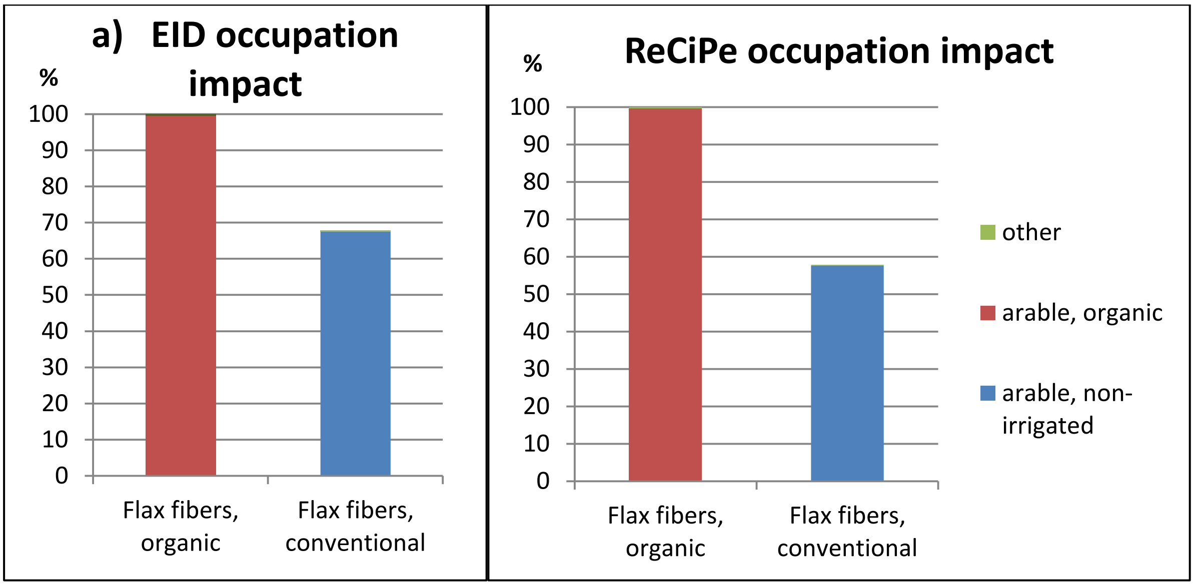
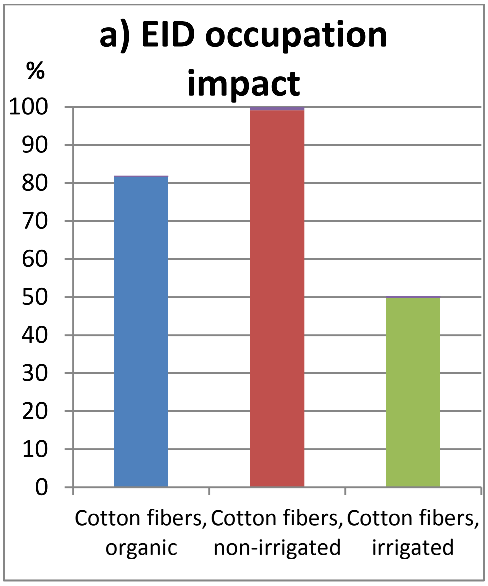


Figure 10. Comparison of occupation impact of organic flax and conventional flax fibers using a) the EID method and b) ReCiPe

The result of the different occupation CFs for organic and conventional agriculture can also be observed when comparing the different cotton cultivation types (figure 11). Because there is no yield difference between organic and non-irrigated conventional cotton cultivation, organic cotton has a lower occupation impact due to the lower CF in both ReCiPe and the EID method. In the EID method, however, damage of organic arable land is relatively lower than in ReCiPe, thus the resulting difference in occupation impact between organic and non-irrigated arable land is larger in the EID method (figure 11).



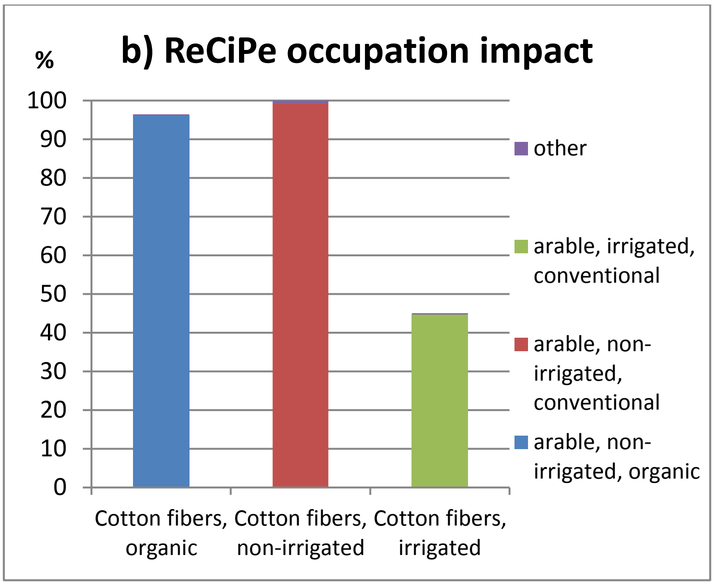


Figure 11. Comparison of occupation impact of non-irrigated organic cotton, non-irrigated conventional cotton, and irrigated conventional cotton using a) the EID method and b) ReCiPe

7.1.1.2 Irrigated versus non-irrigated arable land

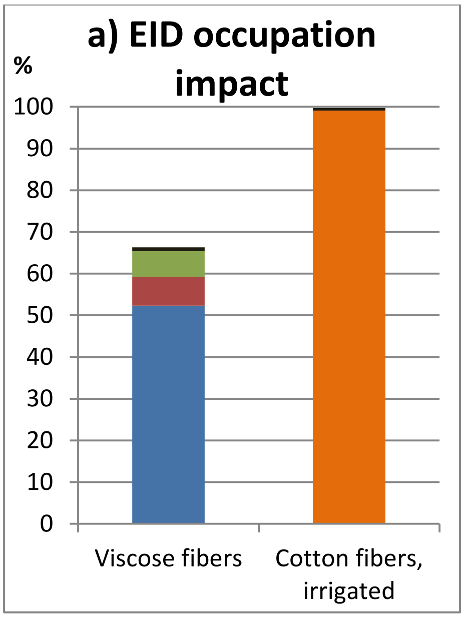
The EID method differentiates between irrigated and non-irrigated arable land, while ReCiPe does not. In fact, irrigated land was not defined in ReCiPe and in this research it was therefore assigned the same CF as *Occupation, arable*, which is the same as the CF for *Occupation, arable, non-irrigated.* Therefore the 55% difference in occupation impact between irrigated and non-irrigated cotton is caused by the 55% lower yield of non-irrigated land and thus higher land requirements for the same functional unit. In the EID method, non-irrigated land has a lower CF than irrigated land, which dampens the effect of higher land requirement, resulting in a 50% difference in occupation impact between irrigated and non-irrigated cotton (figure 11).

7.1.2 Relative damage of forest types

The LCA results indicated a substantial difference between the two methods in the relative damage attributed to forest occupation, which is discussed in section 7.1.2.1, and forest transformation, which is discussed in section 7.1.2.2.

7.1.2.1 Forest occupation

An important difference was found between ReCiPe and the EID method for the damage that was attributed to forest occupation. This can be seen from the comparison of viscose and cotton (figure 12). ReCiPe indicates that viscose has a 14% larger occupation impact than irrigated cotton, while the EID method indicates a 33% smaller occupation impact of viscose. This is a result of the characterization factors used for the main land types contributing to the occupation impact; intensive forest for viscose, and irrigated arable land for irrigated cotton. ReCiPe uses a CF for intensive forest which is 39% smaller than the CF for arable land, while the CFs for forest types in the EID method are 67% smaller than the CF of arable land. This relatively small ecological integrity damage factor of forest leads to a low occupation damage for viscose compared to cotton, while the relatively large damage factor attributed to intensive forest by ReCiPe results in a high occupation impact of viscose. A sensitivity analysis was done to evaluate the effect of attributing a higher damage to forest occupation in the EID method, which is described in section 7.3.1.



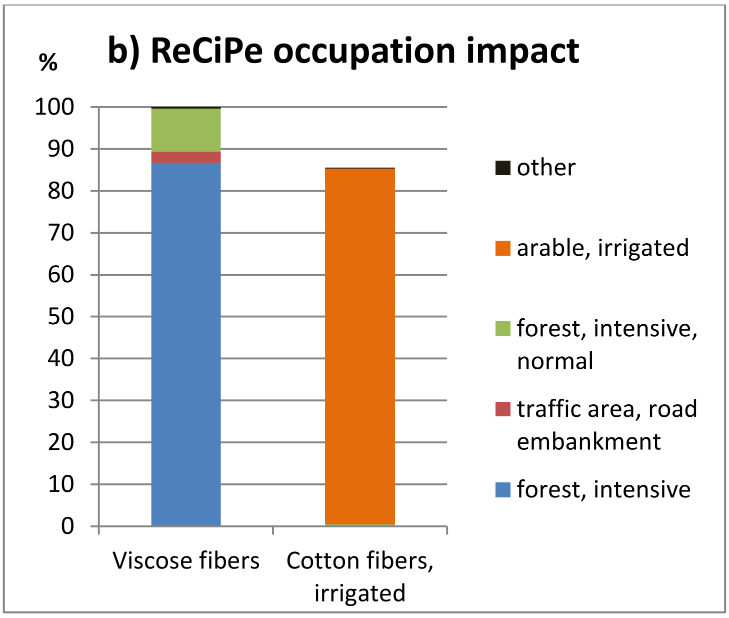
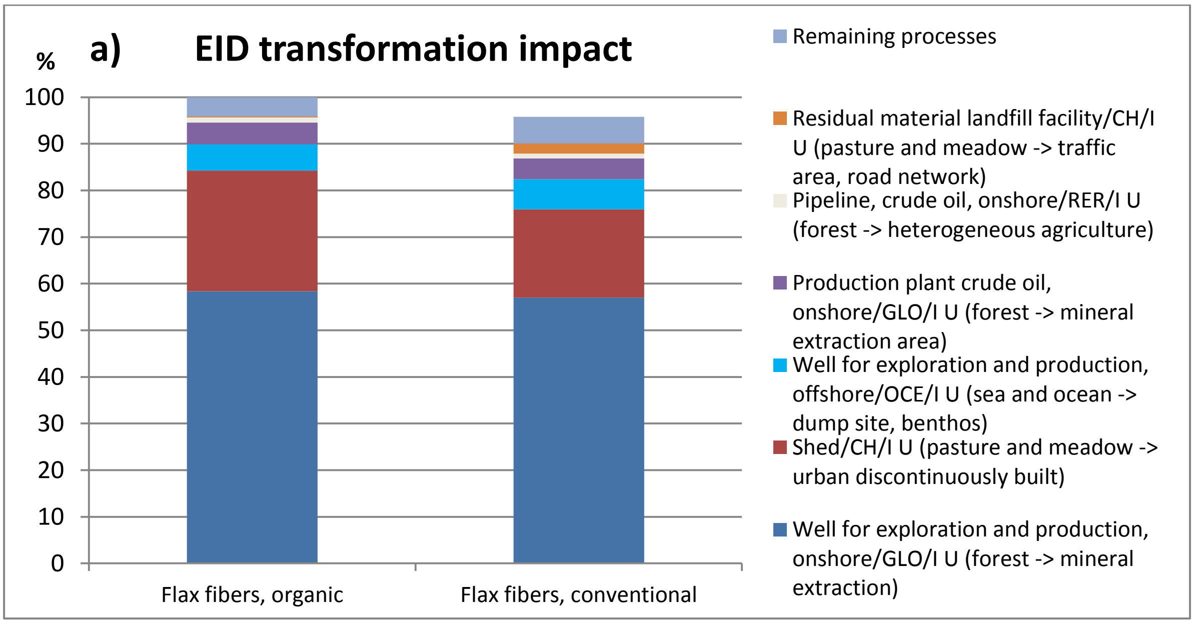


Figure 12. Comparison of land occupation impact of viscose and irrigated conventional cotton using a) the EID method and b) ReCiPe

7.1.2.2 Tropical rain forest transformation

As there were no transformations related to the foreground system, therefore transformation impact was low compared to occupation impact, ranging between 0.1% and 4.1% (see table D1 in appendix D). However, the transformation in the background system, e.g. for production of fertilizers, enabled evaluation of transformation impact of the EID method and ReCiPe.

The most important difference in the calculation of transformation impact lies in the damage attributed to the transformation of tropical rain forest. ReCiPe attributes a 33 times larger impact to transforming tropical rain forest than other forest types, due to the assumed long regeneration time of 3300 years. In the EID method, tropical forest receives the same CF as mixed forest. The effect of this difference can be seen clearly when comparing organic flax with conventional flax: for the production of fertilizers, palm oil is used for which tropical rainforest is transformed, while for organic flax no tropical rain forest is transformed (figure 13).



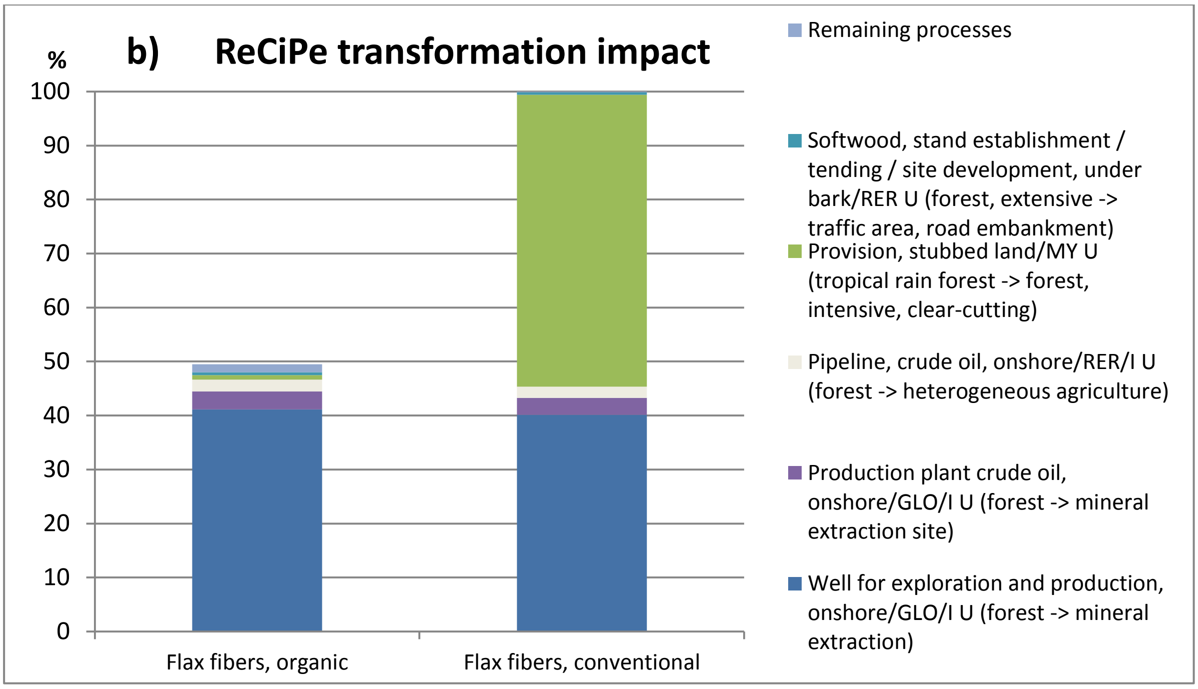


Figure 13. Comparison of transformation impact of organic flax fibers and conventional flax fibers using a) the EID method and b) ReCiPe

Using ReCiPe, approximately half (54%) of the transformation impact of conventional flax comes from the process ‘Provision, stubbed land/MY U’, for which tropical rain forest is transformed (figure 13b). This process is related to the production of palm oil, used for production of the fertilizer triple superphosphate. The second largest transformation impact of conventional flax (40%) using ReCiPe comes from ‘Well for exploration and production, onshore/GLO/I O’, which is used to produce diesel. Diesel is mainly used for tillage, but also for scutching and hackling of fibers, fertilizing and application of pesticides. Transformation impact for organic flax mainly comes from diesel use (‘Well for exploration and production, onshore/GLO/I O’) (83%). This means that the process that caused the largest damage for conventional flax, involving tropical rain forest transformation, is excluded for organic flax, which results in a 50% lower impact.

In the EID method the transformation related to ‘Provison, stubbed land’ is not accounted for at all, because tropical rain forest received the same CF as other forest types, and thus the transformation from *tropical rain forest* to *forest, intensive, clear-cutting* results in a net zero transformation impact. Using the EID method, the processes that contribute most to the total transformation impact are similar for conventional and organic flax fibers (‘Well’ for diesel production and ‘Shed’ for storage of agricultural machinery) (figure 13a). However, for which purposes these processes are used differ. Only approximately a third of the land transformation impact of conventional flax fibers is a result of processes related to the production and application of fertilizers and pesticides, while the largest contribution (73%) is a result of processes related to tillage, scutching and hackling of fibers. For organic flax fibers, most transformation impact is related to mechanical agricultural processes (65% for diesel use, and 27% for storage). Because more land is cultivated for organic fibers, more diesel and storage is necessary for the agricultural practices, which results in a higher transformation damage for organic flax.

Thus, using ReCiPe, the largest transformation impact is a result of fertilizer production, while using the EID method, the largest transformation impact is a result of diesel use and storage. Organic cultivation does not use fertilizers and this results in a lower transformation impact using ReCiPe. On the other hand, organic production uses more diesel for cultivation processes than conventional production, which results in a larger transformation impact using the EID method.

7.1.3 Non-natural transformations

The third large difference between the EID method and ReCiPe is that ReCiPe only takes into account natural land transformation, while the EID method includes all land transformations among the land types in ecoinvent. ReCiPe includes CFs only for transformations to and from *forest*, *tropical rain forest*, and *unknown*, which may lead to underestimation of transformation impacts when transformations occur that involve other land types, especially from high value non-forest land types to low value land types. This can be seen from the analysis of land use impact for organic flax and organic cotton (figure 14).

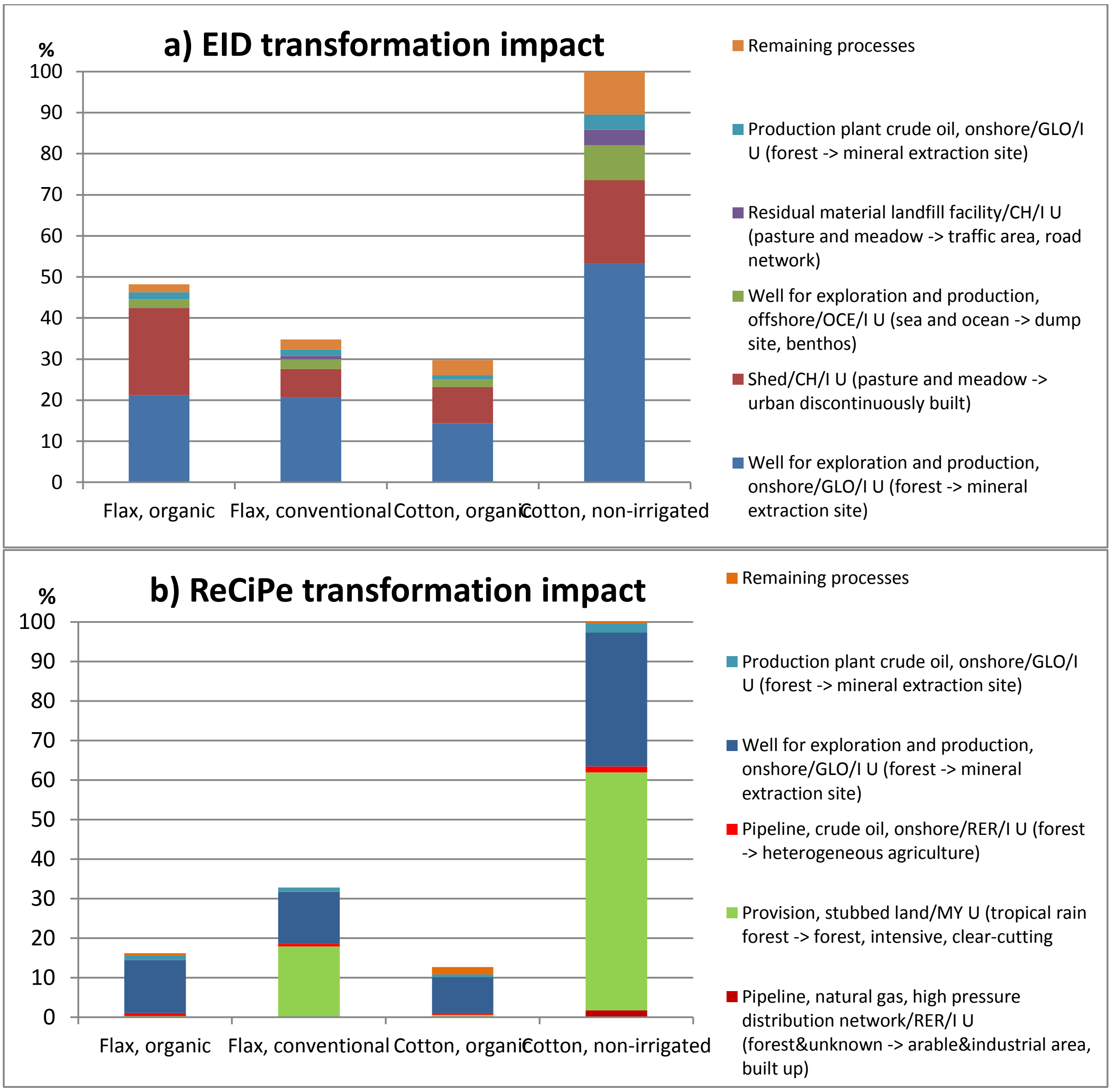


Figure 14. Comparison of transformation impact of organic flax, conventional flax, organic cotton, and non-irrigated conventional cotton using a) the EID method and b) ReCiPe

The transformation from *pasture and meadow* to *urban, discontinuously built* takes place within the process that contributes 19% to the transformation impact of organic flax and 18% for organic cotton when using the EID method ('Shed/CH/I U' - used for storage of agricultural equipment) (figure 14a). This process does not contribute to the transformation impact using ReCiPe, which explains part of the relatively lower transformation impact of organic fibers compared to conventional fibers (figure 14b). Furthermore, it can be seen from figure 14 that using the EID method, the remaining processes[3] contribute between 2 and 10% of the total transformation impact, while in ReCiPe they contribute maximum 2%. This shows that using ReCiPe, most impact comes from a few transformations involving forest types, while several non-forest transformations are excluded from calculation of transformation impact.

On the other hand, the exclusion of CFs for non-forest land types may also lead to an overestimation of transformation impacts. When the land type after transformation is a non-forest land type with a relatively ecological value, this value is not taken into account (e.g. from *forest*, to *grassland*). Only the negative effect of transforming a natural land type is taken into account, but not the ecological value of the land that replaces the former land type. This is the case for the transformations from *forest* to *heterogeneous agriculture* (within the process 'Pipeline, crude oil, onshore') and *forest* and *unknown* to *arable* (within the process 'Pipeline, natural gas') (figure 14b). Because these transformations included relatively small areas compared to the other transformations (related to 'Stubbed land' and 'Well'), this did not lead to large overestimations in this case study.

7.2 Case conclusions

The main conclusions from both methods are that the land occupied by the main land use type (in m^2year) determines the occupation impact, which puts the organic cultivation types and non-irrigated cotton on the bottom of the ranking list, while the transformation of forest and the use of fuel, fertilizers and pesticides determined the transformation impact, and therefore puts cotton and viscose on the botton of the ranking list (table 12). It must be noted that transformation impacts only came from background processes. Table D1 in appendix D shows the occupation and transformation impact of all fibers expressed as ecological integrity damage (EID) and potentially disappeared fraction of species (PDF).

Table 12. Ranking of fiber types from lowest impact (1) to highest impact (6)

	Occupation impact		Transformation impact	
	ReCiPe	**EID**	**ReCiPe**	**EID**
1.	<u>*Flax*</u>	<u>*Viscose*</u>	Organic cotton	Organic cotton
2.	<u>*Irrigated cotton*</u>	<u>*Flax*</u>	<u>*Organic flax*</u>	<u>*Flax*</u>
3.	<u>*Viscose*</u>	<u>*Irrigated cotton*</u>	<u>*Flax*</u>	<u>*Organic flax*</u>
4.	Organic flax	Organic flax	Irrigated cotton	Irrigated cotton
5.	Organic cotton	Organic cotton	Non-irrigated cotton	Non-irrigated cotton
6.	Non-irrigated cotton	Non-irrigated cotton	Viscose	Viscose

7.2.1 Occupation impacts

The final conclusions from both methods agree on the three fiber types with highest occupation impact: non-irrigated cotton, organic cotton, and organic flax (see table 12 and figure 15). The methods differ in ranking of the three fiber types with the lowest occupation impact: flax, irrigated cotton and viscose.

[3] The transformation figures include only the 5 or 6 processes that contribute most to total transformation impact. All other processes are summarized in 'remaining processes'.

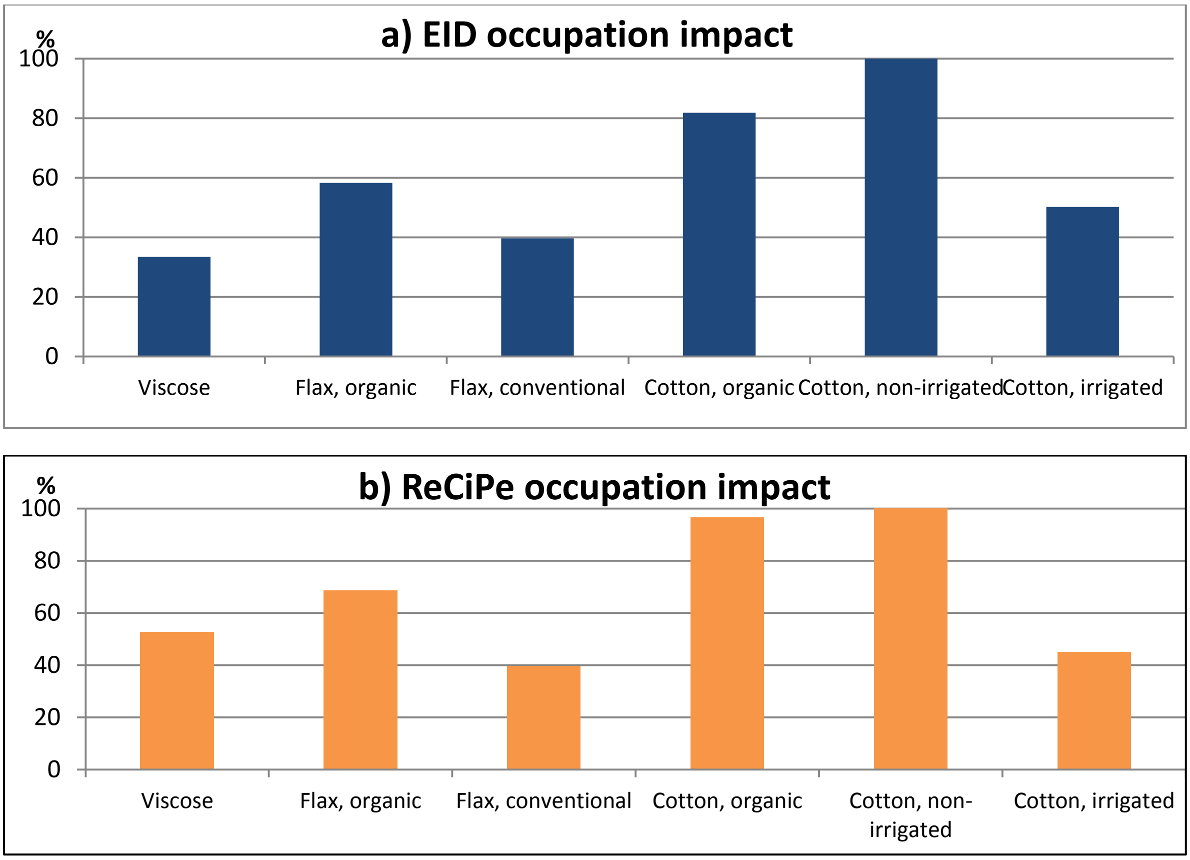


Figure 15. Comparison of occupation impact of all fibers and cultivation types using a) the EID method and b)ReCiPe

The main land occupation land type for viscose (necessary for the production of sulphate pulp) is *forest, intensive*, which occupies 7770 m^2yr. Although this exceeds the area occupied by the main land type of irrigated cotton (7730 m^2yr) and conventional flax (6840 m^2yr), the relatively low damage attributed to intensive forest in the EID method put viscose on the first place with least occupation impact, while the relatively high damage in ReCiPe puts viscose on the third place (see section 7.1.2.1).

Flax needs 12% less land than cotton (in m^2yr) because of the higher yield, and therefore has a lower occupation impact in both methods. Non-irrigated cotton has a higher occupation damage than irrigated cottond due to the lower yield, which causes a higher damage in both methods, although it is slightly dampened in the EID method due to the higher ecological integrity of non-irrigated land compared to irrigated land (see section 7.1.1.2).

The lower impact of organic cotton compared to non-irrigated cotton is for 99% due to the lower CF of organic land in both methods, as there was no assumed difference in yield. The absence of the need for storage of machinery related to pesticide and fertilizer application explains less than 1% of the occupation impact difference. Because organic land scores better on all ecological integrity indciators, as quantified in section 6.1.2, the difference between organic and conventional land is larger when taking into account ecological integrity than just evaluating biodiversity (see section 7.3.1.1).

7.2.2 Transformation impacts

The ranking of the fibers according to their transformation impact does not differ much between the EID method and ReCiPe, except for the place of organic and conventional flax (table 12, figure 16). Section 7.2.2.1 describes the main processes that contribute to the transformation impact of viscose, and section 7.2.2.2 desribes the main processes for the agricultural fibers.

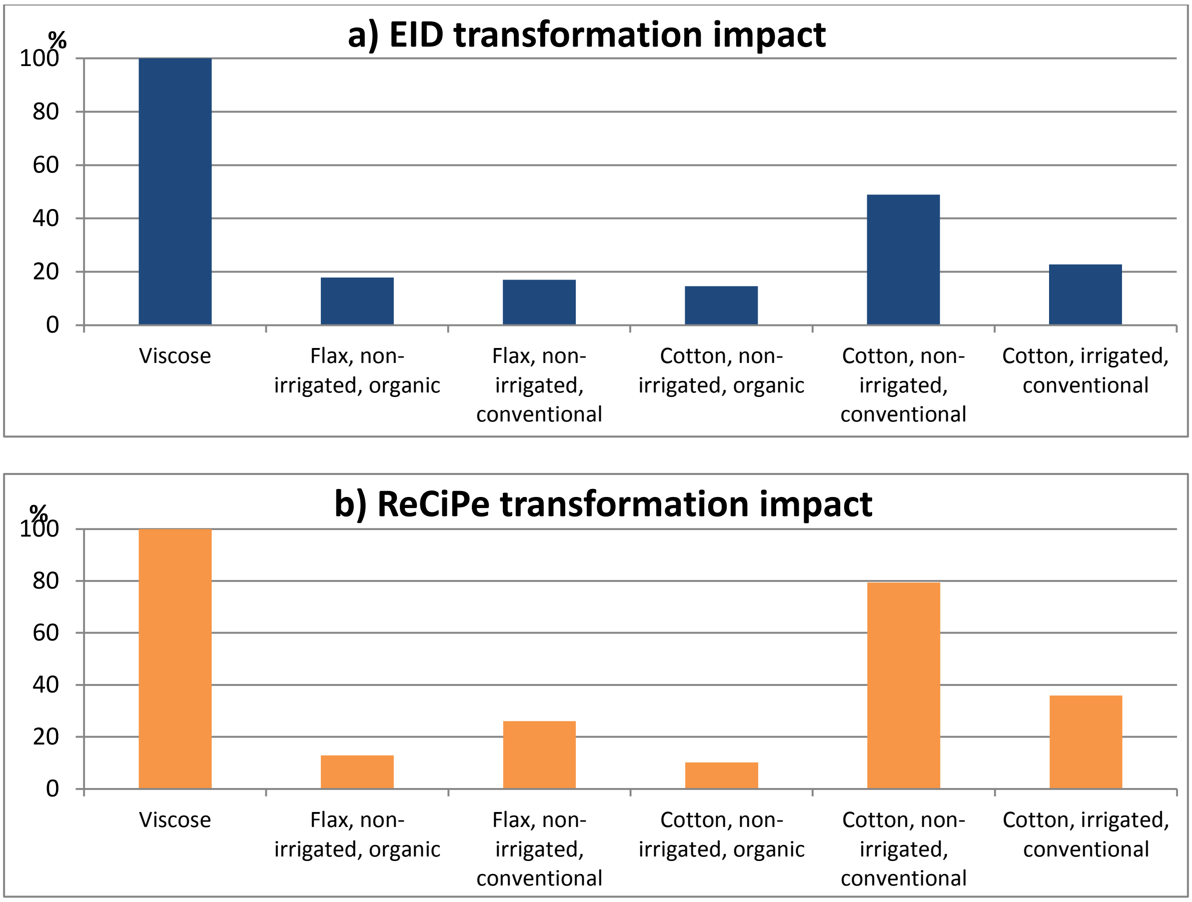


Figure 16. Comparison of transformation impact of all fibers and cultivation types using a) the EID method and b) ReCiPe

7.2.2.1 Viscose

In contrast to the agricultural fibers, viscose included a transformation related to foreground system. The largest transformation, 84.68 m^2 extensive to intensive forest, is related to the main occupation types ('Softwood, Scandinavian, standing, under bark, in forest/NORDEL U' and 'Hardwood, Scandinavian, standing, under bark, in forest/NORDEL U'). Because all forest types were assigned the same transformation CF in the EID method and ReCiPe, a transformation from extensive to intensive forest does not result in a damage. Thus, only the (smaller) transformation from forest types to other land types leads to a damage, and therefore the total transformation damage from to viscose is low compared to the occupation damage. However, viscose has the largest transformation impact of all the analyzed fibers. This is due to the transformation from *forest, extensive* to *road embankment*, which is related to the cutting of industrial wood.

7.2.2.2 Agricultural fibers

For agricultural fibers, the largest part (74-84%) of transformation impacts using the EID method is attributed to the processes 'Well for exploration and production, onshore/GLO/I U' and 'Shed/CH/I U'. The 'Well' includes the transformation from *forest* to *mineral extraction site*, and the 'Shed' the transformation from *pasture and meadow* to *urban, discontinuously built* (see figure 14). The well is used for the production of fuels, and the shed for the storage of agricultural equipment. Fuels are mainly used for the production of fertilizers and pesticides, the application of fertilizers and pesticides, and in lesser degree for tillage and other agricultural practices. Because organic agriculture does not include pesticides and fertilizers, the impact from 'Well' and 'Shed' is lower and total transformation impact is mainly a result from tillage and other agricultural practices. This results in a much lower transformation impact for organic cotton compared to conventional cotton.

Using the EID method, however, organic flax has a larger transformation impact than conventional flax, because transformation impact of flax is mainly determined by the use of agricultural machines, which is larger for organic flax due to the larger land requirements (see section 7.1.2.2, figure 14).

Using ReCiPe, the largest contribution (54-60%) to transformation impact for conventional agricultural fibers (conventional flax, irrigated and non-irrigated conventional cotton) comes from the process 'Provision, stubbed land/MY U', which is related to fertilizer production. This process includes a transformation from *tropical rain forest* to *forest, intensive, clear-cutting*. The second largest contribution (34-40%) to transformation impact of conventional agricultural fibers, and the largest contribution (72-83%) for organic agricultural fibers is from the process 'Well for exploration and production, onshore/GLO/I U', which was the most important process for transformation impact in the EID method (see figure 14). Thus, using ReCiPe, transformation impacts of agricultural fibers resulted mainly from (tropical) forest transformations, which were for a large part related to fertilizer production. This resulted in lower transformation impacts of organic fibers.

In this case study the impact of intensive agriculture is indirectly demonstrated: data of flax fiber production were based on intensive European practices, while data of cotton fiber production were based on less intensive Chinese practices, which included for example hand-picking (Althaus et al., 2007). The higher yield of flax leads to a lower occupation impact than cotton (see figure 15), but the high requirements of diesel for agricultural machines leads to a higher transformation impact, which can be seen clearly when comparing organic flax and organic cotton (figure 16). Due to the use of more pesticides and fertilizers for cotton production compared to flax, the transformation impact of conventional cotton is larger than conventional flax (see figure 16). These differences can be seen more clearly when comparing conventional flax with organic cotton: occupation impact is larger for organic cotton due to the lower yield, but transformation impact is lower, due to the lower diesel requirements for land cultivation in China, where many processes are done by hand (Althaus et al., 2007) (see figure 17).

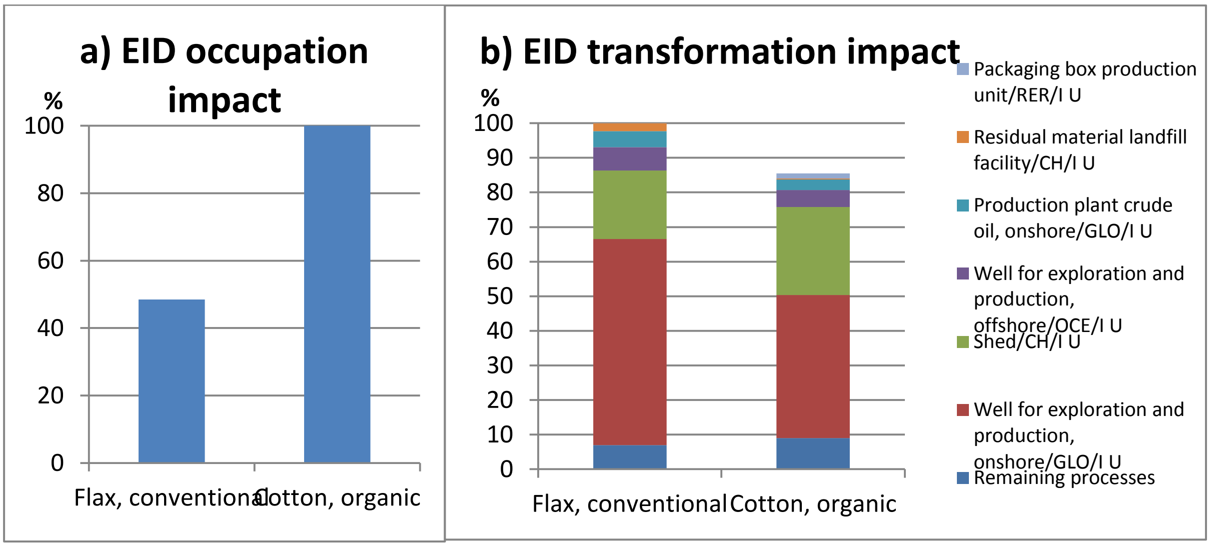


Figure 17. Comparison of a) occupation and b) transformation damage to ecological integrity of non-irrigated conventional flax and non-irrigated organic cotton

7.3 Sensitivity analysis

Analysis of the results indicated several issues that needed further investigation. In this section the results of four sensitivity analyses are discussed: the differentiation between intensive and extensive forest damage (section 7.3.1), the influence of ecological integrity value (section 7.3.2), the effect of differentiated regeneration time (section 7.3.3), and the impact of foreground transformation (section 7.3.4).

7.3.1 Differentiation between intensive and extensive forest

The comparison of the LCA results using the EID method and ReCiPe showed that the EID method attributed a relatively lower damage to forest occupation than ReCiPe, which causes a large difference in the occupation impact of viscose (see section 7.1.2.1). This is mainly due to the fact that no distinction was made between the EID of natural, intensive and extensive forest, and thus all forest types were assumed to have an average damage.

In a sensitibity analysis, the damage of intensive forest was estimated to be 0.3 EID higher than extensive forest, which corresponded with the difference between conventional and organic agriculture. The sensitivity analysis showed that when intensive forest is attributed a CF of 0.9 EID/m^2yr, the occupation impact of viscose increases with 31% and thereby exceeds the damage from conventional flax (see figure 18). However, viscose remains to have a lower occupation impact than irrigated cotton using the EID method, while ReCiPe concludes that viscose has a higher occupation than conventional flax and cotton (figure 18). The ReCiPe CF for intensive forest remains larger (relative to arable land damage) than the increased CF of intensive forest in the EID method.

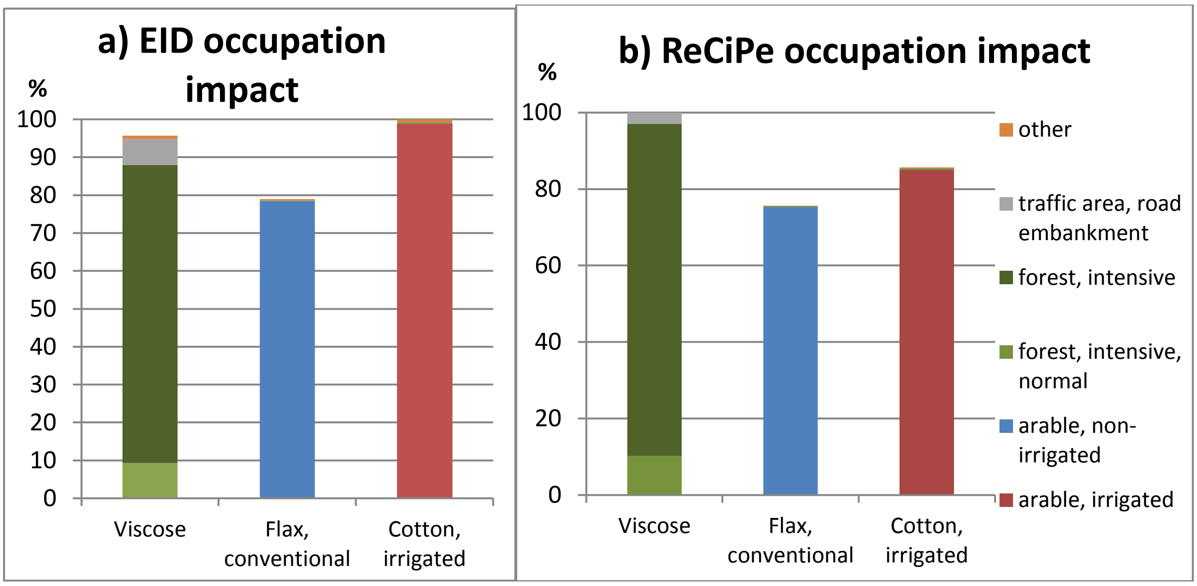


Figure 18. Comparison of occupation impact of viscose, conventional flax and conventional cotton, using a) differentiated CFs for forest types in the EID method and b) ReCiPe

Both ReCiPe and the EID method do not differentiate between the transformation impact of different forest types. After attributing a higher damage factor to intensive forest in the EID method, the transformation impact of viscose increased with 78%, which then comprised 12% of the total land use impact (compared to 4% without differentiating). This is due to the damage of transforming extensive forest to intensive forest, which was zero before differentiating (see section 7.1.2.1).

7.3.2 Influence of ecological integrity value on land occupation impact

More than 99% of land occupation impact of the agricultural fibers (flax and cotton) and 79% of viscose was determined by the main land type of the foreground system. The differences in land occupation impact between agricultural fibers could thus be explained by the land requirement (in m^2year) for cultivation, and the damage factor related to the agricultural type (see section 5.2.2). Sensitivity analysis revealed the influence of the EI value of different agricultural types, when excluding the effect of yield.

7.3.2.1 Irrigated versus non-irrigated arable land

Yield of non-irrigated cotton was assumed to be 55% smaller than irrigated cotton, which resulted in a 50% higher occupation damage (see section 7.1.1). In order to determine the influence of EI values related to non-irrigated and irrigated arable land, irrigated land was given the EID value of non-irrigated land and was compared to the original irrigated cotton record. It can be seen from figure 19 that the low EID value of non-irrigated arable land reduces the occupation impact with 10%. Because no transformation is related to the main land use type, changing the main cultivation type does not influence transformation impact. ReCiPe does not distinguish between impact of irrigated and non-irrigated land and therefore occupation impact of both types is the same (figure 19).

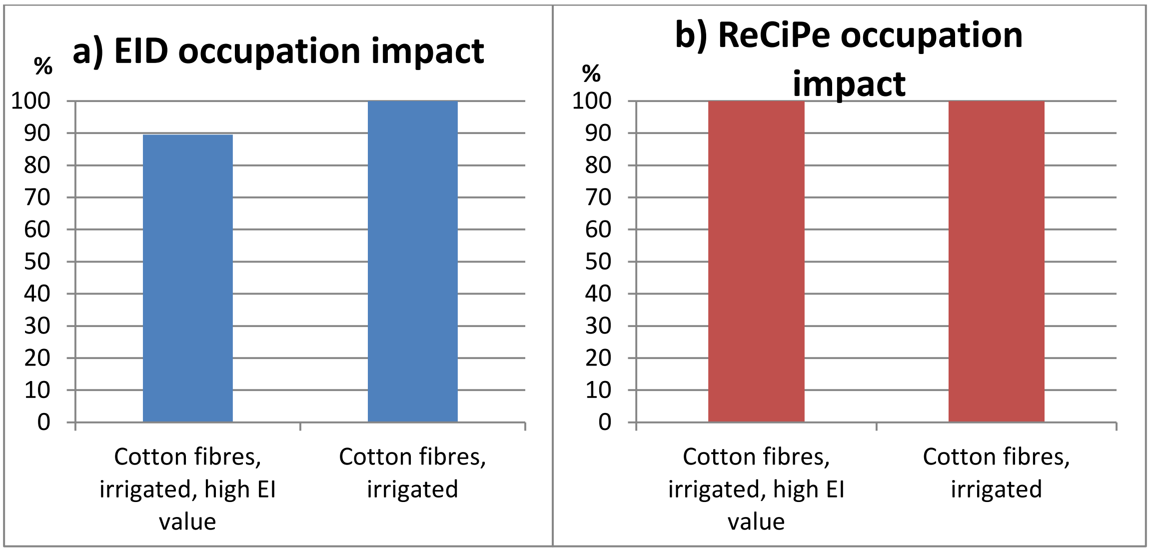


Figure 19. Comparison of occupation impact of irrigated cotton with higher ecological integrity value and regular irritated cotton using a) the EID method and b) ReCiPe

7.3.2.1 Organic versus conventional arable land

To determine the influence of the higher ecological integrity value of organic agriculture, non-irrigated cotton was given the EID value of organic land and compared to the original record of non-irrigated cotton. Figure 20 shows that organic land use results in a 18% lower occupation impact compared to non-irrigated land using the EID method, while using ReCiPe the difference is only 3%.

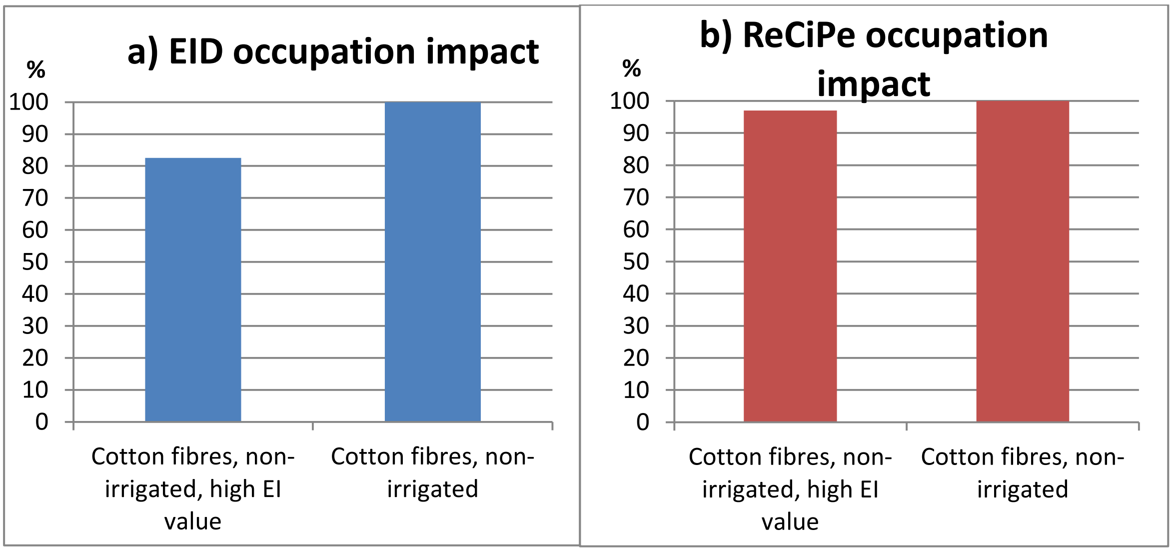


Figure 20. Comparison non-irrigated cotton with high ecological integrity and original non-irrigated cotton using a) the EID method and b) ReCiPe

The negative effect of larger land requirements is thus partly offset by the lower ecological integrity damage of organic agriculture. This means that in theory there is an optimum for land occupation impact of organic cotton: in case organic cotton has yields of at least 83% of conventional cotton yield, the resulting occupation impact is lower (taking into account the small contribution to occupation impacts of background processes). Similarly, non-irrigated cotton with a yield of at least 90% of irrigated cotton has a lower occupation impact than irrigated cotton.

7.3.3 Influence of differentiated regeneration time on land transformation impact

In this research, all transformation CFs were calculated using the same regeneration time of 50 years. It is, however, not realistic to assume that all land use types take the same amount of time to regenerate towards a natural system. Therefore in a sensitivity analysis the effect of differented regeneration time was evaluated: 0 for natural types, 50 years for agricultural types, and 100 years for artificial types. Transformation impacts increase 60-64% for all fibers and cultivation types (table 13). Netto transformation impacts increase, because most transformations are *from* natural land types, for which the CF values decrease (e.g. *transformation from forest* changes from -15 EID/m^2 to -0.7 EID/m^2) and thus the positive contribution decreases, while most transformations are *to* urban land types, for which the CFs increase (e.g. *transformation to urban* changes from 100 EID/m^2 to 220 EID/m^2) and thus the impact increases. The CFs of agricultural types did not change and thus do not influence the change in transformation impact.

Table 13. Land transformation damage to ecological integrity with average regeneration time and differentiated regeneration time

	Unit	Viscose	Flax, organic	Flax, conventional	Cotton, organic	Cotton, non-irrigated	Cotton, irrigated
Land transformation damage with average regeneration time	EID	211.5	37.5	36	31	103.5	48
Land transformation damage with differentiated regeneration time	EID	528	98.4	96.2	82.4	287	133
Increased impact differentiated regeneration time relative to average time	%	60	62	63	62	64	64

The interpretation of changes in transformation damage is difficult, because they are separated in a positive and a negative contribution, which partly cancel each other out (see section 5.2.2). The reason for the small differences in relative increase of transformation impact between the different fibers with differentiated regeneration times can be explained in short by the following: because the CFs of natural land types become smaller and the CFs of urban land types become larger, those fibers which include relatively more transformations involving natural types are associated with a lower increase in damage (viscose > flax > cotton, and organic > conventional), than those fibers which include relatively more transformations involving urban land types (cotton > flax > viscose, and conventional > organic). Transformation impact for the agricultural fibers comes mainly from the same two processes and therefore the change in transformation impact does not differ much between fiber and cultivation type.

7.3.4 Influence of foreground transformation on transformation impact

The records used in this research did not contain transformations in the foreground system, except viscose (discussed in section 7.2.2.1). For all agricultural fiber cultivation it was assumed that this has been the land use for a long time and thus no transformations were related to fiber cultivation (see Althaus et al, 2007). To test the influence of foreground transformation on ecological integrity damage, a transformation was added for the production of cotton, which represented the transformation of forest to the full area needed to produce 1000 kg of cotton fibers[4]. The transformation impact increased from 48 to 232,000 EID, almost 5,000 times as large. The transformation impact exceeded the occupation impact; it is almost 30 times as large. For ReCiPe transformation impact became even 10,000 times as much as cotton without the transformation and was 97 times as large as the occupation impact. This shows that transforming a natural area to agricultural area has a very large effect on ecological integrity according to this method, but an even larger effect on biodiversity using ReCiPe. The larger increase in transformation impact using ReCiPe can be explained by the relatively larger damage attributed to the transformation of forest and not taking into account the value of arable land. This means that only the damage due to the disappearance of forest is taken into account, but not value of gaining arable land.

[4] In this case the transformation was fully attributed to the production of 1000kg fibers, while in reality the agricultural land might be used for a longer time. Therefore an LCI practicioner might choose to allocate a smaller part of the transformation to the functional unit.

8. Discussion

The results that were presented in chapter 7 indicated several strengths and weaknesses of the EID method, their implications are discussed here. First, the selection of indicators and their ability to represent impacts on ecosystem services is discussed in section 8.1. Then the method for quantification of the indicators is evaluated in section 8.2. Finally, 8.3 elaborates upon the added value of the EID method and in which cases application of the EID method is preferred above conventional methods.

8.1 Indicator selection

The selection of appropriate indicators is a crucial aspect of the development of an LCIA method, and should reflect a relevant environmental pathway (ISO 14044:2006). According to the ILCD guidelines, damage to natural environment should reflect changes to the function and structure of ecosystems. To date, the Potentially Disappeared Fraction of Species (PDF) concept was recommended as an endpoint indicator, because it is sufficiently mature for application in LCA (see section 4.4). ReCiPe is a widely used method which applies PDF in its endpoint method. This is, however, only one aspect of ecosystem structure and function, which is why there is a need to include other aspects of ecosystem quality, and in specific, ecosystem services (see section 5.3.4).

The UNEP/SETAC Life Cycle Initiative has developed characterization factors for several ecosystem services: biotic production, erosion regulation potential, freshwater regulation potential and water purification potential (see section 5.3.4). These services are evaluated separately and are not yet combined to one operational integrated method. Furthermore, they do not encompass all ecosystem services (see section 5.3.4). If the goal is to develop an LCIA method which includes impacts on all ecosystem services, the set of indicators would become relatively large. Moreover, there is no agreement on the classification of ecosystem services and thus which ecosystem services should be quantified and which should not, in order to prevent double counting (see section 3.1.2).

The use of ecological integrity indicators provides a solution to this issue: the small set of indicators represents the ecological structures and processes underlying all ecosystem services (see section 3.1.3). Land use types with a high ecological integrity value have a large potential to provide ecosystem services. Expressed as a damage, ecological integrity indicators provides the basis for the land use assessment method that was the aim of this research. The three indicators that are identified by the EC-JRC and UNEP/SETAC as the most relevant to land use are included in the EID method: biotic production, biodiversity and ecological soil quality are represented within ecological integrity indicator set by exergy capture, biodiversity and storage capacity (SOM). The EID method in addition includes biotic water flows, reduction of nutrient loss, entropy production, metabolic efficiency, and abiotic heterogeneity. These indicators represent energy, water and matter balance, and biotic and abiotic structures, and are analogous to supporting ecosystem services (see section 3.2.3).

The strength of the EID method is that it provides a small set of indicators on one conceptual level (ecological structures and processes), which enables the calculation of a total value which can be readily applied in LCA. The indicators are strongly influenced by land use and thus are suitable to reflect differences between land use types (Burkhard et al., 2012, section 3.1.4). Because of the inherent complexity and interdependency of ecological processes, double counting is hard to prevent (see section 3.1.2). It is recognized that the ecological integrity indicator set might be subject of double counting: energy balance is represented by the three indicators exergy capture, entropy production, and metabolic efficiency. It may therefore be more appropriate to use just one of these indicators, this was however not further investigated in this research. Exergy capture is proposed as the most appropriate indicator to represent energy balance, as it is synonym to biotic production, one of the indicators proposed by the EC-JRC (see section 5.2.2).

8.2 Indicator quantification

Data availability constraints the quantification of ecosystem services and ecological integrity on large spatial scales, therefore expert judgment is often applied in addition to quantative data to valuate land use types (see section 3.2.2). In this research the 'matrix model' was used, which provides normalized values based on quantitative data in combination with expert judgment (see section 3.2.2). Jacobs et al. (2015) stress that the rather poor methodological transparency and low reproducibility make the matrix model a risky tool for actual decision support. However, the benefit is that a normalized matrix is easy to interpret and to apply. On a large spatial scale, as in LCA studies, collecting data is challenging (Stoll et al., 2015). Moreover, the urgency for a land use assessment method is large enough to use the knowledge and expertise that is there, rather than first collecting large amounts of data and knowledge (Jacobs et al., 2015; Daily, 1997). The matrix model has been successfully applied in various studies and proves to be useful especially on large spatial scales, and can be combined with quantitative and site-specific data when these are available (Stoll et al., 2015; Burkhard et al., 2009; Vihervaara et al., 2010).

The use of ecological integrity values based on European land use types and data makes the EID method especially relevant for assessing European land use. This is the scale for which most ecosystem services and ecological integrity assessments have been done to date (see section 3.2.2 and 3.2.4). When global data becomes available, this can be integrated within the EID method. Furthermore, biogeographical differentiation or the use of spatially explicit models can provide more detailed LCA results in the future (Koellner et al., 2013a).

8.3 Characterization factors

The EID method was developed to be compatible with the LCI database ecoinvent and thus characterization factors for all ecoinvent land interventions were developed. Because the level of detail from the input data of Stoll et al. (2015) did not correspond to the level of detail and nomenclature used in ecoinvent, there was need for further differentiation. For organic agriculture, indicators were quantified in order to differentiate between conventional and organic non-irrigated arable land. A comparison with ReCiPe by means of a case study helped to evaluate the method. Normalization and weighting was not applied, and is a matter for further research in order to compare land use impact on EI to other impact categories.

The implications of the differences that were found between ReCiPe and the EID method for assessment of agricultural occupation are discussed in section 8.3.1. For other land types, no differentiation based on intensity of use could be made due to time constraints. The implications for the calculation of forest related occupation and transformation impacts are discussed in section 8.3.2. The approach for calculating transformation impacts differs considerably between the EID method and ReCiPe. The implications of this difference are discussed in section 8.3.3.

8.3.1 Agricultural occupation

Ideally, a land use impact assessment method needs to be able to differentiate between the impacts of all different types of land use and intensity of use. This way, a better informed choice can be made between different management practices. For example, organic agriculture generally needs more land area for similar yield to conventional agriculture, but by maintaining ecosystem quality land use impact might be lower on the long term. This need for differentiation between agricultural types in LCA is stressed by the UNEP/SETAC (section 5.3.3). For biodiversity, there is already a certain extent of differentiation, e.g. ReCiPe differentiates between organic and conventional agriculture, and includes a CF for intensive monocultures (Goedkoop et al., 2013). For ecosystem services provision, this differentiation is not yet existent. The ecosystem services CFs of the UNEP/SETAC are defined for roughly the second hierarchical level of the CORINE land classification, only for biotic production (measured in soil organic matter) a distinction is made between several agricultural types (see section 5.3.4).

In the EID method, a lower ecological integrity damage is attributed to non-irrigated versus irrigated arable land, and organic versus conventional arable land (see section 7.1.1). ReCiPe attributes the same biodiversity damage to irrigated and non-irrigated agriculture, which agrees with the biodiversity value determined by Stoll et al. (2015) (see section 7.1.1.2). However, irrigated and non-irrigated agriculture differ in biotic water flow and metabolic efficiency, which results in a different total EID value (see Stoll et al., 2015). Similarly, the difference between conventional and organic agriculture in ReCiPe is based solely on biodiversity damage. The EI indicator values indicate that besides biodiversity, other integrity aspects differ between the two types, which results in a larger total quality difference (see section 7.1.1.1). This implies that when one is interested in solely biodiversity impacts of agricultural types, ReCiPe can be used, while one is interested in ecological integrity and the potential of arable land to deliver services, the EID method is more complete and enables differentiation between different agricultural practices.

8.3.2 Forest occupation

In this research the focus was on agricultural types, and thus no differentiation was made between natural, extensive and intensive forest. Moreover, the forest type nomenclature and definition by Stoll et al. (2015) (CORINE classification) do not correspond with the land interventions in ecoinvent, which hampers the use of the ecological integrity matrix values to calculate forest CFs in an LCIA method. The CORINE classification includes three forest types on the third hierarchical level, based on species composition: *broad-leaved*, *coniferous*, and *mixed forest*. Each forest type includes natural and used forests (Bossard et al., 2000; Koellner and Scholz, 2008). On the contrary, in ecoinvent a distinction is made on the basis of intensity of use (e.g. *extensive*, *intensive*, or *intensive, clear-cutting*), without distinguishing between species composition (Weidema et al., 2013). In addition, *tropical rain forest* is included, which is not defined within the CORINE classification, as this land types does not occur in Europe.

In the EID method, the average of the three forest types from the CORINE classification is used for all ecoinvent forest types. The CF_{occ} of forest types is relatively low (2.6 times smaller than the CF of arable land) when compared to intensive and extensive forest in ReCiPe (1.7 times and 2.2 times smaller than arable, respectively) (see section 7.1.2.1). Unlike the difference between agricultural CFs, this difference is not due to the inclusion of other EI indicators besides biodiversity: the ratio between damage of forest and arable land when just biodiversity damage is calculated from the values of Stoll et al. (2015) is similar to total EID (factor 3). The contribution of biodiversity damage to total ecological integrity damage is similar for both arable land and forest (21% and 20% respectively).

It is suggested here that the difference in relative forest damage between the two methods is a result of the definitions used for forest types: the CFs in the EID method are based on an 'average forest' (including natural, extensive and intensive forest), while ReCiPe assumes all occupation forest types to be 'used forests' (extensive and intensive forest) (Goedkoop et al., 2013). Therefore, larger damage factors can be expected than are attributed to 'average forest' in the EID method. When the average is calculated of the ReCiPe CFs for intensive, extensive and natural forest (the latter with a CF of zero), the ratio between this value and arable land is 2.8. This is comparable to the ratio in the EID method (2.6).

Thus, the differences in relative forest damage are a result of an underestimation of the damage of 'used forest' in the EID method. A sensitivity analysis in which an estimation was made for damage of intensive forest decreased the difference with ReCiPe, but was still relatively lower. This suggests that biodiversity of intensive forest might be relatively low, but intensive forest might be able to deliver several ecosystem services. This needs to be investigated in futher research. At the moment, ReCiPe is more suited to evaluate forest impacts, although it cannot evaluate ecological functioning.

8.3.3 Tropical rain forest transformation

The LCA results indicated that tropical rain forest transformation is related to a relatively high damage in ReCiPe, and a relatively low damage in the EID method (section 7.1.2.2). The EID method and ReCiPe use different assumptions and calculation methods for transformation impact, hence LCA practitioners need to be aware of these when interpreting the results. The method of calculation of transformation CFs by ReCiPe is based on a constant quality difference between natural and non-natural land types, multiplied by the regeneration time of the transformed land type (Goedkoop et al., 2013). In the EID method, CFs reflect a quality difference of each land use type with a natural reference type and the regeneration time towards this natural reference type. So far, a constant regeneration time was used for all land use types and thus the resulting CF of tropical rain forest is equal to other forest types.

While ReCiPe attributes the same occupation CF to extensive forest and tropical rain forest, for the calculation of transformation impact, the long regeneration time for tropical rain forest (3300 years) leads to a large transformation CF compared to other forest types (see also Weidema, 2015). Tropical rain forest is one of the most diverse and complex ecosystems, and therefore regeneration after transformation would take a long time, which justifies the long regeneration time used by ReCiPe (Bastian and Schreiber, 1999). However, this long regeneration time might be longer than the scope of LCA and thus could be considered irreversible within the chosen modelling time (see Koellner et al. 2013b). In addition, several factors impede the regeneration of tropical forests, and thus original ecosystem quality might never be restored within a reasonable timeframe (Hooper et al., 2005b). From the case study conducted in this research, it became clear that tropical rain forest transformations overshadowed other transformation impacts using ReCiPe (section 7.1.2.2). It could therefore be advisable to use a maximum regeneration time of the modelling period (e.g. 500 years, as suggested by Koellner et al. (2013b)) and state irreversible impacts separately. In this way, other potentially important transformations can get more attention.

The effect of a longer regeneration time of tropical rain forest could not be assessed in the EID method, because of the calculation method of transformation[5]. In a sensitivity analysis on the effect of differentiated regeneration times, the regeneration time towards a natural reference situation of all natural land types, including tropical rain forest, was assumed to be zero. The inclusion of an even longer regeneration time for tropical rain forest in the EID method could therefore not be analyzed. Although sensitivity analysis indicated that using differentiated regeneration times in the EID method did not change the ranking of the different fibers, it did cause a substantial increase in transformation impact (see section 7.3.3). Including realistic regeneration times in the EID method is therefore a matter for further research.

8.3.4 Calculation of transformation impacts

Another important difference relating land transformation between the two methods is that ReCiPe only takes into account transformations from or to natural land. This may lead to underestimation or overestimation of transformation impacts, as explained in section 7.1.3. Because the EID method includes transformation CFs for all land interventions, total transformation impact is more specific and comprehensive. Transformations from natural land to human land use, but also natural land to a different quality natural land, or changes within human land use types can be evaluated. This is of value for decision-making, for example for finding the most suitable location for a certain land use or evaluate the impact of active restoration.

[5] Following the UNEP/SETAC guidelines, the regeneration time is based on the regeneration of a certain land type (e.g. tropical rain forest) towards the reference land type (ecosystem with maximum ecological integrity). Therefore tropical rain forest can only be considered of similar ecosystem quality as the reference land type and not exceeding this quality.

9. Conclusion

This thesis linked the concepts of land use and ecosystem services in order to assess potential supply of ecosystem services by different land use types. To date, land use impact assessment in LCA only includes a part of ecosystem quality, mainly biodiversity and soil quality. This leads to incomplete evaluation of biobased products from agricultural and forestry feedstocks and hampers comparison with fossilbased products. Hence, the aim of this research was the development of a Life Cycle Impact Assessment (LCIA) method to evaluate potential impacts of different land use types on ecosystem services.

A review of state of the art LCIA methods pointed out the gaps that still remain within land use impact assessment, which were targeted in this research. The Ecological Integrity Damage (EID) method developed in this research fills the first two of these gaps: 1) it includes land use impacts on potential ecosystem services supply, and 2) includes one set of characterization factors (CFs) representing these impacts, which is compatible with LCI databases and thus easily applicable. The third gap was related to global applicability and biogeographical differentiation. The method is based on European data, and thus might not be accurate for larger or smaller spatial scales. The fourth gap was partly addressed, as a differentiation was made between several agricultural types, but not yet for other land types, most notably forest types. Thus, the EID method fills a large gap which exists in land use impact assessment, and the remaining issues can be addresses in further development of the method.

The use of ecological integrity indicators and the application of the matrix model based on quantitative data in combination with expert judgment is a new approach to assessing land use impacts in LCA. The 8 ecological integrity indicators represent the structural and functional aspects of ecosystem quality (biotic structures, abiotic structures, energy balance, water balance, and matter balance). 'Ecological Integrity' is proposed as a new endpoint indicator representing the Area of Protection 'Ecosystem Quality', to which other impact categories could be linked in future research using the same method proposed in this thesis.

In order to evaluate the EID method, it was applied to a case study on natural fibers. The results were compared to the LCA results using the ReCiPe endpoint method, which evaluates land use impacts on biodiversity. This comparison confirmed that the added value of the EID method is the inclusion of 7 ecosystem quality indicators besides biodiversity: when one is interested in ecosystem quality and impacts on potential ecosystem service supply, the EID method is more complete than LCIA methods that evaluate only biodiversity loss. Furthermore, it enables more differentiation between land use types. This relevance was demonstrated by the differentiation between agricultural types: using just biodiversity indicators, quality differences between agricultural types are small or non-existent, while using ecological integrity indicators the differences become apparent. Another strength of the EID method is the inclusion of characterization factors of all land interventions in ecoinvent, which enables the calculation of all possible land transformations, while ReCiPe enables only the calculation of the impact of natural land transformation.

In conclusion, the EID method provides a way to assess land use impacts on ecological integrity, and thus the potential supply of ecosystem services. It therefore answers the research question: *How can a Life Cycle Impact Assessment method be developed to assess land use impacts on ecosystem services?* The EID method provides a more complete way to assess land use impacts, and enables more differentiation between (agricultural) land types due to the use of 8 ecosystem quality indicators. It thereby enables a more complete impact assessment of biobased products, and other products involving land use. The case study demonstrated that the EID method is able to compare land use impacts of renewable material alternatives (e.g. cotton or flax), and different land management practices (e.g. irrigated versus non-irrigated cotton cultivation). By differentiating between different agricultural types, the method is particularly useful for assessing the trade-off between the amount of land use and the relative quality of a certain land type. This means that larger land requirements do not necessarily lead to a larger impact when a certain land type can maintain ecosystem health and thus ensure the sustainability of land use (e.g. lower yield organic versus high-yield conventional cotton). This more complete assessment of land use impacts improves the comparison of biobased and fossil-based products and can therefore contribute to choosing sustainable product alternatives with lowest environmental impacts.

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Appendix A

Figure A1. CORINE Plus land cover definitions. Italic entries indicate additional types, not included in the original CORINE version (Koellner and Scholz., 2008)

Type		Description (type of intensity)
1 Artificial surfaces		
10 Built up land		Land is totally covered with buildings or artificial infrastructure. (artificial_hi)
11 Urban fabric	111 Continuous urban fabric	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. (artificial_hi)
	112 Discontinuous urban fabric	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. (artificial_hi)
	113 Urban fallow	Remains of demolished houses are removed and areas are leveled. (non-use)
	114 Rural settlement	Village with buildings, streets and gardens. (artificial_li)
12 Industrial, commercial and transport	121 Industrial or commercial units	Artificially surfaced areas (with concrete, asphalt, tamacadam, or stabilized, e.g. beaten earth) devoid of vegetation occupy most of the area in question, which also contain buildings and/or areas with vegetation. (artificial_hi)
	121a Industrial area built up part	Land is totally covered with industrial buildings or infrastructure. (artificial_hi)
	121b Industrial area part with vegetation	High or medium proportion of sealed area; open soils, sometimes contaminated with chemicals. (artificial_hi)
	122 Road and rail networks and associated land	Motorways, railways, including associated installations (stations, platforms, embankments). Minimum width to include: 100 m.
	122a Road networks	Road networks. (artificial_hi)
	122b Road embankments and associated land	Road embankments and associated land. (artificial_li)
	122c Rail networks	Rail networks. (artificial_hi)
	122d Rail embankments and associated land	Rail embankments and associated land. (artificial_li)
	122e Rail fallow	Fallow land with spontaneous vegetation. (non-use)
	123 Port areas	Infrastructure of port areas, including quays, dockyards and marinas. (artificial_li)
	124 Airports	Airport installations: runways, buildings and associated land. (artificial_li)
	125 Industrial fallow	Areas with remains of industrial buildings; deposits of rubble, gravel, sand and industrial waste. (non-use)
13 Mine, dump and construction sites	131 Mineral extraction sites	Areas with open-pit extraction of industrial minerals (sandpits, quarries) or other minerals (opencast mines). Includes flooded gravel pits, except for riverbed extraction. (artificial_hi)
	132 Dump sites	Landfill or mine dump sites, industrial or public. (artificial_hi)
	133 Construction sites	Spaces under construction development, soil or bedrock excavations, earthworks. (artificial_hi)
	134 Mining fallow	Levelled deposits of slag heaps, rubble, gravel and sand. (non-use)
14 Artificial, non-agricultural areas with vegetation	141 Green urban areas	Areas with vegetation within urban fabric. Includes parks and cemeteries with vegetation. (artificial_li)
	142 Sport and leisure facilities	Camping grounds, sports grounds, leisure parks, golf courses, racecourses, etc. Includes formal parks not surrounded by urban zones. (artificial_li)
2 Agricultural areas		
21 Arable land		Cultivated areas regularly ploughed and generally under a rotation system.
	211 Non-irrigated arable land	Cereals, legumes, fodder crops, root crops and fallow land. Includes flower and tree (nurseries) cultivation and vegetables, whether open field, under plastic or glass (includes market gardening). Includes aromatic, medicinal and culinary plants. Excludes permanent pastures.
	211a Intensive	Chemical-synthetic and organic fertilizer as well as pesticides are applied. (agri_hi)
	211b Integrated	Chemical-synthetic and organic fertilizer as well as pesticides are applied. However, the input of these substances is reduced. 21121 Wheat, 21122 Maize (agri_hi)

Type		Description (type of intensity)
	211c Organic	Chemical-synthetic fertilizer and pesticides are not allowed, organic fertilizer is applied. (agri_hi)
	211d Fibre/energy crops	Crops for fibre and energy production: Kenaf (Hibiscus cannabinus L.), Hemp (Cannabis sativa L.), Chinese reed (Miscanthus sp.). (agri_hi)
	211e Agricultural fallow	Agricultural fallow. (non-use)
	211f Artificial meadow	Artificial meadow in rotation system. (agri-hi)
	212 Permanently irrigated land	Crops irrigated permanently and periodically, using a permanent infrastructure (irrigation channels, drainage network). Most of these crops could not be cultivated without an artificial water supply. Does not include sporadically irrigated land. (agri_hi)
	213 Rice fields	Land developed for rice cultivation. Flat surfaces with irrigation channels. Surfaces regularly flooded. (agri_hi)
22 Permanent crops		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. Excludes pastures, grazing lands and forests.
	221 Vineyards	Areas planted with vines.
	221a Intensive	Intensive vineyards. (agri_hi)
	221b Organic	Organic vineyards. (agri_li)
	222 Fruit trees and berry plantations	Parcels planted with fruit trees or shrubs: single or mixed fruit species, fruit trees associated with permanently grassed surfaces. Includes chestnut and walnut groves.
	222a Intensive orchards	Orchards with small growing fruit trees. (agri_hi)
	222b Organic orchards	Orchards with meadows and large fruit trees. (agri_li)
	223 Olive groves	Areas planted with olive trees, including mixed occurrence of olive trees and vines on the same parcel.
23 Pastures *and meadows*	231 Pastures *and meadows*	Dense, predominantly graminoid grass cover, of floral composition, not under a rotation system. Mainly used for grazing, but the fodder may be harvested mechanically. Includes areas with hedges (bocage), e.g. oat grass meadow (Arrhenatherion, Polygono), fertilized moist meadow (Calthion).
	231a Intensive pasture and meadows	Meadows mechanically harvested 3 times or more per year, fertilizer applied, perhaps on former arable land. (agri_hi)
	231b Less intensive pasture and meadows	Meadows mechanically harvested 2 or 3 times per year, reduced input of fertilizer, perhaps on former arable land. (agri_li)
	231c Organic pasture and meadows	Meadows mechanically harvested 1 time per year, no fertilizer, perhaps on former arable land. (agri_li)
24 Heterogeneous agricultural areas	241 Annual crops associated with permanent crops	Non-permanent crops (arable lands or pasture) associated with permanent crops on the same parcel.
	242 Complex cultivation	Juxtaposition of small parcels of diverse annual crops, pasture and/or permanent crops.
	243 Land principally occupied by agriculture	Areas principally occupied by agriculture, interspersed with significant natural areas. (agri_li)
	244 Agro-forestry areas	Annual crops or grazing land under the wooded cover of forestry species. (agri_li)
	245 Agricultural fallow with hedge-rows	Agricultural fallow with hedgerows. (non-use)
3 Forests and semi-natural areas		
31 Forests	311 Broad-leafed forest	Vegetation formation composed principally of trees, including shrub and bush under-stories, where broad-leafed species predominate. (Presence of conifers 0–10%)
	311a Broad leafed plantations	Plantations of fast growing tree species like poplar. (forest_hi)
	311b Semi-natural broad-leafed forests	Natural or semi-natural forests, where broad-leafed species predominate, either moist or arid form. (forest_li)
	312 Coniferous forest	Vegetation formation composed principally of trees, including shrub and bush under-stories, where coniferous species predominate. (Presence of conifers 91–100%)
	312a Coniferous plantations	Plantations of fast growing tree species like Picea abies. (forest_hi)
	312b Semi-natural coniferous forests	Natural or semi-natural forests, where coniferous species predominate. (forest_li)
	313 Mixed forest	Vegetation formation composed principally of trees, including shrub and bush under-stories, where broad-leafed and coniferous species co-dominate. (forest_li)

Appendix B

Table B1. Overview of definition related to ecosystem services used in various papers on ecosystem services

	Costanza, 1997	MA, 2005	De Groot et al., 2002; 2010; 2012	Wallace, 2007	Boyd and Banzhaf, 2007	Fisher et al., 2009
Biodiversity	n.d.	Variability among living organisms from all sources and the ecological complexes of which they are part. Diversity within and between species and diversity of ecosystems. Biodiversity influences ecosystem functions	n.d.	Variety of life forms (plants, animals, fungi, microorganisms, but ecosystems excluded). Generally used for natural biodiversity, but also encompasses cultural biodiversity	n.d.	n.d.
Ecosystem	n.d.	Dynamic complex of plant, animal and microorganism communities and the nonliving environment interacting as a functional unit. Natural and human-modified ecosystems can provide ES	n.d.	Functional entity or unit formed locally by all the organisms and their abiotic environment interacting with each other. Includes at least some natural elements.	n.d.	n.d.
Ecosystem elements/ components	n.d.	n.d.	Biotic and abiotic components	Biotic and abiotic tangible entities described in amount	Resources such as surface water, vegetation types, species populations, including structure	n.d.
Ecosystem composition	n.d.	n.d.	n.d.	Physical organization or pattern of a system/types and abundance of biotic and abiotic elements in a defined ecosystem	n.d.	n.d.
Ecosystem structure	n.d.	Includes diversity	n.d.	Identity and variety of elements in a collection/distribution and arrangement of elements	n.d.	Ecosystem components in Boyd and Banzhaf. Structure provides a platform from which ecosystem processes occur
Ecosystem processes	n.d.	Regulating services regulate ecosystem processes, e.g. primary production and nutrient	Same as supporting services. Results of complex	Complex interactions among biotic and abiotic elements that lead to a definite results. Transfer of energy and	Biological, chemical and physical interactions between ecosystem components. Intermediate to	n.d. (but probably support the definition of MA) e.g. nutrient cycling

		cycling	interactions between biotic and abiotic components of ecosystems through the universal driving forces of matter and energy.	materials. Operations and reactions described in terms of rates.	production of final ES.	
Ecosystem function	Habitat, biological or system properties and processes of ecosystems (e.g. regulation of hydrological flows)	An intrinsic ecosystem characteristic related to the set of conditions and processes whereby an ecosystem maintains its integrity (such as primary productivity, food chain, biogeochemical cycles). Ecosystem functions include such processes as decomposition, production, nutrient cycling, and fluxes of nutrients and energy.	-The capacity of ecosystems to provide goods and services that satisfy human needs, directly and indirectly -Consists of ecological complexity (structures and processes). -One ecosystem function can provide multiple services	Synonym for ecosystem processes, not used	Same as processes	Same as processes
Ecosystem services	the benefits human populations derive, directly and indirectly, from ecosystem functions	The benefits (goods and services) people derive from ecosystems, directly and indirectly.	Generated by ecosystem functions, can be product of one or more processes	Benefits that people derive from ecosystems. Obtained from natural elements of ecosystems. (sometimes also from cultural elements). Includes goods and services.	Final ecosystem services are components of nature, directly enjoyed, consumed, or used to yield human well-being. They are components/ecological things or characteristics, not functions or processes.	The aspects of ecosystems utilized (actively or passively) to produce human well-being (processes/functions when there are human beneficiaries) Ecosystem organization or structure as well as processes/functions might be ES if they have human beneficiaries, e.g. flood regulation, C seq., pollination
Benefits	Costs and benefits associated with human activities can be influenced by changes in quality or quantity of ecosystem services	Same as escosystem services	Actual use of goods or services provides benefits (nutrition, health, pleasure etc), which can be valued in economic terms and monetary terms.	Same as ecosystem services	explicit impact on changes in human welfare total benefit=quantity X values	explicit impact on changes in human welfare. Water for irrigation, drinking water, timber
Human well-being	n.d.	Includes multiple constituents: -Basic material for a good life: secure and adequate	Strongly tied to experience of natural landscapes and species diversity. Nature provides non-material	Adequate resources, protection from predators/disease/parasites, benign physical and chemical	Sources of well-being: aesthetic enjoyment, recreation, maintenance of human health, physical	n.d.

		livelihoods, food, shelter, clothing, access to goods -health, including a healthy environment -good social relations -security, including access to resources, and security from disasters -freedom of choice and action	well-being, besides materials for well-being	environment, socio-cultural fulfilment.	damage avoidance, subsistence or foraged consumption of food and fiber. The goal of social policy is to maximize human well-being as opposed to a purely ecological objective.	

Appendix C

Table C1. Relation of the land types in ecoinvent with the CORINE classes and their ecological integrity value determined by Stoll et al. (2015)

Land type ecoinvent	Land type Stoll et al. (2015)	EI value
agriculture	average of lower hierarchical levels	3.1
arable	average of lower hierarchical levels	3.2
arable, fallow	non-irrigated arable land	3.3
arable, conservation tillage		3.2
arable, conventional tillage	same as *arable*	3.2
arable, flooded crops	rice fields	3.3
arable, greenhouse	same as *arable*	3.2
arable, integrated	same as *arable*	3.2
arable, irrigated		3.1
arable, irrigated, extensive		3.1
arable, irrigated, intensive	permanently irrigated land	3.1
arable, non-irrigated		3.3
arable, non-irrigated, diverse-intensive	non-irrigated arable land	3.3
arable, non-irrigated, extensive		3.3
arable, non-irrigated, fallow		3.3
arable, non-irrigated, intensive		3.3
arable, non-irrigated, monotone-intensive		3.3
arable, organic	estimated in this research (table 6)	3.6
arable, reduced tillage	same as *arable*	3.2
artificial areas	average of lower hierarchical levels	1.0
construction site	construction sites	0.4
dump site	dump sites	0.8
dump site, benthos		0.8
forest		4.4
forest, extensive	Average of *broad-leaved, coniferous and mixed forest*	4.4
forest, intensive		4.4
forest, intensive, clear-cutting		4.4
forest, intensive, normal		4.4
forest, intensive, short-cycle		4.4
forest, used		4.4
grassland	natural grassland	4.3
grassland, for livestock grazing	pasture	3.6
grassland, not used	natural grassland	4.3
grassland/pasture/meadow	average of pasture and natural grassland	4.0
heterogeneous, agricultural	average of lower hierarchical levels	3.0
industrial area	industrial or commercial units	0.4
industrial area, benthos		0.4
industrial area, built up		0.4

industrial area, vegetation		0.4
mineral extraction site	mineral extraction sites	0.5
pasture and meadow	pastures	3.6
pasture and meadow, extensive		3.6
pasture and meadow, intensive		3.6
pasture and meadow, organic		3.6
permanent crop	average of lower hierarchical levels	2.5
permanent crop, fruit	fruit trees and berry plantations	2.9
permanent crop, fruit, extensive		2.9
permanent crop, fruit, intensive		2.9
permanent crop, vine	vineyards	2.1
permanent crop, vine, extensive		2.1
permanent crop, vine, intensive		2.1
permanent crops, irrigated	same as permanent crop	2.5
permanent crops, irrigated, extensive		2.5
permanent crops, irrigated, intensive		2.5
permanent crops, non-irrigated		2.5
permanent crops, non-irrigated, extensive		2.5
permanent crops, non-irrigated, intensive		2.5
sea and ocean	sea and ocean	2.1
shrub land, sclerophyllous	sclerophyllous vegetation	3.1
sparsely vegetated areas, steppe, tundra, badlands	sparsely vegetated areas	1.5
traffic area	road and rail networks	0.5
traffic area, rail embankment		0.5
traffic area, rail network		0.5
traffic area, road embankment		0.5
traffic area, road network		0.5
tropical rain forest	mixed forest	4.4
unknown	average of all values	2.6
urban	average of lower hierarchical levels	0.6
urban, continuously built	continuous urban fabric	0.1
urban, discontinuously built	discontinuous urban fabric	1.1
urban, green areas	green urban areas	2.6
urban/industrial fallow	same as urban	0.6
water bodies, artificial	water bodies	3.3
water courses, artificial	water courses	2.5

Table C2. Characterization factors for *Land occupation damage to EI*

Intervention	CF	Unit
Occupation, agriculture	1.9	EID/m^{2}yr
Occupation, arable	1.8	EID/m^{2}yr
Occupation, arable, fallow	1.7	EID/m^{2}yr
Occupation, arable, conservation tillage	1.8	EID/m^{2}yr
Occupation, arable, conventional tillage	1.8	EID/m^{2}yr
Occupation, arable, flooded crops	1.7	EID/m^{2}yr
Occupation, arable, greenhouse	1.8	EID/m^{2}yr
Occupation, arable, integrated	1.8	EID/m^{2}yr
Occupation, arable, irrigated	1.9	EID/m^{2}yr
Occupation, arable, irrigated, extensive	1.9	EID/m^{2}yr
Occupation, arable, irrigated, intensive	1.9	EID/m^{2}yr
Occupation, arable, non-irrigated	1.7	EID/m^{2}yr
Occupation, arable, non-irrigated, diverse-intensive	1.7	EID/m^{2}yr
Occupation, arable, non-irrigated, extensive	1.7	EID/m^{2}yr
Occupation, arable, non-irrigated, fallow	1.7	EID/m^{2}yr
Occupation, arable, non-irrigated, intensive	1.7	EID/m^{2}yr
Occupation, arable, non-irrigated, monotone-intensive	1.7	EID/m^{2}yr
Occupation, arable, organic	1.4	EID/m^{2}yr
Occupation, arable, reduced tillage	1.8	EID/m^{2}yr
Occupation, artificial areas	4.0	EID/m^{2}yr
Occupation, construction site	4.6	EID/m^{2}yr
Occupation, dump site	4.2	EID/m^{2}yr
Occupation, dump site, benthos	4.2	EID/m^{2}yr
Occupation, forest	0.6	EID/m^{2}yr
Occupation, forest, extensive	0.6	EID/m^{2}yr
Occupation, forest, intensive	0.6	EID/m^{2}yr
Occupation, forest, intensive, clear-cutting	0.6	EID/m^{2}yr
Occupation, forest, intensive, normal	0.6	EID/m^{2}yr
Occupation, forest, intensive, short-cycle	0.6	EID/m^{2}yr
Occupation, forest, used	0.6	EID/m^{2}yr
Occupation, grassland	0.7	EID/m^{2}yr
Occupation, grassland, for livestock grazing	1.4	EID/m^{2}yr
Occupation, grassland, not used	0.7	EID/m^{2}yr
Occupation, grassland/pasture/meadow	1.0	EID/m^{2}yr
Occupation, heterogeneous, agricultural	2.0	EID/m^{2}yr
Occupation, industrial area	4.6	EID/m^{2}yr
Occupation, industrial area, benthos	4.6	EID/m^{2}yr
Occupation, industrial area, built up	4.6	EID/m^{2}yr
Occupation, industrial area, vegetation	4.6	EID/m^{2}yr
Occupation, mineral extraction site	4.5	EID/m^{2}yr
Occupation, pasture and meadow	1.4	EID/m^{2}yr
Occupation, pasture and meadow, extensive	1.4	EID/m^{2}yr
Occupation, pasture and meadow, intensive	1.4	EID/m^{2}yr

Occupation, pasture and meadow, organic	1.4	EID/m^2yr
Occupation, permanent crop	2.5	EID/m^2yr
Occupation, permanent crop, fruit	2.1	EID/m^2yr
Occupation, permanent crop, fruit, extensive	2.1	EID/m^2yr
Occupation, permanent crop, fruit, intensive	2.1	EID/m^2yr
Occupation, permanent crop, vine	2.9	EID/m^2yr
Occupation, permanent crop, vine, extensive	2.9	EID/m^2yr
Occupation, permanent crop, vine, intensive	2.9	EID/m^2yr
Occupation, permanent crops, irrigated	2.5	EID/m^2yr
Occupation, permanent crops, irrigated, extensive	2.5	EID/m^2yr
Occupation, permanent crops, irrigated, intensive	2.5	EID/m^2yr
Occupation, permanent crops, non-irrigated	2.5	EID/m^2yr
Occupation, permanent crops, non-irrigated, extensive	2.5	EID/m^2yr
Occupation, permanent crops, non-irrigated, intensive	2.5	EID/m^2yr
Occupation, sea and ocean	2.9	EID/m^2yr
Occupation, shrub land, sclerophyllous	1.9	EID/m^2yr
Occupation, sparsely vegetated areas, steppe, tundra, badlands	3.5	EID/m^2yr
Occupation, traffic area	4.5	EID/m^2yr
Occupation, traffic area, rail embankment	4.5	EID/m^2yr
Occupation, traffic area, rail network	4.5	EID/m^2yr
Occupation, traffic area, road embankment	4.5	EID/m^2yr
Occupation, traffic area, road network	4.5	EID/m^2yr
Occupation, tropical rain forest	0.6	EID/m^2yr
Occupation, unknown	2.4	EID/m^2yr
Occupation, urban	4.4	EID/m^2yr
Occupation, urban, continuously built	4.9	EID/m^2yr
Occupation, urban, discontinuously built	3.9	EID/m^2yr
Occupation, urban, green areas	2.4	EID/m^2yr
Occupation, urban/industrial fallow	4.4	EID/m^2yr
Occupation, water bodies, artificial	1.7	EID/m^2yr
Occupation, water courses, artificial	2.4	EID/m^2yr

Table C3. Characterization factors for *Land transformation damage to EI*

Intervention	CF	Unit
Transformation, from agriculture	-47.5	EID/m^2
Transformation, from arable	-45	EID/m^2
Transformation, from arable, fallow	-42.5	EID/m^2
Transformation, from arable, irrigated	-47.5	EID/m^2
Transformation, from arable, irrigated, extensive	-47.5	EID/m^2
Transformation, from arable, irrigated, intensive	-47.5	EID/m^2
Transformation, from arable, non-irrigated	-42.5	EID/m^2
Transformation, from arable, non-irrigated, diverse-intensive	-42.5	EID/m^2
Transformation, from arable, non-irrigated, extensive	-42.5	EID/m^2
Transformation, from arable, non-irrigated, fallow	-42.5	EID/m^2
Transformation, from arable, non-irrigated, intensive	-42.5	EID/m^2

Transformation, from arable, non-irrigated, monotone-intensive	-42.5	EID/m^2
Transformation, from arable, organic	-35	EID/m^2
Transformation, from artificial areas	-100	EID/m^2
Transformation, from dump site	-105	EID/m^2
Transformation, from dump site, benthos	-105	EID/m^2
Transformation, from dump site, inert material landfill	-105	EID/m^2
Transformation, from dump site, residual material landfill	-105	EID/m^2
Transformation, from dump site, sanitary landfill	-105	EID/m^2
Transformation, from dump site, slag compartment	-105	EID/m^2
Transformation, from forest	-15	EID/m^2
Transformation, from forest, extensive	-15	EID/m^2
Transformation, from forest, intensive	-15	EID/m^2
Transformation, from forest, intensive, clear-cutting	-15	EID/m^2
Transformation, from forest, intensive, normal	-15	EID/m^2
Transformation, from forest, intensive, short-cycle	-15	EID/m^2
Transformation, from forest, natural	-15	EID/m^2
Transformation, from forest, primary	-15	EID/m^2
Transformation, from forest, secondary	-15	EID/m^2
Transformation, from forest, used	-15	EID/m^2
Transformation, from grassland	-17.5	EID/m^2
Transformation, from grassland, for livestock grazing	-35	EID/m^2
Transformation, from grassland, not used	-17.5	EID/m^2
Transformation, from grassland/pasture/meadow	-25	EID/m^2
Transformation, from heterogeneous, agricultural	-50	EID/m^2
Transformation, from industrial area	-115	EID/m^2
Transformation, from industrial area, benthos	-115	EID/m^2
Transformation, from industrial area, built up	-115	EID/m^2
Transformation, from industrial area, vegetation	-115	EID/m^2
Transformation, from mineral extraction site	-112.5	EID/m^2
Transformation, from pasture and meadow	-35	EID/m^2
Transformation, from pasture and meadow, extensive	-35	EID/m^2
Transformation, from pasture and meadow, intensive	-35	EID/m^2
Transformation, from pasture and meadow, organic	-35	EID/m^2
Transformation, from permanent crop	-62.5	EID/m^2
Transformation, from permanent crop, fruit	-52.5	EID/m^2
Transformation, from permanent crop, fruit, extensive	-52.5	EID/m^2
Transformation, from permanent crop, fruit, intensive	-52.5	EID/m^2
Transformation, from permanent crop, vine	-72.5	EID/m^2
Transformation, from permanent crop, vine, extensive	-72.5	EID/m^2
Transformation, from permanent crop, vine, intensive	-72.5	EID/m^2
Transformation, from permanent crops, irrigated	-62.5	EID/m^2
Transformation, from permanent crops, irrigated, extensive	-62.5	EID/m^2
Transformation, from permanent crops, irrigated, intensive	-62.5	EID/m^2
Transformation, from permanent crops, non-irrigated	-62.5	EID/m^2
Transformation, from permanent crops, non-irrigated, extensive	-62.5	EID/m^2
Transformation, from permanent crops, non-irrigated, intensive	-62.5	EID/m^2

Transformation, from sea and ocean	-72.5	EID/m^2
Transformation, from shrub land, sclerophyllous	-47.5	EID/m^2
Transformation, from traffic area	-112.5	EID/m^2
Transformation, from traffic area, rail embankment	-112.5	EID/m^2
Transformation, from traffic area, rail network	-112.5	EID/m^2
Transformation, from traffic area, road embankment	-112.5	EID/m^2
Transformation, from traffic area, road network	-112.5	EID/m^2
Transformation, from tropical rain forest	-15	EID/m^2
Transformation, from unknown	-60	EID/m^2
Transformation, from unspecified, natural	-52.5	EID/m^2
Transformation, from unspecified, used	-75	EID/m^2
Transformation, from urban	-110	EID/m^2
Transformation, from urban, continuously built	-122.5	EID/m^2
Transformation, from urban, discontinuously built	-97.5	EID/m^2
Transformation, from urban/industrial fallow	-110	EID/m^2
Transformation, from water bodies, artificial	-42.5	EID/m^2
Transformation, from water courses, artificial	-60	EID/m^2
Transformation, to agriculture	47.5	EID/m^2
Transformation, to agriculture, mosaic	47.5	EID/m^2
Transformation, to arable	45	EID/m^2
Transformation, to arable, fallow	42.5	EID/m^2
Transformation, to arable, irrigated	47.5	EID/m^2
Transformation, to arable, irrigated, extensive	47.5	EID/m^2
Transformation, to arable, irrigated, intensive	47.5	EID/m^2
Transformation, to arable, non-irrigated	42.5	EID/m^2
Transformation, to arable, non-irrigated, diverse-intensive	42.5	EID/m^2
Transformation, to arable, non-irrigated, extensive	42.5	EID/m^2
Transformation, to arable, non-irrigated, fallow	42.5	EID/m^2
Transformation, to arable, non-irrigated, intensive	42.5	EID/m^2
Transformation, to arable, non-irrigated, monotone-intensive	42.5	EID/m^2
Transformation, to arable, organic	35	EID/m^2
Transformation, to dump site	105	EID/m^2
Transformation, to dump site, benthos	105	EID/m^2
Transformation, to dump site, inert material landfill	105	EID/m^2
Transformation, to dump site, residual material landfill	105	EID/m^2
Transformation, to dump site, sanitary landfill	105	EID/m^2
Transformation, to dump site, slag compartment	105	EID/m^2
Transformation, to forest	15	EID/m^2
Transformation, to forest, extensive	15	EID/m^2
Transformation, to forest, intensive	15	EID/m^2
Transformation, to forest, intensive, clear-cutting	15	EID/m^2
Transformation, to forest, intensive, normal	15	EID/m^2
Transformation, to forest, intensive, short-cycle	15	EID/m^2
Transformation, to heterogeneous, agricultural	50	EID/m^2
Transformation, to industrial area	115	EID/m^2
Transformation, to industrial area, benthos	115	EID/m^2

Transformation, to industrial area, built up	115	EID/m^2
Transformation, to industrial area, vegetation	115	EID/m^2
Transformation, to mineral extraction site	112.5	EID/m^2
Transformation, to pasture and meadow	35	EID/m^2
Transformation, to pasture and meadow, extensive	35	EID/m^2
Transformation, to pasture and meadow, intensive	35	EID/m^2
Transformation, to pasture and meadow, organic	35	EID/m^2
Transformation, to permanent crop	62.5	EID/m^2
Transformation, to permanent crop, fruit	52.5	EID/m^2
Transformation, to permanent crop, fruit, extensive	52.5	EID/m^2
Transformation, to permanent crop, fruit, intensive	52.5	EID/m^2
Transformation, to permanent crop, vine	72.5	EID/m^2
Transformation, to permanent crop, vine, extensive	72.5	EID/m^2
Transformation, to permanent crop, vine, intensive	72.5	EID/m^2
Transformation, to permanent crops, irrigated	62.5	EID/m^2
Transformation, to permanent crops, irrigated, extensive	62.5	EID/m^2
Transformation, to permanent crops, irrigated, intensive	62.5	EID/m^2
Transformation, to permanent crops, non-irrigated	62.5	EID/m^2
Transformation, to permanent crops, non-irrigated, extensive	62.5	EID/m^2
Transformation, to permanent crops, non-irrigated, intensive	62.5	EID/m^2
Transformation, to sea and ocean	72.5	EID/m^2
Transformation, to shrub land, sclerophyllous	47.5	EID/m^2
Transformation, to traffic area	112.5	EID/m^2
Transformation, to traffic area, rail embankment	112.5	EID/m^2
Transformation, to traffic area, rail network	112.5	EID/m^2
Transformation, to traffic area, road embankment	112.5	EID/m^2
Transformation, to traffic area, road network	112.5	EID/m^2
Transformation, to tropical rain forest	15	EID/m^2
Transformation, to unknown	60	EID/m^2
Transformation, to unspecified, used	75	EID/m^2
Transformation, to urban	110	EID/m^2
Transformation, to urban, continuously built	122.5	EID/m^2
Transformation, to urban, discontinuously built	97.5	EID/m^2
Transformation, to urban, green areas	60	EID/m^2
Transformation, to urban/industrial fallow	110	EID/m^2
Transformation, to water bodies, artificial	42.5	EID/m^2
Transformation, to water courses, artificial	60	EID/m^2

Appendix D

Table D1. Comparison of land use impacts for all different fibers and cultivation types using the EID method and ReCiPe

	Unit	Viscose fibres	Flax fibers, non-irrigated, organic	Flax fibers, non-irrigated, conventional	Cotton fibres, non-irrigated organic	Cotton fibres, non-irrigated, conventional	Cotton fibres, irrigated, conventional
EID method							
Land occupation impact	EID	4930	8590	5855	12070	14737	7410
Land transformation impact	EID	212	38	36	31	104	48
Total EID	EID	5141	8628	5891	12100	14840	7458
Relative contribution transformation to total EID	%	4.1	0.4	0.6	0.3	0.7	0.6
ReCiPe (H) endpoint							
ReCiPe total land occupation impact	species.yr	1.79E-04	2.34E-04	1.36E-04	3.29E-04	3.41E-04	1.53E-04
ReCiPe natural land transformation	species.yr	4.04E-06	0.520E-6	1.05E-06	0.409E-06	3.21E-06	1.45E-06
Total ReCiPe impact	species.yr	1.83E-04	2.35E-04	1.37E-04	3.29E-04	3.44E-04	1.55E-04
Relative contribution transformation to total ReCiPe impact	%	2.2	0.2	0.8	0.1	0.9	0.9