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Contents

0 Executive Summary

The goal of this study (task 8.4) is the methodology development of new environmental impact categories for the evaluation of biobased products, to make life cycle assessments of biobased products more complete, accurate and better comparable. While the study itself has a focus on environmental sciences, this executive summary is written for non- scientists

The general approach of this study is to identify current issues addressed by several policy sources and try to find solutions for these issues, which fit in the state of the art methodologies and are scientifically sound: policy defines the problems, solutions are searched for in science. The scope of the selection procedure for new categories is limited to those that are related with agricultural and silvicultural product systems from which the biobased products originate, and which affect the environment in a different way than fossil products. The practical application of this methodology will be worked out in the other (assessment) tasks of BioBuild.

The policy issues with respect to the environmental assessment of biobased products have been analysed, both from a European and from a global perspective. Summarizing, from the policy there is an actual call for better indicators for Water, Land (-scape) and/or Soil and Ecosystem Services & Biodiversity related impacts and Food security.

Next, it has been analysed to what extent these issues could be addressed with state of the art environmental impact assessment methodologies and where the main gaps in science are located. Limiting uncertainties is important to increase acceptance of the chosen methodology as the outcomes will be used for decision making. End-point indicators introduce many additional uncertainties and the uncertainties in the currently available methods are judged to be too high, either due to lack of inventory data or as a result of limited available quantification methods. Therefore mid-point indicators are preferred to assess bio-related impacts. This decision results in the outcome that biodiversity and food competition will not be quantified, as they both are considered to be end-point indicators.

Landscape was seen as a relevant pressure, but it is to a large extent dependent on the specific geographical location of impact. Due to the local nature of landscape impacts it cannot be integrated in generic life cycle assessments, and is more likely to be an indicator for (local) environmental risk studies. Therefore Landscape impacts were not investigated further in this study.

Hydrology and soil quality are the impact categories that were selected for further investigation in this study. The different methods for environmental impact assessment of hydrology and soil quality were investigated in a descriptive and analytical manner.
Based on this scientific methodology study, the conclusion can be drawn that impact assessment of bio-related impacts is composed of two factors, being quantity and quality.

In hydrology, the main issue is water, as water is relevant for all three Areas of Protection[1]. Water can be assessed in two ways: by assessing the water use or the water stress. Two methods for measuring the impact of water use were discussed (Bayart et al. and Milà). For water stress, only the most prominent method was discussed (Pfister et al.). The conclusion from the analysis is that it would be valuable to add “water” as a new environmental impact category, which is calculated by means of two indicators: the water stress index and water as a resource (the aquifer reservoirs, as calculated in the blue water footprint).

[1] Human Health, Ecosystems and Resources.

Soil quality can be incorporated in environmental impact assessments by methodologies on the basis of either soil erosion or soil organic matter. Several methods were discussed how soil erosion can be included in LCA. Data is available in the form of maps. Regarding soil organic matter, there are also several calculation methods. The method of Mila I Canals et al is recommended by the ILCD. The analysis of all different methods leads to the advice to add “soil quality” as a new environmental impact category, which is calculated by means of two indicators: soil erosion, to be derived from erosion maps or calculated with (r)USLE in specific cases, and Soil organic matter, following the method of Milà I Canals (2007a,b,c) as also recommended by the ILCD Handbook.

By means of the proposed additions to the current impact assessment methods, the policy request for better indicators concerning Water and Soil can be covered. The request for better biodiversity indicators could not be solved because of the additional uncertainties that are introduced in endpoint damage approaches, as described above. For Biodiversity the occupation of land seems to be the most suitable midpoint indicator, as there is enough inventory data available, and it is free of value based weighting. The crude m2 land use indicator can also be used as a mid-point indicator for land-use completion that may threat food security. It is impossible to draw conclusions in this value laden public debate without subjective weighting.

Environmental assessment in BioBuild needs to distinguish between the final assessment at the end of the BioBuild project and the environmental assessment that is executed respectively at the beginning of the project (benchmark) and during the development process (quick-scans for decision making). Because the benchmark study is already finished and quick scans are already serving their purpose, and because the quick scan methodology represent the new bio-related impact themes (i.e. soil quality and hydrology) in a coarse way (i.e. land use and water use), it is decided not to change the quick scan method at this stage of the BioBuild project. This decision will keep the quick scans consistent.

The quantification of land use can be seen as the coarse (quantitative) mid-point indicator for ecosystem services, biodiversity, food competition and soil quality. The final assessment will further elaborate on the midpoint indictors for soil quality, while the end-point indicator biodiversity is assessed in a sensitivity analysis based on the ILCD recommended methodology to express biodiversity in terms of Potentially Disapeard Fraction of Species. Food competition will not be quantified. For food competition, a qualitative description will assess whether it is likely that food competition can become an issue for the product under study.

The quantification of water can be seen as a coarse midpoint indicator for hydrology. In the previous paragraph was concluded that that it would be valuable to add “water” as a new environmental impact category, which is calculated by means of two indicators: the water stress index and water as a resource (the aquifer reservoirs, as calculated in the blue water footprint). However, the current LCO software that is used to perform the environmental assessment in BioBuild is not yet suitable to perform this water assessment. The database supplier (Ecoinvent) has been working on an update of the database that enables water assessment. However the LCA software needs to be adapted to be suitable for this new type of database. The release of the software update is not yet planned. If the new version of the software is available at the time of the final assessment, it will be applied for BioBuild. If not, the final assessment will be based on water quantity only, and a qualitative interpretation will be made on the quality of the used water to describe the (potential) impact.

1 Introduction

1.1 Environmental impact assessment for FP7 BioBuild research

The European Union has set up several research programmes of which the Seventh Framework Programme (FP7) is the most recent. FP7 aims at strengthening the scientific and technologic base of European industry and to encourage its international competitiveness while promoting research that supports EU policies. This framework programme is worked out by several calls for proposals. One of them is call for energy efficient buildings, which calls amongst others for the topic “Materials for new energy efficient building components with reduced embodied energy” (European Commission, 2010). One of the granted projects is *BioBuild*. The focus of BioBuild is on the development of High Performance, Economical and Sustainable Biocomposite Building Materials.

Within the BioBuild project Work package 8 is related to the environmental, economic and social assessment of the BioBuild products to be developed. The environmental assessment is based on the life cycle assessment approach. The current environmental impact assessment methodologies are developed to assess products and processes in the current fossil based economy. As a result the methods cover the most common effects that are caused by the fossil-related processes. Biobased products however can be related to other environmental effects that are not yet covered by the current methodologies. In the BioBuild project, there is room for improvement of the current LCA methodology with respect to biobased materials. Task 8.4 is related to methodology development for new environmental impact categories for the evaluation of bio-based materials. The report you are reading is the final product of this task (Deliverable 8.4). This study covers the scientific methodology development resulting in new definitions of impact analysis for bio-related products. The practical application of this methodology needs to be worked out in the other tasks of BioBuild (task 8.3 quick-scans to support the design process and task 8.5 final assessment).

1.2 Background

1.2.1 Environmental assessment of biobased products

Biobased products rely for their production on the agricultural and silvicultural (forestry) production systems. These agro- and silvicultural systems need land, water, nutrients and energy from the sun as main inputs and thereby affect the environment. The current LCIA methods have focussed on products from industrial production processes and have paid less attention to biobased products. Several European and international sources point out that more research on the specific environmental impacts of biobased products is needed.

The European Union is well aware that current production and consumption have negative environmental impacts and that indicators are important tools to measure and foster progress towards resource efficiency and more environmental practices. It acknowledges, however, that indicators for certain resources such as water, biodiversity, land and soil are currently lacking (European Commission, 2011a). Well-directed environmental policy is therefore hard to develop and implement. Research to increase the understanding of the environmental impacts of biobased materials is thus needed.

The European Union is not alone in this opinion. Also, the International Resource Panel of the United Nations Environment Programme (UNEP) recognises that important environmental impacts – on GHG emissions, water and biodiversity for example – of biofuel production are not covered by LCA methods. The Panel hence concludes that further development of LCA methods is needed (Bringezu, Schütz, O´Brien, Kauppi, Howarth, & McNeely, 2012).

Another voice in this field is the Organisation for Economic Co-operation and Development (OECD). The OECD considers the impacts of agriculture on the environment and the achievement of sustainable agriculture of major public concern in the context of agricultural policy reform, trade liberalisation, and multilateral environmental agreements (OECD, 2001). These specific impacts on the environment should therefore ideally be reflected in Life Cycle Impact Assessment (LCIA) methodologies.

1.3 Goal of this study

The goal of this report, as described in the Description of Work (DoW) of the project, is the methodology development of new environmental impact categories for the evaluation of biobased products. The goal of this study is to select and work out new environmental impact categories which will supplement the current life cycle impact assessment methodology and which will make life cycle assessments of biobased products more complete, accurate and better comparable.

The output of this study is this report, proposing additional impact categories and appurtenant indicators in relation to international policies and to existing methodology. Moreover, this report sketches the following steps necessary for implementation.

1.4 Scope

The sectors focused at in this research are the chemical and construction industries, which make use of biobased products originating from the agro- or silvicultural sector. The scope of the selection procedure for new categories will therefore be limited to those that are related with agricultural and silvicultural product systems. The industrial processes that precede or follow these two production systems, like artificial fertilizer production, are outside the scope of this document because these are already included in the current environmental impact assessment methodologies.

Although the context of the development of these new impact categories is European research, the geographical scope for the development of impact categories is global, in the first place because this is the implicit approach in LCA and secondly because the raw materials originate from in- and outside Europe. The temporal scope of the impacts is from now until 100 years from now, following ISO standard 15804.

2 Methodology

The general approach of this study is to identify current issues addressed by several policy sources and try to find solutions for these issues, which fit in the state of the art methodologies and are scientifically sound. With this approach, only those scientific solutions are identified as high-potential if they are addressing policy-relevant issues; the large amount of scientific ideas that does not directly fit into policy needs is thereby excluded as being insufficiently relevant. In short: policy defines the problems, solutions are searched in science.

The additional impact categories are developed in three steps. First, the state of the art and gaps in environmental impact assessment of biobased materials are identified. The second step is to analyse the options to solve these gaps. The last step is to draw conclusions and do recommendations.

2.1 Step 1: State of the Art and Gaps

The state of the art of and gaps in the environmental impact assessment for biobased materials is inventoried. Both stand-alone topics (as addressed by policy documents) are given as well as the system overview (how are environmental assessments performed, what is the place of biobased issues and what gaps are present).
Summarizing, two subtopics are investigated: first, policy interest and subsequently, impact assessment methodology. This is discussed in the next chapter (3).

2.2 Step 2: Evaluation of potential additional impact categories

The selected areas are now further investigated one by one and potential additional impact categories are evaluated. First, a short definition and description of the area is given, including its position and relations in the overview picture. Next, for each potential additional impact category, its applicability is analysed by means of a matrix containing applicability criteria. This evaluation is done in chapter 4.

2.3 Step 3: Conclusions and recommendations

This final chapter concludes, based on the previous chapters, which categories should be implemented. Additionally it is discussed how these connect to current methodology and a plan for further actions (implementation) is proposed.

3 State of the Art and Gaps

As explained in the introduction, this chapter discusses first the relevant policy issues (paragraph 3.1), then describes the state of the art in environmental impact assessment methodologies (3.2) and ends with concluding (3.3) which impact areas should be further investigated in this study, which will be continued in the subsequent chapter (4).

3.1 Policy: issues in the environmental assessment of biobased products

This paragraph discusses the main international policy issues with respect to the environmental impact of biobased products. As the focus of this study is on Europe, first the policy perspective of the European Union is described (paragraph 3.1.1).

Moreover, there are many intergovernmental institutions, non-governmental organisations and research institutes which have a vision on biobased resource use and which have an influence on European policies. In order to connect also to these global policy issues and methodologies, in the subsequent paragraph (3.1.2) global policy issues are addressed, origination from the FAO, UNEP, OECD and GBEP.

Some environmental issues such as climate change and acidification are well covered in current environmental impact assessment methodologies. These issues are therefore not addressed in this report.

3.1.1 European policy

Hereafter, environmental issues are discussed that are addressed in current policies or European roadmaps. For now it is important to remark that, apart from these policies, the agriculture and forestry sector has the Common Agricultural Policy and the EU Forestry Strategy, which are both currently under negotiation/revision to, amongst other things, better face the environmental challenges in the sector. While the sector has many of the same environmental issues (biodiversity, soil and water quality) on their priority list, it is not clear at the moment how the revised documents are going to relate to the various issue-oriented policies.

Resource efficiency

The European Union is well aware that current production and consumption has negative environmental impacts. These impacts can be lowered by decreasing our resource use. Hence policy is developed on resource efficiency. As part of the flagship initiative 'A resource efficient Europe' the European Commission has proposed a 'Roadmap to a Resource Efficient Europe' ((European Commission, 2011a) and (European Commission, 2011b). This Roadmap is a framework that indicates needed future actions on resource efficiency. It focuses on structural and technological changes that have to come about by 2050 and it indicates milestones that have to be reached by 2020. This Roadmap focuses on the following resources:

- Ecosystem Services
- Biodiversity
- Minerals and Metals
- Water
- Air
- Land and Soils
- Marine Resources

Because several of the above resources can be impacted by agro- and silvicultural systems, it is worth exploring this piece of policy in more detail. Resource efficiency means using the Earth's limited resources in a sustainable manner. We depend on resources like metals, minerals, fuels, water, timber, fertile soil and clean air for our survival, and they all constitute vital inputs that keep our economy functioning. From the above list, the following natural resources are especially relevant for biobased materials: water, land & soils, ecosystem services and biodiversity. This is explained below.

There is a need for policy-relevant indicators of sustainability/vulnerability of *Water* resources (Council of the European Union, 2011). Considering the natural resource Water the diminishing quality and availability of fresh water is highly important. Additional impact categories should address these two impacts (European Commission, 2011d). The EC states that it will propose in the future certification schemes that measure life-cycle impact and virtual water content of products (ibid.).

'Land take', the use of land for e.g. housing and infrastructure and the sealing of the soil surface are the relevant impacts for the natural resource *Land*. Soil sealing means covering of the soil by an impermeable artificial material such as asphalt or concrete. This causes an irreversible loss of soil and its biological functions and loss of biodiversity: directly, and indirectly by fragmentation of the landscape. Soil sealing is therefore also relevant for the natural resource *Soil*. Other main issues are soil loss by erosion and the loss of soil organic matter content (European Commission, 2011a).

Ecosystem Services provide us with services from fertile soil to productive land and seas, from fresh water and clean air to pollination, flood control and climate regulation. Many of these ecosystem services are used almost as if their supply is unlimited. Ensuring a long-term supply of essential ecosystem goods and services implies we must properly value our natural capital (European Commission, 2011a).

In the view of the European Commission, the resource *Biodiversity* is seen separately from Ecosystem Services. For the resource Biodiversity the main challenge is to avoid damage to ecosystems by e.g. excessive use of pesticides and fertilizers in agriculture. Conservation of biodiversity within agricultural areas is also an issue (European Commission, 2011c). In 2005 the Streamlining European Biodiversity Indicators initiative (SEBI, see (European Environment Agency, 2009)) was started, aimed at the development of a set of indicators to measure and help achieve progress towards the European target to halt biodiversity loss by 2010. In total 26 indicators were selected, divided over seven focal areas. In 2010 the process was evaluated and improved in the subsequent SEBI cycle, 2010-2020 (EAA, 2012). The Strategic Plan for Biodiversity 2011-2020 and the EU 2020 Biodiversity Strategy were formulated, and were aligned with SEBI.
According to the European Commission indicators are important tools to measure and foster progress towards the vision and objectives of the resource efficiency flagship initiative. In Appendix 1 the most recent (2010) indicators are listed, organised in a 3 layer approach: a headline indicator (resource productivity), a dashboard of complementary macro indicators (materials, water, land and carbon), and a set of theme specific indicators (which measure the performance on the proposed actions and milestones in the Roadmap). The second layer of indicators is useful for our study.

3.1.2 Global policy

FAO

At a global level, the issue of sustainable resource management is discussed at the FAO, the Food and Agriculture Organisation of the United Nations. Biodiversity, ecosystem services and water scarcity are topics of great importance to the FAO. The FAO says: "The conservation and sustainable use of biodiversity for food and agriculture play a critical role in the fight against hunger, by ensuring environmental sustainability while increasing food and agriculture production. It is imperative to do so in a sustainable way: harvesting resources without compromising the natural capital, including biodiversity and ecosystem services, and capitalizing on biological processes" (FAO). The main focus of the FAO is however to achieve food security for all, and it focuses less on agricultural production for biobased products.

UNEP

Another United Nations (UN) organization that focuses on resource use and ecosystem services is the United Nations Environment Programme (UNEP). Worth mentioning is their International Resource Panel. This Panel was formed in 2007 to develop holistic approaches to global resource management. Recently the Panel has released a report on Measuring Water Use in a Green Economy (McGlade, et al., 2012). In this report, analytical methods and necessary policy frameworks are described to ensure that water use can be properly quantified over the life cycle and integrated into decoupling[2] measures within the green economy. It contains an overview of existing water stress indicators.. Another report of the Panel focuses on the sustainable production and use of resources in relation to biofuels (Bringezu, Schütz, O´Brien, Kauppi, Howarth, & McNeely, 2012). This report acknowledges that important environmental impacts – on GHG emissions, water and biodiversity for example – of biofuel production are not covered by LCA methods. The Panel hence concludes that further development of LCA methods is needed. This endorses the objective of the present study.

OECD

The Organisation for Economic Co-operation and Development (OECD) is another intergovernmental organization that focuses on sustainable agriculture. It performed a study on environmental indicators for agriculture, the results of which are relevant for this study. The Organisation for Economic Co-operation and Development (OECD) promotes policies designed to achieve the highest *sustainable* economic growth and employment and a rising standard of living in Member countries. The impacts of agriculture on the environment and the achievement of sustainable agriculture are of major public concern in the context of agricultural policy reform, trade liberalisation, and multilateral environmental agreements (OECD, 2001).
As part of the OECD's tasks a project on *Environmental Indicators for Agriculture* was executed. A methodological publication (OECD, 2001) was made which aims to review and take stock of progress in developing agro-environmental indicators in OECD countries. This document is seen as highly relevant for the agricultural and silvicultural production systems.

GBEP

The Global Bioenergy Partnership (GBEP) has performed a study on sustainability indicators for bioenergy (Global Energy Partnership, 2011). These indicators are partially relevant for BioBuild; an overview is given in *Table 3.1*. All of these indicators are considered highly or moderately relevant for the environmental impact assessment in BioBuild. The competition of bio(energy) crops with food production is lacking in this table, because GBEP includes it in the set of social indicators. This is a relevant issue for BioBuild as well.
Some of the indicators in *Table 3.1* are already included in current environmental impact categories and do not need further attention, but some really do. This will be further discussed in the next paragraphs.

[2] Decoupling of environmental impact from economic growth

Table 3.1 Relevance for Biobuild of the environmental indicators identified by the Global BioEnergy Partnership (Global Energy Partnership, 2011).

No.	Indicator	Relevance for BioBuild
1	Life-cycle GHG emissions	Moderate
2	Soil quality	High
3	Harvest levels of wood resources	High
4	Emissions of non-GHG air pollutants, including air toxics	Moderate
5	Water use and efficiency	High
6	Water quality	High
7	Biological diversity in the landscape	High
8	Land use and land use change related to bioenergy feedstock production	High

3.2 Impact assessment methodology

3.2.1 Existing methodology

Many methodologies exist to measure environmental impacts. Most of these methodologies have been developed within the field of environmental Life-Cycle Assessment (LCA). For good understanding, the characteristics of impact assessment in an LCA context are explained briefly hereafter.
The explanation is illustrated by Figure 3.1. In life-cycle assessment the environmental impact of a product (or process) is calculated over the entire life cycle of that product, for a certain reference flow. The reference flow is a unit product, e.g. 1 m^2 of biocomposite panel that can fulfil a function defined in the LCA study: the functional unit. In our example this may be ‘protecting 1 m^2 façade from rain’.

Impact assessment consists of three steps. In the first step, an ‘inventory table’ is made, consisting of all interventions in the environment resulting from the life cycle of one functional unit. Interventions are extraction of raw materials and emissions to air, water and soil. Hence, the inventory table contains the amount of each raw material and each emission for one functional unit, for the entire life cycle.

Each use of scarce materials (or other goods such as land surface) contributes to depletion, and each emission can contribute to a number of environmental problems, dependent on the substance and its species. In the second step, the contribution of our product system (for one functional unit) to a number of environmental problems is calculated. This is done by multiplying each of the entries in the inventory table relevant to this environmental problem with an effect factor. The effect factor, also called characterization factor, is a measure for the potential effect of one unit (usually kilogram) of extraction or emission, compared to a reference substance. Methane is known to have a potential effect on global warming 23 times as high, per kilogram emission, as carbon dioxide. The effect factor is 23 kg CO2-equivalents per kilogram.

The indicators describing *potential* environmental impact are denoted ‘midpoints’ in the figure. Midpoint indicators are positioned somewhere in the middle of the cause-effect chain of environmental effects from emission to damage, hence the name. Further down the cause-effect chain, the actual damage takes place. To quantify the damage to ecosystems, human health and natural resources (see graph), information is needed about exposure pathways as well as about the local environment. Some impact assessment methods enable calculation of this third step: from potential effect to damage.

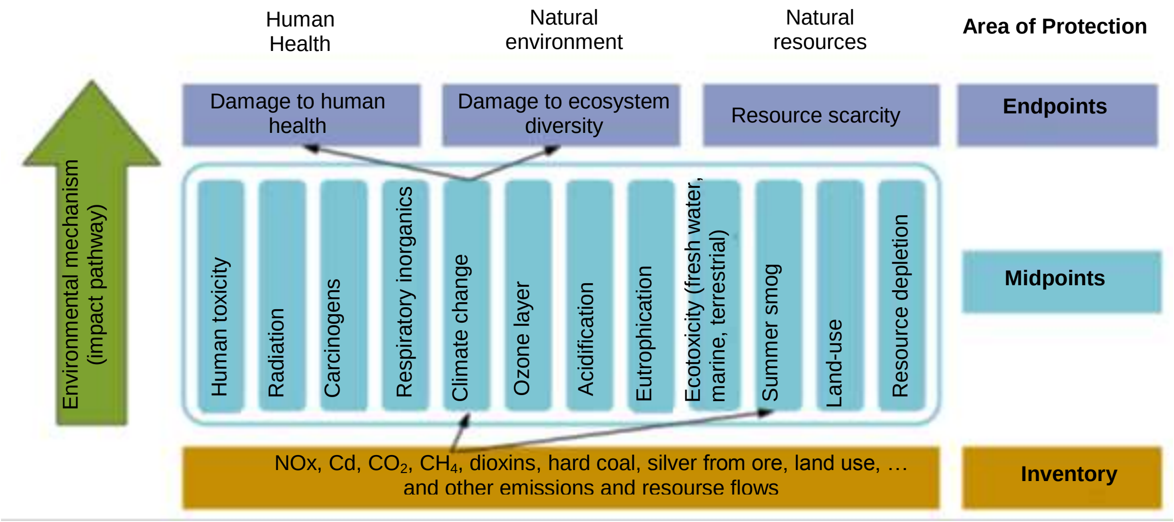

Figure 3.1 Overview of life cycle impact assessment. Source: (European Commission - Joint Research Centre - Institute for Environment and Sustainability, 2010).

In science, many different impact categories are distinguished, usually presented as a set of impact categories which form a consistent methodology. Three state of the art examples of such impact assessment methodologies are CML 2001, Ecoindicator '99 and ReCiPe (European Commission - Joint Research Centre - Institute for Environment and Sustainability, 2010). Table 3.2 gives the overview of which (midpoint) impact categories and appurtenant indicators are included in these three methods.

Table 3.2 Overview of three state of the art impact methodologies: list of used categories and appurtenant indicators. Sources: (Guinée, et al., 2001), (Goedkoop & Spriensma, 2001) and (Goedkoop, Heijungs, Huijbregts, Schryver, Struijs, & Zelm, 2009).

CML		**Ecoindicator '99**		**ReCiPe Midpoint**	
Category	*Indicator*	*Category*	*Indicator*	*Category*	*Indicator*
Abiotic depletion	kg Sb eq	Minerals	MJ surplus	Mineral resource depletion	kg Fe
		Fossil fuels	MJ surplus	Fossil resource depletion	kg oil
				Water depletion	m^3 water
Acidification	kg SO_2 eq	Acidification/ Eutrophication	PDF*m^2yr	Terrestrial acidification	kg SO_2 eq
Eutrophication	kg PO_4^{3-} eq			Freshwater eutrophication	kg P
				Marine eutrophication	kg N
Human toxicity	kg 1,4-DCB eq	Carcinogens	DALY	Human toxicity	kg 1,4-DCB eq
Freshwater ecotoxicity	kg 1,4-DCB eq	Ecotoxicity	PAF*m^2yr	Freshwater ecotoxicity	kg 1,4-DCB eq
Marine ecotoxicity	kg 1,4-DCB eq			Marine ecotoxicity	kg 1,4-DCB eq
Terrestrial ecotoxicity	kg 1,4-DCB eq			Terrestrial ecotoxicity	kg 1,4-DCB eq
Land competition	m^2year	Land use	PDF*m^2yr	Agricultural land occupation	m^2year
				Urban land occupation	m^2year
				Natural land transformation	m^2
Climate change	kg CO_2 eq	Climate change	DALY	Climate change	kg CO_2 eq
Photo-oxidant formation	kg C_2H_4 eq	Resp. organics	DALY	Photochemical oxidant formation	kg NMVOC
Stratospheric ozone depletion	kg CFC-11 eq	Ozone layer	DALY	Ozone depletion	kg CFC-11
		Resp. inorganics	DALY	Particulate matter formation	kg PM_{10}
		Radiation	DALY	Ionising radiation	kg U^{235}

Most of the environmental impact assessment methodologies were developed at the end of the 20th century and are designed from the perspective of a fossil based society. In the assessment of biobased products however, other environmental problems occur which current methodologies are unable to address. Most of the current methodologies lack for example a characterisation of the impacts of degradation of biological resources and depletion of water reserves. And in case bio-related impacts are addressed, the methods introduce uncertainties that do threat the acceptance by decision makers.

Several scientific groups have suggested additional impact categories, meant to improve the currently operational environmental impact assessment methodologies for calculations with respect to biobased products. These suggestions are described per study and per impact category in the appendixes. In the next paragraphs the main findings and conclusions are summarised for landscape, Hydrology and water consumption, and land-use in relation to soil quality, biodiversity and food competition.

3.2.2 Landscape

Landscape is mentioned by several authors as having a certain influence on midpoint and endpoint indicators like biodiversity, hydrology, soil fertility and erosion ((Tscharntke, Klein, Kruess, Steffan-Dewenter, & Thies, 2005), (Oost, Govers, & Desmet, 2000), (Vink, 1980)) and microclimate. Landscape can be seen as an ecological unit with a specific distribution pattern (e.g. landscape mosaic and other patterns), of which the changes have effects on all these mid- and endpoint indicators. Within the landscape itself, complex causal relationships between the life communities and their environment exist (for example, see (Wu, 2006) or (Noss, 1983)).

The place of landscape within the overview picture of environmental cause-effect chains is therefore not unambiguous, nor is the implementation of such a specific element as landscape in the general (and global scaled) methodology of environmental impact assessment. This will be further discussed in the conclusions.

3.2.3 Hydrology / water consumption

Hydrology is defined as the scientific study of properties, distribution and effects of water on the earth's surface, in the soil and underlying rocks, and in the atmosphere. Water-related impacts can generally be divided in two subcategories:

- *Water quality* is related to other substances present in the water state; nutrients; salinity and acidity. Impacts in this subcategory are already addressed by other impact categories, such as eutrophication and acidification.
- *Water use or water consumption* is related to amounts of (fresh) water available. This subcategory is not included in current methodology yet. Water use has a strong relation with policy issues, because it has an impact on all three Areas of Protection.

In general, there are two types of impact assessment methodologies for water use, which are the *water footprint*, and methodologies that aim at measuring *water stress*. There is a potential synergy in using the methodologies together, whereas they serve a different purpose. Therefor both methods may be further investigated.

3.2.4 Land use in relation to soil quality

Soil quality refers to the inherent ability of the soil to provide a growth medium for plants. Soil quality relates to possible functions and uses of soil, but also to location and scale of study. There are already some methodologies to assess soil quality impacts. Two kinds of impact assessment methodologies are available which could be applicable in LCA: methodologies that focus on soil loss or erosion, and methodologies that use the amount of soil organic matter (SOM) as a starting point. Further investigation can point out if these methods are applicable in BioBuild, given the data availability and the need to inform policy makers.

3.2.5 Land use in relation to Biodiversity

Depending on which methodology is used, land use impacts on biodiversity can only be assessed by (very) crude estimates. The more refined the assessment method is, the less data is available. The ILCD Handbook (2002) recommends using the Potentially Disappeared Fraction of species (PDF) concept as an indicator for the natural environment due to its ecological relevance. The additional use of the quality indicator PDF enhances this differentiation between crops but at the costs of transparency and robustness of data. The assessment method that uses PDF.m2.yr as impact indicator tries to address important issues such as local biodiversity and the damage done by specific crops or other interventions. It is in that sense more ecological relevant than an assessment method based upon “just” m2 land use. However, the PDF.m2.yr is very sensitive to local circumstances, especially in combination with valuation. In the currently available methods to quantify the PDF.m2.yr are based on one type of ecosystem only, being Swiss lowlands, and lack representativeness for other ecosystems present over Europe. This lack of representativeness does introduce a large uncertainty that threats wide-spread acceptance of this method.

Instead of the quantification of the biodiversity the land-occupation related value of ecosystem services can be calculated bases on cost factors developed by Costanza. This global estimation of the value of eco-system services can be used to calculated a shadow costs, and thus allows for the comparison of land use with other environmental impact categories. The method by Costanza is internationally accepted.

3.2.6 Land use in relation to Food competition

From a societal perspective, land use competition is subject of heavy debate. Biofuels for example have been proposed as a solution to several pressing global concerns: energy security, climate change and rural development. Several years later, there is growing concern about the role of biofuels in rising food prices, accelerating deforestation and doubts about the climate benefits. This has led to serious questions about their sustainability and extensive campaigns against higher targets. Concern was further raised among policy makers when the paper by Searchinger et al. (2008) asserted that US biofuels production on agricultural land displaced existing agricultural production, causing land-use change leading to increased net greenhouse gas (GHG) emissions (Gallagher et al., 2008).

The food v. fuel discussion however is not reflected in the LCA valuation, since eventually food security will be valued higher than would be expected based on the shadow price of its land use. Furthermore, the intrinsic value of the (natural) environment also is not included as this is difficult to assess with some objectivity.

3.3 Conclusion: identification of gaps

3.3.1 Request by policies

Summarizing, the policy issues related to the environmental impact assessment of biobased products can be analysed either on a European or on a global level.
European policies cover already many environmental issues that are direct effects of agro- and silvicultural systems. On the other hand, there are several policy programmes on specific topics which cover high scale issues like the EU Forestry Strategy and the Roadmap to a Resource Efficient Europe. The former is currently being reformed due to challenges related to greenhouse gas emissions, soil depletion, water & air quality and habitats & biodiversity. The latter explicitly states to the need for better indicators for Water, Land & Soil, Ecosystem Services and Biodiversity.
On a global level, all international organisations like the FAO, UNEP, OECD and GBEP emphasize the need for quantification of the impacts on biodiversity, food security ecosystem services and water scarcity. Some of them also address the need for soil quality, biotic resources and landscape. Summarizing, from the policy there is an actual call for better indicators for Water, Land(scape) and/or Soil and Ecosystem Services & Biodiversity related impacts and Food security.

3.3.2 Midpoint versus Endpoint

Endpoint impact indicators, resulting from a damage-oriented approach, translate environmental impacts into issues of concern such as human health, natural environment, and natural resources. Midpoint impact indicators, resulting from a problem-oriented approach, translate impacts into environmental themes. Endpoint indicators, such as biodiversity, reflect the policy goals in a more direct way compared to mid-point indicators, the. The disadvantage however of end-point indicators is the increased uncertainty compared to midpoint indicators. Midpoint indicators are typically chosen on the level where there is still enough natural science bases knowledge to calculate the indicator score. To calculate endpoint indicators, more assumptions need to be made and value based weighting is introduced.

3.3.3 Gaps in methodologies

The existing environmental impact methodologies are mainly developed from the perspective of a fossil production chain. Guinée *et al*. (the SOWAP project), OECD, Werf & Petit and Garrigues *et al.* do suggestions for extending the existing impact assessment categories. They identified biodiversity, hydrology (both water stress and use) and soil quality (including both fertility and erosion) as problematic topics in impact assessment. Several other sources mention the lack of landscape in environmental impact assessments.

3.3.4 Combining request and gaps

The identified requests by policy and gaps in science are for a very large part overlapping. With these obtained insights, the schematic picture of Guinée *et al.* (Figure 7.3) can be updated (Figure 3.2). Figure 3.2 shows the cause effect chains in the manufacturing of biobased products, including some elements that are underrepresented in existing impact methodology.

The main difference with the original figure is that Soil and Water State are identified as important suggestions for new research. The potential for implementation of these two subjects as new impact categories will be further discussed in the next chapter. The pressure of landscape change or micro-climatic impacts is also taken into account in Figure 3.2. This is however a very difficult impact to include in generic environmental impact assessments, because it is dependent on local factors and cannot be generalized, which is necessary in life cycle assessments. Therefor it is conclude that Landscape is not a useful indicator in life cycle assessments. Landscape effects are more likely to be integrated in (local) environmental risk assessments (which is not the scope of study in BioBuild).

For BioBuild, the choice is made to prefer midpoint indicators instead of endpoint indicators for a number of reasons. The calculation of biobased effects is still a new science field, and is lacking wide spread acceptance. Introducing additional uncertainties by using end-point indicators would further risk the acceptance of the results. In addition, many of the biobased damages would require detailed information on local circumstances, and the data collection for is both not feasible and goes beyond the scope of the study (to executed a life cycle assessment that is representative for the average European situation). This reasoning applies on Biodiversity, which was mentioned both by policy as well as by scientists as a subject which deserves further attention. Another end-point indicator is Human Health, which can be related to the need of Food security and the competition on land use. Both end-point indicators will be discussed based on the outcomes of the midpoint indicator ‘land-use’. Chapter 6 will further conclude on the implications for the environmental assessment in the BioBuild project.

Furthermore, Figure 3.2 shows some midpoint indicators which are already a point of attention in other impact assessment methodologies than for biobased products only. Ecotoxicity, human toxicity and air quality are such points of attention. As they are already involved in other fields of research, these will not be further discussed here.

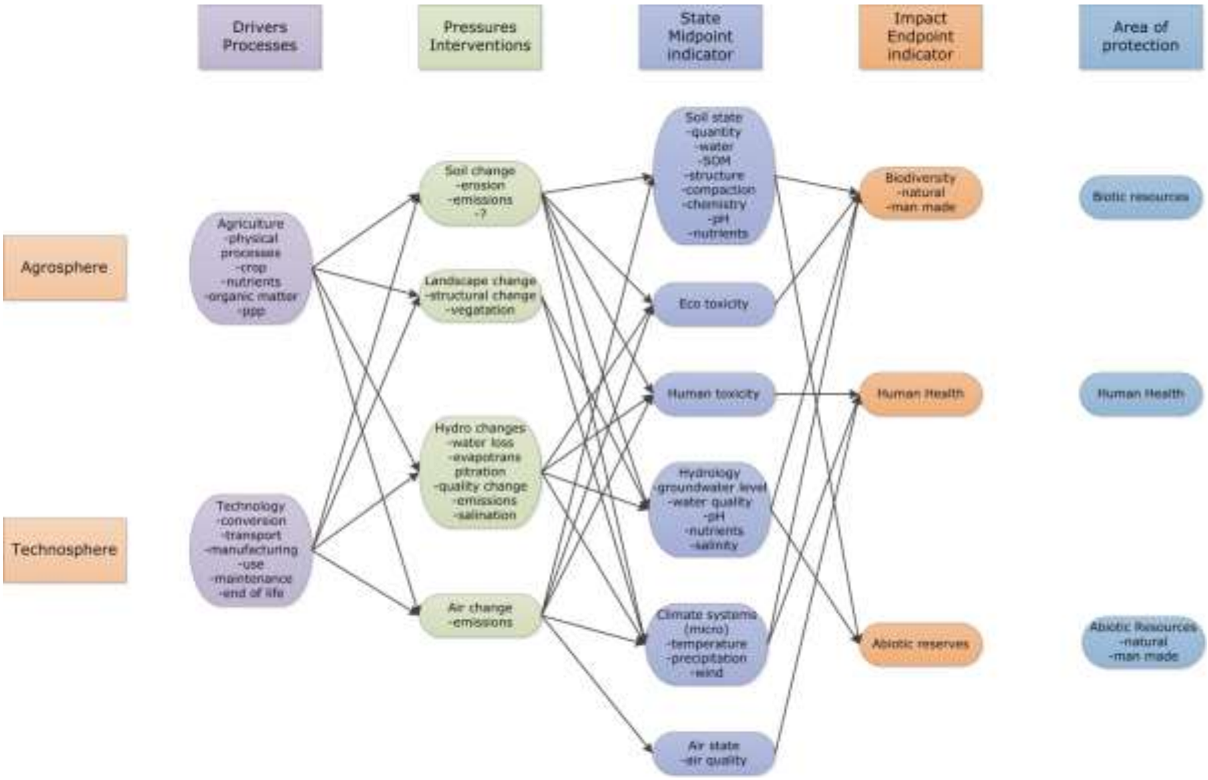

Figure 3.2 Schematic overview of the cause effect chains in biobased product manufacturing. The figure is based on the information of the previous paragraphs and an adaptation of the overview picture of Guinée et al. (see Figure 7.3).

4 Further elaboration on new bio-impact categories

4.1 <u>Hydrology & water consumption</u>

4.1.1 Problem definition & description

In the previous chapter it was concluded that hydrology is an underrepresented category in environmental impact assessments. Hydrology or water-related impacts can generally be divided in two subcategories related to amounts of (fresh) water available, being *Water quality* and *Water use* in terms of quantity. Water as a subcategory is not included in current impact assessment methodology yet. Water has a strong relation with policy issues, because it has an impact on all three Areas of Protection, which will be shortly discussed here.

4.1.2 Overview of potential impact assessment methodologies

Types of impact assessment methodologies for water

In general, there are two types of impact assessment methodologies for water use, which may benefit from each other. On one hand there are methodologies measuring the water footprint, such as the equally named *Water Footprint methodology* developed by Prof. A. Hoekstra (Hoekstra A. , 2011) which encompasses a detailed water accounting framework. Water footprinting is generally aimed at accounting water consumption by businesses. On the other hand methodologies have been described that aim at measuring *water stress*. These methodologies are mainly put forward by the LCA community, emphasizing allocation, definition of system boundaries and impact pathways (Bayart, et al., 2010) (Milà i Canals, Chenoweth, Chapagain, Orr, Antón, & Clift, 2009) (Pfister, Koehler, & Hellweg, 2009).

There is a potential synergy in using the methodologies together, whereas they serve a different purpose. For example the water footprinting methodology has an important use in presenting impacts on maps of water scarcity, the water stress focuses on providing an overview of water volumes in a supply chain, while attributing weights with characterization factors to indicate water scarcity (Milà i Canals, Chenoweth, Chapagain, Orr, Antón, & Clift, 2009). Therefore these will be discussed separately in the next paragraphs. Two methods for measuring the impact of water consumption will be discussed: Bayart's and Milà's method (4.1.3). For water stress, only the most prominent method will be discussed: Pfister's (4.1.4).

4.1.3 Description of water footprint in impact assessment methodology

Definition & general description

The water footprint is an indicator of freshwater use, in which is not only included the direct water of a producer or consumer, but also the indirect water use. The water footprint can be assessed for a product, process, company or even a country.
The water footprint of a product is the volume of freshwater used to produce the product, measured over the full supply chain. It is an indicator showing the volumes of water consumption by source and polluted volumes by type of pollution. The components of a total water footprint are specified geographically and temporally (Hoekstra A. , 2011).

There are three types of footprints you can assess, the blue, green and grey water footprint.

- The *blue water footprint* refers to consumption of blue water resources (surface and groundwater) along the supply chain of a product.
- The *green water footprint* refers to consumption of green water resources (rainwater insofar as it does not become run-off).
- The *grey water footprint* refers to pollution and is defined as the volume of freshwater that is required to assimilate the load of pollutants given natural background concentrations and existing ambient water quality standards.

Consumption refers to loss of water from the available ground-surface water body in a catchment area. Losses occur when water evaporates, returns to another catchment area or the sea or is incorporated into a product.

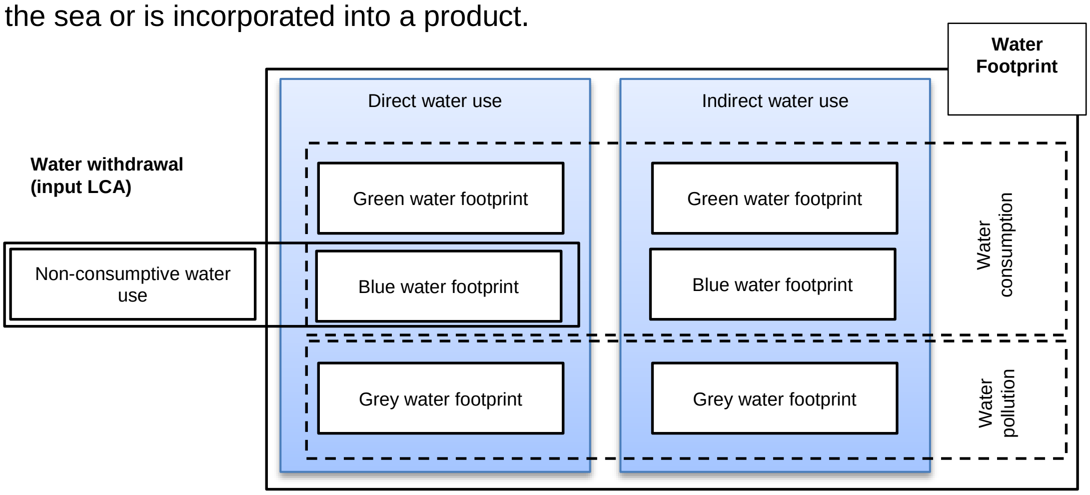

Figure 4.1 Schematic representation of the components of a water footprint. It shows that the non-consumptive part of water withdrawals (the return flow) is not part of the water footprint. It also shows that, contrary to the measure of 'water withdrawal', the 'water footprint' includes green and grey water and the indirect water use component, after (Hoekstra, Chapagain, Aldaya, & Mekonnen, 2011a).

As an indicator of water use, the water footprint differs from the classical measure of water withdrawal in three respects (see Figure 4.1):

- It does not include the blue water use, which is not consumed.
- It includes blue, green and grey water, and is not restricted to the first one.
- It includes both direct and indirect water use, and is not restricted to the first one.

Figure 4.2 shows the green and blue water footprint in relation to the water balance of catchment area. The green water footprint consists of the production-related evapotranspiration and the water contained in the products, with soil and vegetation as the source. The blue water footprint consists of the production-related evapotranspiration, the water contained in products and the water transferred to other catchment areas, with ground and surface water as the source.

Especially for assessing bio-based products the green water footprint is an relevant factor with respect to sustainability of the product. The green water footprint includes the evaporation of water from the soil and the transpiration of water from vegetation which is product related. These are combined into the expression evapotranspiration.

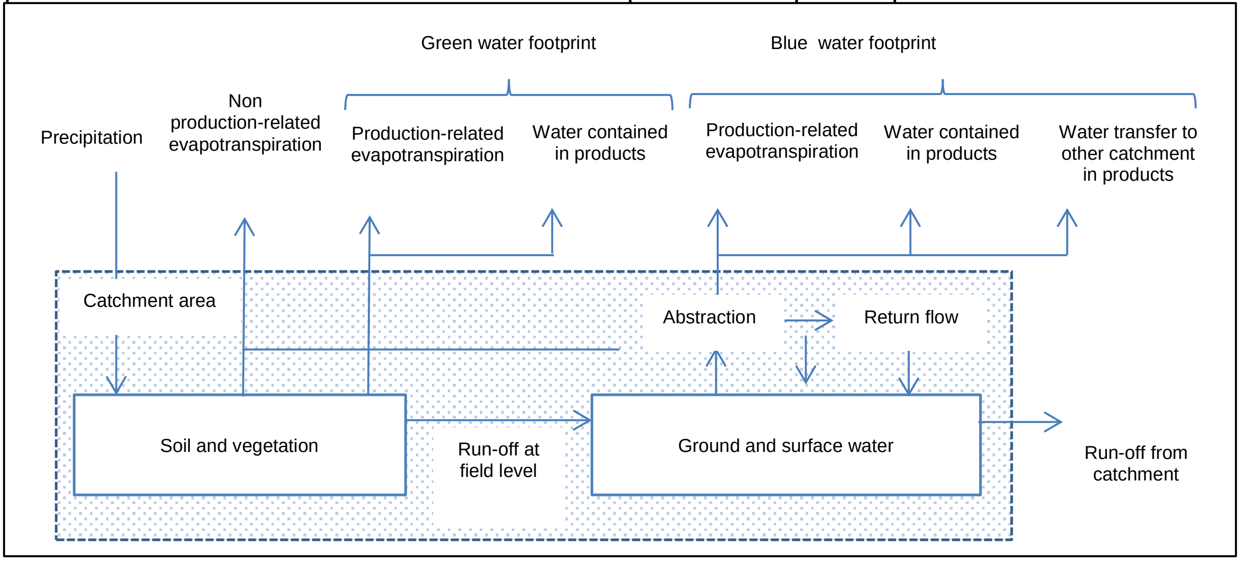

Figure 4.2 The green and blue water footprint in relation to the water balance of a catchment area.

The water footprint of a product is the sum of the water footprints of the process steps that were necessary to produce the product. The sustainability of the water footprint of the product therefore depends on the sustainability of the water footprints of the various process steps. Two methods for water footprinting will be discussed: Bayart (Bayart, et al., 2010) and Milà (Milà i Canals, Chenoweth, Chapagain, Orr, Antón, & Clift, 2009).

Description & indicators for Bayart's method for water footprinting

Bayart et al. (2010) describe cause - effect chains for freshwater use. They define three midpoint impact categories, see Figure 4.3.

- Freshwater deficits for human uses
- Freshwater deficits in ecosystems
- Freshwater depletion

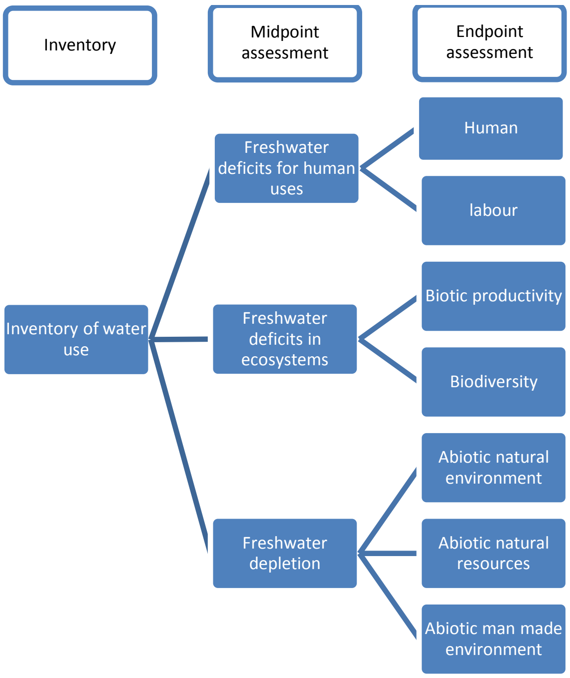

Figure 4.3 Simplified description of relevant cause-effect chains after (Bayart, et al., 2010)

Additional to the simplified cause - effect chains (Figure 4.3) the *freshwater functions* for human activities should be taken into account according to Bayart et al. (Bayart, et al., 2010). Within the technosphere these functions are:

- maintaining human and environmental health
- supporting biotic production and industrial activity
- carrying goods
- playing a psychological role given its aesthetic or cultural value.

Therefore a functional allocation should be applied. Not all types of water can be used for all purposes. When freshwater availability for a certain function is reduced, two scenarios are possible: (1) deficiency with direct effect to human health or (2) compensation, which needs a new LCA to assess impacts.

The impacts of these three types of cause-effect chains, with their functions, were described by Bayart et al. For the first kind of impact pathways (linked to freshwater resource insufficiency for contemporary human users), they described the midpoint indicator as expressed in cubic meters of freshwater equivalent unavailable for downstream users. Operational characterization of the factors is out of the scope, but proposal of principles and parameters to calculate midpoint characterization factors is given in the article. Examples of available methods are the Freshwater Scarcity, Swiss Ecological Scarcity Methodology and Water Stress Index. The impacts can be measured by distance-to-target or functionality, dilution volume or energy requirement for purification.
The second kind of impact pathway (linked to freshwater resource efficiency for existing ecosystems) can be measured by means of the midpoint indicator expressed in cubic meter of freshwater unavailable for ecosystems and the functions they provide. (Pfister, Koehler, & Hellweg, 2009) express this as loss of primary production. Another method is the Water stress indicator – environmental water requirements (Mila I Canals). A third method is to analyse the extraction from groundwater reservoirs cause decline of groundwater level – Potentially Not Occurring Fraction of plant (PNOF) species over a given time period per cubic meter of water use (PNOF*year/m^3).
The last kind of impacts (for future generations, linked to unsustainable use of freshwater) can be described as ‘water depletion’: the volume of water that disappears from a given

watershed for a period of time and refers to both flow and stock resources (cubic meter of freshwater equivalent depleted).

Description & indicators of Milà's method for water footprinting

Milà i Canals (2009) define the following freshwater use impacts:

1. Direct water use is leading to *changes in freshwater availability for humans*, which can lead to changes in human health. The cause – effect chain is not straightforward, human deaths are not directly related to water volumes, but to water quality/sanitation. It is proposed to exclude this impact from LCA
2. Direct water use is leading to *changes in freshwater availability for ecosystems*, which can lead to effects on ecosystem quality (Freshwater Ecosystem Impact, FEI). Only evaporative use should be included in the LCA.
3. Direct groundwater use can lead to *reduced long-term (fund and stock) freshwater availability* (freshwater depletion, FD). Only fund and stock water sources should be taken into account; not river water because this cannot be depleted, there is only competition over its use.
4. Land use changes are leading to *changes in the water cycle* (infiltration and run off), which can lead to changes in freshwater availability for ecosystems, leading to effects on ecosystem quality (FEI).

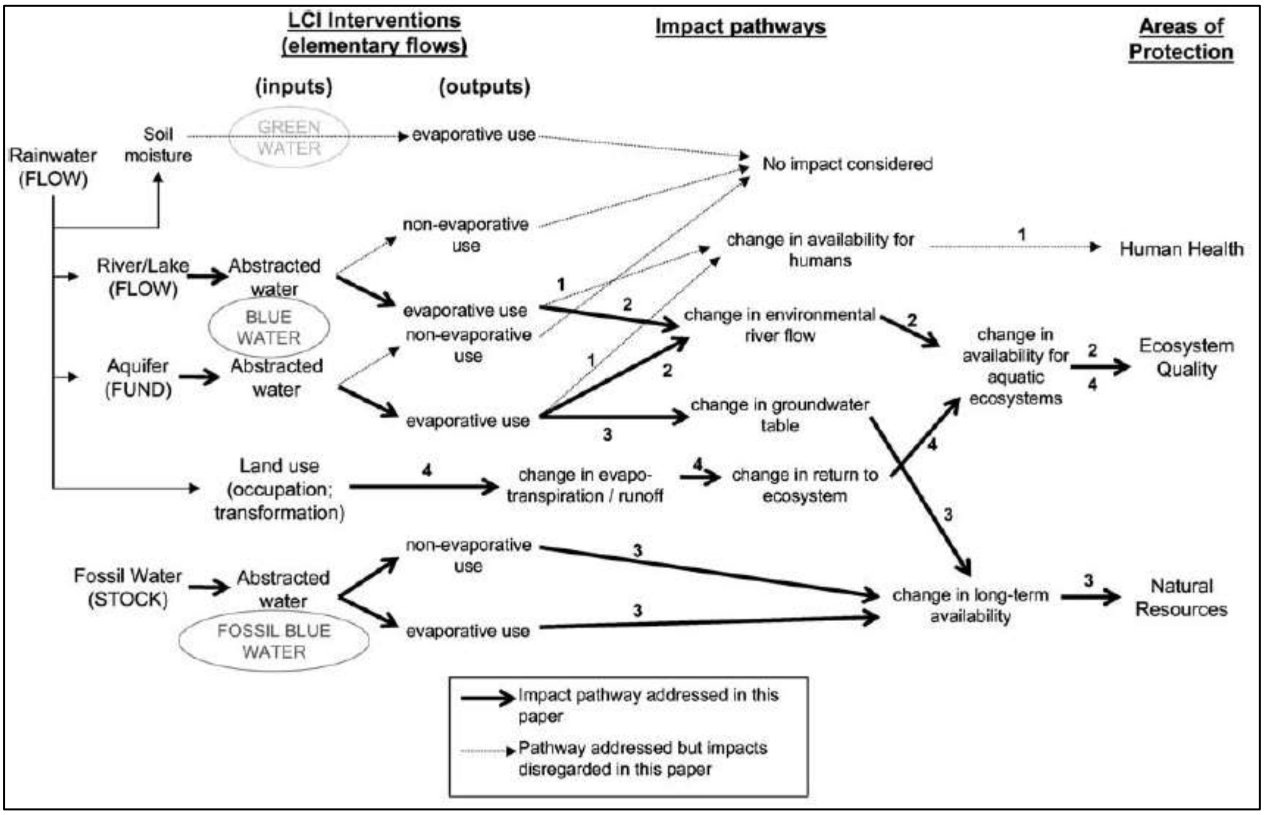

Figure 4.4 Main impact pathways related to freshwater use, only those depicted with solid arrows are considered for LCA by (Milà i Canals, Chenoweth, Chapagain, Orr, Antón, & Clift, 2009). Green water is not taken into account because its use does not lead to environmental impacts.

There are two perspectives from which the (fresh)water flows that should be quantified could be described: from an ecosystem impact point of view, or from a depletion point of view. From the former, the ecosystem impact point of view, the following flows could be quantified:

- Surface and groundwater evaporative uses: in-stream evaporation in reservoirs and power dams and off-stream evaporation of abstracted water through, e.g. irrigation; in cooling towers, etc. in virtual water (VW) terms: evaporative blue water;
- Any type of land use occupation and transformation.

There are two indicators for Depletion Potential (DP):

- Abiotic Depletion Potential (ADP, (Guinée, 2002))

$$ADP_i = \frac{ER_i - RR_i}{(R_i)^2} * \frac{(R_{Sb})^2}{DR_{Sb}}$$

ADP	*= Abiotic Depletion Potential of component i*
ER	*= Extraction Rate of component i*
RR	*= Regenration Rate of component i*
R_{Sb}	*= Ultimate reserve of reference reserve*
DR_{Sb}	*= Deaccumulation rate of reference reserve*

- Back up technology, also called surplus energy for desalinization (84 to 3.5 kWh/m^3, (Stewart and Weidema)).

To quantify freshwater depletion from the ecosystem point of view, only one flow should be quantified: water stocks (groundwater-fossil water) and over abstracted water funds (groundwater-aquifers): both evaporative and non-evaporative uses needs to be quantified.

There are three indicators for Freshwater Ecosystem Impacts (FEI):

- Water resources per capita: WRPC = WR/population (Falkenmark, 1986)
- Water use per resource: WUPR = WU/WR (Rashkin, 1997), water use over water resources
- Water stress index: WSI = WU/(WR-EWR) (Smathkin, 2004), water use over water resources minus environmental water requirements

The last indicator is preferred by Mila I Canals et al. (Milà i Canals, Chenoweth, Chapagain, Orr, Antón, & Clift, 2009).

The Freshwater Depletion (FD) impact is so localized that it will probably only affect known cases of aquifer over-abstraction in foreground system. Modelling is no problem for LCI databases.

Some issues are not yet properly assessed in LCIA by either of the two perspectives:

- Impacts on aquatic water systems, e.g. temperature (water cooling);
- Impacts on human health, microbiological pollution (less developed countries).

Data availability for water footprinting indicators

The water footprinting network publishes number databases on water footprints, water flows in main rivers/water basins, virtual water flows, water scarcity etc. There is a focus on agricultural products. Data is available on a regional (country/region) level. Examples of geographical presentations based on the databases are given in Figure 4.5 and Figure 4..

Figure 4.5 Total water footprint rapeseed (Mekonnen & Hoekstra, 2010)

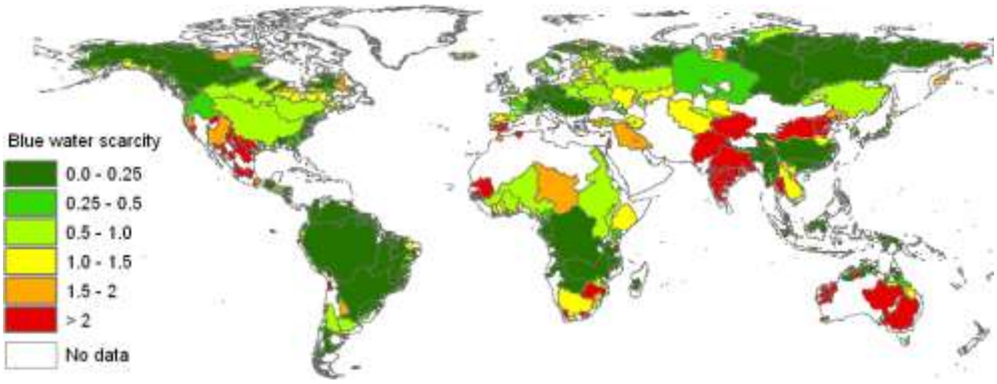

Figure 4.6 Annual average monthly blue water scarcities in the world's major river basins (1996-2005) (Hoekstra & Mekonnen, Global water scarcity: monthly blue water footprint compared to blue water availability for the world's major river basins, 2011)

Categories of data related to water from LCI databases such as Ecoinvent, Idemat 2001 and Industry data 2.0 have been collected and linked to the water footprint methodology. Not all categories can directly be linked, because the categories do not always distinguish between consumptive and non-consumptive use of water. From Figure 4.1 it is clear that the water foot printing methodology only incorporates the consumptive use.

Table 4.1 shows the main water related categories present in LCI databases. The LCI databases generally provide data related to the blue water footprint, i.e. the use of ground and surface water resources. Regarding the blue water footprint, the data concern water withdrawal, not the consumptive use of water. Some of the categories listed in Table 4.1 (indicated with an asterisk) are not directly related to water consumption and hence water scarcity and can possibly left out of the analysis. Water transformation and occupation are related to occupation of a water area and not to water scarcity. The water used in turbines is not consumed, but runs through. In this way it does not contribute to water scarcity theme. Next to the processes which could be ignored in the assessment of water scarcity, the salt water use from oceans can be left out of the analysis, because there is no salt water scarcity.

A number of categories concern the grey water footprint as data reflect contaminants. This impact is treated in other impact categories, e.g. on toxicity. Green water footprint data are usually lacking.

The resulting categories relevant for water scarcity reflect the consumption of high quality water stemming from surface and ground water resources (closely to the definition of the blue water footprint).

Table 4.1 Main 'water related' categories in Life Cycle Inventory databases (e.g. Ecoinvent, Idemat 2001 and Industry data 2.0) and their position in the water footprint. Categories marked with an asterisk are not relevant for water footprinting.

Main category	Definition	Water Footprint
Water, unspecified natural origin,	source and application are not specified	Blue (consumptive use)
Water consumption	water consumption, source is specified	Blue
Water, source	source specified (e.g. lake, river, salt ocean *, salt soil * etc.), application not specified	Blue (consumptive use)
Water, unspecified "water stress"	source and application are not specified, but water stress is related to 'water scarcity'	?
Water, process	process water, source can be specified	Blue (consumptive use)
water, cooling	cooling water, source can be specified	Blue (consumptive use)
Water, transformation	Related to occupation of water area *	not in water footprint
Water, occupation	Related to occupation of water area *	not in water footprint
Water, turbine	turbine use, source is not specified *	not in water footprint (assuming turbine water is not consumed)
Water, contaminants	contamination of water *	Grey
Water, airborne emissions	emission of water into the air	Green and/or blue
Water, waterborne emissions	emission of contaminants into water *	Grey

Missing categories in the LCI database are those water flows which can used as input for the green water footprint. As explained before they are important, especially for bio-based products. The water included in the soil and vegetation is included as an ecosystem service in the land use category. In this very abstract way, independent from crops or locations, and without any specific data on green water footprints, some green water footprint shadow price is attached to a m^2 land use. If one would like to improve on this method, it is proposed to develop a green water footprint method on the basis of land use and/or in relation to crops.

The grey water footprint is related to pollution. The environmental impacts of pollution are already taken into account in other impact categories as toxicity. With respect to water scarcity only the blue and green water footprint should be assessed.

4.1.4 Description of Water Stress in impact assessment methodology

Definition & description

Water stress is commonly defined as the ratio of fresh water use and the fresh water availability. We will discuss here the most prominent method to assess the impact of water stress, as described by Pfister, Koehler & Hellweg (Pfister, Koehler, & Hellweg, 2009).

Pfister et al. propose in their article to take into account:

- Consumptive water use, as it is crucial from a hydrological perspective
- Blue water, as it reflects the consumption of ground and surface water
- Regionalization, water use and availability depend on special factors
- Temporal effects and differentiation in non and strongly regulated flows.

Indicators & data availability

Water stress can be calculated by means of the indicator called "water stress index" (WSI). In the following text, the calculation method for water stress index will be explained step by step. First, the "withdrawal to water availability" needs to be defined:

$$WTA_i = \frac{\sum_j WU_{ij}}{WA_i}$$

WTA = Withdrawal to availability
WU = Water use
WA = Water availability
For watershed i, and user group j

The Withdrawal to Availability (WTA) can be modified to adept the influence of the regulation of the flow by storage structures (dams etc.). This regulation can partly compensate for periods with low precipitation, but can also lead to higher evaporation.

$$WTA^* = \begin{pmatrix} \sqrt{VF} * WTA & for\ SRF \\ VF * WTA & for\ non\ SRF \end{pmatrix}$$

WTA^* = Modified Withdrawal to Availability (WTA)
VF = Variation factor
SRF = Strongly regulated flow

$$VF = e^{\sqrt{\ln(s^*_{month})^2 + \ln(s^*_{year})^2}}$$

s = standard deviation

$$VF_{WS} = \frac{1}{\sum P_i} \sum_{i=1}^{n} VF_i * P_i$$

VF_{WS} = Variation factor water shed
P_i = annual precipitation for grid cell i.

Pfister et al. propose to use the Water Stress Index (WSI) as characterization factor for the midpoint category "water deprivation" in LCIA. The WSI can be calculated with the following formula.

$$WSI = \frac{1}{1 + e^{-6.4 * WTA^* \left(\frac{1}{0.01} - 1\right)}}$$

There are some limitations to this method:
- Current databases show only limited information about water
- Virtual water databases are available for agricultural product, but the data on industrial processes are limited and the supply chain is neglected
- Only taking blue water into account is a simplification; the impact of changes in green water should be addressed in future research

This general concept of water stress calculations can be applied to all three areas of protection (AoPs) and thus to the endpoints. Because this chapter aims at defining midpoint impact categories, this will not be further discussed here, but the information about the relation with the endpoints can be found in the appendix.

Data availability for water stress indicators

The previous paragraph shows that for calculation of Water Stress, information is needed about water use and water availability. The paragraph on water footprinting discussed already that there is general LCI data available about water use. The Water Footprinting

Network (WFN) reports databases and geographical plots of water availability and use of e.g. countries and major river basins in time. This results in location and time specific information, whereas the indicators in an LCA require a higher level of aggregation.

4.1.5 Evaluation of the impact assessment methodologies for water

Table 4.2 shows the overview of the applicability of the impact assessment methodologies for water, as a result of the discussion in the previous paragraphs.

Table 4.2 Overview of the applicability of the impact assessment methodologies for water. Legend: "+" means High/Good; "±" means Moderate; "-" means Low/Poor.

Criterion	Water Footprint		Water Stress	Remarks
	Bayart	*Milà*	*Pfister*	
Relevance for AoPs				
Human Health	-		-	
Biotic Resources	±		+	
Abiotic Resources	+		±	
Measurability	+		+	
Reliability	+		+?	
Sensitivity	+		+	Both geographic and technology sensitive
Transferability/transparency	+		+	
Implementation in LCA	+		±	WSI → local data
Overall applicability	+		+	

With the information from the previous paragraphs and Table 4.2, we conclude that it would be valuable to add "water" as a new environmental impact category, which is calculated by means of two indicators: the water stress index and water as a resource (the aquifer reservoirs, as calculated in the blue water footprint).

4.2 Soil quality

4.2.1 Problem definition & description

Soil quality refers to the inherent ability of the soil to provide a growth medium for plants. In practice, this is likely to be a function of a wide range of different soil properties which individually or in combination affect growth processes such as germination, root elongation and shoot development, tilling, flowering and fruiting. At a general level, however, many of these properties act through their influence on moisture and nutrient supply to the plant.

Soil quality can be described by:

a. Function
b. Use

Concerning a), the Soil Science Society of America officially defines soil quality as "the capacity of a specific kind of soil to function, within natural or managed ecosystem boundaries, to sustain plant and animal productivity, maintain or enhance water and air quality, and support human health and habitation".
Concerning b), "fitness for use".

Thus, soil quality relates to possible functions and uses of soil, but also to location and scale of study. Soil quality can be expressed by:

1. Physical, chemical and biological properties
2. Soil functions
3. Processes that could degrade the soil.

Soils are an essential resource in both managed and natural systems, and maintaining soil quality (change = 0) is critical to the development of sustainable agriculture. The inability to represent impacts on soil quality remains one of the unresolved problems in LCA because of soil's spatial and temporal variability and the complex interactions between soil properties. The ENVASSO program identified erosion (soil loss), decline in soil organic matter (SOM) and biodiversity, contamination, sealing, compaction and salinization as the main threats to soil. A huge variety of MDS have been proposed, in which SOM, texture and density are almost unanimously present.
There is however also some debate on whether soil quality should be (fully or partly) included as an impact category since it is closely related to yield (economic output) and whether loss of soil is to be considered an intervention at all. Alternative approaches include system expansion and change in economic output over longer period of time but this raises many practical problems.

In 1993, the FAO included soil quality in the five criteria on which sustainable land management is based:

1. Productivity
2. Security
3. Protection ("the quantity and quality of soil and water resources must be safeguarded in equity for future generations")
4. Viability (economical)
5. Acceptability (social)

In 1997, the European Commission recognized that the "maintenance of the soil's quality is a prerequisite in defining sustainable agricultural quality" (Garrigues, Corson, Angers, Werf, & Walter, 2012).

There can be trade-offs in policies to encourage environmentally sound management practices. For example, a policy objective to reduce soil loss by encouraging farmers to move from conventional tillage to reduced or no-tillage in crop production, can be achieved if weeds are controlled with herbicides. An environmental side-effect of these tillage practices is a likely change in water movement in the soil, with no-tillage increasing infiltration and percolation of nutrients such as nitrate leaching to the water table when compared with conventional tillage. In addition, the increase in herbicide use may cause pesticide leaching. Thus the objective of lowering soil loss through no-tillage may lead to some negative environmental effects.
At an international level there are no formal agreements or conventions that relate directly to the soil quality issue, although there are various international initiatives to co-ordinate current research in the area[3]. A more recent international development of relevance to soil quality indicators, is the on-going examination of the soil organic carbon issue within the context of the UN Framework Convention on Climate Change (effects to be included in the Global Warming Potential midpoint category) (OECD, 2001).

4.2.2 Overview of potential impact assessment methodologies

There are already some methodologies to assess soil quality impacts. Garrigues et al. (2012) note that the assessment of soil quality depends on criteria that differ according to the method used. The first criterion is the objective of the method. The second criterion is the spatial scale. The complexity of the description of soil functioning and the data measurability and type depend on the third criterion: the target group (researchers, farmers, etcetera). The

[3] Such as the International Soil Reference and Information Centre, and the World Bank Land Quality Indicator initiative.

content of the data set depends on these criteria. Several methods or tools to assess soil quality or impacts already exist, they put forward these examples:

- The IDEA method assigns scores to farmer management practices and behaviour (Vilain, 2008; Zahm et al., 2008).
- The USDA soil quality test kit supplies selected field procedures to indicate the level of one or more soil functions (Seybold et al., 2001; Evanylo and McGuinn, 2000).
- The French BDAT soil analysis database is a tool for a broad-scale study of pedological, agronomic and environmental questions concerning spatial and temporal variability of agricultural soils (Lemercier et al., 2008).
- The Swiss Agricultural Life Cycle Assessment (SALCA) method can be used to assess the environmental impacts of agricultural production. It is composed of different modules, one of which is SALCA-Soil Quality (SQ), focused on impacts of agricultural practices on soil quality (Oberholzer et al., 2006) in Switzerland.

Garrigues et al. (2012) conclude that the data requirements and level of sophistication of these methods differ according to their objectives, but they are not flexible and/or sensitive enough to be applied within the LCA framework.
Two kinds of impact assessment methodologies are available which could be applicable in LCA: methodologies that focus on soil loss or erosion, and methodologies that use the amount of soil organic matter (SOM) as a starting point. The next paragraphs will discuss these methods one by one.

4.2.3 Description of Soil Loss/Erosion in impact assessment methodology

Definition & Description

Soil erosion (soil loss) is the process of wearing of the land surface by physical forces or other natural or anthropogenic agents that abrade, detach and remove soil or geological material from one point on the earth's surface to be deposited elsewhere. Soil erosion is normally a natural process occurring over geological timescales; but where (and when) the natural rate has been significantly increased by anthropogenic activity accelerated soil erosion becomes a process of degradation and thus an identifiable threat to the functions provided by the soil. Types of erosion include water, wind and tillage erosion (European Soil Bureau Network, 2007).
Soil erodibility refers to the susceptibility of the soil to erosion. At a general level, this depends primarily on the structural stability of the soil (and hence its resistance to particle detachment by rain splash or runoff) and on its ability to absorb rainfall (i.e. its infiltration capacity, permeability and transmissivity). These properties, in turn, depend on a number of more basic attributes, including soil texture, organic matter content, carbonate content, salinity and pH (all factors determining soil quality). Other important factors are amongst others stoniness and soil depth. Deep soils typically have a higher water holding capacity, and thus are able to absorb larger rainfall amounts before overland flow is generated. Topography, vegetation cover (due to its management potential) and soil quality are the most important determinants in soil erosion (European Environmental Agency , 1990).

Indicators & data availability

Guinee et al. summarize several existing methods that deal with the issue of erosion in environmental impact assessment (Guinée, Oers, Koning, & Tamis, 2006). According to Cowell & Clift (2000) the loss of soil mass is an indicator for depletion of the soil resource. As a characterisation model, the soil static reserve life is proposed (SSRL = R/E). So the soil static reserve life is a function of global reserves of agricultural soil (R) and current annual global net loss of (top)soil mass by erosion (E). The necessary inventory data to calculate the impact score is the loss of soil mass, either measured or estimated (e.g. using erosion models like USLE, see below). Muys & Garcia Quijano (2002) describe the indicator soil erosion as sub-indicator in the sub-impact category soil. In this method it is proposed to transform the loss of soil mass into a loss of soil depth using the bulk density of the soil.

Finally, the loss of soil depth over a period of 100 years is compared to the total rootable soil depth up to 1m. A complete loss of the soil within a period of less than 100 years leads to the maximum impact score (Erosion risk factor = E * 100 yr/total rootable soil depth (1m)). The necessary inventory data to calculate the impact score is the loss of soil mass, either measured or estimated (e.g. using erosion models like USLE, see below). For both methods, no operational factors were available at the time. The information needed is information on the reserve of the topsoil and erosion data (which is now becoming more and more available).

Erosion can be calculated with the (revised) Universal Soil Loss Equation (rUSLE), and characterization factors can be derived from the world map of Global Assessment of Human-Induced Soil Degradation (Garrigues et al., 2012).

(r)USLE: *A = RKLSCP*
A = computed soil loss
R = rainfall/runoff erosivity factor (climate, seasonal influences!)
K = soil erodibility factor (soil type, also impacted by seasonal influences)
L = length of slope (topography, less sensitive)
S = steepness of slope (topography, more sensitive)
C = cover management factor (land use/crop-related, represents conditions that can be managed most easily to reduce erosion. Use SLRs if available or compute them – indirect relation with SOM present)
P = supporting practices (management-related)

Although the (r)USLE has several limitations, amongst others that it does not represent fundamental hydrologic and erosion processes explicitly, as an empirical equation derived from experimental data, the (r)USLE adequately represents the first-order effects of the factors that affect sheet and rill erosion (Renard, Foster, Weesies, & Porter, 1991).

Another, recently developed method which integrates soil erosion in LCA is described by Nuñez *et al.* (Nuñez, Antón, Muñoz, & Rieradevall, 2012). The authors developed a globally applicable, spatially differentiated LCIA endpoint method to account for land occupation impacts in LCA, focusing on the aspect of soil erosion based on the concept of emergy: loss of energy through loss of soil. The LCI data required (topsoil, calculated by rUSLE and SOC losses) and data sources that can be used to obtain the inventory flows were also identified. Spatially explicit damage factors on a grid cell-level resolution (10*10 km^2) for the entire world were provided for soil resource depletion and ecosystem quality endpoints. The model was successfully applied to agricultural plots in Spain to compare soil erosion-related environmental impacts that may result from substituting traditional food for energy crop rotations.
Maps for (vulnerability to) soil loss due to water erosion can be found online, see for example Figure 4.7, derived from the website of the U.S. National Resources Conservation Service (U.S. National Resources Conservation Service). More detail is found in regional maps (for example the PESERA project, including estimated annual losses of soil by water erosion in Europe in t/ha/yr, averaged over a series of years under current land use and climate, and represented at a resolution of 1 km. The map is intended to provide an objective assessment of current losses of material from hillsides, though sediment delivery through the river system is explicitly not taken into account, and most of the eroded material generally remains close to its source, with significant off-site effects generally confined to a local area. (Kirkby, Irvine, Jones, & Govers, 2008)), see Figure 4.8.

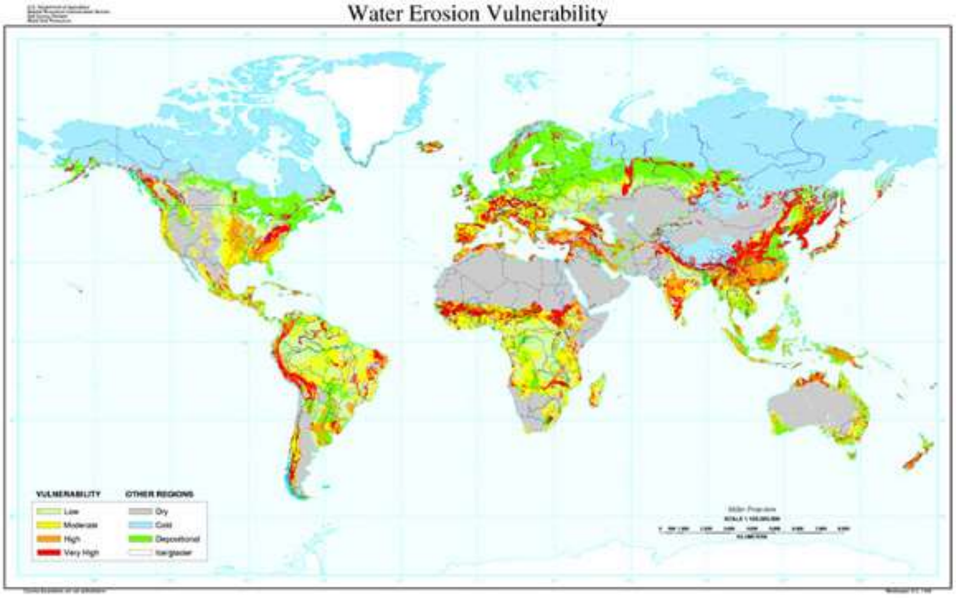

Figure 4.7 Global water erosion vulnerability map, based on soil climate and soil classification (U.S. National Resources Conservation Service).

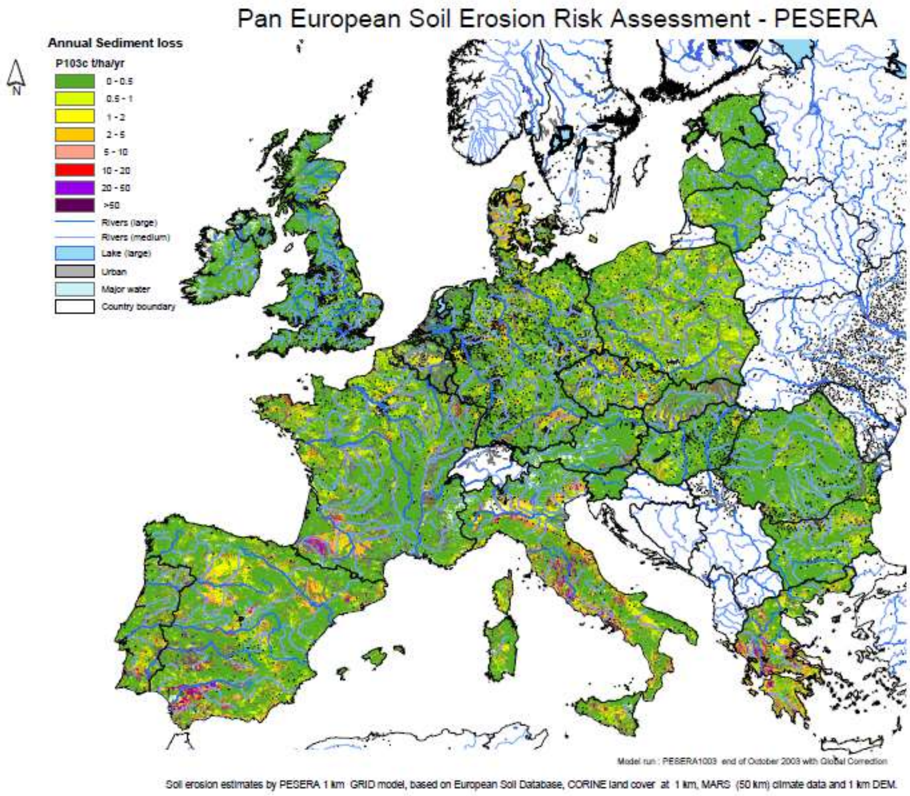

Figure 4.8 Pan European Soil Erosion Risk Assessment map of estimated losses of soil by water erosion in Europe in t/ha/yr[4].

[4] http://eusoils.jrc.ec.europa.eu/ESDB_Archive/pesera/pesera_cd/sect_4_2_1.htm (viewed 18/1/2013)

4.2.4 Description of Soil Organic Matter (SOM) in impact assessment methodology

Definition & Description

The term "Soil organic matter" (SOM) has been used in different ways to describe the organic constituents of soil. SOM was defined by Baldock and Skjemstad (1999) as "all organic materials found in soils irrespective of origin or state of decomposition". It can be divided into three general pools: living biomass of microorganisms, fresh and partially decomposed residues, and humus: the well-decomposed organic matter and highly stable organic material.

Soil organic matter influences (Krull, Skjemstad, & Baldock, 2004):

- Structural stability
- Water holding capacity
- Colour
- Cation exchange capacity[5]
- Buffer capacity and pH
- Adsorption and complexation
- Energy
- Nutrients
- Resilience

Since SOM consists of C, H, O, N, P and S, it is difficult to actually measure the SOM content and most analytical methods determine the soil organic carbon (SOC) content and estimate SOM through a conversion factor[6]. The amount of SOC that exists in any given soil is determined by the balance between the rates of organic carbon input (vegetation, roots) and output (CO_2 from microbial decomposition) (Krull, Skjemstad, & Baldock, 2004). Effectively, a carbon balance between minus 100 kg C/ha and plus 200 kg C/ha is rated optimal for agriculture[7], with scores decreasing in a linear fashion for lower or higher carbon balances (BASF, 2012). A similar pattern is seen in figure 9: when a certain percentage of SOC is reached, soil productivity does not increase any longer.

[5] The cations used by plants in the largest amounts are calcium, magnesium, and potassium. In dry climates, sodium can occupy an important portion of the CEC. The CEC of a soil with pH-dependent charge will increase with an increase in pH (Ketterings, Reid, & Rao, 2007).

[6] A convenient way to calculate SOM is by multiplying the percentage of organic carbon by a factor; however, conversion factors vary between 1.4 and 3.3 (Kuntze, 1988, Rasmussen and Collins, 1991) and this large range is due to the inherent differences between soils. Most commonly, a conversion factor of 1.72 is used (Baldock and Skjemstad, 1999). Therefore, to ensure consistency and allow reliable comparison of data, it is advantageous to report results as SOC rather than as SOM.

[7] Soils with higher carbon balances are often peaty, which is not preferable for agriculture.

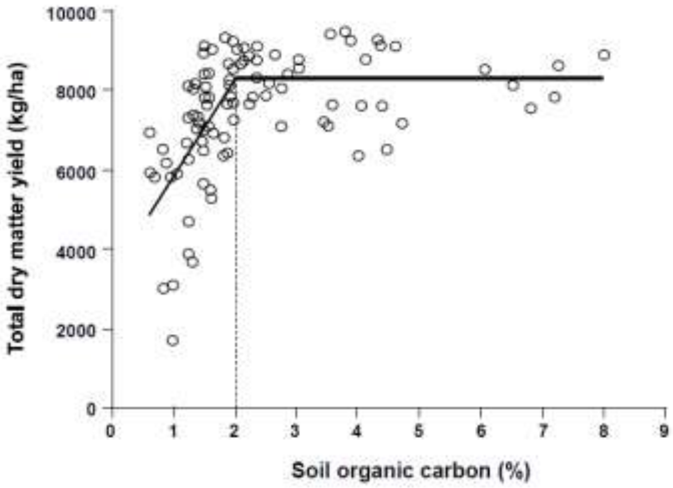

Figure 4.8 Relationship between organic C concentration in the surface 0-15 cm of soil and soil productivity as determined by total dry matter yield at dryland sites in Alberta, Canada (redrawn from Janzen et al. 1992, in (Krull, Skjemstad, & Baldock, 2004).

Different SOM pools influence different soil quality aspects. Therefore, total SOM (or SOC) content is a (too) crude indicator for soil quality, according to Krull et al. However, particularly Mila i Canals et al. (Milà I Canals, Romanyà, & Cowell, 2007b) and others argue that soil organic matter (SOM) can be used as an indicator for soil quality within LCA of agricultural systems: an increase in soil organic matter due to the soil management practices implies a benefit, whereas any decrease in SOM is accounted as damage to the system. The impact is measured as a carbon deficit (or credit, expressed by negative values) with the unit 'kg C/m^2/year', referring to the amount of extra carbon temporarily added to or removed from the soil in the system studied compared to a reference system (Brandão & Milà I Canals, 2012).

Milà I Canals et al. (2007a) describe the selected impact pathways of land use, linking the land use elementary flows (occupation; transformation) and parameters (intensity) registered in the inventory (LCI) to the midpoint impact indicators and to the relevant damage categories (natural environment and natural resources). An impact occurs when the land properties are modified (transformation) and also when the current man-made properties are maintained (occupation). The size of impact is the difference between the effect on land quality from the studied case of land use and a suitable reference land use on the same area (dynamic reference situation). The impact depends not only on the type of land use (including coverage and intensity) but is also heavily influenced by the bio-geographical conditions of the area. The time lag between the land use intervention and the impact may be large; thus land use impacts should be calculated over a reasonable time period after the actual land use finishes, at least until a new steady state in land quality is reached.
The main damages entailed by land use that should be considered by any method to assess land use impacts in LCIA are: biodiversity (existence value); biotic production potential (including soil fertility and use value of biodiversity); ecological soil quality (including life support functions of soil other than biotic production potential). Biogeographical differentiation is required for land use impacts, because the same intervention may have different consequences depending on the sensitivity and inherent land quality of the environment where it occurs.

In the ILCD Handbook, the latter method has recently been classified as 'recommended, but to be applied with caution', with changes in SOM (calculated following Milà I Canals, Romanyà, & Cowell, 2007b) as midpoint indicator for land use (European Commission -

Joint Research Centre - Institute for Environment and Sustainability, 2011). Characterisation factors for certain
land use flows in the background system are provided in Milà i Canals et al. (2007c), and are "work in progress" (see (Brandão & Milà I Canals, 2012).

Indicators & data availability

Guinee et al. summarize several existing methods that deal with the issue of soil fertility (soil quality) in environmental impact assessment (Guinée, Oers, Koning, & Tamis, 2006). Among others, soil organic matter is discussed as an indicator. According to Mila I Canals (Milà i Canals L. , 2003)) the soil organic matter content is an indicator for the long term effects on soil quality and its life support functions. It is proposed to use a SOM model to calculate characterisation factors for several interventions affecting SOM, like emission of crop residues, emission of organic residues (manure etc.) that have a positive effect on SOM and erosion and increased aeration that have a negative effect on SOM. At the time there were no operational characterisation factors. Cowell & Clift (2000) also suggest to use soil organic matter as an indicator. The characterisation model proposed is OM Indicator = M^{-1} in which OM is the tonnes of organic matter added to the system under analysis.

The following values are needed for the assessment of land use impacts, using SOM as an indicator:

- the land occupation due to an activity (Aa) per functional unit (e.g. ha year/f.u.);
- the SOM value at the start and end of land use (SOMini; SOMfin; Qini and Qfin in Figure 4.9);
- the SOM value at each moment of the occupation process (SOMa; Qa in Figure 4.9);
- the SOM value at each moment of the reference situation (SOMref; Qref in Figure 4.9);
- the SOM value at each moment of application of the backup technology during the relaxation to SOMini (Qini in Figure 4.9); and
- the potential SOM value of the site (SOMclimax).

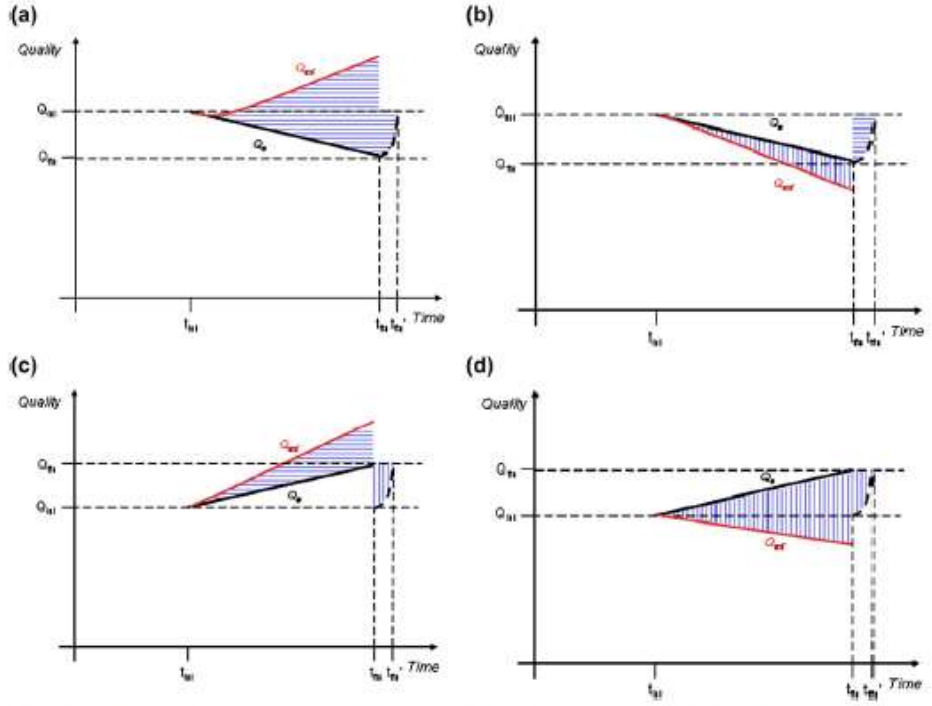

Fig. 5. Graphical representation of land use impact assessment. The horizontal hatching represents impacts due to the studied system, whereas the vertical hatching represents avoided impacts credited to the studied system. In (a) and (c) the studied system has lower quality than the alternative situation (reference), and so a damage accrues due to occupation. In (b) and (d) the reference is lower than the quality of the studied system, and so a benefit accrues due to occupation. In (a) and (b) the studied system reduces soil quality during the land use, and so a damage is considered during its recovery, whereas (c) and (d) improve soil quality and are credited for the avoided recovery.

Figure 4.9 Graphical representation of land use impact assessment (Milà I Canals, Romanyà, & Cowell, 2007b).

SOM content is usually estimated from the analysis of soil organic carbon (SOC), with SOC representing ca. 58% of SOM. In many soil surveys agricultural soils are sampled from 0 to 20 cm in one single soil horizon (Milà I Canals, Romanyà, & Cowell, 2007b).
Maps for organic carbon are available, for example Scharlemann et al. (Scharlemann, Hiederer, Kapos, & Ravilious) present a global map of estimated soil carbon stocks to 1m depth, generated based on the soil organic carbon and bulk density values included in the Harmonized World Soil Database ((IIASA), see Figure 4.10).

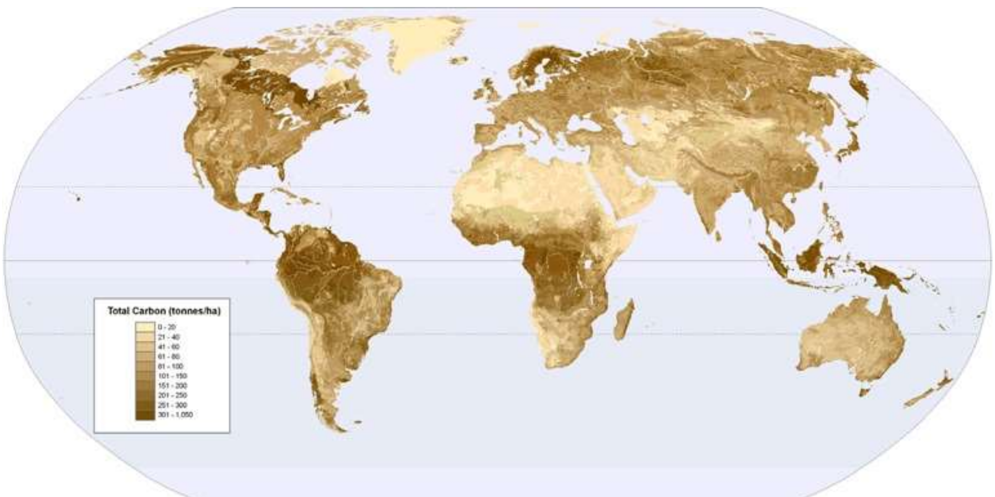

Figure 4.10 Global map of estimated carbon stocks (Scharlemann, Hiederer, Kapos, & Ravilious).

Mila I Canal's method (Milà I Canals, Romanyà, & Cowell, 2007b) is applicable in life cycle impact assessments, but it should be combined with biodiversity indicators for optimal use. Additionally, the LCA practitioner has to know and calculate a set of parameters before use. Some characterisation factors for the background system are available, but the foreground system has to be calculated by the LCA practitioner.

4.2.5 Evaluation of the impact assessment methodologies for soil quality

Table 4.3 shows the overview of the applicability of the impact assessment methodologies for soil quality, as a result of the discussion in the previous paragraphs.

Table 4.3 Overview of the applicability of the impact assessment methodologies for soil quality. Legend: "+" means High/Good; "±" means Moderate; "-" means Low/Poor.

Criterion	Soil Loss/ Erosion	Soil Organic Matter	Remarks
Relevance for AoPs			
Human Health	-	-	Soil is not directly relevant for Human Health
Biotic Resources	+	+	
Abiotic Resources	±	+	
Measurability	+	±	
Reliability	+	±	
Sensitivity	+ (USLE) ± (maps)	+	
Transferability/transparency	+	±	
Implementation in LCA	±	±	
Overall applicability	+	+	

With the information from the previous paragraphs and Table 4.3, we conclude that it would be valuable to add "soil quality" as a new environmental impact category. Since SOM as a (sole) indicator is also often proposed but does not influence erosion enough to represent it fully (let alone other indicators such as salinization) (Garrigues et al., 2012), we propose to calculate by means of two indicators:

1. soil erosion, to be derived from erosion maps or calculated with (r)USLE in specific cases.
2. soil organic matter, following the method of Milà I Canals (2007a,b,c) as also recommended by the ILCD Handbook.

5 Environmental assessment of bio-impacts in BioBuild

The findings on impact assessment of bio-related impacts as discussed in chapter 3 and 4 need to be integrated in the environmental assessment that is executed in BioBuild. There are three different environmental assessment tasks in the BioBuild project, being:

- Task 8.1 benchmark; in this task the environmental performance of competing existing building products is calculated, in order to be able to asses what the difference in environmneteal impact is in cases a BioBuild product will be applied. This task has been finished in December 2012.
- Task 8.3 Quick scan; the goals of this task is to allow for informed decision making during the development & design process.
- Task 8.5: Final assessment. At the end of the BioBuild project the (environmental) performance of the developed product system is assessed.

The Description of Work (DoW) of BioBuild defines what should be assessed in the environmental benchmark and the final assessment. The overview is shown in Table .

In the final assessment, a broad environmental assessment is performed for each case study. The assessment involves an analysis of the embodied energy, a CO_2-footprint, an environmental footprint including new developed impact categories for biobased products and additional environmental information. Both for the Benchmark (finished report of task 8.1) as well as for the Quick scans to support the design process (task 8.3) a shortened version of this assessment is performed. Paragraph 5.1.1. gives a further explanation of the applied methodology for environmental assessment. Cost and Health assessment is outside the scope of this task 8.4 report, and covered by other tasks (8.2 for health assessment and task 8.3/8.5)

Table 5.1 Overview of the sustainability assessments in BioBuild

	Benchmark / Quick scan	Final assessment
Environmental footprint	Not included (Embodied energy as proxy for fossil related impacts)	Categories required by EN ISO 15804 and advised by ILCD.
Additional impact categories	Land use & water use as proxies for new categories	*New categories, developed in Task 8.4.*
Embodied energy	Non-renewable energy use conform CED method	Non-renewable energy use conform CED method.
CO_2 footprint	Not included	IPCC GWP100 (conform EN ISO 15804 and ILCD).
Other environmental information	Not included	Required by EN ISO 15804: resource use, waste and output flows.
Cost assessment	Soft cost assessment	Life cycle costing analysis.
Health assessment	Not included; separate report (D8.2)	Update of information of D8.2.

5.1.1 Environmental footprint

5.1.1.1 Environmental footprint - In the benchmark report and quick scan

An environmental footprint would result in quite a lot (8-15) environmental scores. The complexity of the decision process involving all relevant environmental aspects results often in an unbridgeable gap for designers and decision makers. Weighting is a technique that is applied to overcome this gap, and many weighting methods are available for both the more political oriented decision making as well as for designers (Eco-indicator 95 and 99). However, weighting is not allowed for in comparative (public) LCA studies, As weighting is not allowed for in comparative public studies, an environmental footprint would result in quite a lot (8-15) environmental scores.

For this benchmark report, only a basic environmental analysis will be performed: embodied energy and two new categories will be assessed, because these provide a rather broad environmental overview altogether.

The embodied energy is analysed, which serves as a proxy for other current fossil-economy related impact categories. In addition, water and land use are added indicators to get a first impression on the bio-related environmental impacts during the benchmark and the quick scan.

5.1.1.2 Environmental footprint - In the final assessment

Both the ILCD Handbook and EN ISO 15804 prescribe a different set of environmental impact categories. The set prescribed by EN ISO 15804 is smaller. The set of the ILCD is not mandatory, but directive (“should”). Therefore, the ISO set will be followed in the first place. The required impact categories and the prescribed source for the appurtenant characterisation factors, is shown in Table 4.

Table 4.2 Set of impact categories and the source for their characterisation factors (CF’s), as required by EN ISO 15804.

Impact category	Source for CF’s
Global Warming	ELCD
Ozone Depletion	ELCD
Acidification	ELCD
Eutrophication	ELCD
Photochemical Ozone Creation	ELCD
Depletion of Abiotic Resources (elements)	CML
Depletion of Abiotic Resources (fossil)	CML

The ILCD advices to apply some more categories than the categories that are prescribed by EN ISO 15804. In order to show a broad overview of environmental impacts, these impact categories will be included in the results as well. For these impact categories, the characterization factors of the ILCD will be applied. The overview of the impact categories is given in Table .

Table 5.3 Set of impact categories as advised by the ILCD. For these categories, the characterisation factors of the ILCD will be used.

Impact category
Human toxicity
Respiratory inorganics
Ionising radiation
Ecotoxicity
Land use
Depletion of abiotic resources (renewable)

No weighting will be applied because it is a comparative study which will be disclosed to the public in D8.5 and weighting is not allowed in that case by EN ISO 15804. If necessary for the interpretation, a weighting step might be included in a separate section of the report, which will thus be not conform ISO. This will be explicitly indicated.

5.1.2 Added Bio-related impact categories

5.1.2.1 Additional impact categories - In the benchmark and quick scan

For the final assessment, additional environmental impact categories will be developed; more information on this can be found in the next paragraph. The benchmark (and the quick scan) report however, will be produced too early in the methodology development process and it will not be possible to include these.

However, it is most likely that the new developed categories will involve soil, water and landscape aspects. Land and water use are already present, to a certain extent however, in the lifecycle inventory data and are thus useful indicators for application in the quick scan and benchmark. Land use is not the same as soil use, but it is the closest category that is present in current lifecycle inventories. Therefore, in order to be able to give an indication of the potential effects of on soil and water, “land use” and “water use” will be added to the environmental impact analysis of the benchmark report.

For land use, the already existing characterization method of CML will be used. For water use, all water flows which use fresh water are included, shown in **Error! Reference source ot found.**. Excluded are salt water flows and non-consuming flows like turbine use and cooling water.

Table 5.4 Water flows included in the water use methodology for the benchmark analyses.

Flow name
Water, well, in ground
Water, unspecified natural origin/m3
Water, unspecified natural origin/kg
Water, unspecified natural origin, [country code][8]
Water, river
Water, process, well, in ground
Water, process, unspecified natural origin/m^3
Water, process, unspecified natural origin/kg
Water, process, surface
Water, process, salt, ocean
Water, process, drinking
Water, process and cooling, unspecified natural origin
Water, lake
Water, fresh
Water, unspecified, very high water stress
Water, unspecified, moderate water stress
Water, unspecified, medium water stress
Water, unspecified, low water stress
Water, unspecified, high water stress
Water, unspecified, extreme water stress
Water, fossil

5.1.2.2 Additional impact categories - In the final assessment

In addition to the general impact assessment method in paragraph 5.1.1, new impact categories are developed in BioBuild in order to assess additional effects which play a role in the production of biobased materials. These new impact categories are developed in a separate task (task 8.4) and have a later deadline than the deadline of this benchmark

[8] “Country code” refers to the more than 100 country specifications that are made in the LCA software. Example: “BR” is the country code for Brazil.

report, and are therefore not yet fully developed. More information on this can be found in the D8.4 deliverable when it is finished.

5.1.3 Analysis of embodied energy

For the analysis of Embodied Energy, an existing method was adapted to the specific needs for BioBuild. The method 'Cumulative Energy Demand' of Ecoinvent (Hischier, et al., 2010) is applied. The focus of the environmental analyses will lay on the *non-renewable* energy sources and not on the total embodied energy including the use of renewable sources. This method is used in both the benchmark and the final report.

5.1.4 Carbon footprint

The Description of Work states that a carbon footprint will be calculated. As the carbon footprint is closely related to energy use, it is not calculated for the benchmark report but only for the final assessment.
For the calculation of the carbon footprint, the ELCD characterisation factors will be used, in accordance with EN ISO 15804. Additionally, the ILCD handbook requires the following:

- CO_2 emissions from land transformation will be calculated by means of the characterisation factors that are stated in Annex B of the ILCD Provisions (European Commission - Joint Research Centre - Institute for Environment and Sustainability, 2010).
- Conform the ILCD Provisions 7.4.3.7, carbon uptake by plants is included in the calculations.
- Temporary storage of carbon is not taken into account because this is only a temporary measure.

As the uptake by plants is included in the assessment, the release of the carbon (at end-of-life of the product) is included as well. In this way the temporary storage is not taken into account. In practice, bioproducts can considered to be carbon neutral: the bio-part of the carbon that was once absorbed is also released.

5.1.5 Additional environmental information

The aim of the final report is to contain all Environmental Life Cycle Assessment based information that would be needed for an Environmental Product Declaration (EPD) conform EN ISO 15804; which is all information required by tables 4, 5 and 6 of EN ISO 15804. The additional required I formation that is not covered by paragraphs 5.1.1 to 5.1.4 concerns resource use, waste and output flows.

6 Conclusion and discussion

6.1 <u>Overall conclusion on methodology development</u>

In this research, the policy issues with respect to the environmental assessment of biobased products have been analysed, both from a European and from a global perspective. Summarizing, from the policy there is an actual call for better indicators for Water, Land(scape) and/or Soil and Ecosystem Services & Biodiversity related impacts and Food security.

Next, it has been analysed to what extent these issues could be addressed with state of the art environmental impact assessment methodologies and where the main gaps in science are located. Limiting uncertainties is important to increase acceptance of the chosen methodology as the outcomes will be used for decision making. End-point indicators introduce many additional uncertainties and the uncertainties in the currently available methods are judged to be too high, either due to lack of inventory data or as a result of limited available quantification methods. Therefor mid-point indicators are preferred to assess bio-related impacts. This decision results in the outcome that biodiversity and food competition will not be quantified, as they both are considered to be end-point indicators.

Landscape was seen as a relevant pressure, but it is to a large extent dependent on the specific geographical location of impact. Due to the local nature of landscape impacts it cannot be integrated in generic life cycle assessments, and is more likely to be an indicator for (local) environmental risk studies. Therefor Landscape impacts were not investigated further in this study.

Hydrology and soil quality are the impact categories that were selected for further investigation in this study. The different methods for environmental impact assessment of hydrology and soil quality were investigated in a descriptive and analytical manner.

In hydrology, the main issue is water, as water is relevant for all three Areas of Protection. Water can be assessed in two ways: by assessing the water use or the water stress. Two methods for measuring the impact of water use were discussed (Bayart et al. and Milà). For water stress, only the most prominent method was discussed (Pfister et al.). The conclusion from the analysis is that it would be valuable to add “water” as a new environmental impact category, which is calculated by means of two indicators: the water stress index and water as a resource (the aquifer reservoirs, as calculated in the blue water footprint).

Soil quality can be incorporated in environmental impact assessments by methodologies on the basis of either soil erosion or soil organic matter. Several methods were discussed how soil erosion can be included in LCA. Data is available in the form of maps. Regarding soil organic matter, there are also several calculation methods. The method of Mila I Canals et al is recommended by the ILCD. The analysis of all different methods leads to the advice to add “soil quality” as a new environmental impact category, which is calculated by means of two indicators: soil erosion, to be derived from erosion maps or calculated with (r)USLE in specific cases, and Soil organic matter, following the method of Milà I Canals (2007a,b,c) as also recommended by the ILCD Handbook.

By means of the proposed additions to the current impact assessment methods, the policy request for better indicators concerning Water and Soil can be covered. The request for better biodiversity indicators could not be solved because of the additional uncertainties that are introduced in endpoint damage approaches. There is both a lack in methodology and inventory data to overcome this shortage, and this is the reason why midpoint indicators are still preferred over end-point indicators. For Biodiversity the occupation of land seems to be the most suitable midpoint indicator, as there is enough inventory data available, and it is free of value based weighting. The crude m2 land use indicator can be used in combination with the global estimation of the value of eco-system services made by Costanza in cases a weighting between the environm,ental impact categories is needed for interpretation. The method by Costanza is internationally accepted and gives a global average value of ecosystem services, preventing extreme outliers which occur using the PDF.m2.yr method. Furthermore, it is more transparent, although the assessment of the quality of the land-use is still limited.

The crude m2 land use indicator can also be used as a mid-point indicator for land-use completion that may threat food security. It is impossible to draw conclusions in this value laded public debate without subjective weighting.

6.2 Conclusions and recommendations for environmental assessment of bio-impacts in BioBuild

Based on this scientific methodology study, the conclusion can be drawn that impact assessment if bio-related impacts is composed of two factors, being quantity and quality.

Environmental assessment in BioBuild needs to distinguish between the final assessment at the end of the BioBuild project and the environmental assessment that is executed respectively at the beginning of the project (benchmark) and during the development process (quick-scans). The last two assessments are based on a shortened version of the final assessment. In this shortened simplified approach, a provision was taken to cover the bio-related impacts, being land-use and water-use. Both are limited to the quantification factor only, and lack assessment of the quality of respectively the land use and water use. The benchmark study is already finished, and many of the quick scans are also already executed to support the decision making within the BioBuild project. At this stage of the study, a decision needs to be taken if the method that is applied in the quick scans will be adapted, and what the method for the final assessment will be. As the quick scans show that the most crucial decisions for environmental impact are on fibres, fibre treatment, fibre preforms and resins, the most crucial quick scans for informed decisions making have already been executed. In addition the shortened version of the method does cover the two selected midpoint indicators in a quantitative way, and can therefore be seen a method that is representative for the final assessment. The quantification of land use can be seen as the coarse mid-point indicator for biodiversity, food competition and soil quality, while the quantification of water consumption can be seen as an indicator for hydrology. Based on both arguments, being the fact that the main influential decisions are already taken and the fact that the method does represent the new bio-related impact themes, it is decided not to change the quick scan method at this stage of the BioBuild project. This decision will keep the quick scans consistent.

The quantification of land use can be seen as the coarse (quantitative) mid-point indicator for ecosystem services, biodiversity, food competition and soil quality. The final assessment in BioBuild will elaborate on land use in the following way:

- Soil quality; the quantitative indicator of land use (m2.yr) will be combined with information on soil quality. This new environmental impact category, will be calculated by means of two indicators: soil erosion, to be derived from erosion maps or calculated with (r)USLE in specific cases, and Soil organic matter, following the method of Milà I Canals (2007a,b,c) as also recommended by the ILCD Handbook.
- Biodiversity; As the ILCD recommends to use the Potentially Disappeared Fraction of Species (PDF.m2.yr) to quantify biodiversity, a sensitivity analysis will be made by calculation the PDF.m2.yr using the Recipe method. The risk of this method is that it is known for its extreme outliers.
- Food competition; For food competition, the ILCD does not make any recommendations, and there for a qualitative description will be made on the likeliness that food competition can become an issue for the product under study.
- Comparing land use with other environmental impacts. Weighting is not allowed by EN ISO 15804 in comparative studies disclosed to the public. However is weigting between impact categories is necessary for the interpretation, a weighting step might be included in a separate section of the report, which will thus be not conform ISO. In that case:
  - The value of land-use related eco-system services is based on the the more crude m2 land use indicator combined with the the global estimation of the value of eco-system services made by Costanza for a global assessment. This cost factor is applicable for BioBuild as the origin of the bio-resources is not limited to Europe only. In addition, the method by Costanza is internationally accepted and transparent.
  - A sensitivity analysis is made to find out if different methods for the land-use impacts influence the ranking of the environmental performance of the products. In this sensitivity analysis the shadow costs for biodiversity and soil quality are used.

The quantification of water can be seen as a coarse midpoint indicator for hydrology. In the previous paragraph was concluded that that it would be valuable to add “water” as a new environmental impact category, which is calculated by means of two indicators: the water stress index and water as a resource (the aquifer reservoirs, as calculated in the blue water footprint). However, the current LCO software that is used to perform the environmental assessment in BioBuild is not yet suitable to perform this water assessment. The database supplier (Ecoinvent) has been working on an update of the database that enables water assessment. However the LCA software needs to be adapted to be suitable for this new type of database. The release of the software update is not yet planned. If the new version of the software is available at the time of the final assessment, it will be applied for BioBuild. If not, the final assessment will be based on water quantity only, and a qualitative interpretation will be made on the quality of the used water to describe the (potential) impact.

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Appendix 1 SEBI 2010 biodiversity indicators

CBD focal area	Headline indicator	SEBI 2010 specific indicator
Status and trends of the components of biological diversity	Trends in the abundance and distribution of selected species	1 Abundance and distribution of selected species a Birds b Butterflies
	Change in status of threatened and/or protected species	2 Red List Index for European species 3 Species of European interest
	Trends in extent of selected biomes, ecosystems and habitats	4 Ecosystem coverage 5 Habitats of European interest
	Trends in genetic diversity of domesticated animals, cultivated plants, and fish species of major socioeconomic importance	6 Livestock genetic diversity
	Coverage of protected areas	7 Nationally designated protected areas 8 Sites designated under the EU Habitats and Birds Directives
Threats to biodiversity	Nitrogen deposition	9 Critical load exceedance for nitrogen
	Trends in invasive alien species (numbers and costs of invasive alien species)	10 Invasive alien species in Europe
	Impact of climate change on biodiversity	11 Impact of climatic change on bird populations
Ecosystem integrity and ecosystem goods and services	Marine Trophic Index	12 Marine Trophic Index of European seas
	Connectivity/fragmentation of ecosystems	13 Fragmentation of natural and semi-natural areas 14 Fragmentation of river systems
	Water quality in aquatic ecosystems	15 Nutrients in transitional, coastal and marine waters 16 Freshwater quality
Sustainable use	Area of forest, agricultural, fishery and aquaculture ecosystems under sustainable management	17 Forest: growing stock, increment and fellings 18 Forest: deadwood 19 Agriculture: nitrogen balance 20 Agriculture: area under management practices potentially supporting biodiversity 21 Fisheries: European commercial fish stocks

CBD focal area	Headline indicator	SEBI 2010 specific indicator	
		22	Aquaculture: effluent water quality from finfish farms
		23	Ecological Footprint of European countries
Status of access and benefits sharing	Percentage of European patent applications for inventions based on genetic resources	24	Patent applications based on genetic resources
Status of resource transfers	Funding to biodiversity	25	Financing biodiversity management
Public opinion (additional EU focal area)	Public awareness and participation	26	Public awareness

Appendix 2 Suggestions for Water Stress in relation to End Points

A2.1 Introduction

In chapter 4.1, the water stress index (WSI) is discussed as an indicator to measure the impact on water quantity. The aim of the chapter was to discuss only the relation to midpoint impact categories and not to endpoints. However, water stress has a clear relation with all three areas of protection (AoPs) and thus with the endpoint impacts. These relations are discussed in this chapter.

A2.2 WSI in relation to Human Health

Lack of freshwater for hygiene and ingestion leading to communicable diseases depends on local circumstances which are difficult to assess in LCA. Water shortage for irrigation leading to malnutrition is the focus in the article of Pfifster et al. (Pfister, Koehler, & Hellweg, 2009).

Damage, induced by water consumption in a watershed of country i, is measured in disability adjusted life years (DALY), as in the Eco-indicator-99 method for the assessment of human health effects.

$$\Delta HH_{malnutrition,i} = \underbrace{\underbrace{\frac{WSI_i * WU_{\%,agriculture,i}}{WDF_i}} * \underbrace{\frac{HDF_{malnutrition,i} * WR_{malnutrition}{}^{-1}}{EF_i}} * DF_{malnutrition}}_{CF_{malnutrition,i}} * WU_{consump}$$

$\Delta HH_{nmalnutrition, I}$	= damage to human health
$CF_{malnutrition, i}$	= specific damage per unit water consumed (as specified in LCI-phase) (DALY/m^3)
WDF_i	= water deprivation factor ($m^3_{deprived}$/$m^3_{consumed}$)
EF_i	= effect factor, annual number of malnourished people per water quantity deprived (capita*year/$m^3_{deprived}$)
$HDF_{malnutrition}$	= human development factor, relates human development index to malnutrition vulnerability
$WR_{malnutrition}$	= water requirement to prevent malnutrition (m^3/ year*capita)
$DF_{malnutrition}$	= damage factor (DALY/(year*capita))

WR and DF independent of location

$$HDF_{malnutrition} = \begin{cases} 1 & for \quad HDI < 0.30 \\ 2.03 * HDI^2 - 4.09 \quad +2.04 & for \; 0.30 \le HDI \le 0.88 \\ 0 & for \quad HDI > 0.88 \end{cases}$$

HDI = Human Development Index

For the Netherlands HDI = 0.91 (UNDP, 2011), this means that the HDF = 0. As a result the damage caused by malnutrition due to water shortage is zero. The impacts will be mainly located in developing countries with a low human development index (HDI)

Water requirement to prevent malnutrition ($WR_{malnutrition}$) is set to 1,350 m^3, the minimum direct human dietary requirements. This matches modelled water resource thresholds for food security. The damage factor ($DF_{malnutrition}$) is $1.84*10^{-2}$ DALY/(year*capita)

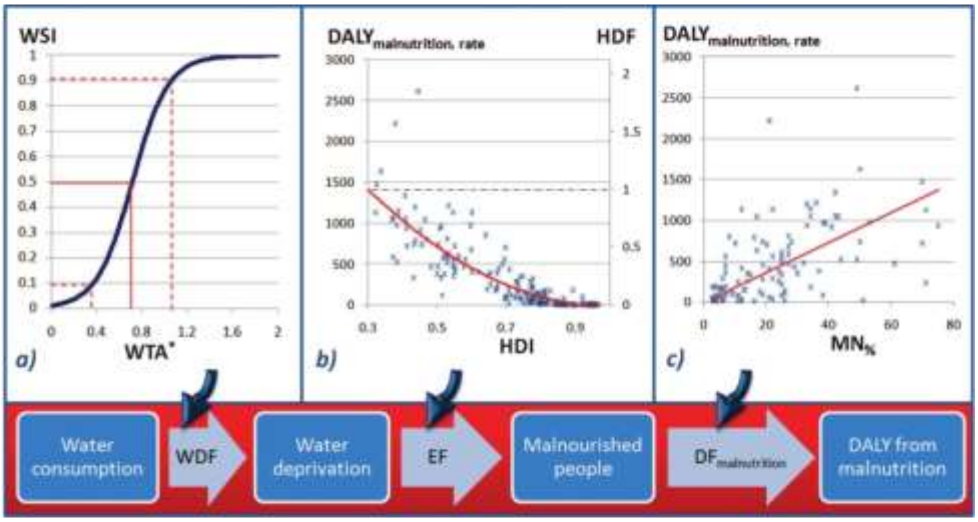

Figure 7.1 Inputs to the impact pathway: a) relation between SI and WTA (blue line, logistic function, b) $Daly_{malnutrition,\ rate}$ for each country (blue stars) and HDF modelled (red line R^2 =0.71) based on HDI, c) $DALY_{malnutrition,\ rate}$ for each country (blue stars) against corresponding $MN_{\%}$ and linear regression (red line, R^2 =0.26)*

Limitations:

- WSI is simple screening indicator
- Human health strongly related to socio economic factors, local situation

<u>A2.3 WSI in relation to Biotic Resources</u>

In places were plant growth is water limited, withdrawal of blue water can reduce the availability of green and diminish vegetation and plant diversity. Riparian and groundwater dependent vegetation is often crucial for ecosystems including birds and insects which are important for the whole ecosystem.

Global specially explicit data for assessing water shortage related vegetation are only available for net primary production (NPP), proxy for ecosystem quality.

$$\Delta EQ = CF_{EQ} * WU_{consumptive} = \underbrace{NPP_{wat-lim,i}}_{PDF} * \underbrace{\frac{WU_{consumptive}}{P}}_{A*t}$$

ΔEQ	= Delta ecosystem quality (m^2*year)
CF_{EQ}	= Ecosystem damage factor (m^2*year/m^3)
WU	= Water use (m^3)
$NPP_{wat\text{-}lim}$	= fraction of net primary production limited by water availability, representing the water shortage vulnerability of an eco-system
PDF	= potentially disappeared fraction of species
P	= annual precipitation (m/year)

Limitations:

- Terrestrial ecosystems are assessed in line with existing methods
- Aquatic ecosystems only partially dependent on quantities, infrastructure and quality can be more important

<u>A2.4 WSI in relation to Abiotic Resources</u>

Back up technology concept as used for abiotic resource depletion in EI99, expressed in “surplus energy” (MJ) to make the resource available in the future is employed to assess the damage to freshwater resources.

$$\Delta R = E_{desalination} * F_{depletion} * WU_{consumptive}$$

ΔR = delta resources
$E_{desalination}$ = energy needed for seawater desalination (MJ/m^3) – sota = 11 MJ/m^3
$F_{depletion}$ = fraction of frashwater consumption that contributes t depletion
WU = water use

$F_{depletion}$ serves as characterization factor for the midpoint indicator “freshwater depletion”

$$F_{depletion,i} = \begin{cases} \frac{WTA-1}{WTA} & for\ WTA > 1 \\ 0 & for\ WTA \leq 1 \end{cases}$$

Limitations:

- Controversial concept, but allows combined approach of stock and flow resources.
- Surplus energy kept constant for all regions, but desalination will be location dependent.
- Impact water consumption on ecosystems quality is larger than on human health and resource depletion. First on global scale, others limited to specific regions.

Appendix 3 Suggestions for weighting of soil impacts

A3.1 Combining soil related indicators
To add several impacts into a midpoint indicator, characterization will be necessary. The AgBalance methodology (BASF, 2012) considers several indicators in its soil impact category which are weighed into a total score:

- soil organic matter balance (14%);
- nutrients balance (14%);
- soil compaction potential (10%);
- and soil erosion (62%).

Within LCA methodology, the nutrients balance impacts are dealt with in the eutrophication impact category and although literature is clear on the importance of soil compaction (for example Mila I Canals et al 2007b), to our knowledge there is no information available on this process on a global scale. Therefore, it is chosen to combine the indicators Soil loss and Soil organic matter into the Soil quality impact category. Scaling these two indicators to 100%, soil loss weighs in for 82% and soil organic matter weighs in for 18%.

A3.2 Suggestions for shadow pricing
A shadow price for top soil would be appropriate to assess the costs of soil quality loss, including both loss due to erosion as well as quality deterioration related to SOM.
The U.S. Natural Resource Conservation Service (website viewed May 21, 2012) estimates the costs of replacing soil functions and remedying off-site damage at $28 per tonne of top soil (USD 2011), see Figure 7.2.

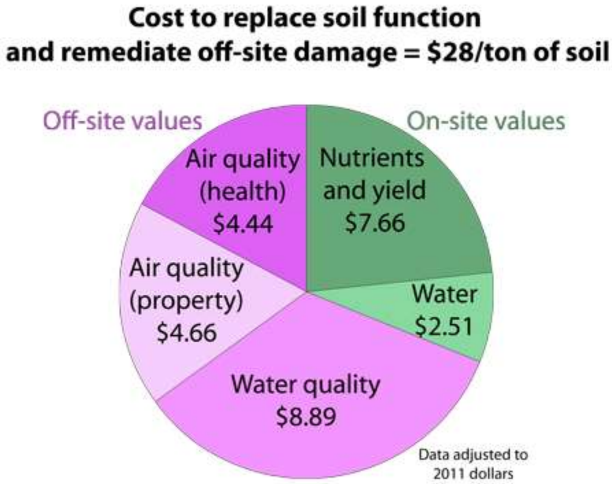

Figure 7.2 The cost to replace soil function and remediate off-site damage (U.S. National Resources Conservation Service).

Other estimates show that the range of uncertainty is large. The EC Joint Research Council (Joint Research Council, 2010) summarizes from the impact assessment document of the Soil Thematic Strategy the costs of organic matter decline estimated at EUR 3.4–5.6 billion/year and erosion at EUR 0.7–14.0 billion/year for the EU.
Yet another approach summarized in Krull et al. (Krull, Skjemstad, & Baldock, 2004) suggests that the application of humic substances (lignite or oxidised coal) would be an economically viable source for rehabilitation of degraded soils as humic substances are relatively inexpensive (US$ 0.5-1.0) and only small amounts (100-300 kg ha-1, depending on substance) are required compared with much larger amounts for farmyard manure applications (50-200 t ha-1). However, Piccolo et al. (1997) also found that there was an upper limit beyond which the beneficial effects of humic substances failed (beyond 0,1 g/kg).

As an upper limit to the shadow price, it is suggested to use the costs of application of the ultimate backup technology. For example, if 1 ha of fertile land is lost, the impacts of constructing an equivalent amount of greenhouses with hydroponics (in terms of yield potential for a similar crop) could be calculated to account for the lost resource functionality for humans (i.e. productivity). The loss of other functions (water cycle, buffer capacity, etc.) should be measured with other backup technologies (Milà I Canals, Romanyà, & Cowell, 2007b).

Appendix 4 Suggestions for Landscape impacts

In this annex, we discuss five interesting sources: Guinée *et al*. (the SOWAP project), OECD, Werf & Petit, Garrigues *et al.* and several sources which mention landscape as a potential addition.

A4.1 Guinée et al. – SOWAP Project

The Soil and Water Protection (SOWAP) project was a collaborative activity by industry, NGOs, academic institutions and farmers to address the concept of conservation tillage in the UK, Belgium and Hungary and the Czech republic. The project focused on issues such as erosion, hydrology, soil fertility and biodiversity. It delivered numerous useful results, like data on soil erosion, water use, nutrients use, quality of the crop, biodiversity, etc. at the experimental farms. The key challenge was to bring these data in an encompassing framework for further assessment and decision support.

Guinée *et al.* (Guinée, Oers, Koning, & Tamis, 2006) presented a study on a life cycle framework for a methodological consistent environmental analysis of agricultural management systems, focusing especially on impact categories that have not yet maturely developed within LCA but are of particular importance in agricultural studies. They explicitely addressed erosion, hydrology/desiccation and soil fertility as problematic topics.

Besides from the SOWAP project, Guinee et al. collected potential indicators from other literature sources (Guinée, Oers, Koning, & Tamis, 2006). An overview of these indicators is shown in Table 7.1.

Table 7.1 Overview of additional indicators for agricultural production systems (based on (Guinée, Oers, Koning, & Tamis, 2006)).

Indicator	Sub-system	Source
Naturalness	Hemeroby	(Brentrup, Küsters, Lammel, & Kuhlmann, 2002)
Vascular plants species density	Biodiversity	(Lindeijer, Biodiversity and life support impacts of land use in LCA, 2000), (Lindeijer, Kok, Eggels , & Alfers, 2002), (Weidema & Lindeijer, 2001)
(f)NPP	Soil fertility	(Lindeijer, Kok, Eggels , & Alfers, 2002)
Soil Organic Matter	Soil fertility	(Milà i Canals L. , 2003)
Organic matter	Soil fertility	(Cowell & Clift, 2000)
Soil compaction	Soil fertility	(Cowell & Clift, 2000)
Regional water balance (change)	Hydrology	(Heuvelmans, Muys, & Feyen, 2005)
Cooling capacity	Exergy	(Wagendorp, Gulinck, Coppin, & Muys, 2006)
Soil static reserve life	Resources	(Cowell & Clift, 2000)
Dynamic water reserve life	Resources	(Heuvelmans, Muys, & Feyen, 2005)

After having studied the possible indicators to assess the environmental impact of agricultural land use, Guinee et al. (Guinée, Oers, Koning, & Tamis, 2006) identify the following for four potential additional impact categories:

2. Biodiversity

- none of the biodiversity impact assessment methods seem to be suitable directly as the method is either (1) too immature to apply in life cycle assessment, or (2) the data collection is not feasible within the BioBuild planning, or (3) the method generated too many uncertainties for decision making.
- the use of land occupation (m^2yr) is in the short term seen as the most basic indicator for suppressing biodiversity, however lacks an indication of the quality of the land-use.
- land transformation impacts must be seen as an economic process with its own interventions.

3. Hydrology
   - the impact assessment method by Heuvelmans et al. (2005) for water use, based on the depletion of water due to water use (water as a resource), seems promising.
   - in the short term collect data on water use and aggregate these without further weighting.

4. Soil fertility
   - soil fertility should not be an impact category as it represents an economic asset.

5. Erosion
   - soil loss as such cannot be the intervention as erosion is a natural phenomenon that happens without any human intervention. Enhanced soil loss is the impact to assess and should be linked to interventions such as cutting hedgerows and ploughing at different depths.
   - in the long term perhaps enhanced erosion can be taken up in LCIA, for the short term the “quick and dirty” approach in terms of soil loss may be adopted.

With all the information collected, Guinée *et al.* made a schematic overview of the cause and effect chains of land use related interventions. It is shown in Figure 7.3 below. The scheme can be seen as an extension to current life cycle assessment methodology as discussed in the previous paragraph. The information from this scheme will be further used in the concluding paragraph of this chapter, but first the other additional theories will be discussed.

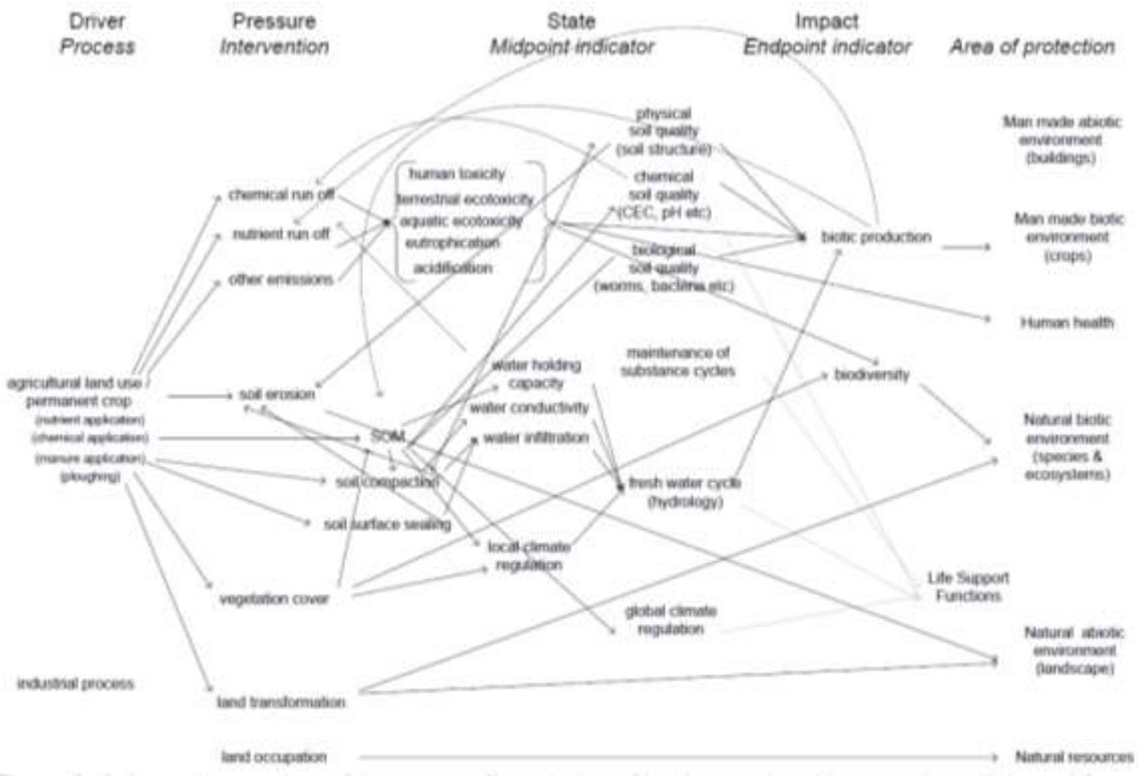

Figure 7.3 Schematic overview of the cause effect chains of land use related interventions (Guinée, Oers, Koning, & Tamis, 2006).

<u>A4.2 OECD</u>

The OECD realised that agriculture is not only important because it contributes to the economy and provides food and feed, but that it also has positive and negative impacts on the environment and on society. To be able to monitor the state of these impacts over different countries and over time environmental indicators were developed ((OECD, 1999) and (OECD, 2001)). These indicators could point out impact categories currently (partially) lacking from the commonly used LCIA methods.

The OECD (2001) defines four major groups of indicators (see Table 7.2) concerning:

1. Agriculture in the broader economic, social, and environmental context
2. Farm management and the environment
3. Use of farm inputs and natural resources
4. Environmental impacts of agriculture

Table 7.2 Complete list of OECD agro-environmental Indicators[1] (OECD, 2001).

I. AGRICULTURE IN THE BROADER ECONOMIC, SOCIAL AND ENVIRONMENTAL CONTEXT		
1. Contextual Information and Indicators		2. Farm Financial Resources
• *Agricultural GDP* • *Agricultural output* • *Farm employment* • *Farmer age/gender distribution* • *Farmer education* • *Number of farms* • *Agricultural support*	• *Land use* – Stock of agricultural land – Change in agricultural land – Agricultural land use	• *Farm income* • *Agri-environmental expenditure* – Public and private agri-environmental expenditure – Expenditure on agri-environmental research
II. FARM MANAGEMENT AND THE ENVIRONMENT		
1. Farm Management		
• *Whole farm management* – Environmental whole farm management plans – Organic farming	• *Nutrient management* – Nutrient management plans – Soil tests • *Pest management* – Use of non-chemical pest control methods – Use of integrated pest management	• *Soil and land management* – Soil cover – Land management practices • *Irrigation and water management* – Irrigation technology
III. USE OF FARM INPUTS AND NATURAL RESOURCES		
1. Nutrient Use	2. Pesticide Use and Risks	3. Water Use
• *Nitrogen balance* • *Nitrogen efficiency*	• *Pesticide use* • *Pesticide risk*	• *Water use intensity* • *Water use efficiency* – Water use technical efficiency – Water use economic efficiency • *Water stress*
IV. ENVIRONMENTAL IMPACTS OF AGRICULTURE		
1. Soil Quality	3. Land Conservation	4. Greenhouse Gases
• *Risk of soil erosion by water* • *Risk of soil erosion by wind*	• *Water retaining capacity* • *Off-farm sediment flow*	• *Gross agricultural greenhse gas emissions*
2. Water Quality		
• *Water quality risk indicator* • *Water quality state indicator*		
5. Biodiversity	6. Wildlife Habitats	7. Landscape
• *Genetic diversity* • *Species diversity* – Wild species – Non-native species • *Eco-system diversity* (see Wildlife Habitats)	• *Intensively-farmed agricultural habitats* • *Semi-natural agricultural habitats* • *Uncultivated natural habitats* • *Habitat matrix*	• *Structure of landscapes* – Environmental features and land use patterns – Man-made objects (cultural features) • *Landscape management* • *Landscape costs and benefits*

1. This list includes all the agri-environmental indicators covered in the Report. For a detailed description of each indicator, see the Annex to this chapter.
Source: OECD Secretariat.

The general objectives of OECD work on agro-environmental indicators are intended to contribute to the demands of policy makers and other stakeholders in three ways (OECD, 2001):

1. By providing information to policy makers and the wider public on the current state and changes in the conditions of the environment in agriculture.
2. By assisting policy makers to better understand the linkages between the causes and impacts of agriculture, agricultural policy reform, trade liberalisation and environmental measures on the environment, and help to guide their responses to changes in environmental conditions.
3. By contributing to monitoring and evaluating the effectiveness of policies addressing agro-environmental concerns and promoting sustainable agriculture.

These indicators were thus not at first hand designed to be used in a life cycle approach. However, they may provide very relevant indicators for the additional impact categories. Their suitability for implementation in a life cycle approach will be discussed in the concluding paragraph of this chapter.

A4.3 Werf & Petit

Werf and Petit (Werf & Petit, 2002) compared and evaluated 12 indicator-based approaches to assessing environmental impact of agriculture at the farm level. Table 7.3 shows an overview of the indicators. It is interesting to notice that the three categories of indicators match with common life-cycle assessment terms ‘inventory’, ‘midpoint impact category’ and ‘endpoint impact category’.

Table 7.3 Input related, emission related and system related indicators as distinguished by Werf and Petit (2002).

Input related	Emission related	System state related
Use of non-renewable energy	Emission of greenhouse gases	Landscape quality
Use of other non-renewable resources	Emission of ozone depleting gases	Natural biodiversity
Soil erosion	Emission of acidifying gases	Agricultural biodiversity
Land use	Emission of nutrifying substances	Total system biomass
Water use	Emission of pesticides	Air quality
Nitrogen fertiliser use	Emission of substances contributing to POCP	Water quality
Pesticide use	Emissions concerning terrestrial ecotoxicity	Soil quality
	Emissions concerning aquatic ecotoxicity	Food (product) quality
	Emissions concerning human toxicity	Animal welfare
	Waste production and utilisation	

Werf and Petit (2002) concluded their paper with guidelines for objectives and indicators of which they consider the most relevant for their study:

- The indicators should cover both local and global effects.
- The procedure used for the selection of objectives should be stated.
- Effect-based[9] indicators are preferred over means-based indicators as the link with the objective is more direct.
- If possible, threshold values should be defined for indicators. (However, the use of thresholds is not a common approach in LCIA.)

The most frequently mentioned indicators in the study of Werf and Petit (2002) that do not occur in the current LCIA methods are: landscape quality, agricultural biodiversity and soil quality. Also natural biodiversity is often mentioned but this indicator has no definitive LCIA methodology yet.

A4.4 Garrigues et al

Garrigues *et al.* (Garrigues, Corson, Angers, Werf, & Walter, 2012) specifically addressed soil quality as an indicator for LCIA. They review a number of available field- or farm-level methods or tools for assessing soil quality and discuss available LCA approaches. According to Garrigues *et al.* soil quality can be expressed by:

- physical, chemical and biological properties;
- soil functions and

[9] Means-based indicators relate to farmer production practices (e.g. fertiliser inputs or erosion control measures). The effects these practices have on the state of the farming system or on emissions to the environment (e.g. nutrient leaching or actual soil loss) are called effect-based indicators. In terms of the DPSIR framework (see 5.3) means-based indicators are related to Pressures and Responses, while effect-based indicators are related to Pressures and Impacts.

- processes that could degrade the soil (e.g., erosion, compaction, salinization, loss of soil organic matter).

Soil properties and functions are difficult to use as inventory items (Garrigues, Corson, Angers, Werf, & Walter, 2012) because of the difficulty in determining how they influence the system functions (e.g., productivity, land use). In contrast, processes that degrade the soil are easier to relate to functional units because many can be expressed as flows and be used to calculate impact indicators.