Assessing the environmental performance of biomass supply chains

Methods, Results, Challenges and Limitations

This report presents state of the art information on environmental impacts of biomass supply chains including how to assess them and identify the challenges and limitations. This publication provides an introduction of the application of LCA to biomass production systems, identifying factors to be taken into account for assessing these biological systems.

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ASSESSING THE ENVIRONMENTAL PERFORMANCE OF BIOMASS SUPPLY CHAINS

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Summary report

KEY MESSAGES

- Life Cycle Assessment (LCA) is a widely accepted tool to assess the environmental impacts of a product or service. Since it has been developed for the assessment of industrial production, the assessment of biomass supply chains is not straightforward.
- If the environmental impacts of biomass-based systems are to be assessed completely, additional non-standard impact categories like land use and land use change, water use, carbon stocks, soils and biodiversity should be part of the assessment.
- This publication gives an introduction to the application of LCA to biomass production and provides recommendations on what should be taken into account in order to assess the environmental impacts of biomass supply chains as thoroughly as possible.

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EXECUTIVE SUMMARY

The current discussion about sustainability of biomass in the on-going public debate and political arena is mainly about environmental performance and more specifically climate impact of biomass cultivation and use. This publication aims to broaden the debate, raise awareness and provide state of the art information on environmental impacts of biomass supply chains, how to assess them and what the challenges and limitations are.

Life Cycle Assessment (LCA) is a widely accepted tool to assess the environmental impacts of a product or service. Since it has been developed for the assessment of industrial production, the assessment of biomass supply chains is not straightforward. Whereas goal and scope of a study determine if an attributional or consequential LCA is appropriate, especially setting the system boundaries needs to take into account the specific characteristic of biomass supply chains.

Forest production, for example, is a long term production system that takes several decades or longer. Since LCA is designed for short-term production systems and their products, it is a challenge to define an adequate temporal system boundary that covers all relevant silvicultural and technical processes necessary to produce roundwood or wood chips. A temporal system boundary that covers a full rotation period and, hence, the complete production process of roundwood, would cover both past and future. Since the spatial, temporal and technical system boundaries need to match, the technical boundary, for example, would need to cover both past and future technology. A solution is needed that covers all relevant production processes with a modest degree of uncertainty. One option is to apply the normal forest model based on yield tables or forest growth models. Across a normal forest all production processes that usually take place in the course of a complete rotation period take place side by side in any one year.

Unlike industrial production systems, biomass supply chains occur in the so-called 'biosphere'. In other words, the production of biomass is first and foremost based on natural production factors like sunlight, soil, water, nutrients and processes like photosynthesis and plant metabolism. Due to complexity, immature methodology and missing data, LCAs of biomass-based products or services usually cover only selected natural material and energy flows. Like the other material and energy flows, these natural flows are either assigned to standard impact categories or impact categories that do not yet belong to the standard impact category set but are necessary to cover environmental impacts relevant for biomass production. This common approach, however, is inadequate for biomass-based systems, since their natural production factors like soil, climate, atmosphere and water are also directly affected by global warming, eutrophication, acidification, management practices, etc. To give a more complete picture, this kind of feedback needs to be quantified. Due to missing data as well as difficulties in quantifying the feedback loops, this has not yet been done in a LCA study.

The Life Cycle Inventory (LCI) is the second step of a LCA study. For each process within the system boundary, this starts with the collection of relevant data necessary for the quantification of inputs and outputs. With respect to biomass-based systems some material and energy flows need to be treated with special attention. First and foremost this applies to atmospheric CO_2 which, in photosynthesis, is transformed together with water into biomass. Since the release of this CO_2 at the end of the life of biomass is considered to have no negative global warming effect if the forest is sustainably managed, it needs to be distinguishable from CO_2 that is released when fossil fuel is combusted. The same applies to inputs and outputs of renewable fuel and energy (electricity, heat). Although neither energy use, energy efficiency nor accumulated energy demand are mandatory impact categories and the use of renewable energy is not *per se* an indicator of environmental friendliness, many LCA studies quantify one of these factors and also indicate the share of renewable energy used or generated in a system.

The selection of impact categories is often determined by the impact category sets provided with a LCA software tool. However, if environmental impacts specifically relevant to biomass-based systems are to be covered, these standard sets are not sufficient because impact categories for impacts caused by land use, land use change as well as impacts on water, biodiversity and nutrient cycles are not included. If the environmental impacts of biomass-based systems are to be assessed completely, these additional non-standard impact categories and category indicators should be part of the assessment. The problem is that there are no standard approaches on how to assess these impacts.

The **first case study** covers the complete life cycle of biomass ranging from cultivation of short rotation coppice, transport and energy generation in a Combined Heat and Power (CHP) plant. It shows that nitrogen fertilisation is the biggest environmental impact of cultivating short rotation coppice. It also shows how to allocate environmental impacts by applying allocation by exergy when biomass is converted into power and heat in a CHP plant. To identify potential advantages of producing electricity from short rotation wood, a comparison was made with fossil electricity production in Germany. Emissions from SRC electricity production are lower in all cases than average electricity production for the German grid. By using electricity from SRC, greenhouse gas emissions can be reduced by 85 to 94% compared to using standard power from the grid. Eutrophying emissions of electricity production from poplar chips contribute more to overall EP in all cases than power from the grid, but the relative contribution to overall eutrophying emissions is quite low. Photochemical ozone creation potential of electricity from wood from unfertilized poplar plantations is higher than from fertilized plantations. This is due to nitric oxide (NO) emissions from fertilizer application which counteract the impact of ozone-creating emissions like nitrogen oxides (NO_x), carbon monoxide (CO) and volatile organic compounds (VOCs).

The **second case study** compares the environmental performance of cultivating conventional agricultural crops – wheat, sugar beet, rapeseed, ley crop, maize and willow – as supply chains for energy generation in Sweden. It allows an easy comparison of the different crops, since goal and scope, system boundaries and impact categories selected are identical. The energy output:input ratio for different cultivation systems is estimated to vary between approximately 5 (rapeseed) and 24 (willow). Thus, there is a significant difference between the different cropping systems in regard to energy and area efficiency. The choice of land use used as a reference is of great importance for the climate performance of bioenergy crops. In many LCAs the biogenic emissions of carbon dioxide have most often not been included, but only biogenic emissions of nitrous oxide. It is clear that perennial crops and crop residues have a much better GHG performance than traditional annual food crops. However, these big differences could, to some extent, be reduced when the feedstock is converted into, for example, biofuels. Thus, it is crucial when evaluating the GHG performance to include the complete bioenergy system from raw material to final energy service and not just the feedstock production phase. Perennial crops and whole crop harvest of sugar beet have a much better eutrophication performance than traditional annual food crops.

The assessment of environmental impacts due to land use and land use change, the use of water, as well as changes in carbon stocks, impacts on soils and biodiversity is still not standard in LCA studies today. This is due to the complexity of the interrelations between production systems and the natural and technical environment as well as the lack of information, which do not allow for easy development of standardized impact categories. However, in order to fully understand the environmental impacts of biomass based production systems, these ‘non-standard’ impacts should be assessed as comprehensively as possible.

Assessing **water use** within LCA is a complex task. There are complex interactions between land use, water cycle and ecosystem function, which can make it difficult to define single cause-and-effect relationships precisely. The methods currently available focus on one special impact category, on special water cycle parameters or on the assessment of special products and they often cannot be combined with each other. Often it is impossible to use one of the methods for a different assessment purpose than it was originally designed for. Sometimes special models are needed which complicates their application to regular life cycle assessment. Due to the strong local and regional dimension of water use, additional data must be collected during the inventory stage. But regional information is not always available. The methods available are in many cases designed for one special application which makes it complicated to compare different product systems such as bioenergy and fossil energy. In addition, the need for comprehensive data makes it difficult in many cases for ordinary LCA practitioners to use the current methods.

Many problems related to productivity of crops, including biomass crops for energy are **soils**-based and include low fertility, physical limitations (e.g. parent material, texture, depth, drainage, moisture content, etc.), chemical restrictions (e.g. cation exchange capacity, alkalinity, acidity, carbon content, etc.). Related problems include slope, soil erosion, compaction, and leaching. Soil quality also co-determines the impact of low water or nutrient availability. How soils can be included in LCAs depends to a large extent how the effects of changes in soil characteristics during bioenergy feedstock production are known, quantified and attributed. While the relevance of soils and their coverage in LCA studies is clear, quantification and allocation of impacts are very difficult to determine. Soil organic carbon (SOC) was used as an indicator of soil quality, and potential changes to SOC linked to different land uses were compiled for the complete life cycle of the products assessed. The results showed that, contrary to assumptions in several other life cycle impact assessment methods, life cycle stages other than cropping may dominate the impacts related to land use, even if cropping still dominates in terms of area per year.

Understanding the global **carbon cycle** and how it is affected by activities associated with bioenergy is important in reviewing the climate change mitigation benefits of bioenergy systems. There are also many challenges for bioenergy LCAs related to how carbon flows are characterized. Similar to LCAs of biofuel production processes that create multiple products, LCAs of bioenergy systems such as those mentioned above face special challenges since they need to consider how the influence on carbon pools and fluxes can be factored in. For instance, when bioenergy systems are part of cascading biomass cycles in which co-products and biomaterials themselves are used for energy after their useful life, space and time aspects need to be considered since GHG emissions and other environmental effects can be distributed over long time periods and take place at different geographical locations. The extraction and use of biomass for energy as part of long-rotation forestry systems represents a specific case in which the dynamics of terrestrial carbon stocks become a challenge for LCA. The temporal imbalance of carbon dynamics is substantially different for bioenergy use on the one hand and decomposition/re-growth processes in the forest ecosystem on the other hand. Depending on system definition, including spatial and temporal scales, and characterization of baseline, LCA results can differ greatly. The net amount of carbon emissions will depend on fossil carbon displacement efficiency and the length of time for the forest regrowth to compensate for the biomass extraction – or, in the case of forest residue extraction where the alternative (reference) situation is to leave the residues in the forest, the residue decay rate.

Land use (as in land occupation) is often included as an inventory flow or impact category in bioenergy LCA studies. However, the integration in LCA procedures of Land Use Change (LUC) and the related impacts has been cumbersome. First, with current data, assessment methods and review procedures, it is difficult to include LUC impacts in LCAs in a timely and cost-efficient way and so that sufficient quality of results can be guaranteed. Second, the complex character of land use change requires multi-disciplinary analytical tools that can evaluate dynamic, multi-scale processes. Assessing land use change from biomass production for energy requires that sufficient information is available on the conditions under which biofuel feedstock is produced. If we want to determine the impact of direct LUC, we need to know (i) which feedstock (e.g. maize grains or palm oil) has been used to produce the biofuel, (ii) where these crops have been grown, and (iii) how the introduction of the biofuel crop has affected land and soil organic carbon. Quantifying the magnitude and location of indirect LUC is even more complicated. In addition to the information we need to determine the impact of direct LUC, we also need to know (iv) what crops, if any, were previously grown on the land where the biofuel crop was cultivated, and (v) how this is affecting land use and land cover. According to the ILCD handbook, LCAs should assess impacts of LUC by means of modelling. It is recommended to use IPCC emission factors for modelling the impacts on CO_2 emissions of changes in soil organic matter from direct land use change. Other GHG impacts of land use (e.g. from burning of litter, soil erosion, nutrient losses) should also be quantified. Including indirect LUC requires that economic or causal descriptive models are integrated into a consequential LCA. However, LCAs and economic models usually have different aims. While LCA is mostly applied to assess a specific production system, product or service, economic models study changes on a global level after which impacts are allocated to single products. Consequently, the two approaches make use of data with different spatial and temporal system boundaries.

The expansive nature of this definition makes **biodiversity** very difficult to quantify directly, so indirect indicators are often used, especially in LCA. These indicators often focus on conditions thought to be important for biodiversity. Many features of ecosystems can be used as the basis for biodiversity indicators, such as a structural component, a process, or any other feature of the system related to the maintenance or restoration of its diversity. Biodiversity can be considered at three different levels: ecological diversity (ecosystems), population diversity (species), and genetic diversity (genes). All these levels have been considered in different LCIA approaches. Additionally, current approaches address three groups of relevant environmental interventions: (1) resource-related (land and water use), (2) pollution-related (acidification, eutrophication, ecotoxicity), and (3) climate change. Still, the major outstanding questions related to the quantification of land use impacts in LCA, including those related to biodiversity, is how to combine generic impact assessment with site-specific assessment. There are also limitations in estimating impacts of land transformation: (1) it must be assumed that land use impacts are reversible in the broad sense and the regeneration time must be determined to estimate transformation impacts, and (2) more accurate and regionalized data for each specific pathway are required. With respect to biomass, differences between biomass production systems used in forests and agricultural systems illustrate the challenges of using a consistent approach for addressing biodiversity in LCA. Despite the difficulties, biodiversity is a key aspect that should be incorporated into life-cycle approaches to reduce the risk of environmental burden shifting across impact categories or across life-cycle stages. Biodiversity should be reflected in the broad suite of indicators assessed within LCA. Site-specific and/or territorial assessment approaches such as EIA are also an essential complementary tool when LCA is applied in the context of biodiversity and can be used to mitigate against inaccurate conclusions.

TABLE OF CONTENTS

EXECUTIVE SUMMARY 3

Table of Contents. 7

1 preface 11

2 LCA methodology and biomass – not the usual story 12

2.1 Goal and Scope 12

2.1.1 Attributional or consequential LCA – a fundamental decision 12

2.1.2 Functional unit 13

2.1.3 System Boundary 14

2.1.4 Cut-Off Criteria 20

2.1.5 Selection of Impact Criteria 20

2.1.6 Allocation Rules 21

2.1.7 Data Requirements 23

2.2 Life Cycle Inventory 23

2.3 Life Cycle Impact Assessment (LCIA) 24

2.3.1 Selection of Impact Categories and Category Indicators 24

2.3.2 Classification 25

2.3.3 Characterization 25

2.3.4 Normalization 26

2.3.5 Grouping 26

2.3.6 Weighting 26

2.4 Life Cycle Interpretation 26

2.5 References 27

3 Case studies 29

3.1 Life cycle assessment of biomass production in short rotation 29

coppice and its use in Germany 29

3.1.1 Specific challenges associated with wood, especially SRC 29

3.1.2 System boundaries 30

3.1.2.1 Functional unit and allocation *32*

3.1.3 Results *32*

3.1.3.1 Energy use *32*

3.1.3.2 Impact assessment results *33*

3.1.3.3 Savings of GHG emissions compared to fossil energy *36*

3.1.3.4 Impacts of land use change *37*

3.1.4 Discussion and conclusions *37*

3.1.5 References *38*

3.2 Bioenergy supply chains in Swedish agriculture 41

3.2.1 Energy and area efficiency *41*

3.2.2 Greenhouse gas performance *42*

3.2.3 Eutrophication performance *45*

3.2.4 Conclusions and discussion *47*

3.2.5 References *48*

4 Assessment of 'non-standard' environmental impacts 50

4.1 Overview of first attempts to integrate impacts of water use 50

into Life Cycle Assessment 50

4.1.1 Background and scope *50*

4.1.2 Difficulties in water use assessment *50*

4.1.3 Requirements for a method for biological production processes *51*

4.1.3.1 Recently developed water assessment methods *51*

4.1.3.2 Impact categories and characterization factors *52*

4.1.3.3 Water footprint *54*

4.1.4 Discussion *54*

4.1.5 References *55*

4.2 Soils 58

4.2.1 Introduction *58*

4.2.1.1 Background: soils around the world *58*

4.2.1.2 The Soil Management Link to Sustainability *61*

4.2.2 Relevance of soils in bioenergy-related LCA studies *62*

4.2.3 Current LCA approaches to soils.. *63*

4.2.3.1 How are soils normally treated in LCA studies? *63*

4.2.3.2 Examples of LCA studies discussing soil quality changes....................... *64*

4.2.3.3 Regional approaches to soil sustainability *65*

4.2.3.4 Observations on soil monitoring approaches..................................... *70*

4.2.4 Discussion.. *70*

4.2.4.1 Difficulties associated with improving assessment of soils in LCA studies..*70*

4.2.4.2 What is the way forward? *71*

4.2.5 Conclusions... *72*

4.2.6 References *73*

4.3 Bioenergy and the carbon cycle 78

4.3.1 Principal carbon pools and fluxes *78*

4.3.2 Influence of bioenergy on carbon stocks and flows................................ *79*

4.3.3 The case of bioenergy in long rotation forestry *81*

4.3.4 Biospheric carbon sinks as alternative to bioenergy *84*

4.3.5 References *86*

4.4 Land use and land use change 91

4.4.1 Land cover and land use *91*

4.4.2 Changes in land cover and land use *93*

4.4.3 Approaches to quantifying land use change *95*

4.4.4 Impacts of land use change.. *96*

4.4.5 Site dependency.. *98*

4.4.6 Integration of LUC into LCA *99*

4.4.7 Conclusions.. *101*

4.4.8 References *102*

4.5 Biodiversity Assessment within LCA of Biomass Harvesting 106

4.5.1 Introduction 106

4.5.2 Biodiversity in the Context of LCIA 106

4.5.3 Indicators of Biodiversity Impacts in LCA 108

4.5.3.1 Impacts of Biodiveristy on Land Use 109

4.5.3.2 Biodiversity of Impacts Related to Water Use 111

4.5.3.3 Impacts of Pollutant Emissions on Biodiversity 112

4.5.3.4 Impacts of Climate Change on Biodiversity 112

4.5.4 Challenges Associated with the Evaluation of Biodiversity Impacts in LCA . 113

4.5.4.1 Challenges Related to LCA Framework 113

4.5.4.2 Challenges Related to the Intrinsic Complexity of Biodiversity 113

4.5.5 Implications for LCAs that Involve Biomass Production Systems 115

4.5.6 Conclusions 117

4.5.7 References 118

1 PREFACE

The on-going public debate on sustainability of bioenergy began in 2007 and 2008 after a couple of papers were published questioning the reported N_2O emissions caused by cultivation of agricultural crops (Crutzen *et al.*, 2007) and the carbon neutrality of biomass use (Searchinger *et al.*, 2008) due to direct and indirect land use change as well as change of carbon stocks. Since these initial papers, many others have been published mainly questioning the positive climate impact of bioenergy. The impact of these papers changed the attitude towards use of bioenergy in the public and political arena. The European Commission, for example, released the Renewable Energies Directive 2009/28/EC (European Parliament, 2009) with sustainability criteria for liquid biofuels, prohibiting the marketing of liquid biofuels that exceed predefined carbon emission saving thresholds and that originate from certain types of ecosystem. Criteria for solid biofuels are currently under development. It is important to mention that these sustainability criteria are valid for biofuel imports into the European Union (EU) as well as for biofuels produced within the EU. Hence, the directive has a global impact.

The current discussion about sustainability of biomass in the on-going public debate and political arena is mainly about environmental performance and more specifically climate impact of biomass cultivation and use. But is impact on the climate all that matters? Are other potential impacts on water, soils or biodiversity negligible?

The authors of this publication don't think so. This publication aims to broaden the debate, raise awareness and provide state of the art information on environmental impacts of biomass supply chains, how to assess them and what the challenges and limitations are.

To assess the environmental performance of biomass supply chains, or any other product or service, Life Cycle Assessment (LCA) is a widely accepted tool. It provides quantitative information on a number of different environmental impacts. However, since LCA was originally designed to assess technical processes, applying it to biological systems is not straightforward. Hence, this publication first gives an introduction to the application of LCA to biomass production. It gives recommendations on what should be taken into account in order to assess the environmental impacts of biomass supply chains as thoroughly as possible.

Two LCA case studies, one on agricultural biomass production in Sweden and one on short rotation coppice for energy use in Germany, show the application of LCA to biomass supply chains. Although both case studies reflect the state of the art they also show the limitations of LCA in assessing environmental impacts of biological production systems. This is why this publication puts special emphasis on those environmental impacts that are not yet assessed within LCA studies on a regular basis. State of the art information on how those impacts are currently assessed is provided in chapters dealing with impacts on water, soils, the carbon cycle, land use change and biodiversity.

The chapters of this publication are thematically linked but self-contained. Each chapter discusses a certain aspect of the assessment of environmental impacts of biomass supply chains.

Finally, as the coordinator of this publication, I would like to thank IEA Bioenergy for its financial support as well as all my IEA Bioenergy Task 43 colleagues and authors outside Task 43 for their efforts to make it happen. Special thanks go to Jim Richardson for proofreading and valuable comments.

Jörg Schweinle
(National Team Leader Germany - IEA Bioenergy Task 43)

2 LCA METHODOLOGY AND BIOMASS – NOT THE USUAL STORY

by Jörg Schweinle

LCA methodology was codified in ISO standards 14040 (ISO, 2006a) and 14044 (ISO, 2006b) in 2006 and since then has been described in a large number of publications. The following section provides summary information on LCA methodology with supplements important for the assessment of biological systems and biomass supply chains.

2.1 Goal and Scope

According to the ISO standard 14040 (ISO, 2006a) each LCA requires a thorough description of its goal and scope. This is to define the framework of the study, its intended application and intended target audience. Following a decision about the purpose of the study, a fundamental decision that needs to be made is regarding the type of LCA which may be either attributional or consequential. If the study assesses direct impacts of a process or product life cycle, an attributional LCA is the method of choice. If the impacts caused by a marginal alteration of a process, product life cycle or a policy are to be assessed, a consequential LCA is more appropriate. In addition to this fundamental decision, the relevant settings of the study need to be defined. These are:

- the functional unit,
- the system boundaries,
- cut-off criteria,
- the impact categories,
- the allocation procedures, and
- data requirements.

Before having a closer look at relevant settings with respect to biomass supply chains, first some words on the major differences between attributional and consequential LCA.

2.1.1 Attributional or consequential LCA – a fundamental decision

As mentioned, the goal and scope of the study determine which type of LCA is appropriate to address the question answered. However, to make this decision, the limits and opportunities of attributional and consequential LCA need to be known beforehand. The following table gives a brief overview of the limits and opportunities associated with each method.

Additional and more detailed explanations of the principal differences between attributional and consequential LCA are given in the following sections describing the structure of a LCA. An outline of the historical development of consequential LCA, summary of methodological advancements and literature review is provided by Earles and Halog (2011).

Once the decision has been taken regarding which type of LCA is most appropriate to meet the goal and scope of the study, the relevant settings need to be defined. Usually, the functional unit is defined first.

Table 0.1: Limits and opportunities of attributional and consequential LCA (Source: based on Brander, M. *et al.*, 2008)

	Attributional LCA	Consequential LCA
Basic goal and scope	Assessment of the environmental impacts of a process, product life cycle or service.	Assessment of the change of environmental impacts due to a marginal change of a process, product life cycle, service due to a policy, or buying decision.
System boundary (technical)	All relevant processes and associated energy and material flows.	All relevant processes and associated energy and material flows affected by the marginal change.
System boundary (temporal)	Assessment of environmental impacts of a process, product or service at a given period of time.	Assessment of the change associated with the marginal change of a process, product or service. Timeframe of the change needs to be defined.
Allocation procedure	Allocation of environmental impacts to co-products based on their mass, energy content or economic value.	System expansion instead of allocation to quantify the environmental impacts of co-products.
Data requirements	Site, process, product-specific or average data.	Marginal data[1].
Indirect effects	Indirect effects of the system studied are not considered.	Indirect effects like indirect land use change due to additional use of biofuel are considered.
Economic effects	Economic effects associated with use of products are not considered.	Economic effects associated with the use of products are considered.
Uncertainty	Impact assessment is based on empirical data regarding production, use and disposal of products.	Impact assessment relies on complex models that are built on empirical data as well as assumptions regarding the development of markets and societies.

2.1.2 Functional unit

In either type of LCA, the functional unit is the reference to which all input or output data as well as impact category values are related. It is usually defined by a reference flow. A comparison of systems is based on the same function(s) quantified by the functional unit(s)

[1] For more information on the different meanings of 'marginal' with respect to LCA modeling please refer to Guinée (ed.), (2002), p. 421 ff)

[2] Exergy is the energy available for work. The exergy content of 1 MJ of electricity is 1, since it is completely convertible into work. The exergy content of steam or any other thermal energy carrier, however, depends on

and their reference flow(s). Although this may sound clear, the selection of a proper functional unit can be difficult. In particular, a comparison of two systems requires a clear understanding of their functions. Consider two systems generating energy: one generates electricity, the other steam. A comparison based on the functional unit of MJ of energy produced is not possible, however, since 1 MJ electricity and 1 MJ steam do not have the same function. Their exergy content is different[2]. One option to compare the two systems would be to define a functional unit where the generated energy is a medium (reference flow) necessary to provide a service (function), e.g. the heating of a room.

If a study aims to generate a generic data set for the product or process studied, all material and energy flows should be related to a physical reference flow. This allows for an easy transformation and use of data in the context of other studies with different goals. A comparison of systems based on reference flows expressed in physical units like mass, volume or energy, however, should be critically verified. The same is true if systems are compared by their products. The following example might help to illustrate this. Assume two chairs, one made of wood and the other one made of steel, are the functional unit. At first sight both chairs have the same functional unit: one can sit on them. But on closer examination of their technical properties, one chair is found to have a carrying capacity of 100 kg and the other one 150 kg. Hence, their functions are not equal. In order to avoid the selection of unequal functional units, the function always needs to be defined as precisely as possible.

2.1.3 System Boundary

The Life Cycle Assessment (LCA) of any process, product life cycle or service is comprehensible only if it is precisely defined and clearly delimited. This delimitation is called the system boundary. The definition of the boundary is a normative setting, justified by the goal and scope. The assessment of the environmental impacts caused by a process, product life cycle, service or their marginal change is based on the material and energy flows quantified within the system boundary as well as crossing it.

Due to their specific concepts, the system boundaries of attributional or consequential LCA are *a priori* differently shaped regardless of the system under study. Whereas the system boundary of an attributional LCA can be either narrow or wide, it is always wide for a consequential LCA, since indirect effects associated with a marginal change of the system are assessed. Figure 2.1 shows the principal differences between the system boundaries of an attributional (left) and a consequential (right) LCA.

The widening of the system boundary is called system expansion. In a consequential LCA, the system boundary is expanded so that, in principle, all indirect effects are covered (Ekvall and Weidema, 2004). For example, in order to cover the indirect effects of biofuel consumption in Europe, the system boundary would include not only Europe but also the

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[2] Exergy is the energy available for work. The exergy content of 1 MJ of electricity is 1, since it is completely convertible into work. The exergy content of steam or any other thermal energy carrier, however, depends on the relation between the temperature at which the thermal energy is available and the surrounding temperature. This difference is also referred to as entropy. On earth, the exergy content of heat is always smaller than 1. To determine the exergy content of thermal energy the Carnot Factor is calculated: $\eta_c = \frac{T - T_u}{T}$

η_c = Carnot factor; T = steam temperature (K); T_u = surrounding temperature (K). The case study on the energetic use of short rotation coppice in Germany later in this publication (Chapter 3.1) shows the application of the Carnot Factor for allocation purposes.

places outside Europe where indirect effects occur. However, to keep a consequential LCA feasible, the number and degree of indirect effects assessed are usually limited (Kløverpris *et al.*, 2008).

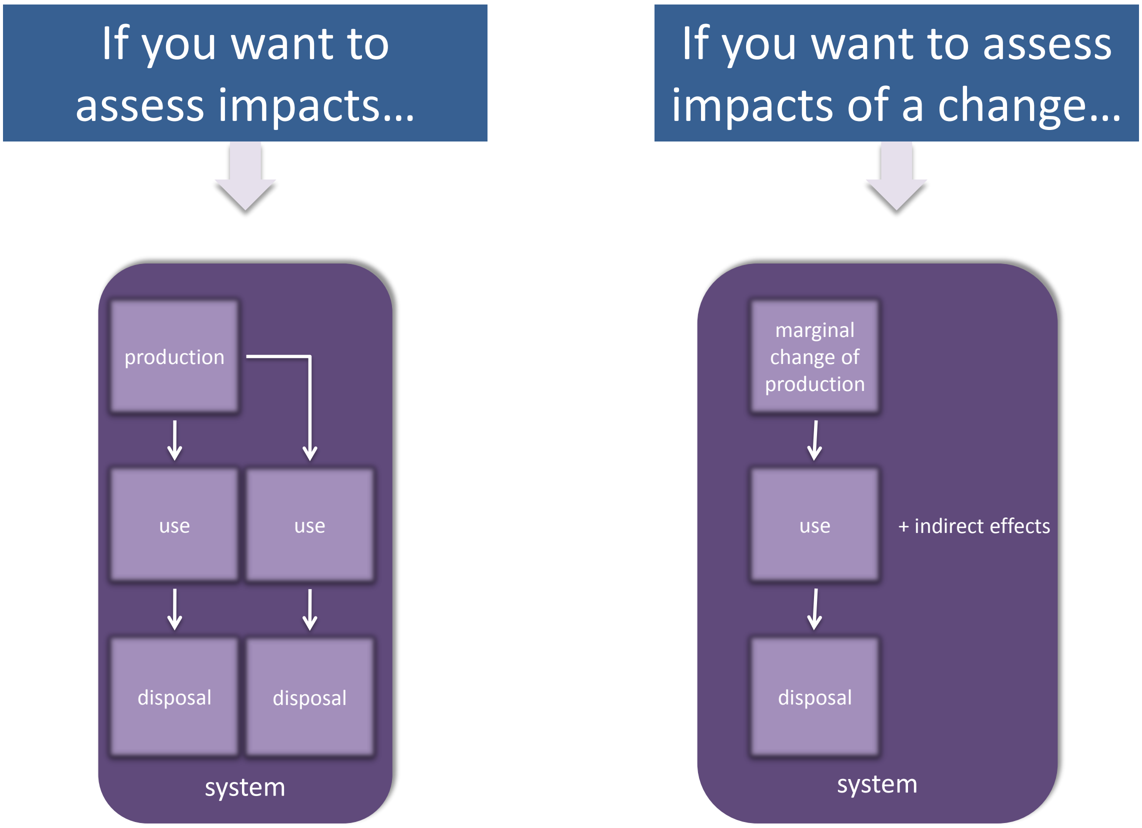

Figure 2.1: Principal differences between symbol boundaries of an attributional and a consequential LCA.

As noted above, the system boundary of an attributional LCA can be either narrow or wide, depending on the goal and scope of the study. A narrow system boundary usually covers a product life cycle. The environmental impacts of co-products are allocated (see 2.1.6 Allocation Rules). To avoid the need for allocation in attributional LCA, the ISO 14040 standard recommends system expansion -- the system is not widened to cover indirect or marginal effects, but to cover the life cycle of the main product as well as the co-product(s). The system boundary around the attributional life cycle of sugarcane-based ethanol, for example, would be expanded to cover energy generation from the co-product bagasse.

Regardless of whether attributional or consequential LCA is used, the system boundary always has three dimensions:

- spatial,
- technical, and
- temporal.

The spatial boundary describes the geographic area in which a system is located. The data collected must be valid for this specific geographic area. If the validity of the data extends beyond that area, this should be stated in the boundary description. Contrary to industrial production systems, the cultivation of biomass is site-specific. Soil, climate, water regime, altitude, latitude, landform, etc. can impact biomass yield as much as or more than management activities. Hence, a thorough and in-depth assessment of biomass production needs to be site-specific and the spatial system boundary needs to be narrow. But what if the goal is to assess a generic biomass supply chain typical for a certain region or to assess marginal change to a biomass supply chain? In that case, the spatial boundary is wide and average or marginal data are used. That implies, however, that based on average data the assessment of site-specific impacts on, for example, water, biodiversity and nutrient cycles is rather uncertain at the site level. This is important to have in mind when LCA results are scaled up or used to define thresholds.

The technical boundary indicates which technologies are applied in a system and its processes. The technological processes, the machinery, materials, energy, etc. should be described as exactly as possible to define the technical boundary. The technical processes and machinery must match the spatial boundary. For example, using data for machinery built for flat terrain in a study on timber harvesting on steep slopes makes no sense. Although this should be self-evident, the careless use of LCA databases is often a cause of mistakes. Therefore, careful selection of appropriate and adequate data sets is a prerequisite for a reliable assessment. In general, a combination of site specific and average data is not desirable. However, in practice this is almost impossible to avoid since data for supply chains for fuel, electricity or raw materials are usually national averages.

The temporal system boundary describes the time period for which the assessment is valid. For short term production systems like industrial processes or most agricultural systems this is not a challenge since all relevant processes take place within a short period of time, usually one year. Forest production, however, is a long term production system that takes several decades or longer. Since LCA is designed for short-term production systems and their products, it is a challenge to define an adequate temporal system boundary that covers all relevant silvicultural and technical processes necessary to produce roundwood or wood chips. A temporal system boundary that covers a full rotation period would cover both past and future (Figure 2.2). Since the spatial, temporal and technical system

Figure 2.2: Temporal system boudary of forest production covering past, present and future.

boundaries need to match, the technical boundary, for example, would need to cover both past and future technology. Whereas acquisition of empirical data for past technology might be possible, actual data for future technology is considered unavailable. To cover future technological progress, the application of learning curves might be a solution here but implies high uncertainty. A solution is needed that covers all relevant production processes with a modest degree of uncertainty. One option is to apply the normal forest model based on yield tables or forest growth models (Schweinle, 2000a). Across a normal forest all production processes that usually take place in the course of a complete rotation period take place side by side in any one year.

Thus, the temporal step by step sequence of silvicultural and technical processes taking place over a complete rotation period is transformed into a temporal and spatially side by side set of processes (Figure 2.3). If relevant information is available, the normal forest model or a forest growth model is applicable to a specific stand on a specific site as well as to the average conditions of a region or country.

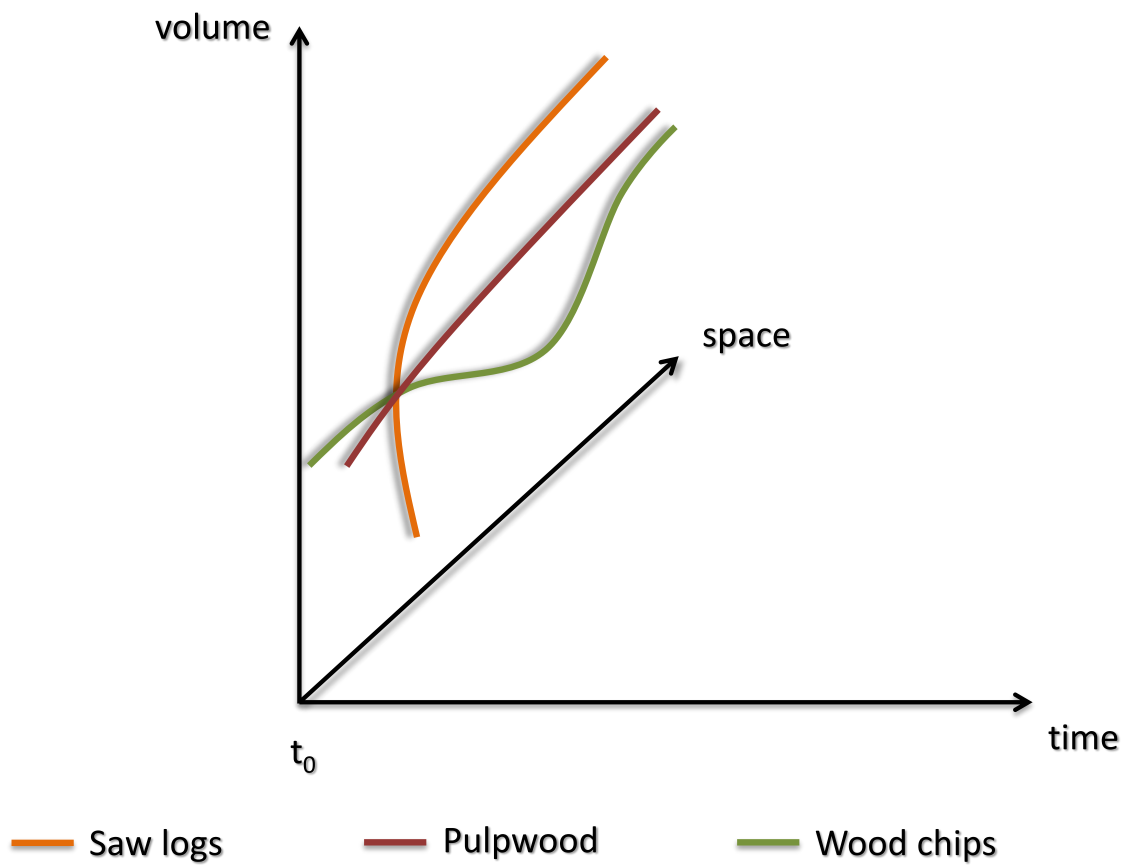

Figure 2.3: Temporal system boudary of forest production whent he normal forest model is applied.

Unlike industrial production systems, biomass supply chains occur in the so-called ‘biosphere’. In other words, the production of biomass is first and foremost based on natural production factors like sunlight, soil, water, nutrients and processes like photosynthesis and plant metabolism. Figure 2.4 shows how biomass production depends on energy and material flows from and to the different compartments of our natural and technical environment. To give a full picture of the environmental performance of biomass supply chains, these material and energy flows need to be quantified and assessed and, hence, be within the system boundary.

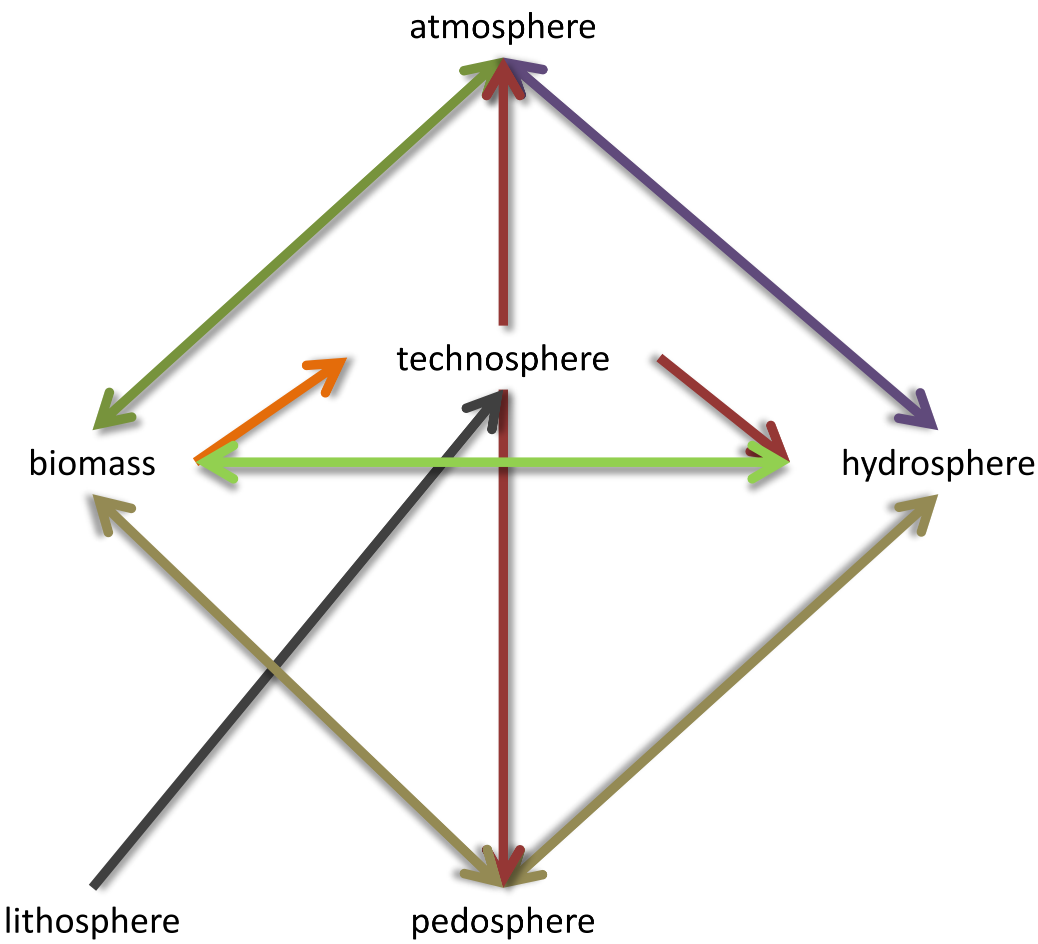

Figure 2.4: Material and energy flows between biomass production systems and the environment.

Due to complexity, immature methodology and missing data, LCAs of biomass-based products or services usually cover only selected natural material and energy flows like sunlight, CO_2, water and nutrient cycles. Like the other material and energy flows, these natural flows are either assigned to standard impact categories or impact categories like land use or biodiversity that do not yet belong to the standard impact category set but are necessary to cover environmental impacts relevant for biomass production (Schweinle, 2000b; Milá i Canals *et al.*, 2007; Saad *et al.*, 2011; Koellner *et al.*, 2013). One example is the (partial) assessment of the nitrogen cycle as illustrated in Figure 2.5. Today, this is state of the art for LCAs of agricultural products. How important it is to assess the nitrogen cycle is shown by the case study on short rotation coppice in Germany in Chapter 3.1. However, even the assessment of the nitrogen cycle still follows the common LCA approach of assessing and evaluating impacts of 'technosphere' on 'biosphere'.

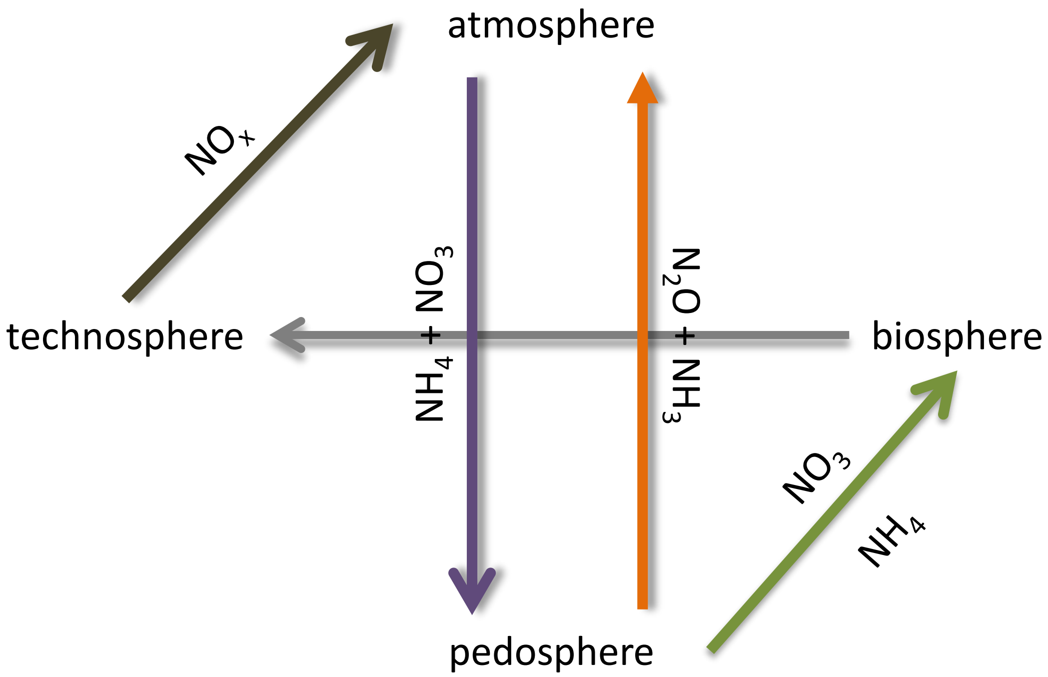

Figure 2.5: Simplified illustration of the nitrogen cycle.

This common approach, however, is inadequate for biomass-based systems, since their natural production factors like soil, climate, atmosphere and water are also directly affected by global warming, eutrophication, acidification, management practices, etc. To give a more complete picture, this kind of feedback needs to be quantified. Due to missing data as well as difficulties in quantifying the feedback loops, this has not yet been done in a LCA study. More information on state of the art assessment of land use, carbon, water, biodiversity, soil and nutrient cycles is given in Chapter 4.

2.1.4 Cut-Off Criteria

Cut-off criteria may be defined for material and energy flows that are not relevant for the outcome of a study. ISO 14044 (ISO, 2006b) mentions mass, energy and environmental significance as the three major cut-off criteria. Cut-off by mass or energy would exclude mass or energy flows that cumulatively contribute less than a certain percentage of the total mass or energy input to the system. Cut-off due to environmental significance would exclude inputs that contribute less than a certain percentage to an environmental impact. If mass is the only cut-off criterion, ISO 14044 recommends that the other criteria should be taken into account to make sure that important inputs are not omitted. Furthermore, ISO 14044 requires describing and evaluating the impacts of cut-off criteria on the results.

2.1.5 Selection of Impact Criteria

The selection of impact categories, category indicators and characterization models must be in compliance with the goal and scope of the study and should ensure a thorough assessment of all relevant environmental impacts associated with the process, product or service analyzed (ISO, 2006b). In practice, the selection of impact categories is often based on the impact assessment methods available in any given LCA software tool. Standard methods like Ecoindicator, CML or ReCiPe (Goedkopp *et al.*, 2013) may be suitable to cover the most relevant environmental impacts of industrial processes, but they are insufficient for biomass production and subsequent supply chains, because the standard methods do not cover environmental impacts that typically occur with cultivation of agricultural crops or forestry. These are impacts caused by land use and land use change as well as impacts on water, soil, nutrient cycles and biodiversity. It appears that impact categories suitable for assessing these impacts are neither generally accepted nor part of standard impact assessment methods. The state of the art of assessing impacts specific to biomass production is presented in later chapters.

2.1.6 Allocation Rules

Many production systems are multi-product systems. In LCA, two options exist to handle the allocation of inputs and outputs to the products of multi-product systems. First, ISO 14044 (ISO, 2006b) requires that allocation must be avoided wherever possible by separating processes into sub-processes or by system expansion. A typical example for a biomass based multi-product system is the production of ethanol and its co-product DDGS (Figure 2.6). Since ethanol is usually the reference flow of the functional unit, subdivision of production processes, if possible, could allow for allocating inputs and outputs directly to ethanol and to the co-product DDGS. If subdivision is not possible because the production processes are not divisible, system expansion by including livestock feed with DDGS would be the second recommended option to avoid allocation.

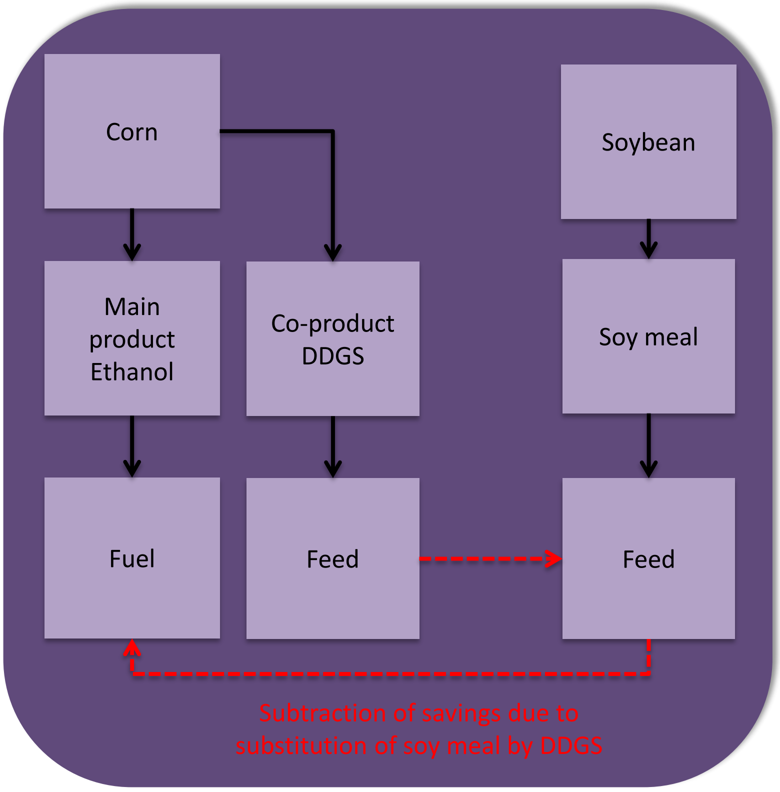

Figure 2.5: System expansion by adding the equivalent process and subtraction of the savings due to substitution of soy meal by DDGS.

Where it is impossible to avoid allocation, ISO 14044 recommends allocation by physical or economic relationships. In our example, this means the inputs and outputs are either allocated to ethanol and DDGS according to their mass portion or according to their economic value. The third option, allocation according to the energy content or exergy of ethanol and DDGS is only theoretical since DDGS, although it has a calorific value, has a different function; it is used as a livestock feed. The option to choose depends on the goal and scope of a study. However, there is often an argument as to whether physical or economic relationships are the 'correct' way to allocate. For generic data sets of co-product systems representative for a certain region, time and state of technology, physical relationships seem to be the first choice. They are easily adaptable and hence transferable and usable in other studies. Allocation by economic relationship is an option reflecting the main economic purpose of a system. It is based on the concept that each production process is not an end in itself but an expression of economic action. In the above example, the main economic purpose is to produce ethanol. Hence, ethanol is the reason for environmental impacts and this is why the impacts should be allocated by the degree of appreciation for ethanol. The degree of appreciation can be expressed as the market price or revenue (Figure 2.7).

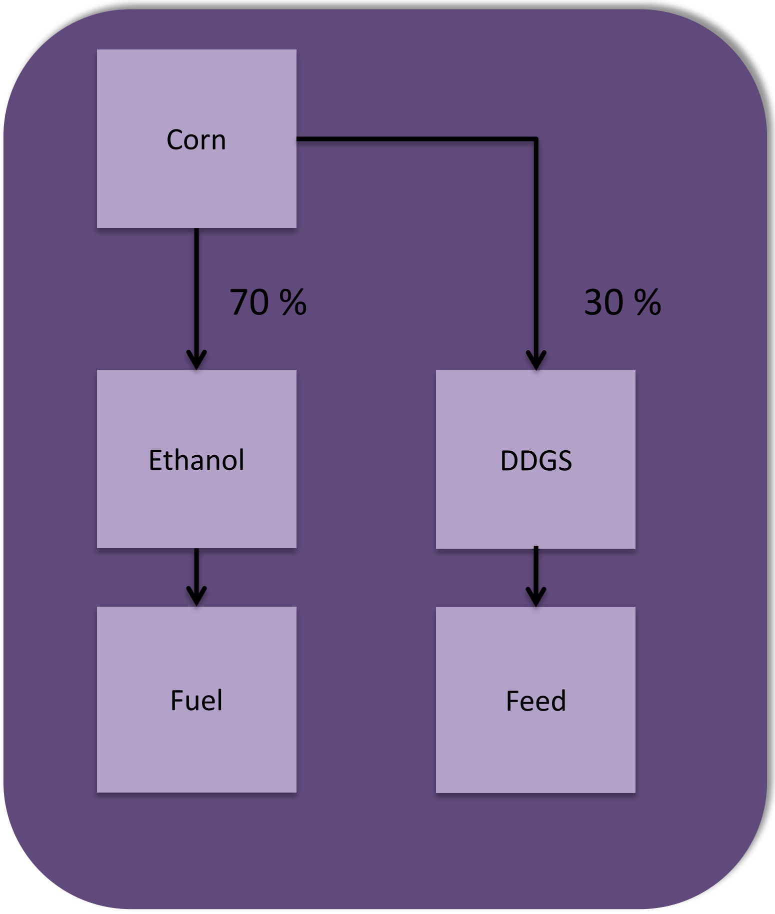

Figure 2.7: Allocation by physical or economic relationship.

2.1.7 Data Requirements

Data requirements very much depend on the goal and scope of a LCA. An attributional LCA aiming to assess a certain product should use, to the extent possible, measured data reflecting production, use, recycling and disposal of this product. If measured data are not available, averages representative for the region where the different life cycle stages of the product are located should be used. For many products and their life cycle stages, open source as well as commercial data are available. The same applies to data for standard supply chain processes such as energy generation and distribution, mining, processing and transport of standard raw materials.

Data requirements for a consequential LCA differ from those for an attributional LCA in two respects. First, the amount of data necessary to conduct a consequential LCA usually is greater since the system boundary is quite wide and the number of processes to be assessed is higher compared to an attributional LCA. Second, a consequential LCA requires data that are able to quantify marginal changes. Hence, data collection might be much more demanding since data on marginal changes in a product life cycle are not readily available in LCA data bases. However, depending on the temporal system boundary, 'marginal' has different meanings. The longer the time perspective, then the wider the temporal system boundary is set, and the more long-term marginal effects need to be covered. This needs to be reflected in the data used. For short-term marginal effects that occur if, for example, an additional unit of a product is produced, data on extra material or energy use as well as the resulting environmental impacts are needed. If the temporal system boundary is wider, data reflecting long-term marginal effects like the extra fabrication capacity and work force needed to produce the extra material are needed as well. Guinée (ed.) (2002, p. 421ff) discusses quite extensively the implications related to average and marginal processes and process data.

2.2 Life Cycle Inventory

The Life Cycle Inventory (LCI) is the second step of a LCA study. For each process within the system boundary, this starts with the collection of relevant data necessary for the quantification of inputs and outputs. To ensure transparency and reproducibility, data sources as well as additional information about data quality need to be documented. ISO 14044 (ISO, 2006b) requires that measures should be taken to achieve a uniform and consistent concept of the modeled system. Hence, a process diagram should be drawn, each process should be described, material and energy flows should be listed for each process, data collection and calculation techniques should be described, and anything special related to the data collected should be documented.

With respect to biomass-based systems some material and energy flows need to be treated with special attention. First and foremost this applies to atmospheric CO_2 which, in photosynthesis, is transformed together with water into biomass. Since the release of this CO_2 at the end of the life of biomass is considered to have no negative global warming effect if the forest is sustainably managed, it needs to be distinguishable from CO_2 that is released when fossil fuel is combusted.

The same applies to inputs and outputs of renewable fuel and energy (electricity, heat). Although neither energy use, energy efficiency nor accumulated energy demand are mandatory impact categories and the use of renewable energy is not *per se* an indicator of environmental friendliness, many LCA studies quantify one of these factors and also indicate the share of renewable energy used or generated in a system. To differentiate between energy types and collect them separately for allocation by exergy (see Allocation Rules), for example, flows of the different energy types should be listed separately.

All relevant information that is needed to calculate material and energy demand on the one side, and supply of products, co-products, generation of waste and emissions to air, water and soil on the other, must be clearly stated and explained. All inputs and outputs are calculated in relation to the reference flow. As a result of the calculations, all material and energy flows are referenced to the functional unit and arranged in input-output tables. On the input side, all energy, material and other physical inputs are listed, whereas on the output side, products, co-products, waste, and releases to air, soil and water are listed. If inputs and outputs need to be allocated, the allocation rules apply as defined in the goal and scope definition.

A Life Cycle Inventory study ends after aggregation of inputs and outputs. A complete LCA continues with the impact assessment, the third step of a standard LCA.

2.3 Life Cycle Impact Assessment (LCIA)

According to ISO 14044 (ISO, 2006b), a Life Cycle Impact Assessment has mandatory as well as optional elements. The mandatory elements are:

- selection of impact categories and category indicators,
- assignment of LCI results to the impact categories (classification),
- calculation of category indicator results (characterization).

Depending on the goal and scope there are optional elements that can be used as well:

- normalization,
- grouping,
- weighting,
- data quality analysis.

2.3.1 Selection of Impact Categories and Category Indicators

As mentioned in section 2.1.5, the selection of impact categories is often determined by the impact category sets provided with a LCA software tool. Usually these are the most common sets such as Eco-indicator 99 (Goedkopp and Spriensma, 2001) or the CML (Guinée (ed.), 2002) indicator set. However, if environmental impacts specifically relevant to biomass-based systems are to be covered, these standard sets are not sufficient because impact categories for impacts caused by land use, land use change as well as impacts on water, biodiversity and nutrient cycles are not included. If the environmental impacts of biomass-based systems are to be assessed completely, these additional non-standard impact categories and category indicators should be part of the assessment. The problem is that there are no standard approaches on how to assess these impacts. To cover these impacts one could define new impact categories and impact indicators or rely on already published approaches. In both cases, ISO 14044 (ISO, 2006b) requires that the underlying environmental mechanisms are empirically sound, understood and reproducible. Chapter 4.1 provides an overview of the latest methodological developments and suggests how to assess land use, land use change, water, biodiversity and nutrient cycles. ISO 14044 gives further very detailed instructions about additional considerations and how to ensure transparency and scientific validity.

2.3.2 Classification

The assignment of LCI results to impact categories is called classification. Generally, emissions to air, water and soil are assigned to impact categories like global warming, acidification, eutrophication, ozone depletion, summer smog, etc. With biomass-based systems special attention should be given to material and energy flows that are not considered to have an impact on the environment like the CO_2 that is converted in photosynthesis into biomass. There are two ways to handle this atmospheric CO_2: (a) it is not assigned to the global warming impact category because, compared to CO_2 emissions from the combustion of fossil fuels, it has no additional global warming effect; (b) it is assigned to the global warming impact category but is treated differently from the fossil CO_2. Chapter 4.3 discusses what these different options are and how they influence LCIA results.

2.3.3 Characterization

The calculation of indicator results is called characterization. In this process LCI results are converted into indicator values per functional unit. Methods are similar for many standard impact categories. According to their specific potential to force global warming, acidification, eutrophication or ozone depletion, emissions are multiplied by a characterization factor and then added up and converted to the total potential indicated in kg-equivalents per functional unit. Figure 2.8 shows the method for calculating the Global Warming Potential (GWP_{100}).

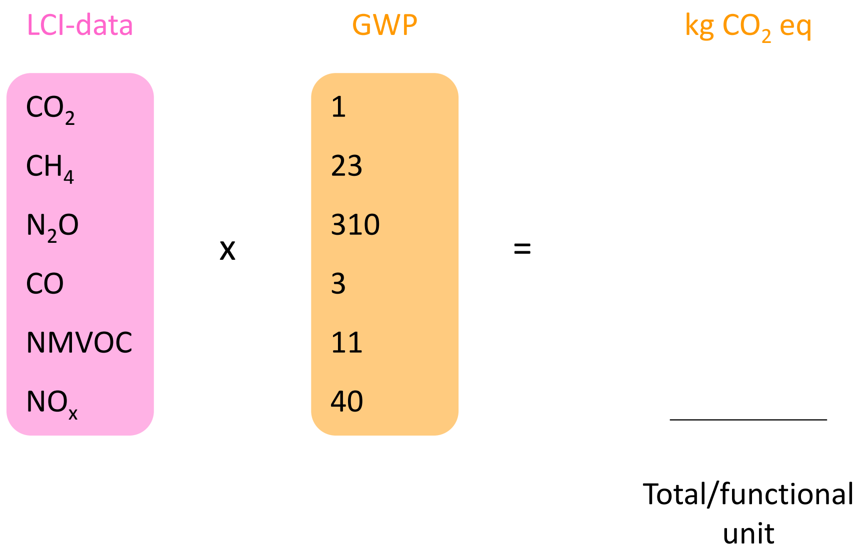

Figure 2.8: General method for calculatin impact categories (example: Global Warming Potential (GWP_{100})

2.3.4 Normalization

The normalization of category indicator results shows the magnitude of these results compared to a reference. Usually this is done by dividing indicator values by reference values. A typical reference value might be the total or total per capita emissions of a particular greenhouse gas in a certain country in a certain year or other time period. The additional information gained from normalizing indicator values could change the conclusion drawn from LCIA. However, whether the additional information - that a system's global warming potential might be 0.00005% of the per capita global warming of a country in the year 2012 – is important or not very much depends on the goal and scope of the study.

2.3.5 Grouping

Grouping of impact categories is optional and may involve sorting into scales or ranking into hierarchies. Sorting and ranking are subjective exercises and are based on values of the individuals who do the ranking and sorting. Impact categories might be ranked according to their geographic relevance or grouped according to priority. Being optional, this element of LCIA is not very often used in LCA studies.

2.3.6 Weighting

The conversion of indicator results using factors based on value choices is called weighting. Depending on the weighting procedure, indicator results of different impact categories might be aggregated to a single figure. Since weighting is purely value-driven, weighting factors must be clearly communicated. Additionally, a sensitivity analysis showing how different weighting factors and methods influence the weighting results is recommended.

2.4 Life Cycle Interpretation

Interpretation of results is the final phase of a LCA study. For biomass-based systems this phase requires no special attention. The underlying principles and procedures are valid for any kind of system and consist of three interlinked steps. First, the points that are significant for the system studied, such as the most prominent processes, inputs and outputs as well as impact categories, are identified. Implications of settings and terms defined in the 'goal and scope' phase are considered in the interpretation of results. Second, according to ISO 14044 (ISO, 2006b), the evaluation of results should consider completeness, sensitivity and consistency checks[3]. The third and final step of the evaluation is to draw conclusions and give recommendations. The conclusions should cover all aspects of the study, including goal and scope, assumptions made, data quality, restrictions and methodological issues. Due to the complexity of LCA studies and the quantity of data and information, the interpretation should clearly highlight which settings and assumptions significantly determine the results and how the results are impacted by any values-based choices made (e.g. weighting). Based on the conclusions drawn, recommendations should be given, e.g. on how to improve the environmental performance of the system under study.

[3] See ISO 14044 (ISO, 2006b) for further information on the checks that should be performed.

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3 CASE STUDIES

The following two case studies are good examples of attributional LCAs of the cultivation of short rotation coppice in Germany and conventional agricultural crops in Sweden. Both represent the state of the art with regard to goal and scope, system boundaries and impact categories selected.

The first case study covers the complete life cycle of biomass ranging from cultivation of woody biomass, transport and energy generation in a Combined Heat and Power (CHP) plant. It shows that nitrogen fertilisation is the biggest environmental impact of cultivating short rotation coppice. It also shows how to allocate environmental impacts by applying allocation by exergy when biomass is converted into power and heat in a CHP plant.

The second case study compares the environmental performance of cultivating conventional agricultural crops – wheat, sugar beet, rapeseed, ley crop, maize and willow – as supply chains for energy generation in Sweden. It allows an easy comparison of the different crops, since goal and scope, system boundaries and impact categories selected are identical.

Both studies consider one of the most critical issues of crop cultivation, the N_2O emissions caused by nitrogen fertilizer transformation in the soil (see Figure 2.5).

3.1 Life cycle assessment of biomass production in short rotation coppice and its use in Germany

by Anne Rödl

The cultivation of short rotation coppice (SRC) has been emerging in Germany over the last several years, supported by some research projects. The cultivation of trees on agricultural land, mainly for fuelwood production, combines aspects of forestry and agriculture. Compared to forests, SRC plantations need more soil preparation, require the planting of a high number of seedlings and in some cases the use of fertilizer or pesticides, and allow the use of agricultural machinery. However, they do not require work and material input every year, leading to reduced soil impact compared to agricultural crops. Growing fast-growing woody species for energy use may become more important in future. In the present study, selected environmental impacts of SRC wood production and its conversion to heat and power are assessed.

3.1.1 Specific challenges associated with wood, especially SRC

The assessment of bioenergy from SRC has some specific characteristics. First, there is the assumption of carbon neutrality of wood which has lately been under discussion. Schulze *et al.* (2012) questioned the common practice of not counting the release of CO_2 from wood as contributing to global warming potential (GWP). They state that bioenergy production from wood might not be GHG-neutral because biomass pools are lowered and carbon emission from soils could occur. These problems might emerge if wood is taken from forests that are not sustainably managed. In the present study, however, where wood is purpose-grown in short rotation plantations this issue is not relevant. Carbon dioxide released during burning of the wood was absorbed from the atmosphere during the years it was growing in plantation. No additional carbon is released.

Further, Whitman and Lehmann (2011) argued that GWP is systematically over- or underestimated if biomass systems are considered to be carbon-dioxide-neutral but other carbon-based GHG emissions are fully counted. They recommend using corrected GWP factors for methane (CH_4) and carbon monoxide (CO) to reflect the neutrality of their carbon atoms if the biomass is judged to be sustainable. For this assessment, regular CML2001 characterization factors (Guinée, 2002) for CH_4 are used. Carbon monoxide (CO) is not counted towards GWP in the CML2001 characterization method. As stated by the authors, CH_4 emissions in industrial combustion systems are quite low because incomplete combustion is avoided by high combustion temperatures. Therefore carbon dioxide which was fixed during the growing of wood is not included towards GWP but the few hydrocarbons which are released are included as usual. There may therefore be slight but negligible overestimations of GWP from methane.

The basic assessment in this study does not include carbon storage or release from soils during SRC cultivation. Nor does it include further impacts of direct land use on biodiversity or water, on which Section 3.1.3.4 provides some further information.

Further challenges occur due to the varying characteristics of wood related to its moisture content. Water included in the wood has to be considered in calculating transport weights, volume expansion and shrinking and heating values. In addition, LCA modeling and its results are mainly based on plantation growth over time and this is not very well documented yet. In particular, there is little experience with growth in the third or fourth rotation cycle.

3.1.2 System boundaries

In this study, the complete biomass production system is considered from soil preparation to biomass harvest, as well as drying, transportation and conversion of biomass to power and heat. Poplar cuttings are planted on a ploughed and harrowed field after spraying with a glyphosate herbicide. One mechanical weeding in the first year is included. In the basic scenario, fertilizing is not included in the model, since some studies (Kauter *et al*., 2001; Boelke, 2006; Knust, 2007) have shown that poplar growth does not respond to fertilizer application, particularly on nutrient-rich former agricultural sites. The pool of nutrients lasts for a long time because removal of nutrients in harvested biomass is low in SRC. However, fertilizer effects are different on poor soils (Kauter *et al*., 2003). On nutrient-deficient sites, poplar growth responds to additional nutrient applications. Differences in environmental impacts between fertilized and unfertilized SRC are analyzed in Section 3.1.3. Harvesting is carried out every four years using a forage harvester with a wood-cutting attachment. In one rotation, the coppice is harvested 5-times and after 20 years the stools are removed to reconvert the field into arable land. All equipment and operational steps used in poplar SRC are shown in Table 3.1.

Freshly harvested poplar chips contain 50% water. After drying the wood chips in piles at the edge of the field during summer time, the water content is reduced from 50% to 25%. Dry mass losses of 3% are assumed due to decay and losses during the removal of the piles. In drying using ‘aeration technology’ wood chips are piled in heaps with exhaust pipes inserted. Drying is driven by self-heating and temperature differences between the inside and outside of the piles. Excess moisture can escape through the pipes (Brummack, 2010). After drying, the wood chips are transported 50 km by truck to the power plant.

Table 3.1: Equipment and fossil energy used in each operational step within the poplar SRC system boundaries.

Operation	Step	Equipment	Fossil energy use [MJ ha^{-1}]		Comments
			unfertilized	fertilized	
Plantation establishment	Soil preparation	Tractor with plough and Tractor with harrow	1,409.2	1,409.2	
	Herbicide spraying	Tractor with sprayer	2,371.4	2,371.4	Glyphosate: 4 l ha^{-1} water: 200 l ha^{-1}
	Planting	Tractor with planting machine	803.9	803.9	10.458 cuttings ha^{-1}
	Fertilizing	Tractor with front loader Tractor with fertilizer spreader		952.3	Fert. after each rotation with ammonium nitrate, potassium chloride, calcium carbonate; incl. fertilizer production chain
	Weed control	Tractor with mechanical hoe	97.3	97.3	
Harvesting and processing	Harvesting	Forage harvester with wood cutting attachment Tractor with trailer	12,012.0	14,935.6	
	Stool removal	Tractor with mulcher Tractor with rotary tiller	19,320.2	19,320.2	
	Drying/storage	Telescopic loader	6,6667	8,333.4	To pile the wood chips
Transport	Road transport	Lorry, 27 t payload	37,313.7	46,642.1	50 km to CHP plant
Total			79,994.4	94,865.4	

Energy conversion

The short rotation wood is assumed to be used for cogeneration where the wood chips are converted into electricity and heat. There are various types of cogeneration (combined heat and power - CHP) plants and several possibilities for cogeneration process control. Biomass cogeneration plants achieve much lower total efficiencies than fossil CHP plants (Kaltschmitt *et al.*, 2009). Electrical and thermal efficiency are closely interconnected in cogeneration technology. The total efficiency is determined by the annual electricity and heat production in relation to the energy input contained in the wood. But heat production is also determined by the consumer heat demand which varies seasonally. If more heat is produced, less electricity can be produced. The assessment results are strongly influenced by the underlying efficiency of the CHP plant.

In order to reflect the variety of cogeneration technologies, two typical examples have been chosen to represent energy generation from wood. One is a medium-sized plant (6 MW_{el}) with an extraction-condensation turbine connected to a district heating system (Mörschner and Eltrop, 2004), which is the most common arrangement for wood combustion and is mostly operated with priority for heat (Witt *et al.*, 2011). It has an electrical efficiency (β) of 20% and a thermal efficiency (α) of 50%. The total efficiency (ω) (the sum of α and β) is 70%. The other technology is a bigger CHP (12 MW_{el}) plant which focuses more on electricity production and much less on heat (Fiedler *et al.*, 2006; ENRO AG, 2007; Stadtwerk Elsterwerda GmbH, 2007). Its total efficiency (ω) is only 34%, with an electrical efficiency (β) of 27% and thermal efficiency (α) of 7%. Figures in Table 3.2 illustrate the total possible energy yield consisting of electricity and heat that could be generated from one kilogram of wood chips.

Table 3.2: Cogenerated electricity and heat and total energy yield from 1kg_{od} wood chips. Heat production is also shown relative to 1MJ electricity.

		Electricity [MJ]	Cogenerated heat [MJ]	Total energy [MJ]
CHP with district heating (ω=70%)	Energy from 1 kg_{od} wood chips	3.70	9.25	12.95
		1	2.50	3.50
CHP with electricity focus (ω=34%)	Energy from 1 kg_{od} wood chips	5.00	1.30	6.29
		1	0.26	1.26

3.1.2.1 *Functional unit and allocation*

The energy conversion process results in two cogenerated products: heat and power. Consumables and emissions from their production are allocated between these products. Because of exergetic quality differences between power and heat, inputs and outputs cannot be allocated equally. Therefore allocation is made according to their inherent exergy. Exergy is the embodied energy which is available to be used. The exergy of a system reflects its maximum capacity to work and is often used to characterize the efficiency of energy conversion processes (Backhaus and Schlichting, 1984). One unit of electrical power embodies one unit of exergy. The exergy contained in the heat depends on its temperature in relation to the temperature of its surroundings. If heat is delivered in a local district heating network at 120°C, the exergy it contains amounts to 0.28 units, assuming a surrounding temperature of 10°C. By applying this allocation heat receives fewer fractions of released or extracted substances.

The functional unit in this study is 1 mega joule (MJ) of electricity. The amount of cogenerated heat is depends on the particular process control and plant configuration. Impact indicator results for both power plants in the study are shown per MJ of electricity (MJ_{el}) and per MJ of heat (MJ_{th}) respectively.

3.1.3 Results

3.1.3.1 *Energy use*

The energy input:output ratio per hectare SRC is presented in Table 3.3. If fertilizer is used to cultivate short rotation coppice the energy input is increased by 77%. Bioenergy production is improved by 25%, but the input:output ratio decreases by about 30%

Table 3.3: Comparison of input:output ration between fertilized and unfertilized SRC cultivation.

		without fertilization	with fertilization
Energy input	[GJ ha^{-1}]	80	141
Biomass output	[t_{od} ha^{-1}]	147	184
Potential energy output	[GJ ha^{-1}]	2,728	3,410
Potential input:output ratio		1:34	1:24

Due to losses during the conversion process, not all the harvested energy contained in the biomass is available after conversion. The input:output energy ratio after biomass conversion depends on the conversion technology. In general CHP technology increases the total efficiency, which is composed of electrical and thermal efficiency and is considerably influenced by the heat use. Table 3.4 shows energy outputs and input: output ratios after biomass conversion to heat and power.

Table 3.4: Comparison of required fossil energy input and derived energy from SRC wood using different cogeneration technologies.

		Electricity		Heat	
		unfertilized	fertilized	unfertilized	fertilized
Input	[GJ ha^{-1}]	80	141	80	141
Output after conversion in CHP with district heating (ω=70%)	[GJ ha^{-1}]	546	682	1,364	1,705
Input:output ratio		1:6.8	1:4.8	1:17.0	1:12.0
Output after conversion in electricity-focused CHP (ω=34%)	[GJ ha^{-1}]	736	921	191	239
Input: output ratio		1:9.2	1:6.5	1:2.4	1:1.7

The total efficiency of the CHP plant influences the amount of energy which can be generated from biomass. A district heating system with good heat use reaches a total input:output ratio of approximately 1:24. A CHP plant focusing on electricity production reaches a total ratio of only 1:12.

3.1.3.2 Impact assessment results

A variety of impact categories are available; four of them have been selected here for impact assessment: global warming potential (GWP), acidification potential (AP), eutrophication potential (EP) and photochemical ozone creation potential (POCP). Figure 3.1 gives an overview of impacts from production and transport of one oven dry tonne (odt) of wood chips from SRC. The effects of fertilizing the plantation are obvious. Wood production in short rotation plantations cause higher environmental impacts in the assessed impact categories than conventional wood production in forests. Compared to chip production from pulpwood (11.9 kg CO_2-eq. odt^{-1}), which was assessed in a former study (Rödl, 2012), wood chip production from short rotation plantations releases twice as much GHG (23.8 kg CO_2-eq. odt^{-1}) even if the plantation is not fertilized.

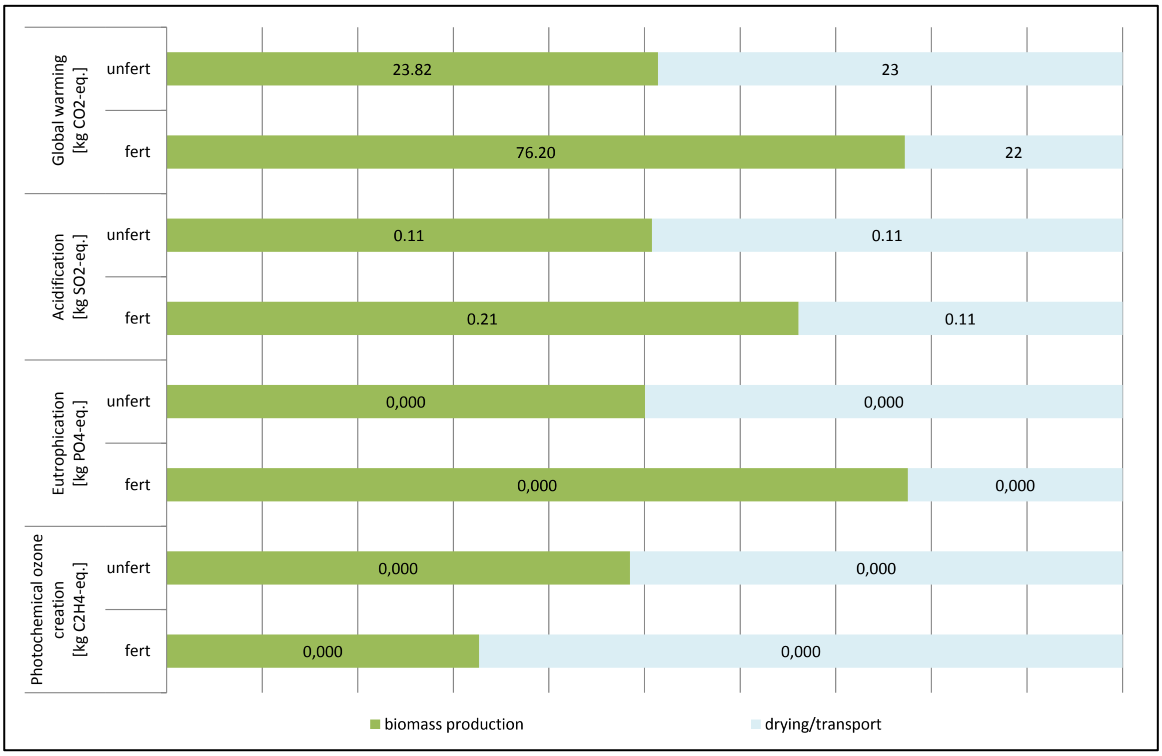

Figure 3.1: Impacts of biomass production, drying and trasnport in unfertilized (unfert) and fertilized (fert) SRC, displayed per oven dry ton (odt) of biomass.

In Table 3.5 impact indicator scores, by MJ_{el} and MJ_{th} respectively, are presented for the two CHP plant types. Due to the allocation procedure and the different amounts of heat and power produced, the impacts per MJ electricity are higher than per MJ heat.

Table 3.5: Impact indicator scores in g equivalents per MJ electricity and heat (SRC cultivation without fertilizer).

		CHP with district heating (ω=70%)		CHP electricity-focused (ω=34%)	
		electricity	heat	electricity	heat
GWP	[g CO_2-eq. MJ^{-1}]	9.01	3.08	9.59	3.75
AP	[g SO_2-eq. MJ^{-1}]	0.16	0.05	0.11	0.04
EP	[g PO_4-eq. MJ^{-1}]	0.039	0.013	0.027	0.011
POCP	[g C_2H_4-eq. MJ^{-1}]	0.014	0.005	0.011	0.004

Impact indicator scores presented in Table 3.5 do not include fertilization of the short rotation plantation. If fertilizer is used, the indicator results change considerably as shown in Table 3.6.

Table 3.6: Changes in impact indicator scores if the plantation is fertilized.

	CHP with district heating (ω=70%)	CHP electricity focused (ω=34%)
Global warming potential	+ 93%	+ 103%
Acidification potential	+ 10%	+ 17%
Eutrophication potential	+ 25%	+ 42%
Photochemical ozone creation potential	-7%	-11%

Impacts related to electricity or to heat do not differ greatly between the two types of power plant. But if the scores are compared on the basis of total energy produced, differences in plant type efficiency become apparent (Table 3.7). The impacts displayed result from dividing total impacts from the combustion by total energy production (electricity + heat) without allocation (see Table 3.2). The higher the plant efficiency the lower are the environmental impacts.

Table 3.7: Impact indicator scores for total energy production in MJ without allocation and in relation to total exergy produced.

		CHP 70%; unfert.	CHP 70%; fert.	CHP 34%; unfert.	CHP 34%; fert.
GWP	[g CO_2-eq. MJ^{-1}]	4.38	8.46	8.16	16.57
AP	[g SO_2-eq. MJ^{-1}]	0.076	0.083	0.094	0.109
EP	[g PO_4-eq. MJ^{-1}]	0.019	0.023	0.023	0.033
POCP	[g C_2H_4-eq. MJ^{-1}]	0.007	0.006	0.009	0.008

In the next stage of analysis, impact indicator scores of electricity production are normalized. The normalized scores (Figure 3.2) show the relative per capita share of 1 MJ electricity on the total annual impact of each selected impact category (Table 3.2). They allow comparison of the relative contribution of the analyzed energy systems to the different total impacts. Normalized impacts from conventional power generation are also included for comparison. As recommended in the ILCD handbook (European Commission, 2010), base references are the total per capita flows for each of the selected impact categories within the EU 25 plus Norway, Iceland and Switzerland for the base year 2000. Updated CML2001 values for the EU 25+3 have been used (Sleeswijk *et al.*, 2008; PE and LBP, 2009), which were calculated from statistical data (EUROSTAT, 2012).

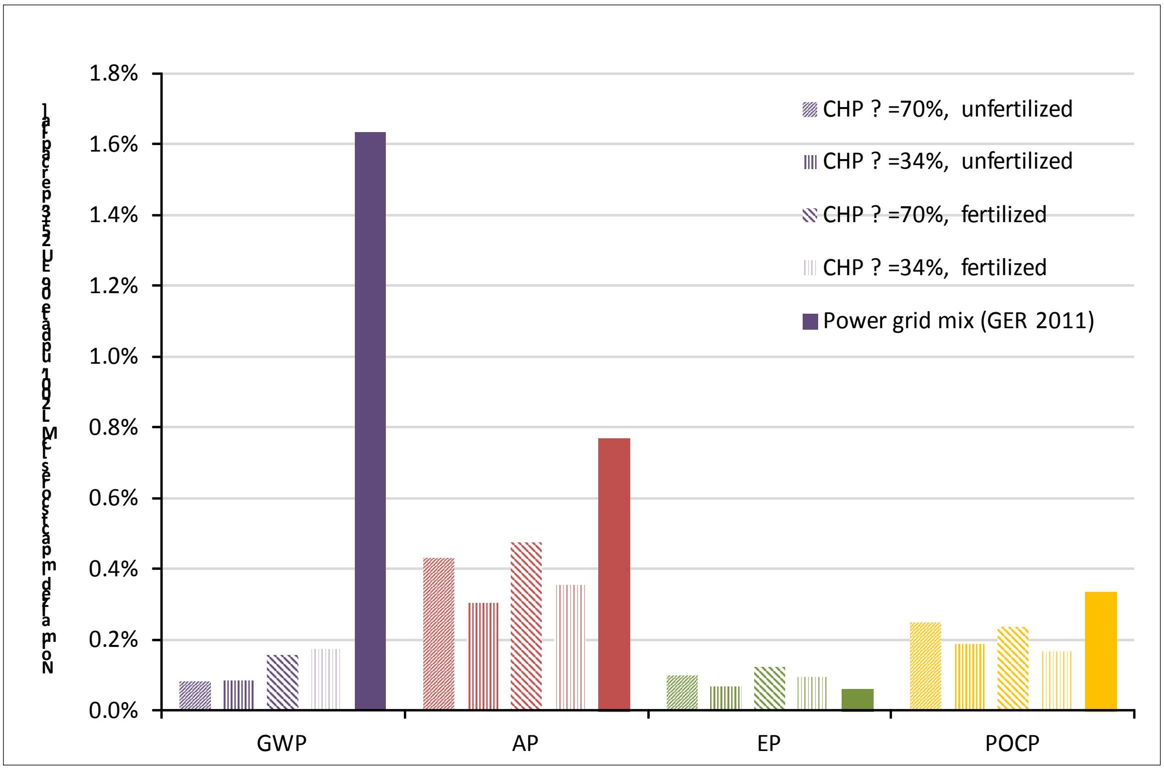

Figure 3.2: Impact indicator results for electricity normalized to CML2001 reference values per capita, Germany.

The analysis shows a higher relative contribution of emissions from biomass electricity production to overall AP than of GHG emissions to overall GWP per capita. Nevertheless the contribution of electricity from SRC wood to total AP and GWP is lower than of electricity from the grid. Eutrophying emissions of electricity production from poplar chips contribute more to overall EP in all cases than power from the grid, but the relative contribution to overall eutrophying emissions is quite low. Photochemical ozone creation potential of electricity from wood from unfertilized poplar plantations is higher than from fertilized plantations. This is due to nitric oxide (NO) emissions from fertilizer application which counteract the impact of ozone-creating emissions like nitrogen oxides (NO_x), carbon monoxide (CO) and volatile organic compounds (VOCs), because it leads to depletion of low level ozone.

3.1.3.3 Savings of GHG emissions compared to fossil energy

To identify potential advantages of producing electricity from short rotation wood, a comparison was made with fossil electricity production in Germany (Figure 3.3). For this comparison, the mixture of fuels for electricity production in 2011 was taken (PE and LBP, 2009). GHG emission differences between the two types of power plant are not very big, but GHG emissions rise if SRC plantations are fertilized. However, emissions from SRC electricity production are lower in all cases than average electricity production for the German grid. By using electricity from SRC, greenhouse gas emissions can be reduced by 85 to 94% compared to using standard power from the grid.

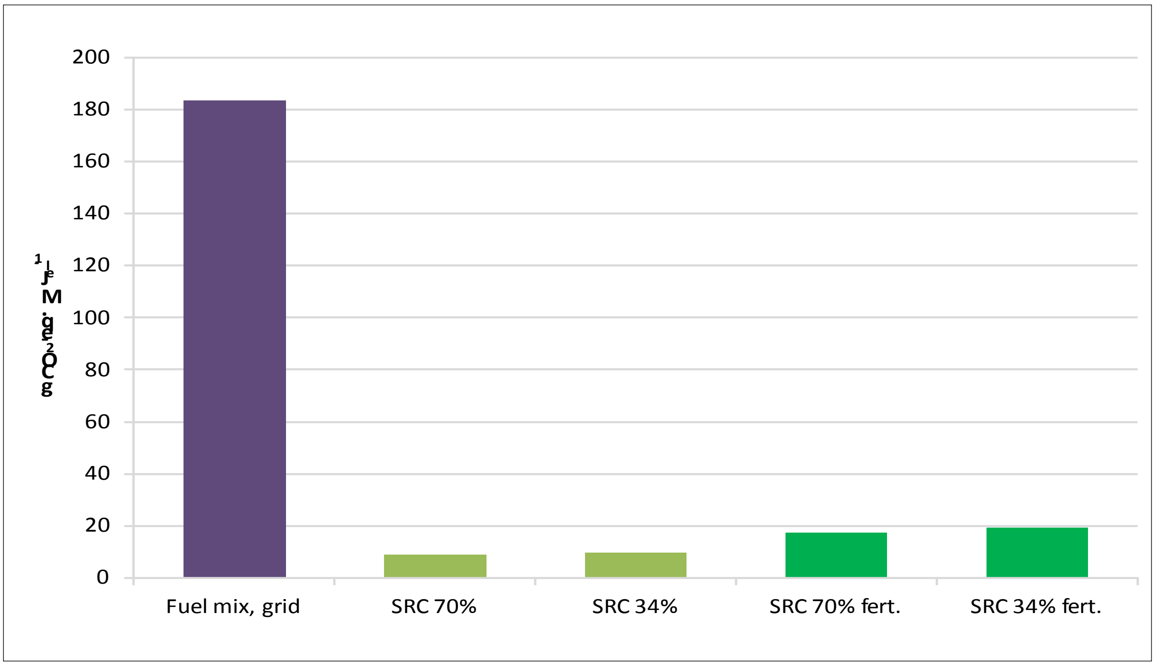

Figure 3.3: Comparison of GHG emission of electricity from SRC and from the 2011 electricity mix of the German grid.

3.1.3.4 Impacts of land use change

Since bioenergy is based on biomass from forestry or agriculture, other impacts from land management, beyond those discussed above, may occur within system boundaries. Such impacts from land use or land use change might include changes in nutrient cycling, soil properties, water balance or species composition. Methods and data are presently insufficient to assess these effects completely and they are therefore not included here. However, some data is available on the evolution of soil carbon stocks under SRC, for which recent studies found an increase of 0.44 t C ha-1 y^{-1} (Don *et al.*, 2011). This amount would more than compensate for GHG emissions from biomass production and distribution. But there is little agreement in the literature. SRC establishment on agricultural land increases soil carbon content mainly in the upper layer. Carbon storage depends on local soil parameters as well as on climate and tree species. During stool removal soil carbon gains might be lost again (Dowell *et al.*, 2009; Guo and Gifford, 2002; Kahle, 2007; Lasch *et al.*, 2010; Morris *et al.* 2007; Saurette *et al.*, 2008; Ulzen *et al.*, 2010).

3.1.4 Discussion and conclusions

The results presented are for selected environmental impacts of heat and power production from short rotation wood. Although typical examples for SRC cultivation and combustion were chosen for modeling, the results vary according to the underlying assumptions. Power plant efficiency, plant operation and the amount of heat used have a strong influence. Impact values are also determined by allocation, which depends on the inherent exergy which represents the importance of both products. Heat production per MJ of electricity produced varies according to the operational management of the CHP. When emissions from the total energy produced are compared per MJ it becomes apparent that the emissions rise with decreasing overall plant efficiency. Because of adjusted assumptions about CHP plant efficiencies, the results presented here differ from those in a previous paper (Rödl, 2010).

Life cycle impacts are also sensitive to changes in rotation length and biomass growth rates. If more biomass were produced per hectare, impacts per functional unit would decrease. If soil carbon changes in agricultural soils were included in the assessment, CO_2-emissions from biomass production and distribution would be overcompensated by this carbon uptake. The impacts of short rotation plantation cultivation on a large scale have not been assessed within this study. Plantation scale could have an influence on water balance, species composition, biodiversity and nutrient cycling at the landscape level.

Comparison of GWP scores for SRC to those for electricity from the grid helps quantify the relationship of bioenergy to conventional energy production. However, it provides no information on marginal effects of bringing SRC electricity onto the market. Reductions in GHG emissions will be greater if inefficient fossil power plants with high CO_2 emissions are replaced. Such reductions might be less if cleaner technologies would be replaced.

3.1.5 References

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3.2 Bioenergy supply chains in Swedish agriculture

by Pål Börjesson

The biomass supply from Swedish agriculture is presently small compared with the biomass supply from the forest sector, equivalent to some 2% of the total biomass delivered (approximately 500 PJ per year) to the Swedish energy system (Swedish Energy Agency, 2011). The supply from agriculture consists of traditional annual crops, such as grain and oil seed for biofuel production (ethanol and biodiesel), ley crops for biogas production, straw for heat production mainly in farm-scale facilities, and willow for combined heat and power (CHP) production in district heating systems (DHS). This chapter describes the environmental performance of bioenergy supply chains in Swedish agriculture, based on previous life cycle assessments. The environmental categories included are energy and area efficiency, greenhouse gas (GHG) performance, and eutrophication performance.

3.2.1 Energy and area efficiency

The average biomass yields and energy inputs for different crop cultivation systems in southern Sweden are shown in Table 1 (Börjesson and Tufvesson, 2011). The biomass energy output varies by a factor of 3, from 80 GJ per hectare per year for rapeseed, up to 240 GJ per hectare per year for sugar beet including tops and leaves. The energy output:input ratio for different cultivation systems is estimated to vary between approximately 5 (rapeseed) and 24 (willow). Thus, there is a significant difference between the different cropping systems in regard to energy and area efficiency. The dominating energy input in sugar beet and ley crop cultivation is diesel, whereas it is fertilizer in wheat, rapeseed, maize and willow cultivation. The energy output:input ratio for straw alone is approximately 35-40.

Table 3.8: Biomass yields and energy inputs for different crop cultivation systems (Börjesson and Tufyesson, 2011).

Crop	Biomass yield[1]		Energy input[2] GJ ha^{-1} yr^{-1}				Energy balance
	Ton dry matter ha^{-1} yr^{-1}	GJ ha^{-1} yr^{-1}	Diesel fuel	Fertilizers	Other	Total	Energy output/input ratio
Wheat	6.4 (4.2-8.6)	120	3.9	7.4	3.9	15.2	7.7
Wheat incl. straw[3]	10.7 (7.0-14.4)	200	5.6	7.4	4.2	17.2	11.3
Sugar beet	11.0 (7.2-14.9)	190	12.8	6.1	1.9	20.8	9.3
Sugar beet incl. tops & leaves[3]	13.5 (8.8-18.2)	240	14.3	6.1	2.1	22.5	10.5
Rapeseed	2.8 (1.8-3.8)	80	4.4	7.2	2.8	14.4	5.4
Rapeseed incl. straw[3]	6.1 (4.0-8.3)	140	5.9	7.2	3.0	16.1	8.7

Ley crops[4]	7.5 (4.9-10.1)	130	5.2	4.0	1.5	10.7	12.3
Maize (whole crop)	9.5 (6.2-12.8)	170	5.9	7.8	1.9	15.6	10.7
Willow[5]	9.5 (6.2-12.8)	180	2.9	4.0	0.6	7.5	24.0

[1] Biomass yields in southern Sweden (including estimated range) and expressed as energy output based on higher heating value.

[2] Expressed as primary energy. Direct use of diesel fuels in field and transportation operations include biomass transport by truck 50 km from the farm gate to a conversion plant. Energy input in the production of commercial fertilizers in the form of N, P and K, expressed as MJ kg^{-1}, is 45, 25, and 5, respectively. The amount of fertilizer supplied, expressed as kg N-P-K ha^{-1} yr^{-1}, is 150-25-10 for wheat, 120-20-40 for sugar beet, 145-25-10 for rapeseed (including preceding crop value), 70-30-40 for ley crops, 140-25-180 for maize and 80-10-30 for willow. Other energy inputs include the production of seeds, pesticides, machinery and transportation vehicles; external drying of wheat and rapeseed is also included.

[3] About 60% of the total amount of straw is harvested in wheat and rapeseed cultivation, and 50% of the tops and leaves in sugar beet cultivation, based on ecological considerations (maintaining the soil carbon content) and practical aspects (harvest losses).

[4] Clover-grass ley.

[5] Short-rotation coppice (*Salix*), harvested every 4 years, for a total of, on average, 24 years.

3.2.2 Greenhouse gas performance

The four main sources of greenhouse gas (GHG) emissions in energy crop cultivation are (i) carbon dioxide (CO_2) from fossil fuels (e.g. diesel in tractors and natural gas in N-fertilizer production), (ii) nitrous oxide (N_2O) from fertilizer production, (iii) biogenic N_2O from the soil, and (iv) biogenic CO_2 from the soil (when direct land use changes occur). Greenhouse gas emissions from potential indirect land use changes are not considered here.

As is clear in Table 3.9 and Figure 3.4 the choice of land use used as a reference is of great importance for the climate performance of bioenergy crops. In many previous LCAs the biogenic emissions of carbon dioxide have most often not been included, but only biogenic emissions of nitrous oxide (Kendall and Chang, 2009; Menichetti and Otto, 2009). Table 3.9 and Figure 3.4 thus also present the results of this traditional calculation method. One criticism that can be levelled at the calculations based on grain cultivation as a reference is that biogenic emissions of nitrous oxide are “excluded” although these are affected by the application of nitrogen fertilizer, at least in the long term. Field surveys show that emissions of nitrous oxide from cropland have little connection to the current nitrogen dosage in the short term, i.e., emissions of nitrous oxide may be as large from unfertilized as from fertilized fields (Kasimir Klemedtsson and Smith, 2011). In this regard, the calculations performed for grain production as a reference may be relevant in the short term (a few years), but in the longer term the emissions of nitrous oxide may be underestimated, as a nitrogen pool of a different size is built up in the soil, depending on the size of the nitrogen dosage.

The calculated biogenic emissions of nitrous oxide (direct and indirect) are based on the current IPCC methodology (IPCC, 2006). One disadvantage, among others, of the IPCC methodology is that it is based on the nitrogen applied (gross supply) and does not take into account how much nitrogen is removed through the harvested crop (net supply). From a nitrous oxide perspective, it is the net input of nitrous oxide that is relevant. A system with high applications of fertilizer but with efficient nitrogen utilization and a large removal of nitrogen can result in a lower net input than systems with lower applications of fertilizer but with low nitrogen efficiency and removal. Another shortcoming of the IPCC methodology is that it does not take into account local conditions such as climate, carbon/nitrogen ratio of the soil, soil water conditions, etc. which have proved to have significant relevance for the risk of formation of nitrous oxide (Kasimir Klemedtsson and Smith, 2011). New methods of calculation need to be developed which, among other things, would take into account the total nitrogen balance of the cultivation system, local soil conditions, etc. In Sweden more site-specific calculation methods for nitrous oxide are currently being developed, in which parameters other than the application rate of nitrogen fertilizer are included (Kasimir Klemedtsson and Smith, 2011). In the future these are expected to replace the IPCC methodology to give more reliable estimates of the size of the biogenic emissions of nitrous oxide.

Thus, the level of biogenic N_2O emissions from the soil is inherently uncertain, since these levels are influenced by a large number of local parameters (Nevison *et al.*, 1996; Bouwman *et al.*, 2002). One crucial parameter of significant importance is, as discussed above, the amount of nitrogen available in the soil. Improved efficiency in the uptake of nitrogen by the crop and more efficient fertilization strategies will therefore lead to a decreased risk of N_2O emissions (Reijnders and Huijbregts, 2008). One option for improving the efficiency of N utilization is to employ so-called 'precision fertilization' which uses geographically explicit field data, weather forecasts, etc. to tailor the N input based on the time-specific N requirements of the crop. This technology is now commercially available and partly utilized in Swedish agriculture. Increased implementation of this technology in the future will reduce the risk of biogenic N_2O emissions and thereby improve the GHG performance of bioenergy crops.

An additional factor of importance in regard to N_2O emissions associated with bioenergy crops is whether the nitrogen fertilizer utilized is produced in fertilizer plants having catalytic N_2O equipment or not (see Table 3.9). Today, approximately half of the nitrogen fertilizer plants in Western Europe have installed catalytic N_2O cleaning equipment, reducing the N_2O emissions by some 80% (Jenssen and Kongshaug, 2003; Mårtensson and Svensson, 2009). In a few years, all plants are expected to have catalytic cleaning, leading to, on average, 3 g N_2O per kg N, compared with the estimated average today of 9 g N_2O per kg N (Table 3.9).

It is important to take the time aspect into consideration when assessing changes in the soil carbon level (Reijnders and Huijbregts, 2008). Based on results from long term field trials, it is estimated that a change from annual to perennial crops on mineral soils with a long history of annual crop cultivation will lead to soil C gains over a period of about 30-50 years before a new soil C equilibrium is reached (Börjesson, 1999) Conversely, the C losses occurring when annual crops are cultivated on former grassland also gradually decrease as a new equilibrium is reached. Similar to biogenic N_2O emissions, changes in soil carbon levels are associated with significant inherent uncertainties since these levels are influenced by a large number of local parameters (see e.g. Kätterer *et al.*, 2004; Röing *et al.*, 2005).

Harvest of crop residues, such as straw and especially tops and leaves from sugar beet, will reduce the risk of biogenic N_2O emissions from the soil due to the increased output of nitrogen from the field (Table 3.9). However, straw harvest, in particular, reduces the input of organic carbon to the soil and this counteracts the positive effect of lower N_2O emissions (Table 3.9). It is here assumed that about 60% of the total amount of straw is harvested in wheat and rapeseed cultivation, and 50% of the tops and leaves in sugar beet cultivation, based on ecological considerations (maintaining the soil productivity) and practical aspects (harvest losses).

It is clear from the results shown in Figure 3.4 that perennial crops and crop residues have a much better GHG performance than traditional annual food crops. However, these big differences could, to some extent, be reduced when the feedstock is converted into, for example, biofuels. Biofuels from wheat and rapeseed will also generate high-value by-products in the form of protein feed used in milk and meat production leading to indirect GHG savings (Börjesson and Tufvesson, 2011). Thus, it is crucial when evaluating the GHG performance to include the complete bioenergy system from raw material to final energy service and not just the feedstock production phase.

Table 3.9: Emissions of greenhouse gases from different crop cultivation systems, expressed as kg CO_2-equivalents per GJ harvested biomass (excluding crop residues) (Börjesson and Turfyesson, 2011).

Crop	***Biomass yield excl. crop res.***	**CO_2 - fossil fuels**[1]	**N_2O – fert. prod.**[2]	**Reference - unfertilized grassland**		**Reference - wheat cultivation**		**Total**		
	GJ ha^{-1} yr^{-1}			N_2O - bio-genic[3]	CO_2 – bio-genic[4]	N_2O - bio-genic[3]	CO_2 – bio-genic[4]	Ref: grass-land	Ref: incl. soil-N_2O excl. soil-C	Ref: wheat cult.
Wheat	*120*	9.7	3.4 (1.1)	7.7	11	0	0	32	21	13
Wheat incl. straw		11		6.2	15	-0.9	4.0	36	21	17
Sugar beet	*190*	8.2	1.7 (0.6)	3.7	6.5	-0.9	0	20	14	9.0
Sugar beet incl. tops & leaves[5]		8.8		2.5	7.4	-2.4	1.0	20	13	9.1
Rapeseed	*80*	14	5.0 (1.7)	10	16	-1.8	0	45	29	17
Rapeseed incl. straw[6]		16		9.0	21	-3.0	5.0	51	30	23
Ley crops	*130*	6.4	1.4 (0.5)	4.6	0	-2.4	-10	12	12	-4.6
Maize (whole crop)	*170*	7.2	2.2 (0.7)	5.8	7.5	0.2	0	23	15	9.6
Willow	*180*	3.2	1.2 (0.4)	2.6	0	-2.7	-7.0	7.0	7.0	-5.3

[1] Life cycle emissions of CO_2 from fossil fuels used including a minor amount of CH_4 and N_2O emissions

[2] Based on current fertilizer production in Western Europe where approximately 50% of the plants have catalytic N_2O cleaning systems installed. Average emissions of N_2O with and without catalytic cleaning are equivalent to 3 and 15 g N_2O kg^{-1} N,

respectively, giving an overall average of 9 g N_2O used in the calculations here. Figures within parentheses represent the average of N_2O emissions when all fertilizer plants have installed catalytic cleaning.

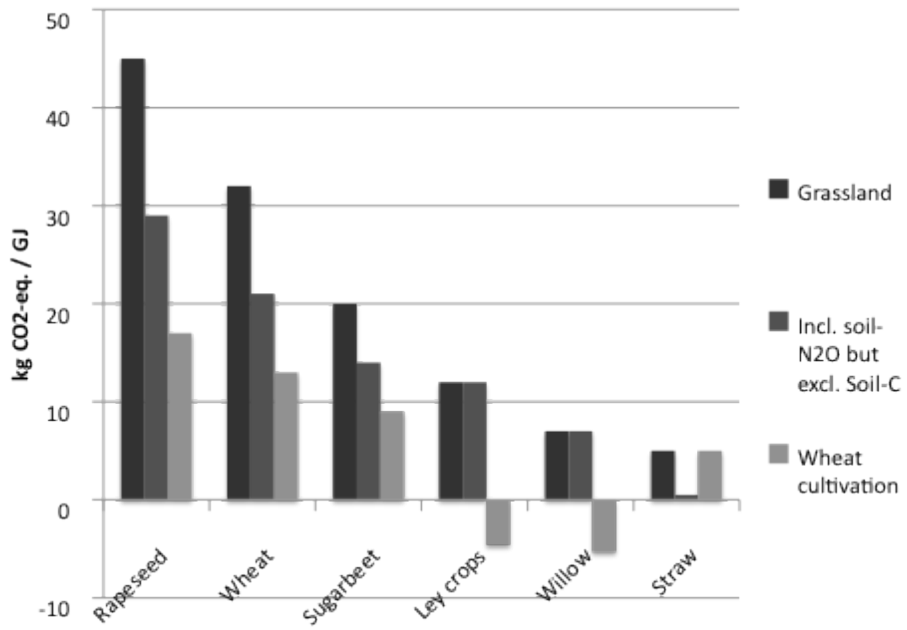

Figure 3.4: GHG performance of various biomass feedstocks in Swedish agriculture, expressed as kg CO_2-equivalents per GJ biomass, including grasslands and wheat cultivation as land use references, and also an additional case where biogenic N_2O-emissions are included but biogenic CO_2-emissions are excluded.

3.2.3 Eutrophication performance

Contributions to the eutrophication potential consist mainly of leaching of nitrate (NO_3^-) and phosphates (PO_4^{3-}) to water and emissions of ammonia (NH_3) to the air from cultivation, and emissions of nitrogen oxides (NO_x) from energy conversion. As can be seen in Table 3.10, contributions to the eutrophication potential are completely dominated by biogenic emissions when unfertilized grassland is used as land use reference. Harvesting tops and leaves in sugar beet cultivation is estimated to reduce the risk of nutrient leaching, due to the high content of nitrogen in this crop residue (Börjesson and Berglund, 2007). Harvesting straw in grain and oilseed cultivation, however, is estimated to have an insignificant overall impact on nutrient leaching. The harvest of straw results in a minor output of nitrogen, leading to a somewhat reduced risk of nitrogen leaching, but this is counteracted by the output of potential soil carbon from the straw which could help to bind the nitrogen released in microbiological processes and soil biomass.

One criticism that can be leveled at the calculations made here regarding nitrogen leakage where grain production is used as a reference - similar to the criticism discussed earlier regarding biogenic nitrous oxide emissions - is that the nitrogen leakage is largely influenced by the amount of nitrogen fertilizer applied (net). In the short term (one year), the estimates made here regarding grain production as a reference may be relevant, but in the longer term the nutrient leakage may be underestimated by this calculation method. When assessing the climate impact of biofuels, including land-use change, biogenic nitrous oxide emissions are often included in all biofuel systems based on crops, to avoid an underestimation of the climate impact. Analogous with this approach and to avoid underestimating the contribution of biofuels to eutrophication, unfertilized grass-covered cropland may be used as reference land in all cases. In this way the differences in the amount of nitrogen applied in each cultivation system are taken into account as well as intrinsic differences in the form of annual or perennial systems.

One conclusion from Figure 3.5 is that perennial crops and whole crop harvest of sugar beet, have a much better eutrophication performance than traditional annual food crops. This benefit will normally also obtain when the complete bioenergy chain is included, e.g. in producing biofuels (see Börjesson and Tufvesson, 2011). However, in the case of biofuels from wheat and rapeseed also generating protein feed as a by-product, indirect benefits in the form of reduced eutrophication may arise, depending on the reference systems replaced. Such potential benefits will then take place in another geographical region where the environmental consequences could be either greater or lower. Thus, there is a geographical aspect to consider when evaluating the importance of the eutrophication performance of biomass energy and biofuels, which is not the case when evaluating the GHG performance.

Table 3.10: Emissions of compounds contributing to the eutrophication potential from different crop cultivation systems, expressed as g PO_4^{3-}equivalents per GJ harvested biomass (excluding crop residues) (Börjesson and Tufyesson, 2011).

Crop	*Biomass yield excl. crop residues*	NO_x emissions - fossil fuels[1]	NO_3^- leaching[2]		PO_4^{3-} leaching[3]		Total	
	GJ ha^{-1} yr^{-1}		Ref. grass-land	Ref. wheat cult.	Ref. grass-land	Ref. wheat cult.	Ref. grass-land	Ref. wheat cult.
Wheat	*120*	5.7	110	0	10	0	130	5.7
Wheat incl. straw		7.2	110	0	10	0	130	7.2
Sugar beet	*190*	8.0	46	-23	6.3	0	60	-15
Sugar beet incl. tops & leaves		8.8	23	-46	6.3	0	38	-37
Rapeseed	*80*	9.1	230	57	16	0	260	66
Rapeseed incl. straw		11.1	230	57	16	0	260	68
Ley crops	*130*	5.4	17	-83	4.6	-5	27	-83
Maize (whole crop)	*170*	5.4	66	-13	7.3	0	79	-7.6
Willow	*180*	2.4	25	-50	3.4	-3	31	-51

[1] Life cycle emissions of NO_x from fossil fuels used (see Table 3.8).

[2] The gross nitrogen leaching is estimated to be: 40 kg N ha^{-1} yr^{-1} for wheat (with and without straw harvest), 30 for sugar beet (20 including harvest of tops and leaves), 50 for rapeseed (with and without straw recovery), 15 for ley crops, 35 for maize, and 20 for willow. The gross nitrogen leaching from unfertilized grassland is estimated to be 10 kg N per hectare and year, based on data from Johnsson and Mårtensson (2002) and Börjesson and Berglund (2007).

[3] The gross leaching of phosphorus is estimated to be, on average, 0.5 kg P ha^{-1} yr^{-1} in the cultivation of annual crops (Flysjö *et al.*, 2008). The corresponding figure for perennial crops is here estimated to be, on average, 0.3 kg P ha^{-1} yr^{-1}, since the

risk of soil erosion (and thereby P leaching) is lower in the cultivation of perennial crops than in the cultivation of annual crops (Börjesson, 1999). Gross leaching from unfertilized grassland is estimated to be 0.1 kg P ha^{-1} yr^{-1}.

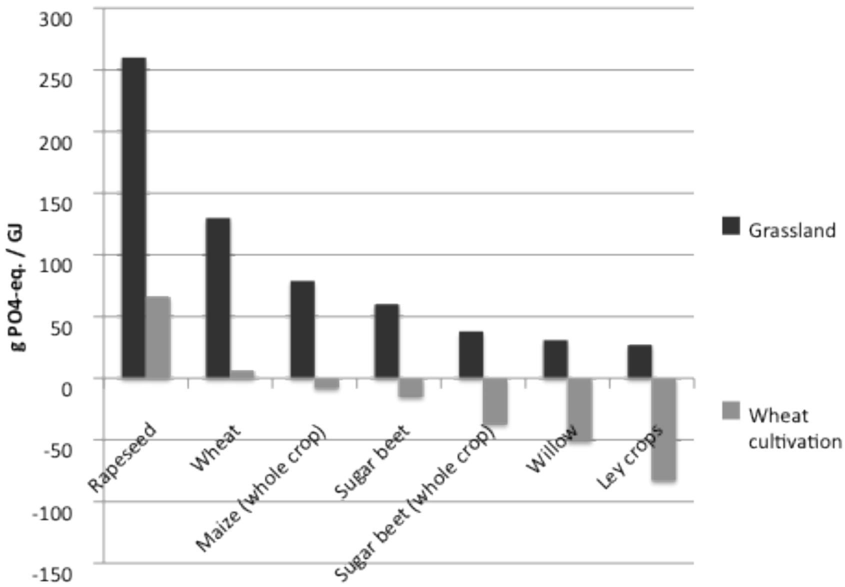

Figure 3.5: Eutrophication performance of various biomass feedstocks in Swedish agriculture, expressed as gPO_4^{3-} equivalent per GJ biomass, including grassland or wheat cultivation as land use references.

3.2.4 Conclusions and discussion

The bioenergy supply from Swedish agriculture comes from a variety of supply chains, but the total energy supply from agriculture is still limited compared to the bioenergy supply from Swedish forestry. Perennial crops such as willow and ley crops, together with crop residues such as straw, normally show higher energy efficiency, lower GHG emissions and lower contribution to eutrophication, compared with traditional annual crops. However, these differences in environmental performance may be somewhat reduced when taking downstream refining producing, for example, biofuels are taken into account. In some cases, valuable by-products, such as protein feed products, may give indirect benefits for biofuels from annual crops. Other environmental aspects, such as biodiversity, pesticide use, etc., are not included here but are crucial in making a comprehensive evaluation from a broad environmentally sustainable perspective.

The importance of considering direct land use changes is obvious since this affects, for example, both GHG and eutrophication performance. Depending on whether traditional cropland or unfertilized grassland is used for bioenergy production, the GHG balance may vary by a factor of two, or even more, depending on how biogenic nitrous oxide emissions are considered, for example. The variation in the contribution to the eutrophication potential will be even larger. Various kinds of uncertainties exist in the calculation of GHG and eutrophication performance of bioenergy supply systems. These uncertainties are both intrinsic (due to specific biophysical and biochemical conditions in the soil) and technical (due to the quality in the measurement, calculation methodology, etc.). One strategy to reduce such uncertainties is therefore to make an assessment based on the specific local and/or regional conditions when possible, since every bioenergy production system is unique in some way. This also implies that we need to have a critical attitude towards some of the life cycle assessments being published and which sometimes receive a lot of media attention. However, with our current knowledge, partly based on life cycle assessments, we can identify the most critical factors determining whether bioenergy supply chains lead to environmental improvement in GHG emissions and eutrophication. This knowledge, together with supplementary information on aspects such as the influence on biodiversity and other environmental and socio-economic conditions, is important in our effort to promote the development of “good” bioenergy supply systems and counteract the utilization of “bad” bioenergy supply systems.

3.2.5 References

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4 ASSESSMENT OF ‘NON-STANDARD’ ENVIRONMENTAL IMPACTS

LCA was originally designed to assess the environmental impacts of industrial production systems which, unlike biomass-based systems, have few interrelations with the natural environment. Hence, the assessment of environmental impacts due to land use and land use change, the use of water, as well as changes in carbon stocks, impacts on soils and biodiversity is still not standard in LCA studies today. This is due to the complexity of the interrelations between production systems and the natural and technical environment as well as the lack of information, which do not allow for easy development of standardized impact categories. However, in order to fully understand the environmental impacts of biomass based production systems, these ‘non-standard’ impacts should be assessed as comprehensively as possible. This chapter describes recent attempts to assess the most important non-standard environmental impacts.

4.1 Overview of first attempts to integrate impacts of water use into Life Cycle Assessment

by Anne Rödl

4.1.1 Background and scope

Water is essential for life. Water scarcity or non-efficient use of water is one of the world’s most urgent problems (OECD, 2008). Agriculture is one of the biggest consumers of water, with 70% of global extraction of fresh water used for irrigating agricultural crops. Growing demand for food and feed, as well as for bioenergy, may increase global water shortage. The UN warns that by 2050 the global agricultural water consumption will increase by 19% in order to secure global food production (WWAP, 2012a). At the same time energy demand, which is closely connected to water demand, will also rise. Water is needed to grow energy crops, as well as for power and heat production from fossil fuels and production of hydroelectric power. Desalination and waste water treatment are also very energy-intensive (WWAP, 2012b). It is therefore desirable to include assessment of water use in LCA studies.

4.1.2 Difficulties in water use assessment

Water use and its impacts are strongly affected by local pre-conditions. Particularly important are water availability and the local water balance, which is mainly influenced by hydrological conditions such as precipitation, temperature and soil properties. The typical local water demand of humans, agriculture and nature must also be considered. Despite local variations, the same volumetric water use may cause different impacts. Therefore it is important to consider not only the volume of used water but also the impacts of its use.

Water being used varies in terms of its origin, its point of use and its dimension. To categorize water use, terms from the water footprint method (Hoekstra *et al.*, 2011) are often used. The term ‘blue water’ includes groundwater resources or surface waters like rivers, lakes or seas. Another water resource - water bound in soils and which can only be extracted by plants - is called ‘green water’. Polluted water, or ‘grey water’, is a preliminary part of water footprint calculations (see Section 4.1.3.3). Points of water use are often 'off-stream' which means that water is removed from its source and after use is released to another water body. Water is also utilized 'in-stream', e.g. for power generation or ship transport. A further classification based on the dimension of water use is possible. Water use is called 'degradative' if water properties are changed due to its use. 'Consumptive' water use means water is removed from its source and is not immediately available afterwards. Water 'depletion' occurs if the removal exceeds the size of the source or its regeneration rate.

All these characteristics require the acquisition of additional data during the life cycle inventory stage. Sometimes the supplementary use of hydrological models is even needed. But it is still difficult to propose one uniform method that suits all cases, regions and conditions.

4.1.3 Requirements for a method for biological production processes

Special requirements arise if biological production processes are examined via life cycle assessment. Here we concentrate on plant cultivation with a special focus on bioenergy crops. Production of algae, and fish or animal farming are also biological production processes but are not be considered here. Plants mostly consume green water during their growth phase. The assessment of green water use has to be distinguished from the assessment of blue or grey water. One reason for this is the close interrelation between green water use and the water cycle. Plants extract water from soils, use it within their metabolism and then release a major part of it back to the atmosphere. Green water use by plants cannot easily be reduced. Trees especially are able to extract water from the groundwater body. Sometimes they even depend on groundwater if soil water is not replenished sufficiently by precipitation. On the other hand, groundwater recharge may be reduced in afforested areas because the extensive root systems of the planted trees hold back more precipitation than other crops. So, by influencing water flows such as evapotranspiration and run-off, trees play a crucial part in the water cycle. It may therefore be problematic to concentrate the assessment on just one water type when analyzing water use in biological production systems. It is also difficult to define suitable boundaries between the production system and its environment. Nevertheless, vegetation in general and trees in particular have a positive effect on water quality and protect soils from erosion and loss of nutrients. All these specific characteristics need to be taken into consideration in the assessment.

4.1.3.1 Recently developed water assessment methods

In recent years, several authors have made new attempts to assess water use within LCA studies. The following section concentrates on the 12 most important recent papers. (A subsequent section (4.1.3.3) covers papers which focus more on water footprint.) These 12 papers (Table 4.1) deal mainly with the evaluation of quantitative water use, mostly in relation to the general water availability in a region, which sometimes requires the additional application of hydrological models. Change in water quality is seldom the focus of the selected assessment methods.

Table 4.1: List of recent publications onw ater assessment methods for LCA.

Author	Impact category: indicator	Short description
Baitz *et al.* (2000)	**Land use**: groundwater recharge, regulation of discharge	Land use is assessed with the help of water indicators (In German with English summary)
Heuvelmans *et al.* (2005)	**Land use, resource depletion, regional water balance**: water resource, precipitation, evapotranspiration	Includes flood and drought risks in impact assessment, method especially designed for assessment of agricultural and silvicultural production
Dewulf *et al.* (2007)	**Resource depletion**: extraction of exergy	Determination of exergy extracted from the environment, for all resources, not just water
Frischknecht *et al.* (2009)	**Resource depletion, water quality**: freshwater use, nitrate load of groundwater body	Calculation of eco-points from the ratio of present to critical state according to legal thresholds (In German with English summary)
Maes *et al.* (2009)	**Land use**: Evapotranspiration	Ratio of evapotranspiration from potential natural vegetation and present land use
Milá i Canals *et al.* (2009)	**Resource depletion, ecosystem health, human health**: freshwater use, water resource, environmental water requirement	Ratio of regional availability and regeneration considering environmental water requirements
Pfister *et al.* (2009)	**Ecosystem function, human health, resource depletion**: freshwater use, water resources, precipitation	Ratio of water use and availability, considering vulnerability of ecosystems to water stress
van Zelm *et al.* (2010)	**Ecosystem function**: groundwater extraction, freshwater use	Assesses impacts of ground water level decrease on vascular plants
Verones *et al.* (2010)	**Ecosystem function**: waste water temperature, freshwater aquatic species	Assesses changes in water quality based on the relationship between water temperature and freshwater aquatic species
Boulay *et al.* (2011)	**Water quality**: quality indicators	Life cycle inventory method, no assessment of impacts
Boulay *et al.* (2011)	**Human Health:** freshwater consumption, water quality	Assesses human health impacts from unavailability or degradation of water
Hanafiah *et al.* (2011)	**Ecosystem function**: river discharge, number of freshwater fish species	Assesses impacts of water consumption on species richness of fish
Lévová and Hauschild (2011)	**Ecosystem function**: freshwater use, water resource, environmental water requirement	Ratio of water use and regional water availability, considering natural water requirements
Saad *et al.* (2013)	**Land Use, ecosystem services:** groundwater recharge, erosion resistance, physiochemical filtration, mechanical filtration	Assesses impacts on freshwater regulation and water purification caused by land use *inter alia* (based on Baitz *et al.* (2000)
Berger *et al.* (2014)	**Freshwater resource depletion:** evapotranspiration, precipitation, consumption, groundwater recharge, surface runoff, evaporation recycling ratio	Presentation of the WAVE-model – water accounting and vulnerability evaluation. Assesses the vulnerability of drainage basins to freshwater resource depletion. Considers the atmospheric evaporation recycling

4.1.3.2 Impact categories and characterization factors

Table 4.1 summarizes methods differing in terms of assessed impact categories and their indicators. Three main categories are employed: resource depletion, ecosystem function and, to a lesser extent, land use. Using these methods mainly quantitative impacts of water use can be assessed. Only Frischknecht *et al.* (2009), Verones *et al.* (2010) and Boulay *et al.* (2011) looked at qualitative changes in water resources.

The withdrawal, and especially the depletion, of freshwater resources affect the living conditions for humans, flora and fauna. Many authors therefore concentrated on the assessment of water use impacts on ecosystem function (van Zelm *et al.*, 2010; Hanafiah *et al.*, 2011; Lévová and Hauschild, 2011; Berger *et al.*, 2014) or human health (Milá i

Canals *et al.*, 2009; Pfister *et al.*, 2009; Boulay *et al.*, 2011). If blue water is extracted by humans it might also affect the availability of green water. Thus freshwater extraction influences aquatic ecosystems and adjacent terrestrial ecosystems. Milá i Canals *et al.* (2009) suggested the impact category "freshwater ecosystem impacts" to assess such effects. They used this category to analyze the damage to ecosystems caused by drawdown induced by extensive human water extractions. Most of the methods assessed blue water use, the extraction of ground or surface water resources (Frischknecht *et al.*, 2009; Milá i Canals *et al.*, 2009; Pfister *et al.*, 2009; van Zelm *et al.*, 2010; Boulay *et al.*, 2011; Hanafiah *et al.*, 2011; Lévová and Hauschild, 2011). Quantitative methods which assess the depletion of natural resources often use the ratio between withdrawn water and water availability as an indicator (Frischknecht *et al.*, 2009). The ability of a water resource or the water demand of adjacent ecosystems to regenerate was also considered (Frischknecht *et al.*, 2009; Milá i Canals *et al.*, 2009). The use of already-established indicators such as WSI (water stress index) for characterization was recommended by the authors. Pfister *et al.* (2009) also assessed changes in ecosystem quality due to water extraction. They related water use to recharge from precipitation and weight it using a factor which represents the limiting effect of droughts on plant growth. The authors provided this factor for bigger catchments and countries in a global grid. The relationship of drawdown to the disappearance of plant species was used by van Zelm *et al.* (2010) who analyzed impacts of freshwater extraction on adjacent ecosystems. Their model used Ellenberg values to determine the possible appearance of certain plant species related to water availability and other side conditions. Other authors employed changes in freshwater species composition as an indicator for assessing impacts of water extraction (Hanafiah *et al.*, 2011) or water quality changes (Verones *et al.*, 2010).

Another group of efforts concentrated on green water use. By building infrastructure, and through agriculture or forestry, humans shape the landscape and so interfere in the natural water cycle. Any land use influences evapotranspiration rates, discharge and infiltration, while crops and trees also consume water. Some authors used the impacts of land use as an indicator to assess in particular water use by agricultural crops (Heuvelmans *et al.*, 2005; Maes *et al.*, 2009). Heuvelmans *et al.* (2005) based their assessment on indicators derived from the water balance, including

flood risk: R_O *(surface runoff),*

drought risk: $P_i - ET$ *(infiltrated precipitation-evapotranspiration)* and

precipitation surplus: $P - ET$ *(precipitation- evapotranspiration).*

Heuvelmans *et al.* (2005) introduced the impact category 'regional water balance' which uses changes in streamflow as an indicator to assess impacts resulting from land uses. On the other hand, Baitz *et al.* (2000) assessed the extent of land use with the help of hydrological parameters including groundwater recharge, discharge regulation and others.

Since assessed changes need to be related to a reference, the authors suggested using the potential natural vegetation (PNV) as a reference land use. Other authors used PNV as a reference to assess land use impacts on the water balance. For example, Maes *et al.* (2009) created an indicator by comparing evapotranspiration and river discharge from present land use with that from the potential natural ecosystem. If evapotranspiration and discharge from present land use equal that from the PNV, the impact is set to be minimal. For a completely sealed surface without plant growth a simplified evapotranspiration of 0 is assumed, which indicates a maximum impact from this land use. The minimal water requirement of adjacent ecosystems is an additional critical value. If present land use exceeds this value it also indicates a maximum impact.

4.1.3.3 Water footprint

Sometimes water use assessment within LCA studies is considered to be the same as 'water Footprint'. But the two differ in several respects. The water footprint method was developed from the Virtual Water Concept (Allan, 1996) which mainly concentrated on the trade of water contained in agricultural products. Hoekstra (2003) played a substantial role in developing this concept further into the water footprint method. Calculation of the water footprint of a product considers total direct and indirect water use over its whole lifetime, distinguishing between blue, green and gray water use. Using the water footprint method, the volumetric use of these different water resources is measured, but this is not followed by an assessment of potential environmental impacts. For a product, consumer, producer or specific region a water footprint can be calculated for a specific time period. It is always specified in terms of volume of fresh water per year (Hoekstra, 2009). Numerous water footprint studies have been made, mostly concentrating on agricultural products (Chapagain *et al.*, 2006; Chapagain and Hoekstra, 2007; Aldaya and Hoekstra, 2010; Drastig *et al.*, 2010) or outlining the water footprints of countries (Hoekstra and Chapagain, 2007).

It is difficult to compare water footprints of products from different sectors because all water types are summed up in a single value. If biological products used mostly green water that could have a different importance than if production required the extraction of slowly-regenerating ground water or its pollution. But it was the intention of the authors to create an indicator primarily to measure quantitative water use. It was not intended to measure the severity of environmental impacts caused by water use but to create a pure indicator without weighting, which would clearly indicate the consumption of a limited resource (Hoekstra *et al.*, 2009). Accounting for green water use is important because water used by one crop is not available for other crops at the same time.

4.1.4 Discussion

Assessing water use within life cycle assessment studies is a complex task. There are complex interactions between land use, water cycle and ecosystem function, which can make it difficult to define single cause-and-effect relationships precisely. The methods presented here each focus on one special impact category, on special water cycle parameters or on the assessment of special products and they often cannot be combined with each other. Often it is impossible to use one of the methods for a different assessment purpose than it was originally designed for. Sometimes special models are needed which complicates their application to regular life cycle assessment. Due to the strong local and regional dimension of water use, additional data must be collected during the inventory stage. But regional information is not always available. Methods which require a lot of special and local input data are those of Baitz *et al.* (2000), Heuvelmans *et al.* (2005), van Zelm *et al.* (2010), Verones *et al.* (2010) and Hanafiah *et al.* (2011). Other methods can be applied more easily because the authors provide calculation factors for broader geographical or political regions (Frischknecht *et al.*, 2009; Maes *et al.*, 2009; Milá i Canals *et al.*, 2009; Lévová and Hauschild, 2011). But the simple implementation of these methods is counterbalanced by fewer possibilities for differentiation within these broad regions, so that regional differences cannot be considered adequately and the results might be of limited relevance.

All the methods described above for assessing water use in relation to the size of local water resources are difficult to implement. On the one hand, some of the studies lack a proper definition of the kind of resource to be considered (Heuvelmans et al., 2005; Milá i

Canals *et al.*, 2009). The scale of the eligible resources may differ considerably which could lead to a variety of results for the same region. On the other hand, it is difficult to obtain suitable data on the amount of available water in a specific region. For example, information on groundwater reserves at the local scale is rarely available.

Another topic of discussion is the inclusion of green water use in life cycle assessment. Green water can only be used by plants. But most of the water taken up is released again in form of water vapor within a short time after withdrawal. Trees or plants in general consume a lot of water but they also release it very quickly into the atmosphere. Trees play an important role within the water cycle and also contribute to the formation of precipitation (Ellison *et al.*, 2011). A water assessment method should therefore also reflect that forests or plantations are not just water consumers. Furthermore, green and blue water are linked into the water cycle. The depletion of green water affects the supply of blue water because less water percolates to the groundwater or runs off into surface water bodies. All these mechanisms should be reflected by water assessment methods so that all impacts related to water use are encompassed.

The selection of a baseline to analyze changes associated with water use is important. The methods presented here either use commonly agreed threshold values, potential natural vegetation (PNV) or the size of water resources as a limit to water availability. Using threshold values could lead to difficulties when water use of two different political regions with different, but equally valid, threshold values is compared.

It could be advantageous to use potential natural vegetation (PNV) as a baseline because it represents the present conditions, including water balance, on each site. However, PNV is a hypothetical concept which does not exist in reality. Land-based references are suitable only for land-based production systems and are not relevant for industrial goods which are produced in an urban environment where the land was sealed many years before. The impact there would be maximal in any case. As noted previously, using resource size as a reference can be problematic if it is not defined properly or if no data are available for the region being studied.

At present it is difficult to assess the impact of water use within life cycle assessment. The methods available are in many cases designed for one special application which makes it complicated to compare different product systems such as bioenergy and fossil energy. In addition, the need for comprehensive data makes it difficult in many cases for ordinary LCA practitioners to use the current methods.

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4.2 Soils

by Daniel G. Neary and Johannes W.A. Langeveld

4.2.1 Introduction

4.2.1.1 Background: soils around the world

Soils are crucial for profitable and sustainable biomass feedstock production. They provide nutrients and water, give support for plants, and provide habitat for enormous numbers of biota. There are several systems for soil classification. FAO has provided a generic classification system that was used for a global soil map (Bot *et al.*, 2000). The USDA Natural Resources Conservation Service has produced a world soils map based on the FAO system (Eswaran *et al.*, 2003, Figure 4.1). An initiative of the Digital Soil Mapping Working Group of the International Union of Soil Sciences is moving forward with making a world digital soil map to assist decisions on food and energy production. Soils are extremely diverse due to the diversity of parent materials, climates, topography, and biota that act over time to produce what we know as soil (Jenny, 1941).

Soils in Latin America are dominated by weathered clay soils in the north-east (Brazil) and more productive soils in the south. Soils in North America are characterized by mostly productive clay soils in the south, glacial-derived soils (containing layers of accumulated soil organic matter) and peat soils in the, poor sandy and clay soils in the east, and the highly productive prairie soils of the mid-continent. Asia has a similar distribution of soils as North America: predominantly peats and spodic soils in the north, less fertile clays in the south-east with large areas of productive clays and sandy soils in the remainder of the continent. Soils in Africa are mostly weathered clay soils in the north and south, with a zone of wetland soils in the middle of the continent from the Sahel to Egypt and large parts of the south-east. Heavily weathered dominate large parts of West and Central Africa. An overview of FAO and USDA soil types and their properties was assembled by Deckers *et al.* (1998).

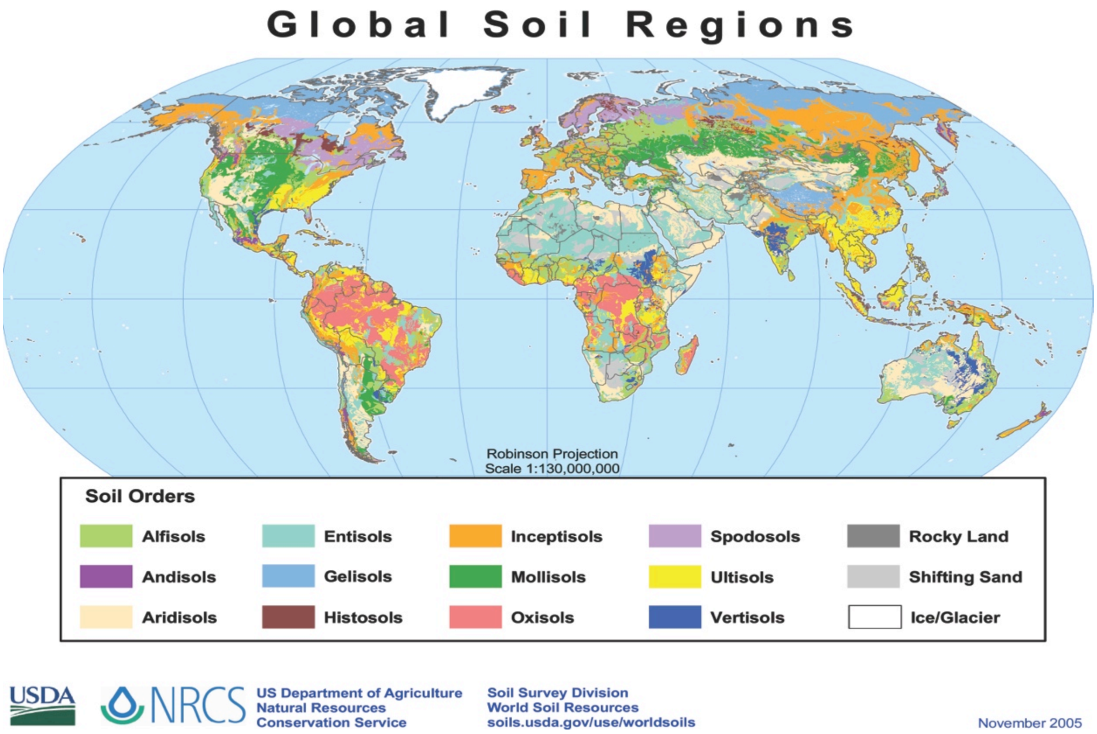

Figure 4.1: The USDA Natural Resources Conservation Service world soils map based on the FAO system and the USA soil taxonomy.

Major problems in soils include poor nutrient status, soil drainage, soil depth, slope, workability, physical limitations, damage due to soil erosion, and chemical restrictions (Figure 4.2). An overview of soil physical and chemical restrictions is given in Table 4.2. Major restrictions are related to high erodibility, aluminium toxicity, shallow soils and problems with soil morphology. Less important problems are low nutrient buffering capacity, phosphorus fixation, cracking and salinity/sodicity. Most problems are rather well distributed around the globe. Exceptions are low capacity for nutrient exchange (mostly restricted to Sub-Saharan Africa), aluminium toxicity (dominant in Latin America and – to a lesser extent – Sub-Saharan Africa) and phosphorus fixation (mostly found in Latin America). Generally, Sub-Saharan Africa and Latin America seem to be endowed with the least favourable soils. Better endowed regions include North Asia (east of the Ural), North America and Europe (Bot *et al.*, 2000).

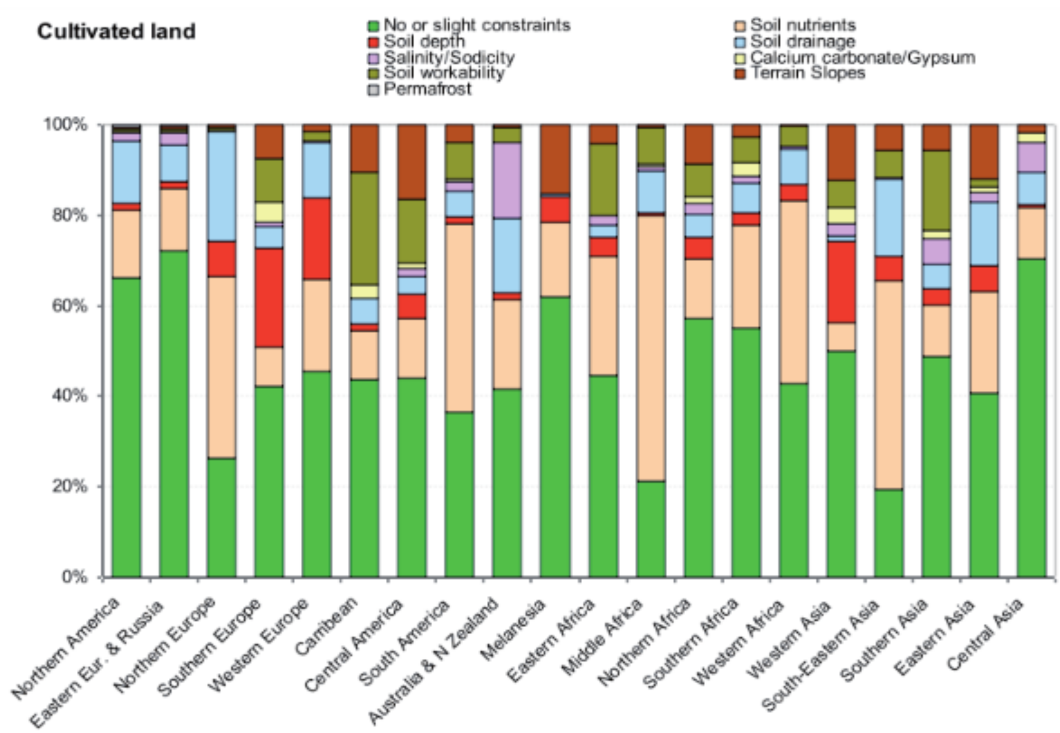

Figure 4.2: Major soil limitations of cultivated land. Source: Fischer *et al*. (2008).

It should be noted that the occurrence of soil limitations does not immediately translate to limitations in crop production. According to Mueller *et al*. (2010), for example, the most production-restricting factor is limited soil moisture which is only indirectly linked to the factors shown in Figure 4.2. The authors further conclude that most soil classification systems provide only limited information on soil productivity. The exact impact of soil-based limitations will depend on local conditions, input level and management style. As a rule, multiple restrictions for crop growth may be cumulative. Some restrictions may be overcome, but options for soil improvement are often limited while costs are often prohibitive. There are three key considerations.

First, soil characteristics are multi-factoral. The yield potential of a soil is the end-result of a combination of soil-related factors, in which the contribution of each factor depends on the total set of soil factors. High organic matter, for example, contributes to good nutrient availability (both inherent and by buffering nutrients that have been applied), good water availability (buffering water), and healthy soils. Any of these capacities can be provided at least partly by other factors: good nutrient availability by rich parent material, good water availability by good soil structure.

Second, soil characteristics are both the end-result and the beginning of dynamic processes. Availability of soil organic matter, for example, is the end-result of previous processes of organic matter contribution, removal of organic matter, and mineralization. Nutrients may be accumulated over many years and sometimes can be lost fairly quickly.

Third, some impacts of soil deterioration are not felt by the one who is cultivating the soil. Sometimes, the damage occurs at a different location. Externality effects related to soil use include mainly soil erosion and the transport and deposition of soil material by sun or wind to other locations.

4.2.1.2 The Soil Management Link to Sustainability

Sustainability is the stewardship goal of crop production and forestry in general, including bioenergy programs. However, a more specific definition of its goals and attributes is complex and open to interpretation (Moir and Mowrer, 1995; Dale *et al.*, 2012). Many ecologists have attempted to answer the “what”, “what level,” “for whom,” “biological or economic,” and “how long” questions of sustainability. Allen and Hoekstra (1994) discussed the emergence of the concept of sustainability and the difficulty in defining it. They point out that there is no absolute definition of sustainability, and that it must be viewed within the context of human conceptual frameworks and societal decisions on the type of ecosystem and the spatial and temporal scales over which attainment of sustainability is to be judged.

Sustainability is also defined in terms of the needs of society, the experiential frame of reference of ecosystem managers, and the ecological models that are used to predict future conditions for natural resources. However, the ability to predict future ecosystem conditions is confounded by the uncertainties of increasing encounters with extreme events, poorly understood ecological processes and linkages, high degrees of natural variability, surprises created by the ‘law of unintended consequences’, the development of critical thresholds, and chaotic system behavior.

Another approach to the definition of sustainability is to define the conditions that warn of or mark ecosystem deterioration into unsustainability (Moir and Mowrer, 1995). For example, although the goals of the Montreal Process and the Santiago Declaration are to ensure management of forest lands for sustainability, the Criteria and Indicators of that process are in essence warning flags to obtain the attention of land managers before ecosystems decline into unsustainability.

Soils are an important, basically non-renewable resource, that provide the physical, biological, chemical, and hydrologic foundation upon which agricultural and forest bioenergy feedstocks grow (Johnson, 1994; Kimmins, 1994; Burger, 2002). Soils are able to renew themselves after being degraded but the time period might be several centuries or even millenia, depending on climate and vegetation. Because of this long time factor, soils are considered to be non-renewable from the viewpoint of human use and management. They are heterogeneous and highly diverse components of ecosystems that form over long time periods under the influence of parent mineral material, climate, landscape position and biological activity.

As the base of bioenergy production system, soils are a major factor in determining crop yields. They provide the physical anchor which tie plants to the earth, supply water and mineral nutrients for plant growth, decompose and recycle organic material and residues, and mediate hydrological processes (Lavelle and Spain, 2001; Neary, 2002; Brooks *et al.*, 2003). Bioenergy feedstock systems are part of agricultural and forest management systems that provide multiple ecosystem products and services, including plant biomass, water flow, water quality control, biodiversity, cultural heritage, and carbon storage (Richardson *et al.*, 2002) (see also Chapter 4.5). Soils are important factors in each of these services (De Groot *et al.*, 2002). Therefore, it is critical that in the process of managing soils for bioenergy production, soils must be managed to sustain all forest values important to human communities.

Table 4.2: Occurrence of major restrictions related to physical and chemical soil characteristics. Source: Bot *et al*. (2000).

Global Location	Area ($km^2.10^6$)	Hydromorphy (%)	Low nutrient exchange (%)	Aluminium toxicity (%)	Phosphorus fixation (%)	Cracking (%)	Salinity and sodicity (%)	Shallowness (%)	Erosion hazard (%)
Sub-Saharan Africa	23.8	8	16	18	4	5	4	13	15
North Africa, Middle East	12.4	1	2	0	0	1	6	23	10
Asia, Pacific	29.0	11	4	14	5	5	11	17	16
North Asia	21.0	27	0	4	0	0	10	13	16
South, Central America	20.5	10	5	39	15	2	5	11	19
North America	21.4	16	0	10	0	1	1	12	18
Europe	6.8	17	1	8	0	1	3	12	20
Total	13.5	13	5	15	4	2	6	14	16

4.2.2 Relevance of soils in bioenergy-related LCA studies

Many problems related to productivity of crops, including biomass crops for energy are soils-based and include low fertility, physical limitations (e.g. parent material, texture, depth, drainage, moisture content, etc.), chemical restrictions (e.g. cation exchange capacity, alkalinity, acidity, carbon content, etc.). Related problems include slope, soil erosion, compaction, and leaching. Soil quality also co-determines the impact of low water or nutrient availability.

Soil compaction, erosion, and organic matter losses are the chief factors that affect decline of ecosystem productivity (Powers *et al*., 1990; Burger, 2002). They can alter ecosystem carbon allocation and storage, nutrient content and availability, water storage and flux, rhizosphere processes, and insect and disease dynamics. The chief disturbances that affect these factors are wildfire, insect and disease outbreaks, climate extremes, vegetation management (wood harvesting and stand tending activities, grazing, prescribed fire, chemical weed control, and manual removal of plant species), and recreation (foot traffic and vehicles) (Hart and Hart, 1993). Management activities that eliminate natural disturbances (e.g. fire suppression, insect control) or alter ecosystem properties can also affect ecosystem sustainability.

As previously indicated, soils are highly variable depending on parent mineral material, climate, landscape position and biological activity. The impact of relatively small changes in soil conditions on crop productivity and long-term sustainability can be huge and often cannot be easily mitigated economically. In addition, disturbances to soils that occur during bioenergy feedstock establishment, harvesting, and transportation can have significant impacts on important environmental parameters such as water quality and ultimately site productivity (Burger, 2002; Neary, 2002). Other elements which play a role in biofuel feedstock cultivation potential include weather, market conditions, energy policy, and societal preferences. Except for weather, these are more or less manageable factors. They may be resistant to manipulation but, compared to soils and weather, they are relatively malleable in the short-term. Soils and weather are physical attributes of bioenergy systems while the other factors are sociological and economic.

4.2.3 Current LCA approaches to soils

4.2.3.1 How are soils normally treated in LCA studies?

How soils can be included in LCAs depends to a large extent how the effects of changes in soil characteristics during bioenergy feedstock production are known, quantified and attributed. While the relevance of soils and their coverage in LCA studies is clear, quantification and allocation of impacts are very difficult to determine. A number of different approaches have been used to assess the effects of bioenergy feedstock production on soils, soil characteristics, and changes in soil quality but direct measures of soil properties and subsequent inferences on sustainability impacts have remained problematic.

Dale *et al*. (2012) discussed many scale issues that need to be considered in the sustainability of agricultural feedstock production systems. They noted the need to quantify the effects of specific management practices on carbon sequestration, nutrients, water, and energy fluxes. An important conclusion was the identification of the need to adopt management practices that reduce pollution, and erosion, reduce wasteful fertilizer and water use, and sustain ecosystem services at field, farm, and regional scales. This means, in essence, adopting Best Management Practices rather than measuring physical, chemical, and biological parameters and inferring sustainability changes.

There is a data dilemma in trying to measure soil characteristics. Many data are needed, which are not easily measured, and correlations with sustainability are problematic, because soils are highly variable and host a number of dynamic processes. Is there a solution to this problem or should a different approach be taken? These questions are difficult to answer. However, estimates of carbon dynamics have been given by IPCC (2006). Generalized concepts and state changes have been developed to evaluate impacts of bioenergy production. These include: Global Warming Potential (GWP); carbon and greenhouse gas (GHG) emissions occurring during production and distribution cycles (Izursa, 2013); Eutrophication Potential (EP); contribution of nutrient emissions to potential eutrophication of surface and groundwater resources (Izursa, 2013); Acidification Potential (AP); and the potential contribution to acidification of land and water (Izursa, 2013), water use (Milá i Canals *et al*., 2008) and ecosystem quality (Núñez *et al*., 2012a) See also Table 4.3.

Table 4.3: **Example of generalized impacts related to soil, water and air quality (adjusted from Izursa, 2013).**

Category Indicator	Impact Category	Description	Unit
Energy Use	Primary Energy Demand	A measure of total amount of primary energy extracted from the earth, expressed in terms of demand from non-renewable (petroleum, natural gas, uranium, etc.) and renewable (hydropower, wind, solar, biomass, etc.) resources.	MJ
Climate Change	Global Warming Potential	A measure of greenhouse gas emissions (CO_2, methane, etc.) which increase the absorption of radiation emitted by the Earth, magnifying the natural greenhouse effect.	kg CO_2 equivalent
Eutrophication	Eutrophication Potential	A measure of emissions causing eutrophication. A stoichiometric procedure which identifies the equivalence between N and P for both terrestrial and aquatic systems.	kg N equivalent
Acidification	Acidification Potential	A measure of emissions causing acidification. Assigned by relating the existing S, N and halogen atoms to the molecular weight.	kg H^+ equivalent
Water Use	Water Use	A measure of water used from sources including surface and ground water	m^3

4.2.3.2 Examples of LCA studies discussing soil quality changes

Soil organic matter (SOM) was addressed by Milá i Canals *et al.* (2007, 2008) who showed a novel approach to addressing land use impacts beyond the inventory indicator "m^2 y^{-1} of occupied land." Soil organic carbon (SOC) was used as an indicator of soil quality, and potential changes to SOC linked to different land uses were compiled for the complete life cycle of the products assessed. The results showed that, contrary to assumptions in several other life cycle impact assessment methods, life cycle stages other than cropping may dominate the impacts related to land use, even if cropping still dominates in terms of area per year. Núñez *et al.* (2012b) used soil and weather data to assess impacts of soil erosion in a spatially explicit LCA analysis by calculating soil carbon loss at grid cell levels for energy crops in Spain (Figure 4.3). Gan *et al.* (2014) included damage due to soil erosion in an economic optimization approach for corn stover removal in a lignocellulosic ethanol production region in Iowa, USA. Cowell (1998) and Cowell and Clift (2000) examined soil compaction effects. Croezen *et al.* (2013) analyzed and presented generalized data on soil carbon changes, nutrient inputs, application of agro-chemicals and water use in major bio-based production chains.

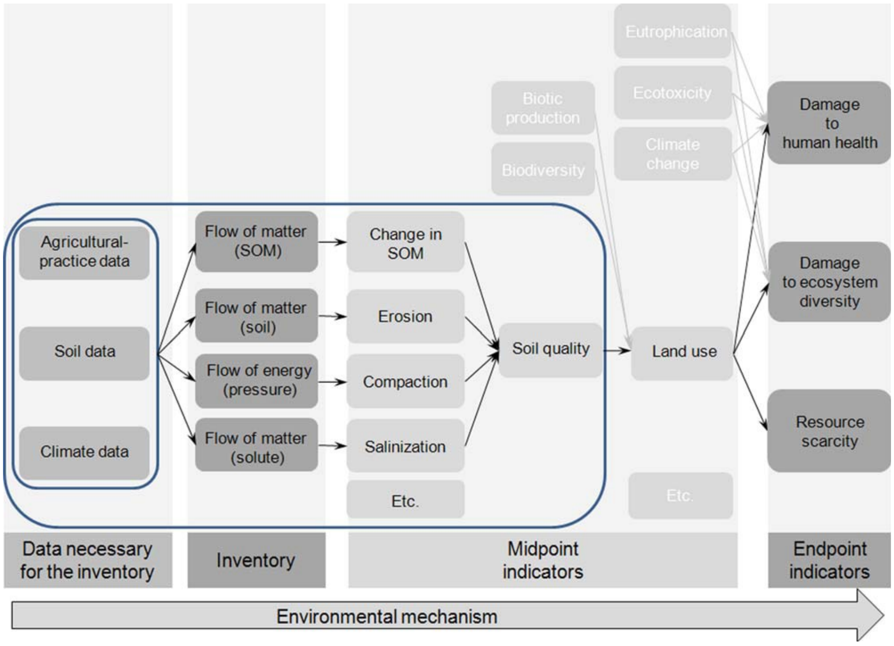

Figure 4.3: Data collection and analysis in LCA (after Garrigues *et al*., 2012).

4.2.3.3 Regional approaches to soil sustainability

Different approaches exist in different parts of the world for addressing soil sustainability in land use and forest management systems. This section discusses specific examples from Canada, the European Union, New Zealand and the USA.

Canada

About 36% of Canada's forests have been certified as being sustainably managed by globally recognized certification standards (Natural Resources Canada 2009). Codes of Forest Practice are in place in Nova Scotia, Ontario, and British Columbia. Canada's forest laws and regulations are considered to be among the strictest in the world. British Columbia has led Canada in developing procedures to ensure forest sustainability. The *Forest Practices Code of British Columbia* of 1996 established the legal framework for monitoring soil disturbances caused by forest operations (British Columbia Ministry of Forests 2001). It has since been augmented by the *Forest and Range Practices Act* of 2002. The Province has an iterative adaptive-management process that provides constant feedback to forest operations and research to improve Best Management Practices and operations planning and execution (Figure 4. 4)

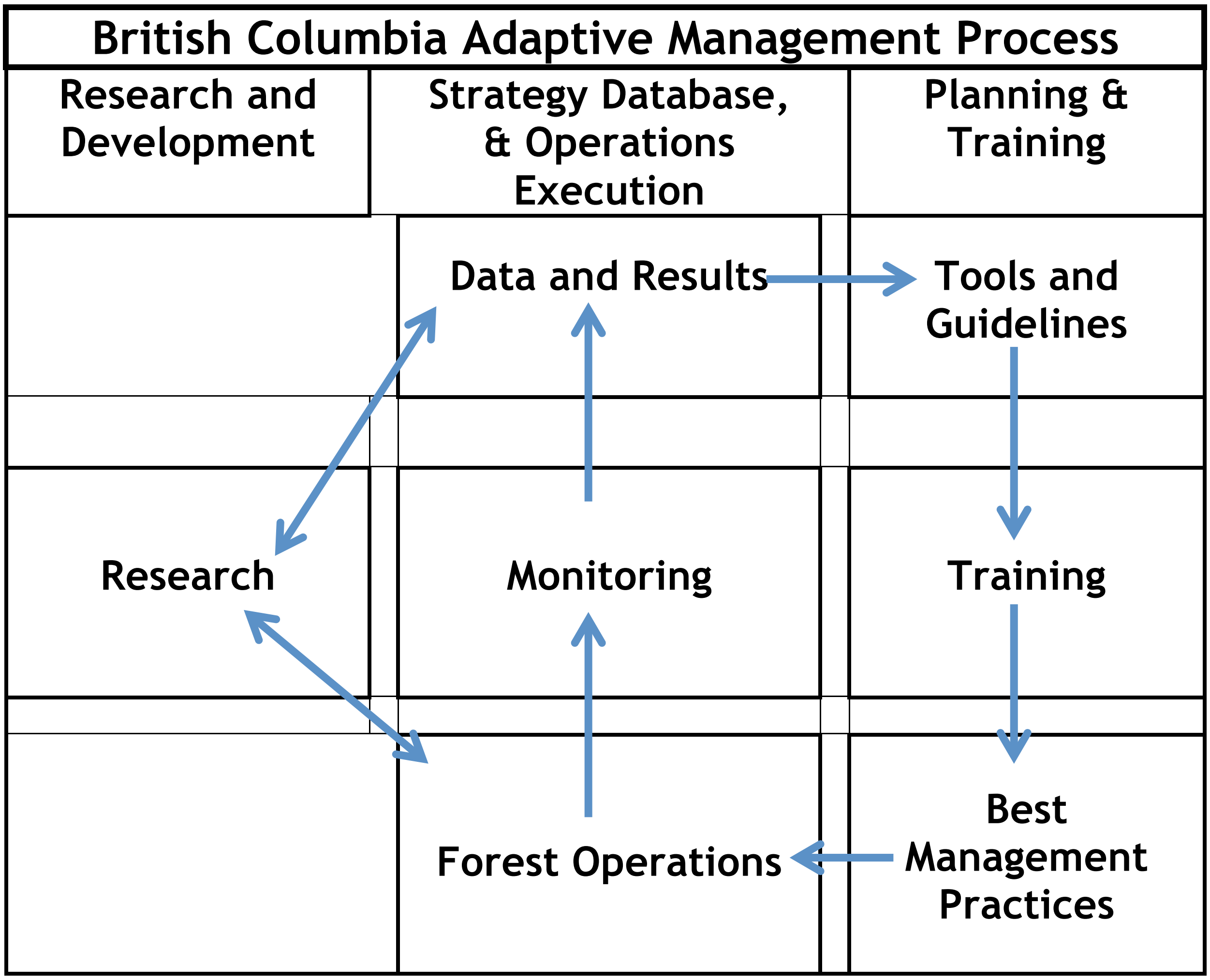

Figure 0.1: British Columbia soil monitoring adaptive management process (adapted from Curran *et al.*, 2005, 2007)

European Union

Forests cover 160 million ha within the European Union (EU), or about 42 percent of the area. Six of the 27 member countries account for two-thirds of the forest area, with Sweden and Finland together accounting for 30% of the total forest area (EUROSTAT, 2009). Official protocols exist in most member countries for soil monitoring (Morvan *et al.*, 2007), but there is a lot of variation in the methodologies used and the intensity of sampling. The EU Monitoring Network has been active since the early 1990s using a 50 x 50 km grid with variable re-measurement periods. Parts of the EU Network have dense sampling grids (United Kingdom, Ireland, Austria, Denmark) while in other areas the network is still sparse (Spain, Italy, Greece) (Figure 4.5). About 90 percent of the EU soils and land cover classes have at least one monitoring site. However, the density of soil monitoring sites within the European Soil Database units is highly variable. Some units (7 percent) do not have any monitoring sites. The highest density of soil monitoring sites is found on pasture lands, as well as on arable land and forests which, however, occupy slightly less area. A grid of 16 x 16 km has been established for forest soils (ICP, 2004). The key soil parameters being monitored in the EU include erosion risk, compaction risk, presence of peat, heavy metals, desertification, and presence of livestock (Morvan *et al.*, 2007).

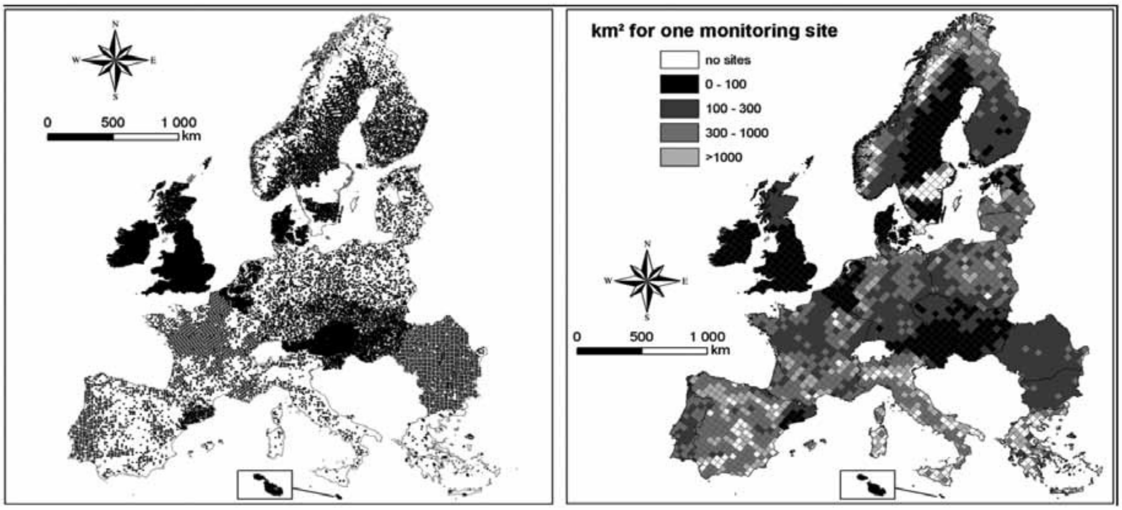

Figure 0.2: European Union soil monitoring network, GIS data subdivision into smaller, more manageable data sets (right) and actual density (left) in km^2 for one monitoring site in grid of soil monitoring sites of the 50 x 50 km Cooperative Program for Monitoring and Evaluation of the Long-range Transmission of Air Pollutants in Europe (EMEP) (Moran *et al.*, 2007).

Other indicators being measured are texture, pH, organic matter, bulk density, cations, and earthworm activity (Morvan *et al.*, 2007). The EU Soil Monitoring Network is basically an inventory system that has no interaction with land management, and does not have any regulatory power. Its needs include adding 4,100 sites in the lower density part of the network and standardizing sampling and analytical methods. Of the countries with mandated soil monitoring (Table 4.4), Sweden requires measurements of soil physical conditions, coarse woody debris, and soil chemistry. Ireland requires measurements of soil condition, soil fertility, erosion, and other parameters as needed. Although the United Kingdom does not require soil monitoring at the present time, changes to codes of forest practice will mandate this activity in the future (Bone *et al.* 2010).

Within the EU, Ireland is a good example because of its well-developed code of forest practice. Over 70 percent of Ireland's 636,164 ha of forests are owned by the Irish Forestry Board (Coillte Teoranta). Soil monitoring in Ireland is included in the country's *Code of Best Forest Practice* and is based on EU and national laws (Ireland Forest Service, 2000). As in a number of other countries, the Irish code is focused on planning, monitoring, and adaptive management rather than punitive regulatory actions.

Table 4.4: Monitoring standards and requirements for conventional forestry and forest bioenergy in some European Union countries.

Country	Forest Harvest Practices Code	Bioenergy Guidelines	Monitoring		
			Required	Type	Soil
Denmark	Yes	No	No	None	No
Finland	Yes	Yes	Yes	Operations	No
Germany	Yes	Yes	Yes	Inventory	No
Ireland	Yes	No	Yes	Operations	Yes
Netherlands	Yes	No	Yes	Operations	No
Sweden	Yes	No	Yes	Multiple	Yes
United Kingdom	Yes	Yes	No	None	No

Monitoring is performed to evaluate the performance of the Irish Code of Best Forest Practice as well as the skills of individual forest harvesting operators. It consists of a self-evaluation impact appraisal carried out by the individual operators and an external assessment by the Irish Forestry Board (Ireland Forest Service, 2000). The Irish impact appraisal evaluates environmental, economic, and social impacts of forestry operations. The Irish focus is on assessing potential impacts in terms of their level, likely consequence, importance, and the length of time for which the impacts will occur. Potential impacts evaluated in the Irish impact appraisal were assessed descriptively or on a 'points' system on the basis of four subjective severity levels (very high, high, moderate, and low), and follow-up mitigation actions are then planned. In the appraisal, soil fertility was evaluated as being at high risk because of the soil type and the whole-tree harvesting planned for the cut block. So the mitigation technique prescribed for this stand was the addition of a nitrogen-phosphorus-potassium fertilizer. The other potential soil impacts were evaluated as being low so no mitigations were planned.

New Zealand

A national-scale soil quality monitoring program was conducted in New Zealand between 1995 and 2001 at 222 sites in five regions (12 soil orders and 9 land-use categories) (Sparling and Schipper, 1998, 2002). Land uses in the survey included arable cropping, mixed cropping, pasture, grassland, plantation forests, and native forest. Sampling of the topsoil (0–10 cm) was done and the properties measured were total carbon (C) and nitrogen (N), potentially mineralizable N, pH, Olsen phosphorus (P), cation exchange capacity, bulk density, total porosity, macroporosity, and total available and readily available water. Seven of these soil parameters (total C, total N, mineralizable N, pH, Olsen P, bulk density, and macroporosity) explained 87 percent of the total variability. Some of the issues that arose during the soil quality sampling were minimum data set, how to stratify, level of precision, cost, centralized data and sample management, re-sampling for trends, and sampling strategy. Important conclusions and recommendations from the program related to methodology were that (1) a precision of 10 percent was impractical due to cost, (2) a precision of 25 percent was more realistic, (3) central storage of data and samples was essential to success of this type of survey, (4) re-sampling needed to be over a 3- to 10-year time period with some re-sampling every year, and (5) financial constraints prohibited random sampling.

Key findings from the New Zealand Soil Quality Survey were:

- Soil order had a strong effect on the results.
- Land use accounted for only 21 percent of total C variability.
- There was no evidence of acidification under exotic tree species. Changes in soil quality between land uses can be detected.
- Biochemical parameters and total C indices are more sensitive to land management differences than physical parameters.
- Soil quality of mature pine plantations, before and after logging, was similar to that of native forests or low-productivity pasture.
- Many research needs were identified to make a national-scale soil quality survey a viable management tool.
- Changes in soil quality characteristics can be detected, but there is a general lack of a scientific framework to define acceptable and unacceptable ranges of soil quality parameters.

USA

The Natural Resources Conservation Service (NRCS) and the Agricultural Research Service (ARS) of the USA conduct research and development activities related to soil quality and soil monitoring (Doran and Parkin, 1994; Doran and Jones, 1996; Karlen *et al.*, 1997; Doran *et al.*, 1998; USDA NRCS, 2001). The ARS has also developed standardized methods for monitoring grassland, shrubland, and savanna ecosystems (Herrick, 2005a, b). Soil monitoring conducted by the Forest Inventory and Analysis (FIA) Program of the US Forest Service is discussed in detail by Amacher and Perry (2010) and by O'Neill *et al.* (2005a, b). Forest industries such as the Weyerhaeuser Company are committed to soil productivity. Weyerhaeuser uses a two-step strategy (Heninger *et al.*, 1997, 2002). First, its operations use equipment and operations practices that are appropriate to the soil, topography, and weather to minimize erosion and harmful soil disturbance. Secondly, the company employs forestry practices and technology to retain organic matter and soil nutrients on site.

A reliable monitoring protocol has been identified as a critical component of any adaptive management process for forest and rangeland soil conservation programs (Curran *et al.*, 2005). Uniform and unambiguous definitions of soil disturbance categories must also relate forest productivity and hydrologic function (Curran *et al.*, 2007). A soil monitoring protocol must incorporate a statistically rigorous sampling procedure and firm definitions of visually observable soil disturbance categories

A soil monitoring protocol, first developed by the US Forest Service in its Region 1 and Rocky Mountain Research Station, incorporates soil quality monitoring efforts pioneered in the Pacific Northwest (Howes *et al.*, 1983) and is a multi-faceted approach to soil disturbance and forest sustainability issue. The protocol uses visual soil disturbance classes (Howes *et al.*, 1983; Page-Dumroese *et al.*, 2006) and a standard inventory, monitoring, and assessment tool. It employs common terminology and has an accessible database. The visual disturbance considerations are soil resilience, degree of disturbance, duration, distribution, and location in relation to other resources.

4.2.3.4 Observations on soil monitoring approaches

In general, the soil monitoring systems discussed in this section give a clear definition of system boundaries. While visual disturbance classes can be well-documented, there appears to be a considerable amount of common methodology. This makes it relatively easy to train field personnel. On the other hand there are difficulties in assessing the importance of effects due to variation in time and space. Moreover, correlations to productivity change are not always given. As a rule, monitoring systems are reactive, and procedures for mitigating negative effects are not always clear or in place.

4.2.4 Discussion

4.2.4.1 Difficulties associated with improving assessment of soils in LCA studies

Notwithstanding a considerable amount of overlap, a large variation in soil databases between countries still exists. This is related to the fact that soil classification systems still differ. The EU and USA classification systems cover the largest land masses, but there are still differences between them. Systems like that developed by the FAO attempt to merge these differences and highlight the similarities (Deckers *et al.*, 1998; Bot *et al.*, 2000).

High spatial variability is another major problem in the inclusion of soils in LCA studies. The scale of variation is such that units normally covered by an LCA study are bound to have significant differences in major soil characteristics, e.g. soil organic carbon content, inherent nutrient fertility, nutrient buffering capacity, and water absorption ability. This may well be one of the most difficult and specific aspects of soils-based LCAs. Focusing on soil erosion as an example, the impact of crop cultivation will depend on topography, (i.e. the location of the field or plot on the slope). Higher-located plots may affect growing conditions on lower plots when they are causing soil erosion. In a similar way, water extraction in a given plot close to a river may affect water availability in other plots farther from the river, or farther from the origin of the river. How should this be handled in LCA methodology? Are there different thresholds for those plots that are affected by other plots? Or, does the analysis move to a larger scale and evaluate the entire slope or watershed?

Other interactions may give very conflicting outcomes. These include higher yields that lead to soil water depletion, favorable N responses associated with high water demands, and nitrate leaching versus nitrous oxide volatilization.

In addition to problems arising from high soil spatial variability, there is also a temporal variation problem. Each year the impact of cultivation and harvest on the soil can be different as impacts are co-dependent on weather conditions. This phenomenon also impacts other elements of LCA analyses such as economic aspects. Consider, for example, allocation procedures depending on economic values of (co-)products. These will have temporal variability as well, and this is considered one of the main reasons for not applying allocation to economic values in LCA studies.

4.2.4.2 What is the way forward?

This publication addresses the environmental performance of bioenergy feedstock supply chains. The way forward relative to assessing soils impacts of bioenergy systems and the sustainability of biomass production rests with three approaches that could be used individually but are more likely to be employed in some combination. These approaches are: (1) utilizing characteristics that can be quantified in LCA studies by software, remote sensing, or other accounting methods such as GHG balances and energy balances; (2) measuring and monitoring ecosystem characteristics that can be evaluated in a more or less qualitative way (e.g. maintaining SOC) which might provide insights on potential productivity and sustainability; and (3) employing other proactive management characteristics such as Best Management Practices that are aimed at preventing environmental degradation.

Life Cycle Assessment has been used to estimate the environmental impacts of biomass energy uses. Typically this includes GHG emissions, CO_2 emissions, energy balances and some indirect effects. Cherubini and Strømman (2011) reviewed 94 LCAs, most of which were published in scientific journals. More than half of the studies were from North America and Europe, with others from South Asia, Africa and South America. About 50% of the studies limited the LCA to GHG and energy balances without considering contributions of bioenergy programs to other impact categories such as soils and water. Cherubini and Strømman (2011) concluded that there are a number of issues and methodological assumptions in currently used LCA approaches that make it impossible to quantify environmental impacts from bioenergy programs.

Some of the key indirect effects strongly depend on local operations, vegetation, soil, and climate conditions that make accurate assessment of environmental effects very difficult. Although policy makers claim that methods exist for assessing environmental impacts on soil and water, the scientific foundation for estimating indirect effects of bioenergy programs is constrained by the lack of adequate validation research, accurate assessment methods, and the relative infancy of the LCA process. Cherubini and Strømman (2011) clearly pointed out that determination of environmental outcomes of bioenergy production is complex and can lead to a wide range of results. They stated that indirect environmental effects are not ready for immediate incorporation into LCA methodology, but remain the next research challenge.

In regard to the second approach suggested above, soil quality monitoring was developed as a means of evaluating the effects of forestry and agricultural management practices on soil functions that might affect site productivity (Doran and Jones, 1996; Neary *et al.*, 2010). A number of soil physical, biological and chemical parameters which have linkages to soil productivity have been proposed as a minimum monitoring set for screening the condition, quality, and health of soils relative to sustaining productivity (Doran *et al.*, 1998; Burger *et al.*, 2010; Johnson, 2010). Evaluation of soil condition would lead to a time-trend analysis that could in turn be used to assess the sustainability of land management practices and bioenergy programs. However, even though sustainability is the stewardship goal of land management, more specific definitions of its goals and attributes are often complex and open to wide interpretation (Allen and Hoekstra, 1994; Moir and Mowrer, 1995). As noted earlier, many scientists have attempted to answer the “what”, “what level”, “for whom”, “biological or economic”, and “how long” questions of sustainability. Allen and Hoekstra (1994) clearly pointed out that there is no absolute definition of sustainability, and that it must be viewed within the context of the human conceptual framework, societal decisions on the state of ecosystem to be sustained, and the temporal and spatial scales over which sustainability is to be judged. In short, this approach is loaded with considerable uncertainty and lack of consensus.

Absent some breakthrough in validating a key set of soil parameters that will predict soil productivity and sustainability trajectories, the most sensible approach is the third, specifically the development, implementation, monitoring, and assessment of Best Management Practices (BMPs). A large number of BMPs for forestry and agriculture have been developed throughout the world because of national regulatory demands and the international development of codes of land management practice". The BMPs in the codes and regulations cover traditional forestry and agricultural activities. New BMPs have been developed for bioenergy applications such as energy production facilities, ash recycling, and short-rotation cropping. Best Management Practices were originally developed in the 1970s for water quality protection but now extend to other environmental concerns such as sustainability. An important part of BMP utilization is the cycle of application, monitoring, evaluation, refinement, and re-application. Research and development studies play a key part in the refinement and communication of improved BMPs. Existing studies of BMP effectiveness have demonstrated that most BMPs, if applied correctly, are very effective in mitigating or preventing adverse soil and water quality impacts. Some jurisdictions have mandatory BMPs but others operate completely under voluntary systems.

The key components of successful BMP-based codes of practice for bioenergy systems, whether voluntary or mandatory, revolve around the cyclical strategy of planning, implementation, monitoring, evaluation, adaptation and renewed implementation. The minimum number of BMPs needed should come out of the planning process and is dependent on resources to be protected, site physical characteristics, regulatory requirements, and overall organization and operation goals. These will vary by site, region, country and organization. Life cycle analysis should always be included in order to identify all water and ecological impacts. The next step is crucial: monitoring and evaluation should be conducted routinely in order to decide if selected BMPs are effective and can be reapplied, or if they need to be modified, researched further or discarded. Research and development studies play a key part in the refinement and communication of improved BMPs. They are also crucial in validating the effectiveness of BMPs. This is especially important where local environmental conditions or operational standards are unique. Best Management Practices ensure that bioenergy programs can be a sustainable part of land management and renewable energy production.

4.2.5 Conclusions

Soils are crucial for profitable and sustainable biomass feedstock production. They provide nutrients and water, give support for plants, and provide habitat for enormous numbers of biota. Sustainability is the stewardship goal of crop production and forestry in general and specifically of forest bioenergy programs. Soils are an important, basically non-renewable resource, that provide the physical, biological, chemical, and hydrologic foundation upon which agricultural and forest bioenergy feedstocks grow. Thus sustainable soil management is a prime goal of forest bioenergy programs. Soils are able to renew themselves after being degraded but the time period might be several centuries or millenia, depending on climate and vegetation. Because of this long time factor, soils are considered to be non-renewable from the viewpoint of human use and management. Soils are heterogeneous and highly diverse components of ecosystems that form over long time periods under the influence of parent mineral material, climate, landscape position and biological activity. As the basis for bioenergy production, soils are a major factor in determining the sustainability of biomass energy systems.

4.2.6 References

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4.3 Bioenergy and the carbon cycle

by Göran Berndes and Annette Cowie[4]

Bioenergy is different from other renewable energy technologies in that it is a part of the terrestrial carbon cycle. The CO_2 emitted due to bioenergy use was previously sequestered from the atmosphere and will be sequestered again if the bioenergy system is managed sustainably, even though emissions and sequestration are not necessarily in temporal balance with each other. Besides influencing the carbon pools and fluxes associated with the ecosystems that are managed and harvested, bioenergy use may also affect the global carbon cycle by causing land use change – directly, as when a forest is converted to cropland for biofuel feedstock production, or indirectly, as when farmers who lose their cropland to bioenergy projects re-establish their cultivation elsewhere by converting other land to cropland. Land use change has contributed roughly one- third (more than 150 Pg C) of the accumulated anthropogenic carbon emissions to the atmosphere since 1850 – primarily associated with the conversion of forests to agricultural land (Houghton, 2008). This is further discussed in Chapter 4.4 which focuses on land use change.

Understanding the global carbon cycle and how it is affected by activities associated with bioenergy is important in reviewing the climate change mitigation benefits of bioenergy systems. There are also many challenges for bioenergy LCAs related to how carbon flows are characterized.

4.3.1 Principal carbon pools and fluxes

The world has five principal carbon pools – fossil resources, the atmosphere, the ocean, the biosphere[5] containing all ecosystems, and the pedosphere, which is the free layer of soils above the bedrock (see Figure 1). The above-ground terrestrial part of the biotic pool contains about three times less carbon than what is stored as soil organic carbon in the soil pool and about seven times less carbon than what is stored in the fossil carbon pool. The carbon exchange rate between the biosphere and the atmosphere is on the other hand relatively high: photosynthesis fixes about 120 Pg of atmospheric carbon each year and a similar amount is transferred back to the atmosphere via plant and soil respiration.

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[4] This section is based on two reports produced for IEA Bioenergy ExCo (Berndes *et al.*, 2011; Cowie *et al.*, 2013)

[5] The biosphere is referred to here as consisting of the terrestrial biotic pool and the soil organic carbon component of the pedologic pool, consisting of humus and charcoal carbon, including plant and animal residues at various stages of decomposition; substances synthesized from the decomposition products; and the living micro-organisms and small animals with their decomposition products. The aquatic biomass (plankton, algae, etc.) is also part of the biotic pool, but since this publication concerns terrestrial ecosystems only, 'biotic pool', 'biosphere', etc. are used only in reference to the non-aquatic part of the system.

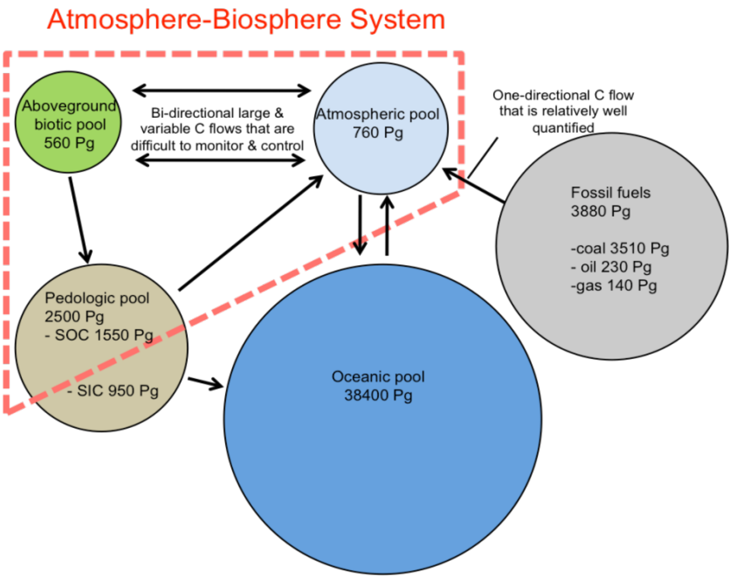

Figure 0.3: The five principal carbon pools and fluxes between them (SOC = soil organic carbon, SIC = soil inorganic carbon). The biosphere and the atmosphere together make up the atmosphere-biosphere system, which is characterised by large bi-directional flows that are highly variable from year to year, difficult to quantify, and expected to be influenced by climate change in ways not yet well understood. In contrast, the flow of carbon from the fossil pool to the atmosphere that is caused by the use of fossil fuels is uni-directional on relevant time scales and better quantified. Atmospheric carbon can – at least temporarily – be re-allocated to the biosphere, but this does not solve the problem of climate change, which is caused mostly by the transfer of fossil carbon into the atmosphere-biosphere system (Berndes et al., 2011).

About 330 Pg of fossil carbon has been emitted to the atmosphere since 1750, with two-thirds of these fossil carbon emissions taking place since 1970 (Canadell *et al.*, 2007; Friedlingstein *et al.*, 2010; Houghton *et al.*, 2012). A small proportion of anthropogenic carbon emissions to the atmosphere comes from cement production. The CO_2 fertilisation effect – elevated CO_2 levels in the ambient air stimulating plant growth – results in a portion of the carbon emissions being transferred from the atmosphere to the biosphere. Part of the carbon that is emitted to the atmosphere is also assimilated in the biosphere due to reforestation in some parts of the world. Forests in Europe and North America, for example, presently function as a carbon sink (Denman *et al.*, 2007). Quantifications of the effects of reforestation and CO_2 fertilisation are uncertain, but estimates indicate that the biotic pool as a whole is presently a net sink of carbon, despite the biospheric carbon losses associated with land use and land use change (Denman *et al.*, 2007).

There are other naturally occurring processes that remove carbon from the atmosphere, in particular the uptake of atmospheric CO_2 by the ocean, which has taken up approximately 40% of anthropogenic-sourced CO_2 from the atmosphere since the beginning of the industrial revolution (Reid *et al.*, 2009). However, the long time-scales characterising such processes make them insufficient to counter-balance the effects of human activities influencing different carbon pools (Canadell *et al.*, 2007; Cotrim da Cunha *et al.*, 2007).

4.3.2 Influence of bioenergy on carbon stocks and flows

The production and use of biomass to displace fossil fuels reduces the transfer of fossil carbon to the atmosphere but, as noted above, bioenergy systems can also influence carbon pools and fluxes in other ways. Besides the influence on carbon pools and fluxes associated with the ecosystems that are managed and harvested for bioenergy – and the influence associated with bioenergy-induced land use change, discussed in Chapter 4.4 – there are several options for removing carbon from the atmosphere that can include a bioenergy component:

- Land application of bio-char produced via slow pyrolysis can be combined with biofuel production and can also improve the structure and fertility of soils. The stability of the bio-char will depend on the type of feedstock and production conditions as well as soil properties. However, large amounts of bio-char-derived carbon stocks remaining in Amazonian dark earth soils today indicate possible residence times of many hundred years (Lehmann *et al.*, 2006; Fowles, 2007; Lehmann, 2007; Gaunt and Lehmann, 2008; Laird, 2008; Major *et al.*, 2010)
- The production of long-lived structures from biomass, such as buildings and furniture, provides an option for removing and keeping carbon away from the atmosphere for a time period that is long enough to be relevant to near-term GHG targets, but is commonly shorter than what should be required from long-term storage options. One advantage, however, is that the products can be used to generate bioenergy, replacing fossil fuels when they have served their original purpose. However, the total carbon storage potential is small compared to the estimated carbon mitigation requirements associated with ambitious stabilisation levels for atmospheric CO_2 (Nunery and Keeton, 2010; Ingerson, 2011; Earles *et al.*, 2012; Stockmann *et al.*, 2012).
- When combined with technologies for carbon capture and storage (CCS), bioenergy can provide energy services while generating so-called 'negative CO_2 emissions', i.e. a net flow of carbon from the atmosphere to geologic CO_2 storage reservoirs. Capturing CO_2 from biomass-based processes, such as sugarcane-based ethanol mills and chemical pulp mills, is one possibility that has been suggested, and biomass could also be used as fuel in power generation in association with capture. However, since the economics of CCS assume large-scale units and high thermal efficiency, CCS applications may not be straightforward if considering biomass-only fired units, due to the logistic and other challenges associated with managing large biomass flows and the risk of high-temperature corrosion under conditions needed to reach high conversion efficiency (Rhodes and Keith, 2005; Azar *et al.*, 2006; Azar *et al.*, 2013; Ricci and Selosse, 2013; van Vuuren *et al.*, 2013).

Similar to LCAs of biofuel production processes that create multiple products, LCAs of bioenergy systems such as those mentioned above face special challenges since they need to consider how the influence on carbon pools and fluxes can be factored in. For instance, when bioenergy systems are part of cascading biomass cycles in which co-products and biomaterials themselves are used for energy after their useful life, space and time aspects need to be considered since GHG emissions and other environmental effects can be distributed over long time periods and take place at different geographical locations.

Bioenergy systems may also be associated with specific cases of land use intended to address specific concerns. For instance, in forested lands susceptible to periodic fires, good silviculture practices can lead to less frequent, lower intensity fires and can improve site conditions for replanting leading to higher growth and productivity, i.e. accelerated forest growth rates and soil carbon storage. Using biomass removed in such practices for bioenergy can reduce emissions of greenhouse gases and particulates by controlled burning of biomass that might otherwise burn in open-air forest fires. LCAs may here need to consider in a baseline situation the long-term fate of biomass that is not harvested for bioenergy. Delayed greenhouse gas emissions and sequestration of carbon in continously growing trees might make a baseline lacking silvicultural fellings beneficial from the perspective of near-term net carbon balances. However, fires, insect outbreaks and other natural disturbances can quickly convert a forest from net sink to emitter. In one noteworthy example, mountain pine beetle-killed wood in North American forests is a fire hazard and will – if not harvested – either burn in wild fires or decay and release carbon back to the atmosphere. The removal of such wood can instead provide a feedstock for bioenergy applications, but may come out as non-beneficial from a carbon balance point of view if the baseline assumes a relatively slow release of carbon to the atmosphere through natural decomposition processes.

4.3.3 The case of bioenergy in long rotation forestry

The extraction and use of biomass for energy as part of long-rotation forestry systems represents a specific case in which the dynamics of terrestrial carbon stocks become a challenge for LCA. The temporal imbalance of carbon dynamics is substantially different for bioenergy use on the one hand and decomposition/re-growth processes in the forest ecosystem on the other hand. Depending on system definition, including spatial and temporal scales, and characterization of baseline, LCA results can differ greatly.

As an illustration of the carbon dynamics of long-rotation forestry systems, Figure 4.7 shows the modelled *differences* in accumulated harvested biomass and soil carbon change between one scenario in which only stems are harvested and two scenarios in which slash and stumps are also extracted. The net carbon effect of changing the biomass extraction to include felling residues that can be used for energy is demonstrated. The figure shows the carbon dynamics at the stand level as well as how the carbon dynamics for individual stands are averaged to create steady trends at the landscape level (i.e. a forest estate with equal areas of equal age class).

As can be seen, the landscape-level soil carbon stocks initially decline in response to intensified harvests but stabilise over time. Compared to the carbon in the additional accumulated harvest, there are no major changes in soil carbon stocks at the landscape level. The losses in ecosystem carbon are considerably less than the carbon in corresponding withdrawals. If the additional biomass that is extracted in the scenarios with intensified harvests displaces fossil fuels, the benefit of increased harvest occurs immediately and grows linearly over time whereas the associated loss of soil carbon has a declining response at the landscape level. From a climate change mitigation perspective, the relative attractiveness of increased biomass extraction depends on the fossil fuel displacement efficiency associated with the use of the harvested biomass.

Figure 0.4: Effects on the carbon balance of increased removal of felling residues in a Norway spruce forest in south Sweden. In the "Stems Only" scenario, harvest residues are left on the ground after both thinning and final felling. The "Stems & slash" scenario involves extraction of 80% of the logging residue after thinning and final felling and the "Stems, slash & Stumps" scenario includes in addition the removal of 50% of stumps-coarse root systems at final felling. The increased residue removal continues over the whole 300-year period. Upper panes show the amount of removal in comparison to the "Stems Only" scenario and lower panes the corresponding variation in soil C. Single stands are plotted behind the landscape averages in the foreground. The sharp declines in stand level soil carbon shown at each harvest are caused by the removal of residues, reducing litter addition to soil C. Source: Eliasson et al. (2011).

As shown in Figure 4.7, increasing extraction from forests of biomass for any purpose (not just bioenergy) may cause a short-term decrease in carbon stocks over the business as usual situation, unless it accompanied by activities that promote regeneration, enhance growth or decrease decomposition rates. Thus, although carbon stock losses are small compared to the increase in accumulated harvest in the longer term, evaluations that assess the contribution of individual forest bioenergy projects to nearterm GHG reduction targets may well come to the conclusion that it would be better to leave the biomass to decay in the forest and to continue using fossil fuels. The project level approach to evaluating different options, such as LCA, has limitations and needs complementary consideration of forest bioenergy and associated carbon flows at the landscape level as well as consideration of how forest management is affected by the promotion and growth of bioenergy. Forest management will likely change in response to changing demand for forest biomass (including for bioenergy). This influences forest carbon flows and can lead to increased or decreased forest carbon stocks. Active forest management can ensure that increased biomass output need not take place at the cost of reduced forest stocks.

Shortening the length of forest rotations to obtain increased output of timber and biomass fuels leads to decreased carbon stocks in living biomass (all things being equal). Intensified biomass extraction in forests, e.g. for bioenergy, can lead to a decrease in the soil carbon or dead wood carbon pool compared to existing practice. Conversely, if changed forest management employing intensified extraction also involves growth-enhancing measures, forest carbon stocks may increase (Sathre *et al.*, 2010). For example, site preparation increases forest productivity by drastically shortening the regeneration phase, but may also lead to faster decomposition of soil organic matter (Johansson, 1994). Fertilization is another management option that has long been used to increase forest production; experiments show that stem volume production, even in already highly productive forests, can be more than doubled with optimal fertilisation and, when needed, irrigation (Bergh *et al.*, 1999). This increased stem volume production should result in an approximately equal increase in litter production (although fine roots may not respond as much) and a similar long-term increase in soil C. It has also been shown that fertilisation slows down decomposition in boreal forests contributing further to increase in soil carbon stocks (Ågren *et al.*, 2001; Franklin *et al.*, 2003). Other studies of possibilities to intensify forest management (particularly in the boreal forests) confirm that stand management can increase the carbon stocks as well as the biomass and timber production on the same piece of land (Alam *et al.*, 2010; Routa *et al.*, 2011a; b).

To the extent that increased demand for forest bioenergy makes such measures feasible (i.e., they would not have taken place in a scenario without bioenergy demand), effects of changes to forest management practices should be considered when evaluating the climate change mitigation benefit of forest bioenergy. This has implications for how evaluation frameworks should be designed. Commonly, the time period for the evaluation starts at the point of biomass extraction, with the consequence that forest management prior to extraction is not considered. The two cases of forest carbon accounting in Figure 4.8 illustrate how this accounting design prevents a holistic evaluation of forest bioenergy operations and how management of long rotation forests contributes to climate change mitigation. The two cases start at different years and both use a 20-year time frame:

- in A, the starting point for accounting allows consideration of the higher growth rate – and consequently higher carbon sequestration – achieved from fertilization (the unfertilized forest might be used as a reference case). The biomass extraction during thinning operations is quite soon compensated for due to the rapid growth rate.
- In B, the accounting starts at the time when final harvest takes place. If a sufficiently large part of the extracted biomass is used for bioenergy or in short-lived products, the accounting will in most cases (depending on energy system configuration) show a large net carbon emission to the atmosphere, since there is little time for forest regrowth before the accounting period ends.

The net amount of carbon emissions will depend on fossil carbon displacement efficiency and the length of time for the forest regrowth to compensate for the biomass extraction – or, in the case of forest residue extraction where the alternative (reference) situation is to leave the residues in the forest, the residue decay rate (Melin *et al.*, 2009; Palviainen *et al.*, 2010). However, “Case A” in Figure 4.8 would clearly appear to be much more favorable for the climate than “Case B” in an evaluation that narrowly considers a distinct forest bioenergy project (either A or B) and that uses a relatively short time horizon. Yet, both “Case A” and “Case B” are components of the same forest management regime that have indisputable net substitution benefits (lower diagram in Figure 4.8).

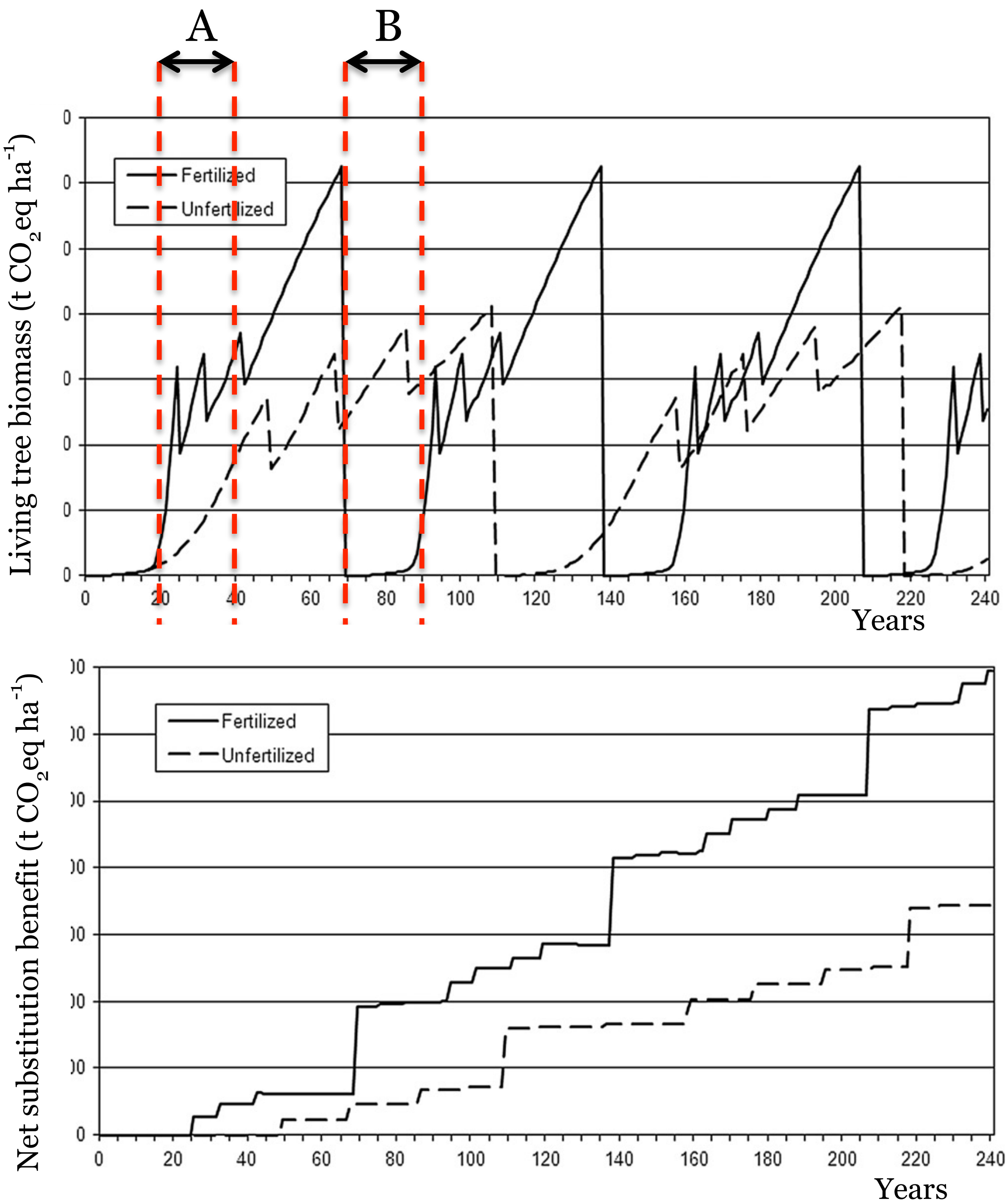

Figure 0.5: Development of carbon stocks and GHG flows over a 240-year period for typical fertilized and unfertilized stands in northern Sweden. The top diagram shows living tree biomass and the bottom diagram shows net substitution benefits of wood product use assuming coal reference fuel, with deductions made for N_2O, CH_4 and fossil CO_2 emissions. The dynamics of carbon in soils and dead biomass (not shown) are highly influenced by forest management but occur at a smaller scale (fluctuations are within 250 t CO_2 ha^{-1}). A and B denote two possible cases of forest bioenergy accounting (see text). Source: based on information in Sathre *et al*. (2010).

4.3.4 Biospheric carbon sinks as alternative to bioenergy

Society can also employ other approaches to actively relocate atmospheric carbon to the biosphere. The conversion and management of ecosystems with the aim of creating so-called biospheric carbon sinks for assimilation of atmospheric carbon has often been proposed as a land use option for climate change. Examples of this would be afforestation of sparsely vegetated areas and various approaches to cropland management to increase soil carbon content. Such options are sometimes proposed as an alternative to using land for bioenergy and in LCAs it can be relevant to consider the creation of carbon sinks as one alternative when defining the baseline land use.

Using the creation of carbon sinks as a baseline in bioenergy LCAs might be correct in cases where this would be a more likely future land use than the continuation of the present state. However, developing such a baseline involves significant challenges related to how the two principal land use options – bioenergy and the creation of carbon sinks – differ in their contribution to climate change mitigation and in relation to other aspects of land use.

First, the carbon sinks option differs from the option to produce biomass for fossil fuel displacement in that it does not prevent emission of fossil carbon to the atmosphere but rather relocates carbon from the atmosphere to the biosphere. The carbon is still kept within the unstable and dynamic part of the atmosphere-biosphere system. Due to the difficulties of monitoring and quantifying biospheric carbon stocks and flows – and the risk that future events such as climate change impacts, fires, and future LUC may lead to sequestered carbon being transferred back to the atmosphere – there is concern and debate regarding the permanence of different biospheric carbon sink options and how they can be verified (Watson *et al.*, 2000; Smith, 2005; Galik and Jackson, 2009; van Kooten, 2013) There are also diverging views about whether temporary carbon storage in sinks contributes to climate change mitigation (Kirschbaum, 2003a, b; Kirschbaum, 2006; Dornburg and Marland, 2008). Some critics further state that carbon sinks distract from the necessary transformation of energy systems rather than buying time for developing emission reduction technologies (Kaiser, 2000; Smith, 2008).

Secondly, land cover changes may alter the surface albedo and evapotranspiration, which influence the climate system on varying scales. Global warming is also influenced by non-CO_2 greenhouse gases. Thus, considering only the net carbon effect can lead to wrong conclusions concerning the climate change mitigation benefits of different land use options. Tropical forest systems in particular appear to have significantly reduced capacity to reduce global warming potential (GWP) as carbon sinks due to N_2O emissions, possibly from rapid N mineralisation under favourable temperature and moisture conditions. Net GWP contributions from wetlands are large, primarily due to CH_4 emissions (Dalal and Allen, 2008). The cooling effect that is associated with evaporation of water to the atmosphere is another factor. Especially in tropical areas, this evaporative cooling may compensate for the albedo change effects of afforestation/deforestation.

In regions with seasonal snow cover or a seasonal dry period, e.g. savannahs, reduction in albedo due to the introduction of perennial green vegetative cover can counteract the climate change mitigation benefit of the associated carbon sequestration and/or bioenergy production. The land becomes darker, i.e. less reflective, so albedo is reduced and more solar energy is absorbed leading to increased warming. Under specific circumstances, the warming effect of albedo changes associated with afforestation counteract the cooling effect of most of the carbon sequestered in the forest. Conversely, albedo increases associated with the conversion of forests to agricultural land (annual crops and grasses) may counter the global warming effect of CO_2 emissions from the deforestation (Bonan, 2008; Arora and Montenegro, 2011; Betts, 2011; Anderson-Teixeira *et al.*, 2012; O'Halloran *et al.*, 2012; Hallgren *et al.*, 2013)

Thirdly, as for bioenergy, there are concerns about effects on local livelihoods as well as the biodiversity impacts of reforestation and forest management when prioritising carbon sequestration (Krcmar *et al.*, 2005; Ninnik and Bizikova, 2008). As with bioenergy plantations, biospheric carbon sinks that are developed with close attention to local environmental and socio-economic circumstances, and are suitably integrated with existing agricultural activities can provide additional benefits such as improved livelihoods, biodiversity preservation, reduced erosion and eutrophication load from agricultural land, improved soil and water quality, rehabilitation of degraded ecosystems and increased crop yields (van Wesemael and Lambin, 2001; Updegraff *et al.*, 2004; Mayer *et al.*, 2007; Palma *et al.*, 2007; Berndes *et al.*, 2008; Dimitriou *et al.*, 2011; Reubens *et al.*, 2011; Pawson *et al.*, 2013).

Thus, the carbon sinks option differs in many important ways from the bioenergy option in regard to how it influences climate change and, depending on how sink projects are developed, their influence on other environmental and socio-economic conditions can vary widely. Defining a carbon sinks baseline therefore involves many speculative assumptions and the outcomes of bioenergy LCAs are obviously sensitive to how these assumptions are made.

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4.4 Land use and land use change

by Serina Ahlgren and Johannes W.A. Langeveld

Land use (as in land occupation) is often included as an inventory flow or impact category in bioenergy LCA studies. However, the integration in LCA procedures of Land Use Change (LUC) and the related impacts has been cumbersome. In this section, two major reasons for this are identified. First, following Wolf *et al.* (2012), it is stressed that with current data, assessment methods and review procedures, it is difficult to include LUC impacts in LCAs in a timely and cost-efficient way and so that sufficient quality of results can be guaranteed.

The second reason for difficulties in integrating the impacts of LUC in LCAs is the complex character of land use change which requires multi-disciplinary analytical tools that can evaluate dynamic, multi-scale processes. Although some inherent uncertainties, such as predictions of the future will never be overcome, steady progress can be expected in the field of data collection and analysis and reviewing procedures. In anticipation of that, this section provides a thorough overview of present insights in land use, both in a broad sense, related to the role that land plays in the production of food, feed, products and services, and in the context of LCA.

We first provide an overview of global land cover and land use (4.4.1). Next, we define types of land use change (4.4.2), and explain how it is quantified using models (4.4.3). We then explain the impact land use change can have (4.4.4) and how impacts of biofuel production depend on local conditions (4.4.5), discuss how LUC can be incorporated in LCA studies (4.4.6), and finally draw some conclusions and discuss how negative impacts of LUC can be prevented (4.4.7).

4.4.1 Land cover and land use

Studying land cover is the primary tool for assessing changes in land use, helpful in determining how natural resources management relates to land, but land use also affects vegetation, biodiversity, water and other natural resources. Over time, changes in land cover have played a major role in discussions related to processes of deforestation, biodiversity conservation, climate change, and water management that are central to the way human life is supported by natural resources on this globe.

Relating to global land cover patterns, main types of land cover identified in the land resources database of the FAO (http://faostat3.fao.org/faostat-gateway/go/to/search/land%20cover/E), include the following types of land cover: forest, inland water, agricultural land and other land. 'Forest' refers to land with a permanent forest cover; 'other land' may include roads and cities, but also mountains, land with permanent ice cover and deserts. 'Agricultural land' includes permanent grassland, arable land and land with other permanent crops. It may be idle, cultivated or under fallow (Figure 4.9).

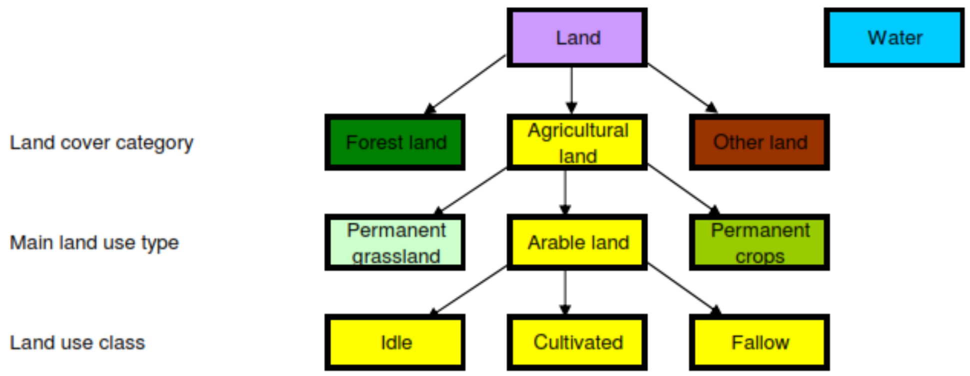

Figure 4.9: Major types of land cover and land use.

In this section, forest land, agricultural land and other land are referred to as types of '***land cover***'. '***Land use'*** refers to land use categories, which may be classified by crop type (arable, grassland or permanent crops), forest type (commercial, natural, etc.), or other types of land use (urban, infrastructure, etc.). Land use types are more or less stable in time; changes in actual land use are tracked by '***land use class***' (Figure 4.9). Land cover and land use types are mutually exclusive. For example, land can only be classified as 'agricultural land', 'forest' or 'other land'. In practice, however, this can lead to difficulties as the distinction between forest and agricultural is not always clear. Forests may be used for the collection or cultivation of food crops or to herd cattle; agricultural and other land may have trees that are used for collection of wood or other products.

The total area of the globe covered by the main land cover types is presented in Table 4.5. The earth surface contains 135 M km^2 of landmass, of which 5 M km^2 is inland water and some 130 M km^2 land. Most of the land area is classified as agricultural land (49 M km^2). Forest and other land types each cover about 40 M km^2.

Table 4.5: Global area covered by main land cover types.

	Area (M km^2)	Definition
Forest land	40.0	Permanent forests for non-agricultural use. Includes non-natural forests for commercial use. May be difficult to distinguish from forested agricultural land.
Agricultural land	49.0	Mostly covered by crops for food, feed or industry, mainly used for agricultural or industrial purposes.
Other land	41.0	Cities, infrastructure, other industrial areas (e.g. mines), nature areas, permanent ice, mountainous areas, deserts. May be difficult to distinguish from extensively used agricultural land, especially when using remote sensing.

Source: FAOSTAT

Table 4.6 presents an overview of agricultural land use categories. Most agricultural land is permanent grassland, i.e. it is covered with grasses or other plant species used for grazing of animals. Less than one third of all agricultural land is classified as arable land. Permanent crops are generally perennial crops used to produce food, food or industrial feedstock, and include fruit and nut tree crops, coffee, tea and other beverage crops. Distinguishing between main land use types is not always straightforward. Arable land, for example, may include rotational grassland; arable land left under fallow can develop a cover of wild grassland species.

Table 4.6: Use of agricultural land worldwide.

Land use	Area (M km^2)	Type of use	Remarks
Permanent grassland	33.6	Unused, extensive (low animal density, no fertilization) or intensive use (active management and/or high animal density)	Distinguishing between unused and extensively used is very difficult.
Arable land	13.8	Unused, extensive (<1 harvest per year), normal (one harvest) or intensive (>1 harvest)*	Includes fallow (temporally unused), set aside (formalized fallow), abandoned land and non-permanent grassland
Permanent crops	1.5	Unused, extensive (collection) or intensive (actively managed)	Includes industrial tree and fruit crops, used or abandoned

Source: FAOSTAT. *Average number of crops per year is referred to as 'Cropping Intensity' (CI; area harvested/area of arable land). CI can vary between 0 and 2 or 3 but in practice, it usually is between 0.3 and 0.9.

4.4.2 Changes in land cover and land use

It can be very difficult to distinguish clearly between different land cover or land use types. Among other reasons, this is due to the fact that land use can be very dynamic, often showing considerable changes both within a single year and between years. Land management is a continuous process, including major decisions (e.g. what crop to grow, where and when to grow it) and minor decisions (how many inputs to use and how and when to apply them). The combined impact of these decisions determines how many hectares of a given crop are cultivated (harvested) and what the yields may be.

Farmer decision making is influenced by a range of factors including: alternative opportunities for land, time, cash and machines. In practice, decision-making on crop cultivation and management may be less straightforward than sometimes is expected. There are several reasons for this:

1. Some countries offer programs to keep land under fallow, or under restricted cultivation. The Conservation Reserve Program (CRP) in the USA, for example, provides compensation to farmers who abstain from growing a particular crop in a given year. Objective of this program, which covered 12.7 M ha in 2010 (Westcott, 2011) is to reduce soil erosion while protecting long-term food production capacity and improve water quality. A program for voluntary fallow in the European Union covered 6.9 M ha in 2007 (Areté and University of Bologna, 2008). Similar programs are available in other countries, including the Russian Federation.
2. In many regions, rainfall and weather conditions are unpredictable or even erratic and decision-making on land preparation and, especially, sowing or planting is a process with an uncertain outcome. It may lead to land that is not (timely) ploughed at the right time, crops that are sown late, harvests that are delayed or crops that are not harvested at all. This situation is common in drought-prone regions but may also occur in more humid areas (e.g. following heavy rains, storms or hail events). There is evidence that the frequency and extent of extreme weather events is increasing (IPCC, 2012).
3. Under favourable ecological and weather conditions and when sufficient water and other inputs are available, farmers may be able to realise multiple harvests in one year. This situation is found in the humid tropics, where two or three crops often

are cultivated in succession, in subtropical regions when water is not limiting and even in temperate zones (e.g. relay cropping or in horticulture). As a rule, the decision to grow a second or third crop is only made after the first crop has been harvested.

4. Crop cultivation in many countries is an activity determined by economics. If the expected economic return is considered insufficient, farmers may limit their efforts in time devoted to, or inputs applied to the crop. Such inputs affecting yields include plant density and the application of fertilizers or crop protection agents. These often represent considerable costs, which farmers obviously want to recover. They may also abstain from cultivation if they feel the risk of losing money is too high.

By influencing decision-making about crop cultivation and management, these processes interfere with other factors that determine the prospects for farming. Thus, the response of farmers to policy measures may be less straightforward and predictable than expected. This may affect the impact of the growing demand for bioenergy, as observed by several authors, including Bauen *et al.* (2009), IPCC (2011) and Lambin and Meyfroidt (2011). As a rule, increasing demand for bioenergy crops leads to changes in land use (LUC) and crop management. The former may include changes in land cover which generally represent state changes, e.g. from forest to grassland or from grassland to arable land. Such changes are usually irreversible and may have dramatic impacts on the quality and composition of the land, e.g. number and type of crop species, amount of soil organic matter, water cycle.

Changes in land cover therefore need to be accounted for when the impacts of bioenergy projects are evaluated. The exact effect of changes in the demand for bioenergy products depend on (i) decision-making by farmers on land use and crop management (determining biomass availability) and (ii) the behaviour of consumers and companies that use biomass for non-bioenergy applications (determining the demand for bioenergy feedstock). Although the concept of LUC was already discussed in the 1990s (Leemans *et al.*, 1996; Marland and Schlamadinger, 1997), debate on the impact of LUC on bioenergy GHG balances peaked following the publication of two studies in 2008. Both Fargione *et al.* (2008) and Searchinger *et al.* (2008) demonstrated that indirect LUC could increase carbon emission following biofuel expansion to such a level that the studied biofuels should be considered net emitters of GHGs at a level even higher than that of fossil fuels.

Two types of land use change can be distinguished: direct land use change (dLUC) and indirect land use change (iLUC) (Figure 4.10). An increasing demand for Crop A will have a range of consequences. First, higher demand can be expected to affect the market price. Higher food prices could reduce the demand for this crop. Market prices could also affect decision-making by farmers about land use and crop management, e.g. a higher price for Crop A can lead to intensification of production and/or expansion of production on previously unmanaged land such as permanent grassland or unmanaged forests. Crop A could also displace other land uses such as pasture or cultivation of another crop (Crop B). These land use changes are *direct* land use changes and are associated with the location where cultivation of Crop A is taking place.

All other land-use-related impacts of expansion of Crop A are considered *indirect* land use changes (iLUC). These include market chain reactions following the displacement of previous activities, e.g. when Crop B in its turn is displacing other crops. Indirect land use change thus is steered by physical processes (bringing a crop to a plot where it was not cultivated before), economic processes (reacting to or anticipating crop price changes) or both. It may occur in the same location where Crop A expanded or elsewhere, through a web of reactions.

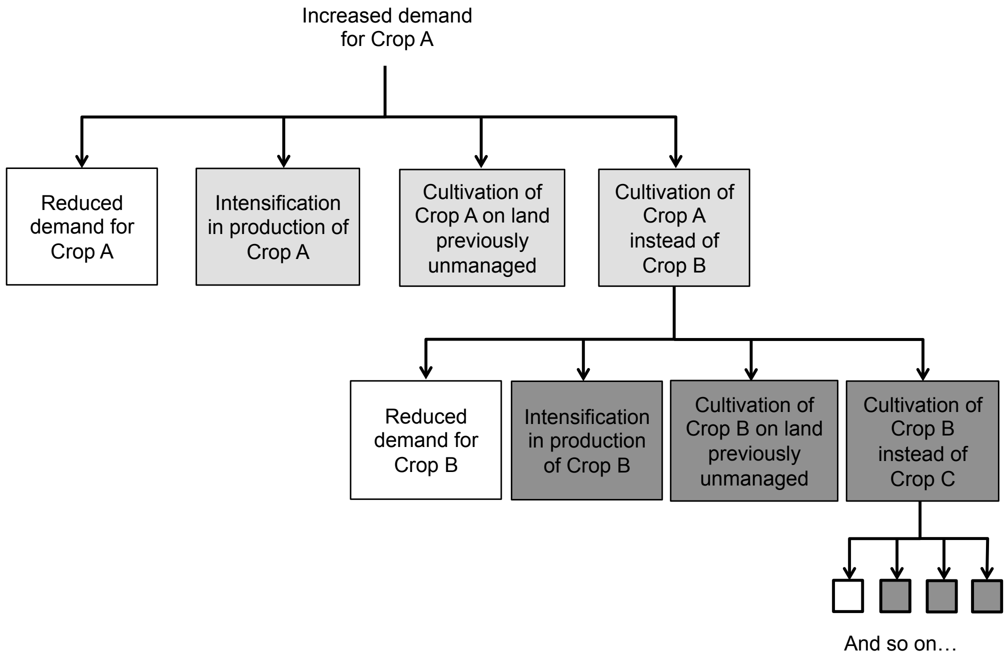

Figure 4.10: Impact of increased demand for a crop on direct (light grey boxes) and indirect (dark grey boxes) labnd use change.

It is important to note that by-products generated during biofuel production also can lead to land use change. LUC caused by changes in by-product availability may have a large impact. DDGS (Distillers Dried Grains with Solubles), for example, is a by-product of ethanol production from cereals, and is used as animal feed. Expansion of ethanol production from maize increases the availability of DDGS, which then will replace other types of animal feeds, including soybean-based feeds. Replacement of those feeds will reduce the land area required for soybeans. This positive effect should be incorporated in LUC calculations of the ethanol production process that is generating the DDGS.

Similar reasoning could (and should) also be applied to LUC caused by the production of food, feed, solid bioenergy, etc. Even the mining of fossil fuels may cause direct and indirect land use change, as when large areas of land are disturbed, e.g. for open-pit mining).

4.4.3 Approaches to quantifying land use change

Assessing land use change from biomass production for energy requires that sufficient information is available on the conditions under which biofuel feedstocks are produced. If we want to determine the impact of dLUC, we need to know (i) which feedstock (e.g. maize grains or palm oil) has been used to produce the biofuel, (ii) where these crops have been grown, and (iii) how the introduction of the biofuel crop has affected land and soil organic carbon.

In practice, it often is extremely difficult to find out where a specific unit of biomass has been grown. Biofuel feedstock can be used for different applications, but generally crop products are first collected from farmers in a central location, and then distributed to different places where they are converted into food, feed, or fuels. This makes it almost impossible to know exactly on which piece of land a crop was grown. In some cases producers are therefore required to have a system for tracking and tracing of the feedstock they use. This is the case within the EU as required by the EU Renewable Energy Directive.

Quantifying the magnitude and location of iLUC is even more complicated. In addition to the information we need to determine the impact of dLUC, we also need to know (iv) what crops, if any, were previously grown on the land where the biofuel crop was cultivated, and (v) how this is affecting land use and land cover. If sugar cane is expanding at the expense of grassland, we need to determine how farmers are now feeding their livestock. Do they open new lands to graze their animals, or do they apply more fertilizers to increase yields of remaining grasslands? Things become even more complicated if farmers purchase more hay or cereals to make up for the lost grazing. Then we need to determine where these feeds are cultivated, and how this affects land use in that region.

As indirect land use changes occur in response to a complex set of interconnected economic, biophysical and institutional factors (as explained in the beginning of section 4.4.2), quantification and establishment of causal chains remain difficult and uncertain. Existing analytical methods either use complex economic models (market analysis) to determine how crop cultivation and land use are influenced by the increasing demand for biofuels, or are based on more straightforward generic causal calculation methods (causal analysis). Both methods have their advantages and disadvantages.

Economic equilibrium models are used to study changes in economic development at the national or international level (general equilibrium models), or in a specific economic sector such as agriculture (partial equilibrium models). All equilibrium models are based on the assumption of perfect market organisation and that equilibrium is reached when demand equals supply in the studied economy. Economic models have been developed and used by researchers for many years. However, most economic models are complex and require in-depth understanding of the way they are organised (Fritsche *et al.*, 2010; Nassar *et al.*, 2011). Transparency is further reduced for some models which are not publicly available or lack proper documentation (van Tongeren *et al.*, 2001).

Economic models typically assess changes associated with the implementation of policies such as biofuel policies under a set of assumptions for population growth, economic growth, demand for food and other social and economic parameters. Land use change is often calculated as the difference between two scenarios, one with and one without implementation of a policy that stimulates biofuel production. Such a model attributes all land use change occurring over a period of time due to the specific policy, which means it cannot differentiate between dLUC and iLUC.

Several casual descriptive models have been developed. One special case is the method described by Bauen *et al*. (2010) in which land use scenarios are created by a reference expert group. Other studies have used statistics of past land-use to predict future land use changes (e.g., Tipper *et al.*, 2009; Fritsche *et al.*, 2010). Just as the economic models, outcomes of causal descriptive models are highly variable.

4.4.4 Impacts of land use change

The models described above can evaluate the expected amount of LUC in different regions. However, the *impact* of land use change needs to be further assessed. For this, other spatial allocation models can be used together with assumptions of carbon stock changes. Due to the complexity in the modeling the resulting emissions from iLUC show large variations (Figure 0.6). As a comparison, life cycle emissions from 1 MJ fossil fuel are 83.8 g CO_2e MJ^{-1} according to the EU Renewable Energy Directive 2009/28, but including indirect emissions from production and distribution could lead to emissions from petrol, up to 195 g CO_2e MJ^{-1}, according to Liska and Perrin (2009).

There are many reasons for the large divergence of the model outcomes. The choice of model is highly relevant; models have different geographical and sectoral scope as well as different start and end points. It is also of importance to know how a model incorporates changes in demand for and supply of biomass unrelated to biofuels, e.g. changes in human consumption, intensification of agricultural production. Models also vary in the way changes in availability of biofuel by-products are treated. Another important element is which data sources are used to determine prices, elasticity of demand, harvest yields, biofuel yields etc. Further, the type of land that is assumed to be affected, greenhouse gas emissions associated with the land use changes, and the number of years the emissions are allocated over are vital to the outcome of iLUC assessment studies. Many published studies explore the differences in model assumptions, e.g. Prins *et al*. (2010), Khanna and Crago (2011), Nassar *et al*. (2011), Broch *et al*. (2013), Warner *et al*. (2014).

There is no doubt that indirect land use can occur and can be relevant. If iLUC is not considered in LCA, it could lead to misleading conclusions, which could cause the promotion of policies provoking more emissions than had been anticipated. On the other hand, an overestimation of GHG emissions caused by iLUC could seriously limit the prospects for bioenergy production. The overall challenge is to provide a solid and reliable estimate of the amount of indirect land use change and its implications.

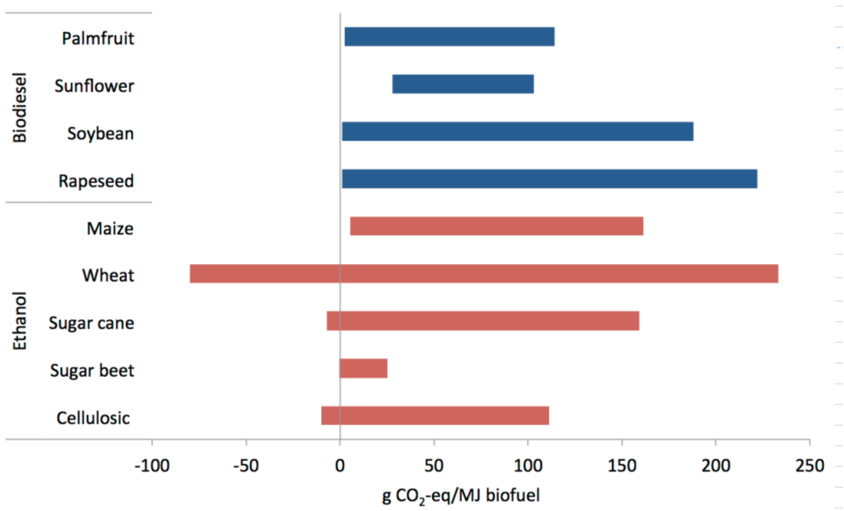

Figure 0.6: Ranges of model-based quantifications of LUC (dLUC + iLUC) emissions associated with the expansion of selected biofuel/crop combinations amortised over 20 years. Based on literature review of LUC modelling in Ahlgren and Di Lucia (2014). Note that there is no statistical processing of data involved, this only shows the range of the reported results in literature, including extreme values. Cellulosic ethanol refers to modelling of switchgrass, miscanthus and maize stover.

Land use (both dLUC and iLUC) causes various environmental impacts. At the moment research and policy mainly focus on greenhouse gas emissions related to land use, but other effects such as changes in carbon and water cycles and storage, impact on soil quality and soil net productivity, and consequences in biodiversity loss are growing in importance (Mattila *et al.*, 2011)

4.4.5 Site dependency

Many factors help to determine how much direct or indirect land use change will be caused by a given change in biofuel production. The area of land that is involved and the subsequent changes in land use (local or elsewhere, direct or indirect) by and large depend on:

1. Land use characteristics in the biofuel production area, e.g. the ratio between agricultural and other land use types, the share of grassland in agricultural areas, cropping intensity and options to increase crop production.

2. General farming conditions (intensive vs extensive farming practices, amount of fallow, commercial vs subsistence market orientation).

3. Food and land market dynamics (increasing vs declining demand for food, changes in food patterns).

The way in which these factors can cause or impact land use change are complex and interdependent. Some general rules can, however, be determined. First, direct land use change is likely to be smaller if (i) the cropping intensity is sufficiently low (allowing more intensification), (ii) the share of grassland in agricultural area is high (allowing conversion to arable land), (iii) the share of fallow land is high (e.g. under specific fallow programmes) and/or (iv) farming is extensive (allowing intensification).

Direct land use change is likely to be higher in regions where options for intensification are limited, i.e. fallow area is small or cannot be reduced, cropping intensity is high, share of grassland in agricultural land is low. Under these conditions, increasing demand for biofuel feedstock may lead to land use changes elsewhere.

Indirect land use change is likely to occur if:

1. Domestic opportunities to increase biomass production are limited, or at least small in comparison to the additional amount that would be required to generate biofuels.

2. Availability of alternative feedstock which can be used in the production of the biofuel or in replacing the biofuel feedstock in other (food, feed, fibre, etc.) markets is limited.

3. Import of biomass feedstock is relatively cheap in relation to the costs of domestic feedstock production, and is easy.

4. Quality of imported feedstock suitable for biofuel production is relatively high in comparison to domestically produced feedstock.

Thus, the likelihood that an increase in biofuel production in a given country will lead to significant indirect land use change depends on conditions related to local land use, local economic conditions, and local biofuel policies (blending obligations, subsidies, import levies, etc.). This makes it very difficult to model or project iLUC using generic models.

Given the dynamic character of land use, the large number of processes that influence decision-making in land use, and the complex and dynamic character of land itself, it becomes clear that no simple rules can be used to model land use changes. This also holds, to a certain extent, for policy making in regard to land. Regulation of land cover change in practice is one of the most difficult policy areas, and worldwide examples of land grabbing, illegal logging, over-exploitation, and corruption are amply available.

4.4.6 Integration of LUC into LCA

How to integrate LUC into LCA exercises is discussed extensively e.g. by JRC-IES (2010a). Following a concept defined by Weidema and Lindeijer (2001), a distinction is made between changes in land use (referred to as ´land occupation´) and changes in land cover (´land transformation´). Land occupation is expressed in units of area x time (e.g. $m^2 y^{-1}$), land transformation is measured in area of converted land (e.g. m^2 coniferous forest converted to arable land; see also Mattila *et al*. (2011)).

According to the ILCD handbook, LCAs should assess impacts of LUC by means of modelling. If these impacts are not included, a justification for excluding them should be presented (JRC-IES, 2010b, p. 45). It is recommended to use IPCC (2006) emission factors for modelling the impacts on CO_2 emissions of changes in soil organic matter from direct land use change. Other GHG impacts of land use (e.g. from burning of litter, soil erosion, nutrient losses) should also be quantified. IPCC parameter values are specified in the annex to the ILCD handbook (JRC-IES, 2010b, p. 113-6). The ILCD Handbook does not recommend including indirect land use change, since there is no methodology available to deal with such change. However, it could be included in the future if such methodology were to be developed. More information on integration of LUC into LCA is given in Weidema and Lindeijer (2001). Definition of indicators to assess implications of changes in land use on biodiversity and soil quality is discussed in Milá i Canals *et al*. (2006). Integration of land use into LCA is also discussed by OECD (2008).

Table 4.7 evaluates the suitability of LCA methodology to address land use issues related to biofuels. Some impacts are difficult to integrate into LCA due to lack of reliable data, indicators or consensus. Further elaboration of methodology is required for most elements, especially related to processes determined by soil and water flows (OECD, 2008).

Table 4.7: Suitability of LCA to address environmental impacts of direct land use for biofuel feedstock production.

Land use issue	**Suitability of LCA to address issues**	**Reference**
Land occupation	Yes	JRC-IES (2010a,b)
Emissions from production and use of fossil fuels and fertilizers	Yes	OECD (2008)
Soil carbon stock changes	Yes	JRC-IES (2010a,b)
Soil quality preservation	No (no impact indicator)	OECD (2008)
Water management	Partly (as water consumed and depleted)	OECD (2008)
Water pollution	Partly (not at local level)	OECD (2008)
Biodiversity	Partly (no consensus on impact indicator)	OECD (2008), Curran *et al*. (2011)

Source: adapted from OECD (2008, Table 1.3)

All the environmental impacts from a bioenergy plantation (direct land use) listed in Table 4.7 also occur due to iLUC. Including indirect effects, however, is one of the most difficult and controversial issues to be dealt with in LCA (Mattila *et al.*, 2011). It requires that economic or causal descriptive models are integrated with LCA models. So far, the focus has been mainly on GHG emissions. Many studies have simply added emissions from iLUC generated from economic modelling on top of the LCA results (sometimes referred to as iLUC-factor). Others have tried to integrate economic models with LCA in a more advanced way (Earles and Halog, 2011).

There are a number of aspects that need to be considered in combining economic models with LCA models. First of all we need to be aware that the evaluation of existing production chains and the assessment of indirect impacts need to be addressed by different types of LCA. Attributional LCAs (aLCA) are generally used in accounting for emissions of existing production systems, while consequential LCAs (cLCA) can be used to assess expected or predicted changes in production systems (see Chapter 2 for details on different LCA types and their application).

Major features of aLCAs, cLCAs and economic models are outlined in Table 4.8. An aLCA assesses the GHG balance of existing systems (or future existing systems) and does not include changes due to the introduction of a new product or service. iLUC should, therefore, in principle not be included in an aLCA (Biodiesel TechNotes, 2011). On the other hand, many economic models use average data in modelling which makes it difficult to use the results in cLCA where the purpose is to reflect the future marginal change (Brander *et al.*, 2009).

Further, LCAs and models have different approaches. While LCA is mostly used to calculate the emissions from a specific production system, economic models study changes on a global level after which impacts are allocated to single products. Consequently, the two approaches make use of data with different spatial resolutions; the applied time perspective often differs also (Table 4.8).

Table 4.8: Features of two types of LCA model and economic models.

	Attributional LCA approach	Consequential LCA approach	Economic equilibrium models
Type of question the model aims to answer	What are the total emissions from production of 1 MJ biofuel?	What is the change in total emissions when producing 1 MJ additional biofuel?	What is the land use change due to the implementation of a biofuel policy?
Scope	Specific process	Specific process	Global/regional
Perspective	Current/Future	Future	Future
Optimisation function	Not included	Not included	Profit function (often), welfare function
Marginal/Average input data	Average	Marginal	Average and/or marginal
Handling of by-products	Allocation/System expansion	System expansion	System expansion

Changing land use or land cover can have major impacts on the social, legal and cultural well-being of inhabitants. This is easy to imagine where deforestation affects indigenous tribes as is the case in the Amazon and the Far East, but in other situations also expansion of biofuel crop production may positively or negatively affect the livelihoods of many people (Sawyer, 2008; Mançano Fernandes *et al.*, 2010). According to Mattila *et al* (2011), social and economic sustainability should not be incorporated into an LCA procedure. Instead, they recommend an external certification procedure in which expert judgments are used to assess overall sustainability of changes in land use systems.

4.4.7 Conclusions

Integration of the evaluation of indirect land use change into bioenergy LCA studies is presently far from common practice. It would require adjustments to current LUC modelling approaches, which are mainly based on integrated economic modelling or more basic logical analytical tools. Both have their limitations. Fully integrating LCA with (i)LUC evaluation needs to:

1. Catch and represent dynamics of land use decision-making.
2. Model better links between land use practices and land cover change.
3. Combine elements of attributional and consequential LCA approaches and economic modelling.
4. Integrate social, legal, and cultural impact evaluation of bioenergy production.

While it remains difficult to design and implement methods to provide LCA assessments with full integration of iLUC impacts, many attempts have been made to identify biofuel production practices that limit or avoid negative impacts of indirect land use change (Gallagher, 2008; McCormick and Athans, 2010; Berndes *et al.*, 2011). Biofuel production could, of course, source feedstock from available crop wastes, residues or other non-land-intensive commodities (e.g. algae). Pressure on land use could also be limited by integrating bioenergy systems into existing land use practices while aiming at improvements in farm productivity. Another strategy could be to select regions where expansion of land use would have the least impact on the environment (e.g. cropping zones; see Dehue *et al.*, 2010; Wicke, 2011). Other options include more strict regulation of bioenergy systems through, for example, international land use agreements, global CO_2 taxes or emission rights trading systems.

Finally, it would be sensible to identify and stimulate biofuel production routes, such as production on degraded land, that can have positive impacts on land use change. Dedicated bioenergy plantations could help to sequester carbon in soils and vegetation compared to current land use while increasing soil characteristics like water holding capacity and soil fertility. The field of increasing soil quality, including fertility, has been addressed in literature related to biofuel feedstock crops like short rotation coppice (Börjesson, 1999; Langeveld *et al.*, 2012) and in more generic evaluations like IPCC (2011) which showed how soil fertility could be increased by reducing soil erosion and eutrophication or by improving biodiversity. Exciting research on this topic has been undertaken for crop residue management options in the USA, where certain levels of maize stover removal can apparently have positive impacts on nutrient leaching, soil biota and initial spring crop growth (Mann *et al.*, 2002; Cibin *et al.*, 2011).

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4.5 Biodiversity Assessment within LCA of Biomass Harvesting

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4.5.1 Introduction

Energy policy to develop alternatives to fossil fuels has stimulated interest in the use of biomass as feedstock for bioenergy. It has been argued by many that it is necessary to examine the full life cycle of biomass-derived fuels to understand the implications of their production and use (e.g. Cherubini *et al.*, 2009; Dandres *et al.*, 2012; Gaudreault *et al.*, 2012), typically using Life Cycle Assessment (LCA). It is increasingly recognized that a comprehensive set of environmental indicators should be considered when evaluating biomass-derived fuels but existing LCA studies have mostly focused on greenhouse gases and energy (Cherubini and Strømman, 2011). Land use and biodiversity are of particular importance for bioenergy systems based on agricultural or forestry feedstock (Michelsen, 2008; Cherubini and Strømman, 2011) but are very rarely considered in LCA studies (Cherubini and Strømman, 2011; Koellner *et al.*, 2013).

This section discusses approaches for assessing biodiversity impacts in LCA and implications for evaluating biomass production systems.

4.5.2 Biodiversity in the Context of LCIA

The ISO 14044 Standard (ISO, 2006b) recommends that a comprehensive set of environmental issues be considered when performing LCIA. The ISO 14047 Technical Report (International Organization for Standardization (ISO), 2012) that accompanies the ISO 14044 Standard lists nine commonly used impact categories: climate change, stratospheric ozone depletion, photo-oxidant formation, acidification, nitrification, human toxicity, ecotoxicity, depletion of abiotic resources (e.g. fossil fuels, minerals), and depletion of biotic resources (e.g. wood, fish). The Technical Report also specifies that this list is not complete and thus other impacts might also be of interest, such as those related to land use.

The Life Cycle Initiative, a joint project between the *United Nations Environment Programme* (UNEP) and the *Society for Environmental Toxicology and Chemistry* (SETAC) proposed a comprehensive LCIA framework (Jolliet *et al.*, 2004) that combines midpoint-oriented and damage-oriented approaches, i.e. through to the potential impacts for a comprehensive set of environmental indicators related to various areas of protection (AoPs) including human health, biotic and abiotic natural environment, biotic and abiotic natural resources and biotic and abiotic man made environmental. The biotic natural environment AoP quantifies the effects of exposure to chemicals or physical interventions on the function and structure of natural ecosystems, including impacts on biodiversity (EC-JRC, 2011).

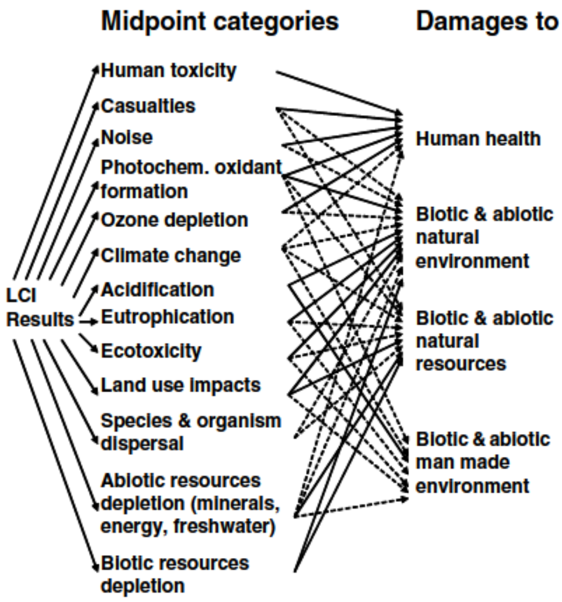


Figure 4.12: UNEP/SETAC Life Cycle Initiative Framework for Life Cycle Impact Assessment (from Jolliet *et al.*, 2004).

A number of midpoint impacts (e.g., climate change, ozone depletion, ionising radiation, photochemical ozone formation, acidification, eutrophication, ecotoxicity and land use) can potentially contribute to impacts on the biotic natural environment (Figure 4.12); however, not all these links are modelled in current LCIA methodologies.

The most common definition of biodiversity is that of the Convention on Biological Diversity (UNEP, 1992):

> *'Biological diversity means the variability among living organisms from all sources including, inter alia, terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are part; this includes diversity within species, between species and of ecosystems'*

The expansive nature of this definition makes biodiversity very difficult to quantify directly, so indirect indicators are often used, especially in LCA. These indicators often focus on conditions thought to be important for biodiversity. Hansson (2000) suggested that many features of ecosystems can be used as the basis for biodiversity indicators, such as a structural component, a process, or any other feature of the system related to the maintenance or restoration of its diversity. The UNEP-SETAC guideline on global land use impact assessment on biodiversity and ecosystem services in LCA (Koellner *et al.*, 2013) suggested that assessment of the impact on biodiversity should consider the protection of global species diversity and the functional diversity of species in ecosystems.

Because biodiversity is a complex issue, addressing it in LCIA is a challenge. Biodiversity can be considered at three different levels: ecological diversity (ecosystems), population diversity (species), and genetic diversity (genes). All these levels have been considered in different LCIA approaches but currently only those addressing population diversity are sufficiently mature for application in LCIA, according to the ILCD Handbook, which provides technical guidance for detailed LCA studies and the technical basis for deriving product-specific criteria, guides, and simplified tools (EC-JRC 2011). It is concluded in this handbook that the concept of Potentially Disappeared Fraction of species (PDF), which is species diversity-oriented and integrates the potentially lost fraction of natural species over area and time, is the only one among those investigated that is currently practical;

however, this simplified approach discounts the possibility of species gains and does not account for the many complexities of biodiversity response (see Sections 4.5.4 and 4.5.5 for further discussion).

4.5.3 Indicators of Biodiversity Impacts in LCA

Curran *et al*. (2011) and de Baan *et al*. (2012) extensively reviewed approaches and indicators for modelling biodiversity impacts in LCA, and Curran *et al*. (2011) identified three groups of relevant environmental interventions[6]: (1) resource-related (land and water use), (2) pollution-related (acidification, eutrophication, ecotoxicity), and (3) climate change. Proposed indicators are summarized in Table 1 and discussed below.

Table 4.9: Examples of Biodiversity Indicators Proposed for Use in LCA.

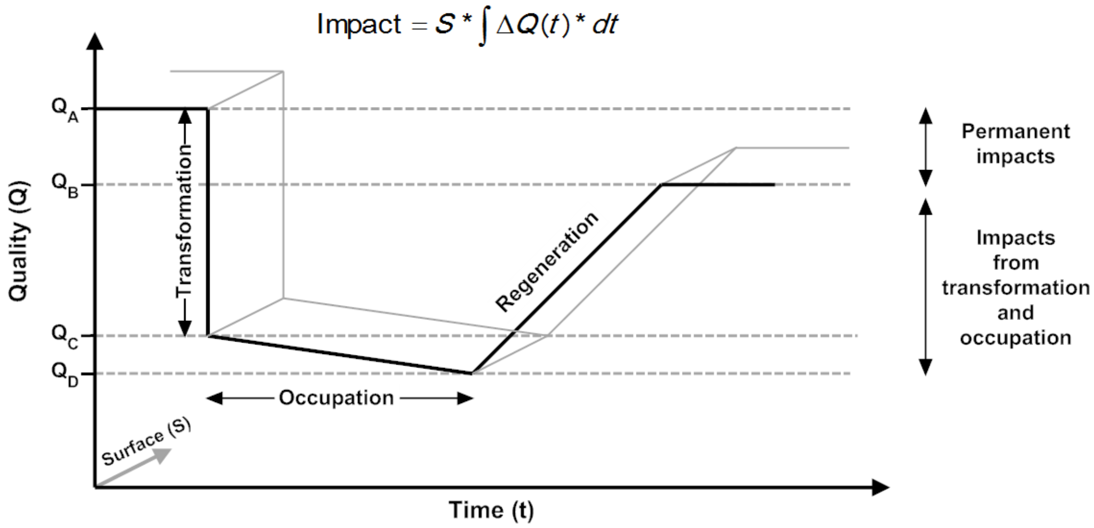
Environmental Intervention	Indicator	Reference
Land use	species richness	Lindeijer (2000), Köllner (2000), Goedkoop and Spriemsma (2001), Köllner (2003), , Koellner and Scholz (2008), Schmidt (2008), Geyer *et al*. (2010)
	species-area relationship	Köllner (2003), Koellner and Scholz (2008), Schmidt (2008), De Schryver *et al*. (2010)
	number of threatened species	Koellner and Scholz (2008), Schmidt (2008)
	regional pool of species	Koellner and Scholz (2008), Schmidt (2008)
	Fragmentation	Jordaan *et al*. (2009)
	net primary productivity	Lindeijer (2000), Weidema and Lindeijer (2001)
	potentially disappeared fraction	Eco-indicator 99 (Goedkoop and Spriemsma, 2001)
	ecosystem damage potential	Ecological Scarcity (Frischknecht *et al*., 2009)
Water use	vascular plant species diversity	Pfister *et al*. (2011)
	Potentially Not Occurring Fraction	van Zelm *et al*. (2010)
	volumetric water footprint	Water Footprint Network (Hoekstra *et al*., 2011)
Emission of Pollutants	percentage of threatened or endangered species	Eco-Indicator 99 (Goedkoop and Spriemsma, 2001), Swedish EPS methodology (Steen, 1999)
	fraction of affected species), or disappeared species	Eco-Indicator 99 (Goedkoop and Spriemsma, 2001), Payet (2006), IMPACT 2002+ (Jolliet *et al*., 2003), ReCiPe (Goedkoop *et al*., 2009)
	Potentially Disappeared Fraction	Eco-Indicator 99 (Goedkoop and Spriemsma, 2001)
	Potentially Not Occurring Fraction	ReCiPe (Goedkoop *et al*., 2009), van Zelm *et al*. (2007)
	fraction of affected species (or fraction of disappeared species)	USETox (Rosenbaum *et al*., 2008)
Climate change	increased extinction risk	de Schryver *et al*. (2009)

[6] 'Environmental intervention' describes the physical interaction between a product system and the environment. It is defined in terms of the extraction of resources; emissions to air, water or land; space occupied by waste or structures; or area of disturbance.

4.5.3.1 Impacts of Biodiveristy on Land Use

Land use is often identified as the main driver of biodiversity change (Chapin *et al.*, 2000; Millennium Ecosystem Assessment, 2005; de Baan *et al.*, 2013; Koellner and Geyer, 2013). Land use impacts relate to both the occupation and the transformation of land, the latter often referred to as 'land use change'. Occupation is the anthropogenic use of the land for a specific purpose such as agriculture, waste disposal or harvesting. Occupation-related impacts have been described to occur 'because ecosystem quality is kept at a different level than would naturally/otherwise be present' (Koellner *et al.*, 2013). Land occupation as an inventory parameter is generally measured in surface-time units, e.g. 1 m^2 of managed forest used for a period of 1 year = 1 $m^2 \cdot y$. Transformation occurs when land goes from one type of occupation, including no occupation, to a different type of occupation. It is generally measured in surface units, e.g. transformation of 1 m^2 of natural forest into managed forest.

Milá i Canals *et al.* (2007) proposed a framework for evaluating land use impacts in LCIA that reflects a general consensus amongst LCA experts. In this framework, the magnitude of land use impacts is calculated by the area between the curves shown in Figure 4.13, which expresses change in land quality over time relative to a reference state - the natural reference state Q_A or the potential natural vegetation Q_B. Two simplifying assumptions are made: first, linear shapes are considered appropriate for describing the evolution of soil ecological quality over time; and second, the decrease in quality during the occupation phase is considered to be comparatively small relative to the quality drop during the transformation phase and thus is assumed to be negligible (i.e., $Q_C = Q_D$). A specific quality curve can be developed for each impact indicator, including biodiversity, and each land use type. Information on the ecosystem type supporting a given activity (soil parameters and reference state) must be collected if assessments are spatially explicit. In 2013, this framework was refined by providing more recommendations on spatial differentiation, data collection and impact calculations (Koellner *et al.*, 2013).



(Adapted from Milá i Canals *et al.* (2007))

Figure 4.13: Framework for the Evaluation of Land Use Impacts in LCIA.

Curran *et al.* (2011) summarized current approaches for quantifying potential impacts of land use on biodiversity in LCA. These approaches mainly use indicators at the local community level, primarily species richness (Köllner, 2000; Lindeijer, 2000; Goedkoop and Spriemsma, 2001; Köllner, 2003; Koellner and Scholz, 2008; Schmidt, 2008; Geyer *et al.*, 2010). The species-area relationship (SAR) (Rosenzweig, 1995) is an example of an - indicator based on species richness that has been used to compare the species richness of standardized sampling areas in different land classes (Köllner, 2003; Koellner and Scholz, 2008; Schmidt, 2008; De Schryver *et al.*, 2010). Total species richness, however, has limitations as an indicator of biological diversity; for example, it fails to account for taxonomy or for whether the species involved are of conservation significance, and it underestimates (1) the conservation value of important but naturally species-poor habitats, and (2) other compositional, structural, and functional aspects of biodiversity (Ferris and Humphrey, 1999; Smith *et al.*, 2008).

Other proposals for characterizing potential impacts of land use on biodiversity include: (1) using the number of threatened species as an indirect indicator of ecosystem diversity and land use value (Koellner and Scholz, 2008), (2) predicting the effect of ecosystem level changes on the regional pool of species (Koellner and Scholz, 2008; Schmidt, 2008), (3) using fragmentation as an indicator of biodiversity (Jordaan *et al.*, 2009), and (4) using net primary productivity (NPP) as a functional indicator for ecosystems (Lindeijer, 2000; Weidema and Lindeijer, 2001). All of these approaches contribute to an understanding of biodiversity response to land use but none of them (or any other single metric) is an appropriate proxy for biodiversity impacts because biodiversity is inherently a multi-dimensional concept (Ferris and Humphrey, 1999).

Only biodiversity indicators related to land use have been incorporated into the four LCIA methodologies typically used by LCA practitioners - Eco-indicator 99, IMPACT 2002+, ReCiPe and Ecological Scarcity 2006 and these are described below in more detail. All of these methodologies use only single indicators, however, and the limitations of using a single indicator as a proxy for impacts of land use on biodiversity have been touched on above.

Eco-indicator 99 (Goedkoop and Spriemsma, 2001) uses species diversity expressed as the percentage of species threatened to disappear from a given area during a certain time (Potentially Disappeared Fraction, PDF*m^2y) as proposed by Köllner (2000). The potential impact on ecosystem quality is calculated as a function of the difference between species numbers on reference and occupied/transformed land; the area occupied, transformed or affected in the surrounding region; and the occupation or, in the case of transformation activities, restoration time. Eco-indicator 99 includes estimated numbers for species on various land types based on observations in different types of land cover in Europe defined by the CORINE inventory (Commission of the European Communities, 1991a, b). The reference land type is defined as the natural state of the land for *occupation* activities, and as the original state of the land for *transformation* activities. The surface occupied and transformed and the time occupied are estimated by the LCA practitioner during the inventory (using either site-specific information or databases). Eco-indicator 99 also includes estimated restoration times for converted land.

IMPACT 2002+ (Jolliet *et al.*, 2003) uses the same framework as Eco-Indicator 99 for impacts of biodiversity on land use but does not include transformation impacts.

ReCiPe (Goedkoop *et al.*, 2009) uses a species-area relationship (SAR) to develop a species diversity indicator (species*y) based on de Schryver *et al.* (2010). Ecosystem response to land occupation is estimated using the difference in species richness between the reference land type and the occupied area, together with the occupation time; the reference land is Europe-average undisturbed woodlands. Transformation activities are estimated using the difference in species richness of the area from before until after the transformation and the assumed restoration time. Two different models are used, depending on assumptions made regarding the regional effects of occupation or transformation: one model assumes that the occupied or transformed land is isolated from other land types, and the other model assumes that the occupied or transformed land is connected to other areas of the same land use type. Species richness parameters that are specific to each land type and estimated restoration times for each are also available. The area occupied or transformed and the time occupied are estimated by the LCA practitioner during the inventory.

Ecological Scarcity 2006 (Frischknecht *et al.*, 2009) applies a method which assesses the various types of land cover according to their plant biodiversity (Köllner 2003; Koellner and Scholz, 2007, 2008,). The method generates Ecosystem Damage Potential (EDP) factors, which are developed primarily from the Swiss Biodiversity Programme. These factors are based on the predicted number of species and the actual number of species found on a specific land type compared to the regional average. Positive EDP factors for land use imply that there has been ecosystem damage because plant biodiversity is below average and negative factors indicate improvement because plant diversity is above average. Land types are defined using the CORINE inventory (Commission of the European Communities, 1991a, b). EDP factors are a logarithmic function of relative species richness, i.e. number of vascular plant species on occupied land relative to an average standardized number of species in the region calculated by correlating the size and species number for all local and regional plots of Switzerland). Parameters in the logarithmic function are based on results of an expert survey on the expected functional form of the relationship between biodiversity and ecosystem function (Schläpfer, 1999). The LCA practitioner specifies time of occupation and surface occupied. The Ecological Scarcity 2006 methodology does not include characterization factors for land transformation.

4.5.3.2 Biodiversity of Impacts Related to Water Use

Estimating impacts of water use in LCA is an emerging topic. The review by Curran *et al.* (2011) emphasized that water use reduces regional water availability and thus impairs the functioning and diversity of water-dependent terrestrial and freshwater ecosystems; it should therefore be considered when assessing potential biodiversity impacts in LCA. Existing LCIA methodologies do not integrate endpoint indicators related to water use but several papers have modeled how terrestrial and aquatic ecosystems respond to water use: Pfister *et al.* (2011) modeled the relationship between terrestrial biodiversity and water use and assumed a correlation between vascular plant species diversity and water-limited net primary production; van Zelm *et al.* (2010) calculated terrestrial biodiversity response to lowering of the groundwater table by withdrawal and assumed the disappearance of terrestrial plant species, expressed as Potentially Not Occurring Fraction of species; and Hanafiah *et al.* (2011)..

According to Kounina *et al.* (2012), methods proposed by Pfister *et al.* (2009), van Zelm *et al.* (2011) and Hanafiah *et al.* (2011) provide complementary assessments of biodiversity response and can thus be used in parallel to assess biodiversity loss related to water use. Authors favoring volumetric water footprint indicators, such as the method from the Water Footprint Network (Hoekstra *et al.*, 2011), claim that global freshwater appropriation is more important than local impacts, easier to determine, and less error-prone than attempting to model complex ecological interaction (Berger and Finkbeiner, 2012). However, numerically smaller volumetric footprints can cause greater impacts in local areas suffering from water scarcity and/or hosting sensitive ecosystems.

4.5.3.3 Impacts of Pollutant Emissions on Biodiversity

Impacts of pollution on biodiversity have traditionally been accounted for through use of midpoint-level ecotoxicity indicators in LCIA methodologies, such as IMPACT 2002+, TRACI (Bare *et al.*, 2003) and CML (Guinee *et al.*, 2002). The ecotoxicity potentials of various pollutants are expressed in these methodologies relative to a reference substance by modelling the risks and potential impacts of a given pollutant, taking into account its fate resulting from multimedia and spatial transport and effects on various species (Jolliet *et al.*, 2003). This is generally done at a continental or country level. Ecotoxicity potentials are not, however, direct indicators of biodiversity response.

Several LCIA methodologies have proposed models for the cause-effect chain up to endpoints. Among them, ecotoxicity is modeled up to the endpoint in Eco-indicator 99 , IMPACT 2002+, ReCiPe and LIME (Itsubo and Inaba, 2003) methodologies by combining results from multimedia fate models with ecotoxicological effect data extrapolated from three different phyla. Potential impacts on terrestrial eutrophication are modelled based on the percentage of threatened or endangered species caused by eutrophying (and acidifying) emissions for the Netherlands (Eco-indicator99) and Sweden (Swedish EPS methodology (Steen, 1999)). Similarly, aquatic eutrophication links the fraction of affected species or fraction of disappeared species to phosphorus exposure for fresh water (ReCiPe, Eco-indicator99, and the work of Payet (2006) integrated in IMPACT 2002+), and to nitrogen exposure for marine water bodies (LIME, EPS). Acidification is modelled as the change in the Potentially Disappeared Fraction of species for each marginal change in deposition of acidifying substances (in Ecoindicator99) or in the Potentially Not Occurring Fraction of plant species for each change in soil base saturation (in ReCiPe according to van Zelm *et al*, (2007)).

The ILCD Handbook (EC-JRC, 2011) concluded that none of the current methods are mature enough to be recommended for modelling endpoint indicators for biodiversity impacts. The most advanced characterization indicators and models associated with biodiversity are ecotoxicity as proposed by USETox (Rosenbaum et al., 2008) and acidification as proposed by van Zelm *et al.* (2007), as used in ReCiPe.

4.5.3.4 Impacts of Climate Change on Biodiversity

Climate change will likely cause some terrestrial extinctions because of changing temperature, precipitation, and seasonality (Thomas *et al.*, 2004; Thuiller *et al.*, 2006) and therefore must be considered when assessing the biodiversity impact in LCA (Curran *et al.*, 2011). However, current LCIA methodologies do not consider the link between climate change and biodiversity and only a few propose to do so. De Schryver *et al.* (2009) proposed modelling the effect of climate change on terrestrial biodiversity based on the increased extinction risk associated with changes in distributions of individual species under future climate scenarios. Many approaches have been proposed for predicting future extinction risks resulting from climate change but several authors have argued that these methods are generally either inappropriate for this purpose or untested (Botkin *et al.*, 2007; Loehle, 2011; Loehle and Eschenbach, 2012).

4.5.4 Challenges Associated with the Evaluation of Biodiversity Impacts in LCA

4.5.4.1 Challenges Related to LCA Framework

The methodological framework of LCA poses particular difficulties for incorporating biodiversity considerations. LCA aims to cover the entire life cycle of a product or service and information on where and when environmental interventions occurred is often partially or totally missing. Impact characterization is therefore typically generic in space and summed across a time horizon. Indeed, according to Koellner and Geyer (2013), one of the major outstanding questions related to the quantification of land use impacts in LCA, including those related to biodiversity, is how to combine generic impact assessment with site-specific assessment. Moreover, impacts from emissions and resource consumption are linked to a functional unit (see Section 2.1.2), which contrasts with other methods developed to assess the potential impacts of a specific project or chemical localized in space and time (mainly Environmental Impact Assessment (EIA) and Chemical Risk Assessment). The borders between LCA and EIA become less distinct when land use issues, and more specifically biodiversity, are considered in LCA, and it may be appropriate to adapt EIA and other methodologies for use in LCA to better account for local specificities and site-specific biophysical processes. EIA and other tools may be more adequate than LCA for some types of decisions (Milá i Canals *et al.*, 2007).

There are also limitations in estimating impacts of land transformation: (1) it must be assumed that land use impacts are reversible in the broad sense and the regeneration time must be determined to estimate transformation impacts, and (2) more accurate and regionalized data for each specific pathway are required.

In this context, recent effort and initiatives such as the project for land use impacts on biodiversity and ecosystem services in LCIA within the UNEP/SETAC Life Cycle Initiative and the LC_IMPACT project demonstrate the growing interest and research activity around spatially-differentiated LCIA (Koellner and Geyer, 2013). It is therefore expected that the next generation of LCIA methodologies, such as IMPACT World+ (http://www.impactworldplus.org/en/), will systematically address spatial differentiation and include uncertainty information that encompasses both spatial variability and model uncertainty. This will allow application of more environmentally relevant characterization factors by addressing regional assessment of geo-referenced emissions, but the resolution of these characterization models will remain too coarse to perform site-specific assessment.

Another key factor in assessing impacts of land use occupation in LCA is that a reference state needs to be defined. Using the potential natural state of the land as a reference is most commonly suggested (Milá i Canals *et al.*, 2007; Koellner *et al.*, 2013). Koellner and Geyer (2013) recognized, however, that another reference state may be more appropriate, depending on the goal and scope.

4.5.4.2 Challenges Related to the Intrinsic Complexity of Biodiversity

The review of Curran *et al.* (2011) found serious conceptual shortcomings in the way models of biodiversity change are constructed: (1) scale considerations are largely absent, (2) there is a disproportionate focus on indicators that reflect changes in compositional aspects of biodiversity (usually as changes in species richness), (3) functional and structural attributes of biodiversity are largely neglected, (4) taxonomic and geographic coverage is problematic because the majority of models are restricted to one or a few taxonomic groups and geographic regions, and (5) only three drivers of biodiversity loss (habitat change, climate change and pollution) of the five identified by the Millennium Ecosystem Assessment are included in current impact categories but two (invasive species and overexploitation) are not. In addition to the shortcomings mentioned by Curran *et al.* (2011), there are other key problems with current biodiversity metrics used in LCA because of assumptions around (1) species richness, (2) reference land, (3) uni-directional approach, (4) fragmentation, and (5) use of single metrics.

Existing land use LCIA methods were mainly developed for one specific region, often Europe, and use species richness of vascular plants as an indicator (Curran *et al.*, 2011; de Baan *et al.*, 2012). Plants are an important component of terrestrial ecosystems but they only make up an estimated 2% of all species (Heywood *et al.* 1995) and their reaction to land use is not necessarily representative of the potential impacts on other species groups (de Baan *et al.*, 2013). Michelsen (2008) goes further in suggesting that vascular plant diversity is an inappropriate indicator of biodiversity mainly because several studies showed no correlation between species richness in one taxonomic group and species richness in other taxonomic groups. In that context, Failing and Gregory (2003) argued that total species richness provides no information as to which species are present, nor is biodiversity a collection of similar species that respond to habitat changes in a linear manner. Assumption of linearity can lead to over-simplification of biodiversity metrics used to assess the impact of a specific land use action on AoPs. Some of these challenges were at least partially addressed by de Baan *et al.* (2013) and de Souza *et al.* (2013). de Baan *et al.* (2013) proposed a method that combines biodiversity surveys and national biodiversity monitoring data to assess biodiversity land use impacts across multiple taxonomic groups, using a set of species-based indicators. de Souza *et al.* (2013) proposed an indicator based on functional diversity, which is a measure of the range and value of the quantifiable aspects of species—such as feeding behavior, quantity of resources consumed, phosphorus uptake, etc. Although those two approaches are a significant step forward from early biodiversity indicators, they still have some limitations. For instance, de Baan *et al.* (2013) concluded that 'the presented characterization factors for BDP [Biodiversity Damage Potential] can approximate land use impacts on biodiversity in LCA studies that are not intended to directly support decision-making on land management practices. For such studies, more detailed and site-dependent assessments are required.'

Occupied or transformed land is compared to reference land using biodiversity response metrics in several LCA methodologies (Eco-indicator 99, IMPACT 2002+ and ReCiPe. However, natural disturbances such as fire, windstorms, ice storms, alluvial processes, and landslides are important processes in ecosystems (White 1979; Pickett and White 1985) and usually lead to forest landscapes that are a dynamic mosaic of forest ages and conditions. Each patch in the mosaic can be characterized by a unique but potentially overlapping assemblage of fauna and flora, and this makes it difficult to determine what constitutes appropriate reference land. LCIA method developers and LCA practitioners therefore need to take great care when choosing reference land, and it may be necessary to consider a range of landscape conditions rather than a single ecosystem state as a reference in LCIA methodologies for biodiversity.

Many biodiversity assessment methodologies used in LCA have a uni-directional focus on loss, damage and extinction but the interconnection between landscape components and biodiversity is highly complex and in many cases can be bi-directional. This can make indicators difficult to interpret within an LCA context. For example, forest disturbance can have negative influences on species groups or positive effects on landscape heterogeneity and species diversity (Huston, 1999; Robbins *et al.*, 2006; McWethy *et al.*, 2010) and there can be a wide variation in effect duration (Grime, 1973; Huston, 1999) because many species are adapted to or rely upon forest conditions that develop following disturbance (Litvaitis, 2001; Kimmins, 2003).

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

Finally, many indicators can be used to describe ecosystem quality and in particular to measure change in biodiversity (Duelli and Obrist, 2003; Milá i Canals *et al.*, 2007). The use of multiple measures will likely be required to fully capture the complexity of biodiversity and provide the information necessary to understand the implications of trade-offs. For instance, Michelsen (2008) and Michelsen *et al.* (2012) proposed a methodology that considers ecosystem scarcity, ecosystem vulnerability and a suite of key biodiversity factors. Biodiversity indicators can be (1) 'direct' indicators that are biological or taxon-based (e.g. indicator species, richness of functional groups or guilds) or (2) 'indirect' or vegetation structure-based indicators that reflect local or landscape-level habitat conditions such as forest stand structural complexity or measures of landscape structure (Lindenmayer *et al.*, 2000; McElhinny *et al.*, 2005; Rossi, 2011). The usefulness of taxon-based indicators depends upon the taxonomic resolution and taxonomic groups considered, and richness among different taxonomic groups is sometimes not strongly correlated. Habitat-based indicators may have a narrow scope and be related only to certain taxa or influenced by the values of LCA practitioners (Duelli and Obrist, 2003; Rossi, 2011). Failing and Gregory (2003) encourage users of biodiversity indicators to clarify the value-oriented basis for selection of biodiversity indicators and to design indicators that are concise, relevant and meaningful to decision makers. Because of issues such as those identified in this section, Penman *et al.* (2010) proposed establishing a scoring system that results in a single biodiversity metric for use in LCA based on expert opinions about a series of questions related to potential or expected impact of a process on biodiversity. As with any scoring system, this proposed approach is potentially influenced by its subjective aspects, and its ultimate usefulness may depend on issues such as how differences among groups of experts are handled so that comparisons can be made among LCAs, what questions are used, how near-term versus long-term effects of a process are accounted for, and how scores for different questions are weighted.

4.5.5 Implications for LCAs that Involve Biomass Production Systems

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

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

Another barrier to incorporating biodiversity response into LCAs that involve biomass production systems is lack of knowledge. Few field studies have investigated this question, and Campbell and Doswald (2009) note in their review of the topic for liquid biofuels that 'more research is needed, especially at the local level since much of the current literature reviewed focuses on global overviews'. However, recent meta-analyses of manipulative and observational field studies provide insight into potential biodiversity responses to practices associated with intensive biomass production systems in North American forests (Riffell *et al.*, 2011a; Riffell *et al.*, 2011b; Verschuyl *et al.*, 2011; Riffell *et al.*, 2012). Biodiversity responses varied among taxa and production systems reviewed. Most taxa responded positively to thinning treatments (Verschuyl *et al.*, 2011). Diversity and abundance of birds were substantially and consistently lower in treatments with lower amounts of downed coarse woody debris (CWD) and/or standing snags, as was biomass of invertebrates (Riffell *et al.*, 2011a). Other taxa did not respond strongly to reduced downed CWD and/or snags, but these conclusions were based on fewer studies. A recent review of amphibian response to down wood retention levels found a generally positive correlation between volume of down wood retained and amphibian abundance (Otto *et al.*, 2013). The authors note empirical support remains limited for the oft- cited dependence of terrestrial salamanders on retention of CWD in harvested systems, and variation in species' response is high. Little is currently known about biodiversity response to harvest of fine woody debris. If reductions in coarse woody debris from actual harvests are less than the 70 – 95% used in experimental studies, then overall biodiversity responses may be minimal.

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

Results from studies of biodiversity response to intercropping of native, warm season grasses in commercial forests are only now emerging. Marshall *et al.* (2012) recently reported initial effects on rodents of removing woody biomass after clearcutting and intercropping switchgrass for 2 years post-treatment in regenerating pine plantations in North Carolina. Species richness and diversity of rodents did not change due to switchgrass intercropping or biomass removal. However, abundance of two species differed between the treatments. *Peromyscus leucopus* was more abundant and had the greatest survival in treatments without switchgrass while the invasive *Mus musculus* was most abundant in treatments with switchgrass. On this same study site, Homyack *et al*. (2013) found that neither intercropping switchgrass with pine nor removal of harvest residuals caused herpetofauna diversity or abundance of common species to differ from that in traditional pine plantation management during the first 2 years following treatment establishment. They concluded that biofuel production in loblolly pine plantations, as implemented in their study, is unlikely to have short-term effects on herpetofauna relative to traditional pine management. Riffell *et al*. (2012) noted that research with grasses in row crop agriculture suggests that effects will likely vary with habitat needs of individual species and communities and that intercropping regimes favouring mixed native warm-season grasses over switchgrass only, spring harvests over fall, and rotational harvests producing mosaics of grass heights would likely benefit biodiversity.

4.5.6 Conclusions

Integrating biodiversity considerations into the methodological framework of LCA for biomass production systems poses particular challenges. Many proposed approaches rely on a single indicator of biodiversity, although biodiversity is a multi-dimensional concept that can never be fully represented by a single number. Reliance on a single metric over-simplifies 'biodiversity' and will undoubtedly lead to inappropriate conclusions in LCA, thereby failing to support decision-making on local land management practices.. In addition, the interconnections between landscape components and biodiversity can often be bi-directional and yet many of the current methodologies for biodiversity assessment within LCA are unable to incorporate positive effects because of a uni-directional focus on loss, damage and extinction. The empirical basis for addressing site-specific biodiversity in LCA is limited because of the lack of field research investigating response of biological diversity to actual biomass production practices (Riffell *et al*., 2011a, b; Verschuyl *et al*., 2011; Riffell *et al*., 2012).

LCA is not currently suited to providing reliable site-specific assessment results in regard to the complexities of biodiversity, and probably never will be because of the global and comprehensive nature of LCA. Nonetheless, biodiversity is a key aspect that should be incorporated into life-cycle approaches to reduce the risk of environmental burden shifting across impact categories or across life-cycle stages. Biodiversity should be reflected in the broad suite of indicators assessed within LCA. Site-specific and/or territorial assessment approaches such as EIA are also an essential complementary tool when LCA is applied in the context of biodiversity and can be used to mitigate against inaccurate conclusions. This type of paired assessment allows for acknowledgement of the relevance of potential biodiversity-related impacts within the context of LCA, while recognizing that effective examination of the complexities of biodiversity responses requires significant additional site-specific analysis.

4.5.7 References

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IEA Bioenergy

IEA Bioenergy is an international collaboration set up in 1978 by the IEA to improve international co-operation and information exchange between national RD&D bioenergy programmes. IEA Bioenergy's vision is to achieve a substantial bioenergy contribution to future global energy demands by accelerating the production and use of environmentally sound, socially accepted and cost-competitive bioenergy on a sustainable basis, thus providing increased security of supply whilst reducing greenhouse gas emissions from energy use. Currently IEA Bioenergy has 22 Members and is operating on the basis of 13 Tasks covering all aspects of the bioenergy chain, from resource to the supply of energy services to the consumer.

IEA Bioenergy Task 43 – Biomass Feedstock for Energy Markets – seeks to promote sound bioenergy development that is driven by well-informed decisions in business, governments and elsewhere. This will be achieved by providing to relevant actors timely and topical analyses, syntheses and conclusions on all fields related to biomass feedstock, including biomass markets and the socioeconomic and environmental consequences of feedstock production. Task 43 currently (Jan 2011) has 14 participating countries: Australia, Canada, Denmark, European Commission - Joint Research Centre, Finland, Germany, Ireland, Italy, Netherlands, New Zealand, Norway, Sweden, UK, USA.

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