Development of life cycle assessment for residue-based bioenergy

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(Ontwikkeling van levenscyclusanalyse voor op residuen gebaseerde bio-energie)

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Dedicated to Bapak, Ibu, Istri, Kakak, Adek, and Keponakans

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Chapter 1

General introduction

1.1 Sustainability of bioenergy

Bioenergy is expected to play an increasingly significant role in the development of sustainable energy supplies. This, consequently, incites concerns regarding the sustainability of biomass supply for bioenergy feedstock, choices of conversion technologies, and policies on the future energy mix. In that context, the impacts of bioenergy systems must be meticulously assessed before a decision can be made to develop them on a more extensive scale. In order to make a determination, a reliable sustainability assessment framework is required to evaluate the environmental, social, and economic performances of a bioenergy system. Potential unfavorable impacts of a bioenergy system include land use changes, biodiversity loss, water availability, and threats to food security. These risks must be weighed against the potential benefits such as a reduction in global warming, increased energy security, income generation, and rural development. This thesis focuses on specific elements of the sustainability criteria for a bioenergy system, i.e., environmental sustainability.

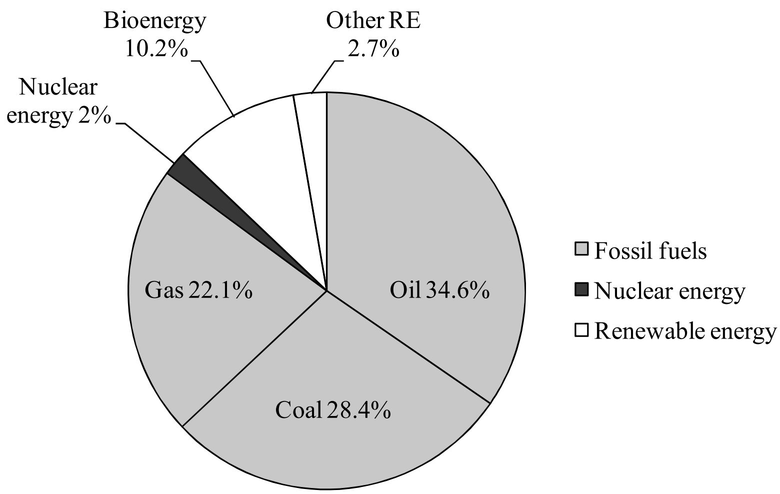

Figure 1.1. Global energy mix (based on IPCC, 2011; RE = renewable energy; the largest contribution is from fossil fuels, 85%).

As indicated in Figure 1.1., the recent global energy mix continuous to be dominated by fossil fuels consisting of oil, coal, and gas. Bioenergy places second, followed by other renewable energy, and nuclear energy. It is clear that bioenergy plays a role as the greatest contributor to renewable energy. In the future, production and exploitation of bioenergy is also predicted to sharply increase. For example, in 2008, the total primary energy supply from biomass in the forms of electricity, heat, combined-heat and power, and transport fuels amounted to 11.3 EJ; meanwhile, future deployment of bioenergy of any forms could reach 100 EJ to 300 EJ by 2050 (IPCC, 2011), noting that there is large uncertainty associated with these estimated potential figures. The numerous increases in the future estimate for bioenergy production express substantial current concerns for the continuing depletion of fossil oil.

There have been ongoing debates concerning the environmental status of bioenergy systems. On the one hand, bioenergy is believed to possess significant greenhouse gas (GHG) mitigation potential. For example, IPCC (2011) stated that bioenergy has the ability to reduce emissions by 80% to 90% compared to the fossil energy baseline, provided that the biomass is sustainably prepared and efficient bioenergy systems are employed. On the other hand, bioenergy has also been suspected of increasing GHG emissions due to the substantial loss of carbon stocks as a consequence of land use changes (Searchinger et al., 2008). Furthermore, intensive agriculture practices within bioenergy systems could potentially affect soil fertility which subsequently influences future biomass productivity. This cause-and-effect interaction is still insufficiently understood and is likely to exhibit strong regional differences. With such contradicting statements, it is difficult for policy makers to make beneficial decisions on the future development of bioenergy.

Scientific analysis aided by analytical tools such as life cycle assessment (LCA) can be employed to assist in solving the problem. LCA is a tool to evaluate the inputs, outputs, and the potential environmental impacts of a product system throughout its life cycle (ISO, 2006). The tool has been applied quite extensively to bioenergy systems but, again, with ambiguous or even contradicting answers (van der Voet et al., 2010; Cherubini and Strømman, 2011). The issue is that LCA has not been mature enough to be able to answer questions in an emerging field such as bioenergy and even more so for those bioenergy based on biomass residues. A significant amount of efforts is essential in order to harmonize these diverging results (van der Voet et al., 2010; Cherubini and Strømman, 2011). This thesis discusses the development of LCA as an assessment tool for residue-based bioenergy.

1.2 Residue-based bioenergy

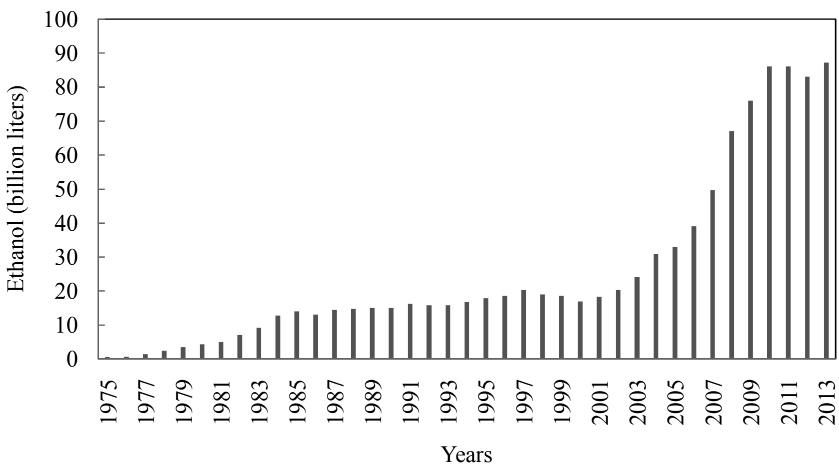

Figure 1.2. World production of bioethanol (based on Sorda et al., 2010; Gupta and Verma, 2015).

The term biomass in general is often used in a broad meaning to include plant materials, animal product and manure, food processing and forestry by-products, and urban wastes (Lal, 2005). In this thesis, bioenergy feedstocks are categorized into biomass products (food products, energy crops) and biomass residues. More specifically, biomass residues are referred to as non-edible portions of plants that are generated in fields or post-harvest processing plants. They are called residues because not intentionally produced for certain purposes, instead, as co-products of other product systems (for example, food, feed, or fibers). From an environmental perspective, biomass residues could be exploited as an alternative feedstock with less risk than biomass products. The International Energy Agency categorized biomass residues as one of the potential candidates for sustainable bioenergy feedstocks (IEA, 2009). Such preference for a more sustainable bioenergy could become a major driver for an increase in demand for biomass residues in the future.

This thesis generally refers to bioenergy as being in the forms of solid (wood chips or pellets), liquid (ethanol or biodiesel), and gas (biogas). In addition to discussing bioenergy in general, special emphasis is given to second-generation bioethanol, a major potential driver for the future increase in demand for biomass residues as bioenergy feedstock. It refers to bioethanol derived from non-edible lignocellulosic biomass (Bright and Strømman, 2009; Garcia et al., 2009; Slade et al., 2009).

As indicated in Figure 1.2., over the last decade, there has been a dramatic increase in global bioethanol production with a total amount of 87 billion liters in 2013. This contribution is dominated by the USA and Brazil producing bioethanol based on corn grain and sugarcane syrup, respectively. The bioethanol is primarily employed as an alternative transportation biofuel to substitute for gasoline. This development, however, has been hindered by the burgeoning concerns regarding the competition for raw materials with food sectors and the consequences related to land-use change (de Oliveira et al., 2005). The competition with food sectors, for example, has been partially blamed for the increase of food prices in the USA between 2003 and 2008 (Sorda et al., 2010).

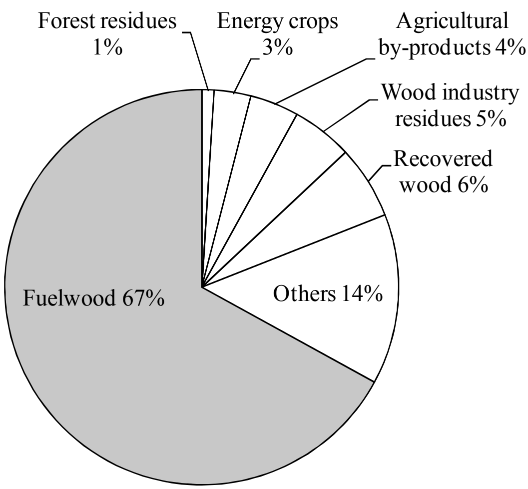

Figure 1.3. Shares of global non-edible biomass for bioenergy (based on IPCC, 2011; Others = charcoal, animal by-products, municipal solid wastes and landfill gas, black liquor; shaded = the largest utilization is for traditional cooking).

The competition with food sectors has encouraged the exploitation of non-food resources for use as bioenergy feedstocks. This non-edible biomass is often exceedingly available in many regions of the world which shares for global bioenergy are depicted in Figure 1.3. It illustrates the different types of biomass used for bioenergy. The figure also indicates that wood biomass has been primarily used for traditional cooking. This fact shows that there continuous to be significant opportunities for improved utilization for bioenergy that is more efficient and environmentally friendly.

Although there is extensive potential for the feedstocks of second-generation bioethanol, the production levels in many countries has been, thus far, discouraging. Expensive production costs prohibit employing bioethanol as a transportation biofuel (Schnoor, 2011). In addition, extensive removal of biomass residues from agricultural fields is not without dilemmas.

Biomass residues fulfill an important role for agricultural lands. Their stock of nutrients provides a basic recycling mechanism that maintains soil fertility, and they can improve the structure of soil for enhanced aeration and water management. This is the primary issue related to the removal of biomass residues from plantation fields. Excessive removal of biomass residues may degrade soil quality and further potentially influence future crop yields (Wilhelm et al., 2004; Lal, 2005; Cherubini et al., 2009). This can be a serious detriment to the long-term sustainability of residue-based bioenergy and should, therefore, be considered as a component of the sustainability assessment. One of the challenges is to compensate the substantial nutrient export with additional fertilizers. However, this will eventually bring further environmental and financial cost that needs careful evaluation.

Another feature associated with biomass residues is concerning the treatment of biogenic carbon in life cycle inventory (LCI). In this thesis, biogenic carbon is defined as carbon contained in the biomass that is accumulated during plant growth involving photosynthetic processes. Currently, Divergence in ways to develop biogenic carbon inventories is extensively found in the scientific literature of bioenergy LCA (Johnson, 2009; Haberl et al., 2012; Levasseur et al., 2013; Downie et al., 2014). In addition, it is speculated that different system boundaries, forms of carbon emissions, and valuation of biogenic carbon will pose certain consequences on the LCI of bioenergy systems (Heijungs and Wiloso, 2014). This thesis elaborates the inventory aspect of the above parameters.

1.3 LCA of residue-based bioenergy

In addition to the general problems in conducting an LCA of product-based bioenergy, those generated from biomass residues add another layer of complexity requiring specific approaches. Generic bioenergy systems consist of typically three main stages, i.e., agricultural processes to produce biomass, conversion into bioenergy, and use of bioenergy. The agricultural processes will in general coproduce biomass products (which are harvested) and biomass residues (which are typically left in fields) which respectively have different economic values. If these two biomasses are to be utilized as bioenergy feedtocks, they would in general undergo the same evaluation schemes, i.e., an LCA procedure involving stages in the generic bioenergy system mentioned above. Feedstocks from biomass residues however require additional considerations. For example, in addition to the bioenergy option, they could also decompose in fields, emitting GHG into the atmosphere and adding organic carbon into soil. These latter options are unique to biomass residues which bring specific consequences. Such consequences will be elaborated further in this thesis.

The majority of LCA studies on bioenergy have initially focused on systems based on food products or energy crops (Cherubini and Ulgiati, 2010; Cherubini and Strømman, 2011). Recent development indicates that more LCA studies have also been expanded toward the utilization of biomass residues conventionally left on fields (Cherubini and Ulgiati, 2010; Cherubini and Strømman, 2011). These studies are often assumed to employ only the surplus biomass or to utilize a greater fraction of residues, but with fertilizer compensation to maintain soil fertility. In certain forest and crop management practices, biomass residues are burned following harvest. In this later case, it can be assumed that the burned biomass residues will not significantly alter carbon flow entering the soil carbon pool (Cowie et al., 2006).

The effect of removing biomass residues from soil has been increasingly recognized as an important component in LCA studies (Mckechnie et al., 2011; Repo et al., 2011 and 2012; Liska et al., 2014). However, at the same time, they are often disregarded because methods to quantify their effects are inadequate, or, if methods are available, they are not easy to implement (Cherubini and Strømman, 2011). These various practices in handling biomass residues in LCA may lead to different results, thus, they must be harmonized for improved characterization of residue-based bioenergy.

As previously indicated, biomass residues are differentiated from biomass products as the former is not intentionally produced for certain purposes. The common criteria in the valuation of biomass residues are that coproducts provide relatively similar proceeds as the primary product, by-products have less value than coproducts, while waste has a negative value (Singh et al., 2010). However, in the LCA community, by-products are not typically differentiated from coproducts. Rather, all economic outputs other than the main product are considered coproducts, but with different values. This valuation scheme can be employed as a criterion to distinguish the impacts of bioenergy based on product- and residue-based bioenergy. This criterion in LCA is referred to as partitioning or allocation methods, which application in bioenergy systems has been discussed thoroughly by Guinée et al. (2009).

Potential supplies of biomass as indicated in Figure 1.3 may not all be entirely available for only bioenergy purposes. There are other potential, competing uses. The feasibilities of the preferred options depend on many factors including new development in emerging technologies such as second-generation bioethanol. In a palm oil system, empty fruit bunches which were initially desired only as mulch are now increasingly explored as a potential feedstock for compost, bioenergy, biochar, biooil, and syngas (Hansen et al., 2012). Considering enormous variations in possible technological options and in the valuation of biomass residues, significant divergence in LCA results are often encountered in practice. In this regard, the development of a specific method to accurately compare and interpret the LCA results is warranted.

1.4 Objective and structure of the thesis

This thesis aims at identifying the key issues in conducting an LCA of residue-based bioenergy. More specifically, the thesis focuses on four primary characteristics associated with the raw materials (biomass residues), i.e., excessive removal from plantation fields which can affect soil fertility; valuation (relative to biomass products); competing uses (bioenergy, feed, fiber, fertilizers); and treatment of biogenic carbon (assumptions of carbon neutrality vs. complete inventory). These unique features require specific LCA approaches which vary from those of conventional product-based bioenergy.

This thesis also proposes improvement in LCA procedures, specifically in the areas of LCI, life cycle impact assessment (LCIA), and methodological choices in the comparative LCA. In that regard, the specific goals of the thesis are to propose the following approaches: [1] solutions to the existing dissimilar practices in the LCI of biogenic carbon; [2] an LCIA method of removing biomass residues from soil on biomass productivity; and [3] methodological choices in comparative LCA of biomass residues utilization. To accomplish the above objectives, the following research questions are addressed:

1. What are the key issues in conducting an LCA of bioenergy systems?
2. What is the environmental sustainability status of second generation bioethanol?
3. How much would the final results of biogenic carbon neutrality assumptions deviate from the true values based on a complete inventory?

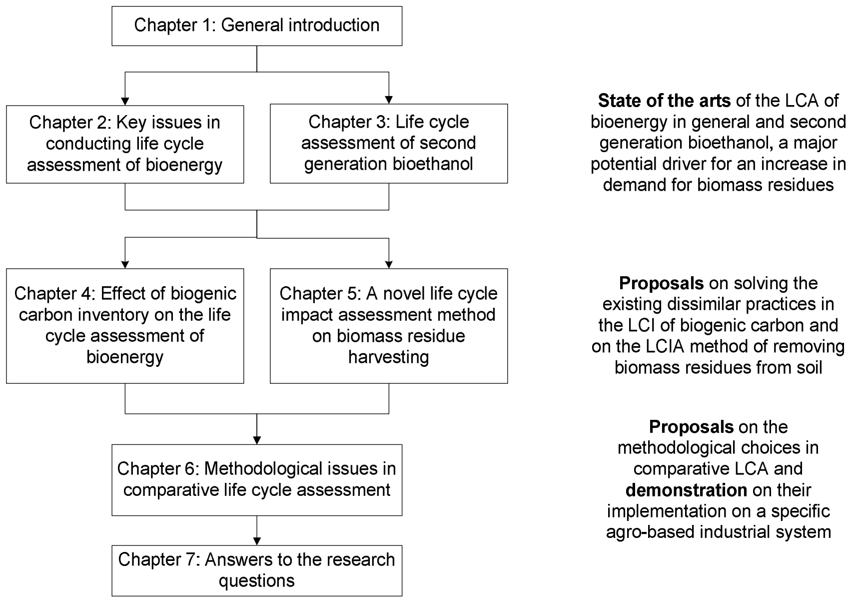

Figure 1.4. Structure of the thesis.

4. How to assess the impact of removing biomass residues from soil on biomass productivity?
5. How to compare the impact of various treatment options on biomass residues in a palm oil system?

These research questions are discussed in seven chapters whereby the structure is exhibited in Figure 1.4. The thesis initially incorporates background information in terms of statistics on the global energy mix, bioenergy in general, and residue-based bioenergy in order to justify the importance of the subject (Chapter 1). Further, it identifies the state-of-the-art of the LCA of bioenergy in general (Chapter 2). Subsequently, it is followed by a review on the environmental sustainability status of second generation bioethanol, a major potential driver for an increase in demand for biomass residues (Chapter 3).

1.5 Outline of the thesis

Chapter 1 describes the background information on the statistics of bioenergy, the LCA of residue-based bioenergy, the research questions, and the structure of the thesis. The primary issues are emphasized and methods to address the associated problems are discussed.

Chapter 2 identifies key issues in conducting an LCA of bioenergy systems. There is an ISO-standardized method for conducting an LCA, but its application to bioenergy is not free from ambiguity. Tremendous numbers of LCA studies on bioenergy have been conducted, however, it continuous to create difficulty when drawing general conclusions on its environmental sustainability due to large variations in the outcomes. Sources of these variations include differences in product systems, data uncertainties, and methodological choices. In particular, bioenergy poses more methodological challenges than other renewable energy because the feedstocks are derived from agricultural chains. In this sub-system, the methodology has not been unambiguous in terms of system boundary, direct and indirect land-use change, treatment for biogenic carbon, and regional variability. Therefore, it is important to define the key issues in conducting an LCA of bioenergy systems, and employ this information as a framework for future studies. Such a framework is beneficial, for example, when analyzing a specific technology like second-generation bioethanol in terms of net-energy output, greenhouse gas emissions, and hotspots along the supply chains.

Chapter 3 identifies the sustainability status of second generation bioethanol, a major stimulator for an increased demand for biomass residues. It reviews the LCA literature on second generation bioethanol and identifies the issues to be resolved to improve LCA practices. The reviews focus on discrepancies in methodological and practical approaches. Emphasis has been placed on system definitions in relation to feedstock specifications (energy crops, biomass residues, and biomass wastes), levels of bioethanol conversion technology (current and future scenarios), bioethanol use as transportation fuel, functional units, allocation methods, and impact categories. The outcome aims at providing decision makers with an increased understanding of the status of second generation bioethanol based on the most studied sustainability aspects, net-energy output and global warming. It may also aid researchers in developing a framework for the bioethanol LCA with correct parameters considering typical problems encountered in the agricultural and bioethanol production chains.

Chapter 4 proposes a solution to the existing dissimilar practices of the LCI of biogenic carbon in bioenergy systems. Biogenic carbon is defined as carbon contained in biomass that is accumulated during plant growth. In spite of considerable progress towards the inventory of biogenic carbon in the life cycle assessment (LCA) of bioenergy in policy guidelines, many scientific articles tend to give no consideration to biogenic carbon, due to the neutrality assumption, rather than employing a complete inventory according to the LCA principles. Meanwhile, the assumption of biogenic carbon neutrality has been previously challenged on the basis of changes in soil carbon stock due to land use change and carbon storage capacities of long-rotation trees or wood products. Supporting this argument, we investigate three other inventory aspects, namely, differences in framing system boundaries (cradle to grave vs. cradle to gate), forms of carbon emissions (carbon dioxide vs. methane), and valuation of biogenic carbon (biomass products vs. biomass residues). Referring to a generic bioenergy system, our analysis is focused on eight scenarios of various carbon flows encompassing biomass decomposition in fields and its alternative utilization as bioenergy feedstocks. These scenarios are applicable to both biomass products and biomass residues, which impacts proportionally depend on the chosen allocation criteria between the two.

Chapter 5 proposes a novel life cycle assessment method for biomass residue harvesting in bioenergy systems. Bioenergy plays a role as the largest global contributor to renewable energy as alternative sources for heat, electricity, and biofuel. In addition to reducing dependence on fossil fuels, it is believed that bioenergy can reduce GHG emissions as targeted by many countries around the world. Concerns regarding global competition with food sectors have encouraged the exploitation of non-food resources as feedstocks, for example, for developing second generation bioethanol. One promising source of feedstocks, from an environmental perspective, is biomass residues which are defined as the non-edible parts of plants that are left on the fields following a harvest. However, excessive removal of these residues may negatively affect soil quality and hamper future harvests. In order to estimate this specific impact, it is essential to develop a new characterization model within the LCA framework based on the assumption that the residues also serves as nutrient stock to improve soil quality. The proposed model must also consider organic carbon flows into the soil matrix which will affect future biomass productivity.

Chapter 6 proposes methodological choices in comparative LCA and demonstrates their implementation on a specific agro-based industrial system. Palm oil systems generate substantial amounts of biomass residues which are from an agronomy perspective preferably returned to plantation in order to maintain soil fertility. However, there are often variations in this practice. Diverse treatment options and differences in economic status determine the preferences of performing an LCA, leading to a divergence in results. Consequently, difficulties encountered when comparing LCA results based on literature are not unusual due to dissimilar approaches. The objectives of this chapter are to provide guidelines for methodological choices. The guidelines enable a systematic comparison of diverse scenarios for the treatment and valuation of empty fruit bunches and to explore the effects of, for example, different scenarios on the environmental performances of a palm oil system.

Chapter 7 summarizes the answers to the research questions that are discussed in Chapters 2 through 6. This chapter further discusses the potential application of the developed approaches as well as areas of future research and is finally ended with concluding remarks.

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Chapter 2

Key issues in conducting life cycle assessment of bioenergy

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Abstract

Although there is an ISO-standardized method for conducting life cycle assessment (LCA) studies, its application to renewable energy sources, in particular to bio-based renewable energy (bioenergy) involving agricultural chains, is not straight forward. There are theoretical and practical issues in goal and scope definition, functional unit, inventory analysis, and impact assessment. The debate between attributional LCA and consequential LCA is, for bioenergy, even more crucial than for ordinary products, especially when it comes to either direct or indirect land-use change. Data are often highly variable, and system boundaries are quite arbitrary. For bioenergy from biomass residues, allocation and recycling provide complications. The treatment of biogenic carbon is of particular interest. The choice of impact categories and the necessity of a regionalized impact assessment are another problem. This chapter provides a systematic overview of these topics.

2.1 Introduction

Our economy has long been dependent on non-renewable energy carriers, especially on fossil energy. The high dependence on non-renewable energy sources developed over a relatively short period of time. From the middle of the nineteenth century, there was a rapid increase in the use of fossil fuels. These non-renewables replaced wood and soon became the basis of an exponential growth in energy use associated with a number of novel energy-demanding activities (Sørensen, 2002). Early man was only capable of causing environmental disturbance on a local scale; however, man has currently achieved a technological level, enabling him to convert energy at rates that are responsible for climate change over extended areas. With 81 % of recent global energy use originating from fossil fuels, 6 % from nuclear, and 13 % from renewable energy (IEA-Bioenergy, 2009), it is understandable that human societies have recently begun to reconsider the use of renewable sources. In light of this development, we are now, along with other environmental impacts, facing two major problems: depletion of fossil resources and an increase in anthropogenic levels of carbon dioxide.

Alternative options that are available to reduce our dependence on nonrenewable sources and simultaneously mitigate climate change are already in development. The use of bio-based renewable energy (bioenergy) is now deemed to be one of the most promising renewable energy alternatives. Reasons typically given for why bioenergy should be promoted are diverse. Bioenergy is considered carbon neutral, it is made from renewable resources, it stimulates the agricultural sector, and it may be produced domestically in many countries, hence diminishing political and economic dependency on other countries (Guinée et al., 2009). However, criticisms have also developed against biofuels, particularly on their role in the food price spikes and the nature of land-use change. A specific example of this case is the maize to bioethanol for transportation fuel in the United States that induced land-use impact, direct and indirect (Harvey and Pilgrim 2011). WRI (2005) indicated that land use (18.2 %) and agriculture's (13.5 %) contribution to greenhouse gas emissions (GHGs, including N_2O and CH_4 in addition to CO_2) are globally estimated to be at least twice the amount of the total emissions from global transport (13.5 %). This assessment indicates the importance of the potential contribution of the land-use aspect to the overall environmental burden of bioenergy systems. Major activities related to these land-use-related impacts are deforestation that releases carbon dioxide from burning or decomposing biomass and oxidizing uncovered humus. In addition to other impact categories such as biodiversity loss and soil quality degradation, all these emissions may negate any GHG benefits of biofuel systems for decades to centuries (Tilman et al., 2009). In this regard, these same authors proposed that biofuels should receive policy support as substitutes for fossil energy only when they make a positive impact on four important objectives: energy security, GHG emissions, biodiversity, and the sustainability of the food supply.

Bioenergy is presently the largest global contributor (77 %) to renewable energy and has contributed significantly to the production of heat, electricity, and fuels for transport (IEA-Bioenergy, 2009). Therefore, in the following parts of this chapter, discussion will be focused on bioenergy as the dominant fraction of renewable energy. The main feedstocks for bioenergy are biomass residues from forestry, agriculture, and municipal waste. Only a small portion of sugar, grain, and vegetable oil are used for the production of liquid biofuels (IEA-Bioenergy, 2009). There are many technological routes available to convert biomass feedstock into final bioenergy products. Several conversion technologies have been developed to adapt to the unique physical nature and chemical composition of various biomass feedstocks. These include direct combustion (heat), co-firing/combustion (heat/power), gasification (heat/power), anaerobic digestion (heat/power/fuel: methane), fermentation (fuel: bioethanol), trans-esterification (fuel: biodiesel), and photosynthesis (fuel:

hydrogen) (IEA-Bioenergy, 2009). These various conversion technologies will dictate overall environmental performances. For example, ethanol production through biochemical or thermochemical conversions is expected to result in different levels of decreasing GHG emissions. However, these conversion-related differences are likely to be small in relation to those associated with feedstock production (Williams et al., 2009). In addition, emissions of methane or nitrous oxide from agricultural field and indirect land-use change may contribute to a more complicated overall picture (Cherubini and Strømman, 2011). Side and rebound effects, as well as market mechanisms, of large-scale production of biofuels also affect food markets, resource scarcity, and environmental quality, while these factors are often left out in a sustainability assessment (Guinée et al., 2011; van der Voet et al., 2010). Moreover, bioenergy systems may involve a unit process with input–output flows, which often make it difficult to differentiate between economic (products) and elementary (resource use or emissions) flows.

Recently, there have been tremendous numbers of LCA studies describing bioenergy in order to support policy making. The growing debate on bioenergy and other bio-based products contributed to the acceleration of the development of LCA methodology. However, it is difficult to draw general conclusions from the set of studies due to large variations in outcomes. Sources of these variations include real-world differences, data uncertainties, incompleteness of included impacts, and methodological choices (van der Voet et al., 2010). More specifically, the methodological choices are related to the selection of a functional unit, system boundary, land-use aspects, biogenic carbon, treatment of multi-functional processes, data variability, and regionalized impact assessment (Cherubini and Strømman, 2011; van der Voet et al., 2010; Guinée et al., 2009; Finnveden et al., 2009). This indicates that bioenergy poses more methodological challenges than other renewable energy. Moreover, these issues are insufficiently comprehensively addressed by current LCA studies.

This chapter is aimed at providing a systematic overview on the above-mentioned key issues in conducting LCA of bioenergy. Detailed comparison of methodological choices among different LCAs of bioenergy systems can be found in recent surveys such as those of Cherubini and Strømman (2011), van der Voet et al. (2010), Wiloso et al. (2012), and Singh et al. (2010). The structure of this chapter will follow the first three phases of the LCA framework (ISO 2006), including goal and scope definition, inventory analysis, and impact assessment as follows:

- Goal and scope definition:
    - Attributional and consequential LCA
    - Functional unit
- Inventory analysis:
    - System boundary
    - Land use and land-use change
    - Biogenic carbon
    - Treatment of multi-functional processes
    - Data variability
- Impact assessment:
    - Impact categories
    - Regionalized impact assessment

A generic bioenergy system that spanned from cradle-to-grave boundaries is presented in Figure 2.1 The system covers biomass production, biomass transport, biomass conversion, and bioenergy distribution and use. In the upstream chain, the production of biomass feedstock is connected with agricultural land use, direct and indirect. The association of the biomass feedstock with land-use aspects is currently recognized as the central feature in conducting an LCA of bioenergy systems.

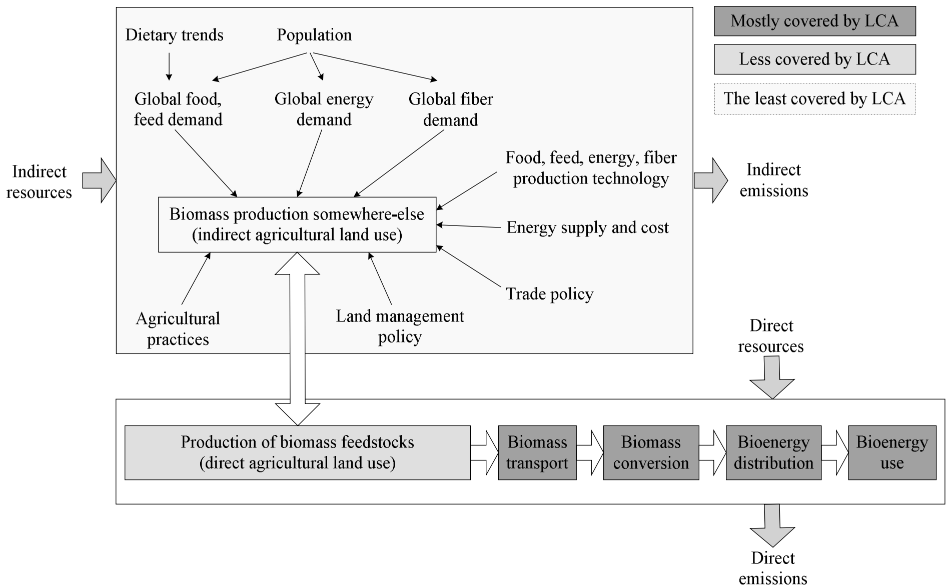

Figure 2.1. Direct and indirect effects of a generic bioenergy system (modified from Sheehan, 2009). Different shading intensity indicates present coverage in LCA studies.

2.2 Goal and scope definition

Questions related to the overall objective of LCA studies should be formulated in the goal and scope definition. The goal is closely related to the context in which an LCA study is done, and the scope includes making choices concerning the methodology to use in the subsequent modeling (Baumann and Tillman, 2004). Goal and scope definition is an important initial step since the choice of methodology used depends on the purpose of the individual study. These methodological choices include system boundary, treatment of multi-functional processes, types of required inventory data, and functional unit. The first three topics are described in the following section (attributional and consequential LCA), while the last one is described separately.

2.2.1 Attributional and consequential LCA

A clear definition of the goal and scope should specify the types of LCA needed. They can be attributional or consequential (ALCA and CLCA for short). In general, the goal of ALCA is to assess the environmental burden of a product, assuming a status quo situation, while the goal of CLCA is to assess environmental consequences of a change in demand (Thomassen et al., 2008). These different LCA principles require a systematic approach to reduce uncertainty due to freedom of choosing the methodology (Finnveden et al., 2009).

ALCA describes the environmentally relevant flows to and from a life cycle and its subsystems (Finnveden et al., 2009). The attributional method is less important for policy decisions as its purpose is not to support changes. ALCA, however, is useful in obtaining insight into the main environmental impacts related to existing bioenergy products. This is done to better describe the effect of changing feedstocks, changing production processes, or improving efficiency. Another type of application with a more direct relevance to a bioenergy system is the use of ALCA to identify main hot spots in the life cycle chain, the share of certain emissions, or flows to an impact category. This can be a first step in realizing process improvements from a sustainability point of view. An example of this is the LCA study of a generic life cycle of bioenergy with a boundary system as shown in Figure 2.1 but without involving the indirect effects. This is in contrast with a CLCA concept, which also includes the indirect effects.

CLCA describes how relevant environmental flows (resource use and emissions) will change in response to possible decisions (Finnveden et al., 2009). Referring to this definition, Cherubini and Strømman (2011) concluded that the CLCA appears as the most broadly applied in bioenergy systems as compared to ALCA. They revealed that almost three-fourths of the reviewed studies compare the environmental impacts with those of a fossil reference system. This is done to address the needs of policy makers in order to decide on relevant bioenergy options. The assessment, however, needs further clarification since not all comparison studies necessarily qualify as consequential.

A distinction between foreground and background systems is especially useful in the CLCA approach. Background systems are often based on databases representing average data of aggregated industrial processes, such as electricity. When a chain of processes are being considered as a foreground system, the proposed technology needs to be specifically known and marginal data are required (Finnveden et al., 2009). Other distinctive characteristics of CLCA are that unit processes within a system boundary are included to the extent of their expected change caused by a demand and that co-products are handled by system expansion (Weidema, 2003). To summarize the main characteristics of these approaches, a comparison between ALCA and CLCA is given in Table 2.1.

Table 2.1. Main characteristics of ALCA and CLCA (based on Thomassen et al., 2008).

Characteristics	ALCA	CLCA
Synonym	Status quo, descriptive	Change-oriented
Type of questions	Accounting	Assessing consequences on changes
Type of required inventory data	Average, historical	Marginal, future
Knowledge on the cause-effect chains	Physical mechanisms	Physical and market mechanisms
Functional unit	Represents static situation	Represents change in volume
System boundaries	Static processes	Affected processes by change in demand
Treatment of multi-functional processes	Co-product allocation (partition)	System expansion
Assessment quality	Sensitive to uncertainty	Higher sensitivity to uncertainty

CLCA is, in principle, only preferable within certain limits since the uncertainties in the modeling stage may outweigh the insight gained from it (Cherubini and Strømman, 2011). This is related to the fact that the reference system should always refer to the scope and context of the study. For example, the bioenergy system is typically compared with a fossil reference system producing the same amount of products and services. In most cases, however, studies use conventional extraction of crude oil as a benchmark, thereby ignoring the increasing carbon footprint arising from the extraction of non-conventional oil such as oil sands, shale oil, and deep-ocean drilling (Harvey and Pilgrim, 2011). Similarly, when the bioenergy pathway delivers some co-products able to replace existing products, the reference to the substituted products should also be defined in the fossil reference system. The same applies to the case when the production of feedstock for biofuels uses land that was previously storing carbon such as forests. In this case, the previous land use should be taken into consideration for the determination of carbon emissions due to land-use change (Singh et al., 2010). Also, when the same feedstock is used for another function, the reference system should include the alternative biomass use. In our view, this last example is the crucial aspect of CLCA in the case of a bioenergy system. This requires a CLCA approach to include the production of biomass feedstocks, resulting in a wider system boundary. This feedstock, consequently, is no longer available for other purposes (such as food, feed, or fiber), so new land to produce an extra feedstock may be needed. The above requirements may increase the uncertainty of the assessment; hence, the adoption of CLCA approach must be treated carefully.

A famous issue in CLCA is the coverage of indirect land-use in biofuel system. Based on the study of Searchinger et al. (2008), Zamagni et al. (2012) pointed out that most of the previous LCA studies provided only a limited analysis to the life cycle of biofuel system. They failed to account for the indirect effects (i.e., those taking place outside the biofuel value chain) by excluding emissions from land-use change. As shown in Figure 2.1, indirect effects may result from the competition for land currently used for food, feed, or fiber to fuel production (Hedegaard et al., 2008). Interaction between various influencing factors and displacement mechanisms can occur in many forms. The main challenge now is how to quantitatively measure the indirect impact of biofuel development on other chains (food, feed, and fiber) that is modeled based on global economic interaction. A CLCA was also used to address problems like the environmental consequences of including the production of second-generation biofuels from biomass residues compared to a current palm oil biodiesel production system in Malaysia (Lim and Lee, 2011) or to investigate the expected indirect effects of the development of a grass biomethane industry in Ireland (Smyth and Murphy, 2011).

Currently, there is no clear distinction between ALCA and CLCA in most policy guidelines of a country or region, partly due to unresolved debate in framing direct/indirect effects and allocation of co-products (Brander et al., 2009; van Dam et al., 2010). This conclusion is based on at least three policy guidelines (UK-Renewable Transport Fuel Obligation (RTFO), EC-Renewable Energy Directive (RED), and US-Renewable Fuel Standard (RFS)) that tend not to distingusih ALCA and CLCA. For example, EC's RED and UK's RTFO include only direct land-use change, while US's RFS includes both direct land-use change and indirect land-use change; EC's RED is based on energy allocation, while UK's RTFO and US's RFS prefer system expansion (van Dam et al., 2010). These conditions may result in a combination of the two approaches within a single analysis and, consequently, an unfair comparison of results derived from different methods (Brander et al., 2009).

2.2.2 Functional unit

A product system is defined based on a functional unit of a product, specified in relation to the nature of a system, geographical, and time boundary. The main role of a functional unit is to be used as a reference to quantitatively connect inputs and outputs of a life cycle inventory (LCI). In this way, LCA results of the same functional unit can be compared between one another provided that, among other things, the system boundaries are similar and the scales are normalized. A proper functional unit that positively reflects the reality is very important in LCA studies. This is important since different choices of functional units from the same system may result in different outcomes when compared to each other. A nice illustration on the effect of different functional units on the results of biofuel LCAs is given by van der Voet et al. (2010).

Theoretically, a functional unit in the form of one MJ would be more appropriate to compare the best use of biomass feedstock for bioenergy of different forms (heat, electricity, biofuel). However, in practice, the functional units vary among studies. Based on current reviews, typical functional units commonly used in LCAs of biofuel systems are volume or mass of input biomass feedstock, volume or mass of biofuel, caloric value of biofuel, driving distance of a car, and agricultural land area (van der Voet et al., 2010; Cherubini and Strømman, 2011; Wiloso et al., 2012). These choices of functional units are driven by the main questions or goals of the LCA study. For example, to compare the benefit of gasoline and biofuel systems as transportation fuels will lead to a functional unit in terms of 1 km driving distance. Land area required to produce biomass feedstock is an extremely important parameter since bioenergy can compete against food, feed, or fiber under land availability constraints. However, there are only a few bioenergy LCAs based on this parameter. One of them is the study by Lim and Lee (2011) that used a one-year use of one-hectare palm oil plantation as a functional unit to produce both biodiesel and bioethanol.

2.3 Inventory analysis

An LCI of a product or process quantifies economic and environmental inputs and outputs around the system boundary. It is constructed as a flow model of a technical system according to the system boundary decided in the goal and scope definition. The model is basically a mass and energy balance over a system, but only environmentally relevant flows are considered. Activities of the LCI also include data collection of all activities in the system and calculation of the environmental loads in relation to the functional unit (Baumann and Tillman, 2004). There are five key aspects specific to bioenergy systems that need further elaboration, i.e., system boundary, land use, biogenic carbon, multi-functional processes, and data variability.

2.3.1 System boundary

In LCA of bioenergy studies, the choice of system boundary is often arbitrary. With the basic cradle-to-grave principle of LCA, everything should be included; however, in practice, many processes are left out for different reasons. System boundaries define what are to be included and what are not. In general, capital goods and wastes as an input feedstock are cut off from the system. This implies that the emissions by the production of capital goods and wastes are not taken into account.

As previously indicated in Sect. 2.2.1, one of the main issues related to CLCA is the identification of the processes to be included in the analyzed system, which implies the way in which boundaries are defined. In the case of biofuels, for example, the system boundary is expanded to include emissions and resources used, directly and indirectly, as a result of the consequential effects of introducing biofuels to the global economy. In this regard, the rule is to include only relevant affected processes, defined as those that respond to changes in demand or supply driven by the decision at hand (Zamagni et al., 2012). In doing so, the resulting functional unit of the whole system may consist of multiple functions, including the main system and those processes added into the system boundary. However, when a comparative analysis must be conducted, it may be difficult to guarantee the functional equivalency between the systems compared since the processes included could serve different functions. Such a resulting multi-functional system raises some concerns about whether it can still be considered a functional unit (Zamagni et al., 2012). In this case, differences in system boundaries are rather crucial. Therefore, they must be specified, unambiguous, consistent, and in-line with the actual goal and scope of the study (van der Voet et al., 2010). This may be the most difficult problem to address.

The cradle-to-gate approach is sufficient for comparing various production technologies to make the same biofuel from different feedstocks, while the cradle-to-grave is the best approach for comparing, for example, the utilization of certain biofuels with fossil fuels (Singh et al., 2010). Cradle-to-gate studies are performed by excluding the use and waste treatment stages, but it is, of course, admissible only when there is no difference between these stages. To illustrate this, a comparison between a plastic cup and a paper cup for drinking tea can be used. In this case, the upstream stages (the growing of tea plants, the processing of tea leaves, and the boiling of water) are likely the same, but the waste treatment of plastic cups is obviously different from that of paper cups (Heijungs and Wiloso, 2012).

The same system boundary with a difference in functions will have a different basis of comparison. For example, electricity generated from municipal solid waste is not very efficient and usually shows no improvement over a fossil fuel alternative. However, when a waste management aspect is included in the electricity generation, this extended new waste-to-energy system boundary will likely favor over the waste management alone (without electricity generation) or over a fossil fuel system (van der Voet et al., 2010).

2.3.2 Land use and land-use change

Although the majority of global GHG emissions have been blamed on the use of fossil fuels, there has recently been growing recognition that land use also significantly contributes to the emissions. The increased understanding of the effects of land-use change needs further consideration in bioenergy systems. In this regard, a UNEP-SETAC guideline on land-use impacts (soil quality, biodiversity, and ecosystem services) has been proposed (Koellner et al., 2012), but there is currently no widely acceptable way to incorporate land-use impacts in an LCA study. The main reason may be that this aspect is very difficult to quantify.

In agricultural land use, there are three time periods in examining the long-term consequences of agricultural activities, i.e., the period before (transformation), during (occupation), and after (restoration) agriculture (Milà i Canals et al., 2007). Based on these time frames, one may refer to land use as an activity during the occupation period and land-use change as land transformation or a change in the properties of the land surface area. This could be a new type of land use at a single point in time such as deforestation or agricultural expansion (Milà i Canals et al., 2007). Similarly, IPCC refers to land-use change as land conversion but also, interestingly, as changes in carbon pools without land conversion (IPCC, 2001). In fact, the precise place of land use and land-use change in the LCA framework is not clear. For example, besides as an activity, land use can also be an inventory item, just like CO_2 (certain land area occupied for certain period of time). Additionally, land-use change can be an activity (a unit process, e.g., clearing of forest) (Heijungs et al., 1992). Even impacts of land use or land-use change are frequently indicated simply with the term land use.

Mitigating the competition for land can only be established if the complexity of the competition dynamics is fully addressed. Each of the contributing factors (energy, food, feed, and fiber demand) cannot be treated in isolation (Harvey and Pilgrim, 2011). All these factors are intimately interconnected, particularly in large-scale development of bioenergy (McKone et al., 2011). Although the competition of land used for food, fiber, and energy was recognized a long time ago, quantification attempts involving competition aspects have been made only quite recently (Searchinger et al., 2008). Drivers for increased bioenergy use (e.g., policy targets for renewables) can lead to increased demand for biomass, leading to competition for land currently used for food production and, possibly, indirectly causing new and sensitive areas to be converted into arable land (IEA-Bioenergy, 2009). These interconnected factors in the complexity of direct and indirect land use are previously illustrated in Figure 2.1, while activities, resource use, and emissions typically involved in land use and land-use change are shown in Figure 2.2.

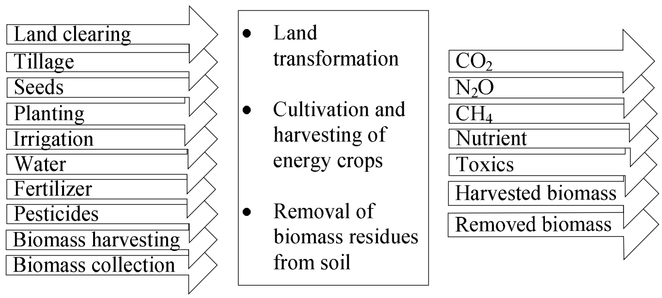

Figure 2.2. Inventory of activities, resource use, and emissions in the agricultural chain of biomass feedstock.

2.3.2.1 Direct impacts

Land use and land-use change, in relation to biomass supply for bioenergy, are characterized as having various input–output inventories, resulting in different contributions to impact categories that affect different areas of protection. Relevant impact categories include global warming, eutrophication, acidification, toxicity, water use, and land use. These impacts are induced by input–output components and activities in the agricultural chain including land transformation, cultivation of energy crops, and removal of biomass residues from soil, as shown in Figure 2.2. Typical inventories include, for example, the use of fossil fuels in tractors for land clearing, tillage, planting, and harvesting; the application of seeds, fertilizer, and pesticides; and the use of water for irrigation. Important GHG emission species related to agricultural activities are N_2O and CH_4 in addition to CO_2. Land-use related activities may directly affect the quality of land (natural environment) as an area of protection. This quality in terms of ecosystem services include soil quality, biomass productivity, and biodiversity (Milà i Canals et al., 2007). The characterization of these land-use impact categories, however, is less developed compared to other categories.

2.3.2.2 Indirect impacts

In principle, indirect land use will have the same inventory components and relevant impact categories as that of direct land use. Indirect land use refers to the changes in land use that take place elsewhere as a consequence of the development of bioenergy systems. In the LCA methodology, this indirect impact may have a broader meaning, including any relevant effects to different chains, for example, if large-scale bioenergy production affects food production chains. As an illustration, if fertile land previously used for food crops (such as corn, soybeans, or palm) is transformed to produce bioenergy, this could lead to farmers clearing wild lands elsewhere in the world to meet the displaced demand for food crops (Tilman et al., 2009).

The paper by Searchinger et al. (2008) has pointed out the significant contribution of indirect impacts on the LCA of bioenergy systems. The authors argued that, based on a sustainability criterion, fuel oil is better than most biofuels. There are two connected arguments put forth. First, biofuel development provoked a rise in the price of food, leading to the stimulation and expansion of food production. Second, the subsequent displacement of food production into new areas of cultivation (indirect land-use change) resulted in a release of CO_2 into the atmosphere. It holds biofuel production responsible for global climate change in ways not measured by previous LCA studies (Harvey and Pilgrim, 2011). The above explanation on indirect impact changes the entire nature of LCA to one which must be able to model global economic interaction (Sheehan, 2009). In addition to indirect land use, other types of indirect impacts may be needed to properly assess the total GHG emissions implications of substituting biofuels for gasoline. In this regard, Liska and Perrin (2009) illustrated that livestock and military security also had a significant impacts in the case of the US bioenergy system. The inclusion of these indirect effects in the bioenergy system understudy can change the direction of the final results. There is, however, much scientific uncertainty in measuring these indirect emissions related to both bioenergy and fossil oil systems, thus creating a problem on how to properly calculate them.

2.3.3 Biogenic carbon

One of the important aspects in bioenergy systems is related to biogenic (shortcycle) carbon. Although under debate, it has been recognized that bioenergy is not carbon neutral since it requires a significant input of fossil fuels. In practice, many studies exclude biogenic carbon from biofuel LCAs, rather than including it initially as an extraction and later as an emission. This convention is so widespread that in the majority of biofuel LCA case studies, the aspect of biogenic carbon is not even mentioned (van der Voet et al., 2010).

The neutrality of biogenic carbon is part of the natural carbon cycle over a relatively short period of time, resulting in stable atmospheric carbon. As illustrated in scenario 1 of Figure 2.3, this is the case when the emission of biogenic carbon in the form of naturally decayed or burned biomass is compensated by the same amount of photosynthetic carbon sequestered by naturally grown vegetation. However, this cycle can no longer be 'neutral' if the input–output inventory is out of balance. This occurs, for example, when large-scale bioenergy systems introduced are involving significant amounts of fossil fuel and agricultural input (scenario 2). In this case, the bioenergy system may emit more total carbon than the sequestration capacity of trees, resulting in net accumulation of CO_2 in the atmosphere. On the other hand, carbon emissions from fossil fuel combustion are considered as an irreversible one-way process (scenario 3). It transfers geological carbon, locked underground, over long-term geological time into the atmosphere. This process increases atmospheric carbon levels with time. Therefore, to properly assess the benefit of the bioenergy system over fossil fuel systems, it is necessary to account for all relevant input–output flows in the inventory phase of LCA studies, including carbon sequestration and carbon emissions of both biogenic and geologic sources. From various options provided in LCA studies, the bioenergy product with the larger GHG saving, among other criteria, would be the preferred energy system.

There are at least two points to make with respect to carbon neutrality of bioenergy. First, if there is anything neutral, it is LCA, as an analytical tool, that makes the conclusion.

If biofuels are carbon neutral, this will result from the LCA study instead of being a starting point of the LCA study. Second, there are several situations where the carbon neutrality of bioenergy is challenged. One of these occurs in the situation when some of the CO_2 absorbed is not released as CO_2, instead, as CH_4, a greenhouse gas that is much stronger than CO_2. This may happen, for instance, when the biotic feedstock is subject to a process of incomplete burning or anaerobic decomposition with leakages occurring along the way. Another case is a plantation with two co-products (e.g., palm oil and palm kernel oil) where part of the absorbed CO_2 is allocated to each of the co-products. In chains with only one product, exclusion of biogenic carbon can result in the same outcome as long as the issue of CH_4 does not arise. However, in cases of chains with co-products, it makes a difference. Allocation may place the credits for extracted CO_2 in a different part of the multi-product chain, while ignoring biogenic CO_2 would not have this effect (van der Voet et al., 2010). A recent review indicated that carbon sequestration, if included at the biomass generation stage, can offset the GHG emissions from all parts of the life cycle chains at a high ethanol percentage (C85 %) (Wiloso et al., 2012). A final example challenging the bioenergy neutrality is the fact that there is a time difference between CO_2 fixation and release. A specific dynamic LCA method has been developed to account for such situations.

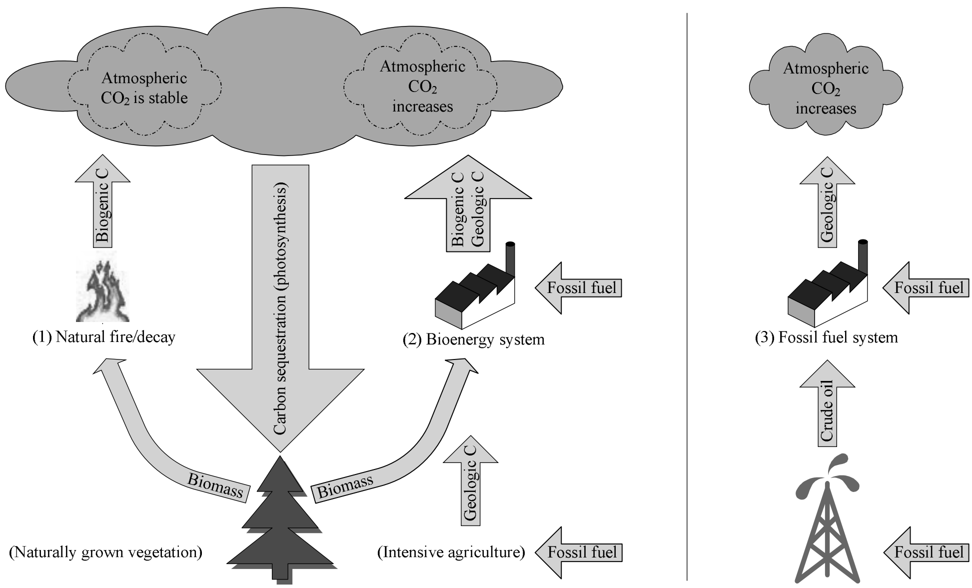

Figure 2.3. Biogenic-carbon cycle versus 'irreversible' geologic-carbon emission.

The most appropriate way to treat carbon cycles is to view them as genuine cycles. During tree growth, a certain amount of atmospheric CO_2 is fixed but is ultimately released as CO_2 or CH_4 when the wood is landfilled, is incinerated, or decays naturally. At the systems' level, the fixation of CO_2 during tree growth is subtracted from the CO_2 emitted during waste treatment of discarded wood (Guinée et al., 2009). For fossil fuels, carbon fixation has taken place as a natural process millions of years ago, but carbon emissions occur immediately when these fuels are burned.

The rationale behind different treatments between biogenic carbon and geologic carbon is because, for example, forestry (the process that fixates the CO_2) is considered as a unit process. It is an intentional activity, controlled by humans, requiring inputs and producing outputs. The creation of fossil fuels is a spontaneous process without human intervention. The forestry is, thus, an activity that should be included in the flow diagram of an LCA study, whereas the process of fossil fuel formation should not (Guinée et al., 2009). Wegener Sleeswijk et al. (1996), in their report on the application of LCA to agricultural products, propose to not include biogenic carbon dioxide in the analysis if the entire life cycle is being analyzed. If the study is based on cradle-to-gate analysis, carbon sequestration must either be included, or it must be explicitly stated that this fixation is being excluded from the study. If this is not done, there is a danger that if other researchers use the results of the study, they will include, say, the emission of CO_2 during combustion of biodiesel fuel, while fixation of CO_2 was omitted in the cradle-to-gate analysis.

There is currently no consensus regarding how to treat biogenic carbon at the policy level. The Intergovernmental Panel on Climate Change (IPCC) currently considers biomass to be carbon neutral, suggested by the adoption of a stock-change method rather than an input–output flow approach in carbon accounting (Levasseur et al., 2012a). In this case, if biogenic carbon is released later in the life cycle, CO_2 emissions are not accounted for to avoid double counting. As discussed in Johnson (2009), a life cycle-based method such as the British specification PAS 2050 (BSI, 2011) suggests the same approach as IPCC, not considering biogenic carbon uptakes and emissions, while the International Reference Life Cycle Data System ILCD (EC-JRC-IES, 2010) recommends the opposite. Similar to PAS 2050, EU Directive (2009) also excludes the capture of CO_2 in the cultivation of biomass and emissions from biofuel use from the calculation of GHG emissions by setting their values equal to zero. The rationale behind these differences is the argument that the combustion or decay of woody biomass is simply part of the global cycle of biogenic carbon, and over a long period of time, it does not increase the amount of carbon in circulation due to compensation by photosynthetic processes. Meanwhile, in the conventional LCA practices, all flows including carbon uptake and emissions should be accounted for in the inventory stage without considering the time scale. To deal with this time frame issue, Levasseur et al. (2012a) proposed to treat biogenic carbon as temporary storage with dynamic LCA. The argument behind this approach is that the concentration of CO_2 in the atmosphere is temporarily reduced and some radiative forcing is avoided. This is favorable in the short term as it also allows 'buying time,' while technology develops in the field of GHG emission reduction and mitigation (Levasseur et al., 2012b).

2.3.4 Treatment of multi-functional processes

Various forms of bioenergy products are ideally derived from feedstocks produced with much lower life cycle GHG emissions than traditional fossil fuels and with little or no competition with food production. According to Tilman et al. (2009), feedstocks in this category may include perennial plants grown on degraded lands, crop residues, sustainably harvested wood and forest residues, double crops and mixed cropping systems, municipal and industrial wastes. These various feedstocks and bioenergy products in LCA should be treated with proper allocation and recycling procedures to attribute environmental burden of multi-functional processes to their input or output flows.

A multi-functional process is a unit process, yielding more than one functional flow including co-production, combined waste processing, and recycling. Coproduction is a multi-functional process having more than one functional outflow and no functional inflow. Recycling is a multi-functional process having one or more functional outflows and one or more functional inflows. Combined waste processing is a multi-functional process having no functional outflow and more than one functional inflow. The most relevant multi-functional processes in bioenergy systems with reference to the types of input and output inventory are the first two cases as illustrated in Figure 2.4. Guinée (2002) distinguishes two steps in solving the multi-functionality problem. The first concerns avoiding burden allocation in accordance with the ISO preference. This is done by specifying the system boundary to a unit operation level (e.g., individual machines) to reduce the number of multifunctional processes or by system expansion. It is accomplished by extending the analyzed product system to include additional functions related to the co-products or recycled wastes. The system then includes more than one functional unit. The term system expansion is sometimes used to refer to the substitution method. The second step concerns solving the remaining multi-functionality problems by allocation on the basis of mass, energy, or economic values. Further discussion on the procedure to deal with allocation procedures and system expansion can be found in Tillman et al. (1994) and Heijungs and Guinée (2007).

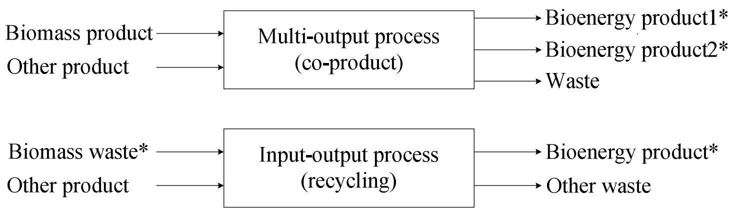

Figure 2.4. Relevant multi-functional processes in bioenergy systems (*= functional flows).

If some waste streams from agriculture are used to make bioenergy products, how the waste was produced is not included in the inventory. It is assumed that its production is free of environmental burden. This, however, requires a clear distinction between products and wastes. To distinguish products from wastes, the economic value of flows can be used as the determining factor. A product is a flow between two processes with a positive economic value, whereas a waste is a flow between two processes with a negative economic value (Guinée et al. 2009). However, there are quite a few cases where we do not know for certain if the price of an agricultural residue is positive or negative, especially when it remains within one company or farm. An example of this is that someone may pay to have their residues picked up, while someone else must pay to receive it. Further, due to technological developments, fluctuations in markets, and governmental intervention, goods may rapidly turn into a waste or the other way around (Heijungs and Wiloso, 2012).

2.3.5 Data variability

An LCA depends on a large number of input elements, and these elements are often based on data of varying quality. The variability in input quality will, in turn, influence the robustness of outcome estimates. This is an important issue that deserves more attention in LCA. A strong challenge for LCA in addressing uncertainty is to provide and track metrics of data quality with respect to how data are acquired (measurements, assumptions, expert judgment), to what extent the data have been validated (checked with respect to mass and energy balance), and how well the data capture technological, spatial, and temporal variations. Some of these uncertainties and variabilities cannot be reduced with the current knowledge (through improvements in data collection or model formulation) because of their spatial and temporal scale and complexity (McKone et al., 2011).

When developing LCIs, one needs quantitative data on the inflows and outflows of the included processes such as resource use, emission data, energy use, and waste production. The limited accuracy and availability of LCI data are generic problems of LCAs. Uncertainty can be due to various reasons that may stem from geographical, temporal, and technological differences. In the case of bioenergy systems, common sources of uncertainty include variability in agriculture yield as it depends on soil conditions, weather, and agricultural practices; variability in biomass conversion technology at different development status; and regional variability as the data are known only for certain countries (Heijungs and Wiloso, 2012). Despite the above difficulties, doing LCA is now much easier than ten years ago since there are now a number of online data repositories for different continents. Some of these databases are quite extensive, though mostly for the USA and EU. The Ecoinvent database, for example, contains thousands of processes from electricity production to transport by truck and from palm oil production to pesticide production (Ecoinvent, 2010).

2.3.5.1 Agricultural process variability

Data variability in the agricultural chain of bioenergy systems is an issue in LCI. For example, there are a large number of potential biomass feedstocks with different characteristics. This presents substantial challenges for current LCA approaches because of the vast scope of information needed to address so many alternatives (McKone et al., 2011). The production of biomass feedstock is likely to involve hundreds to thousands of decision-makers, unlike oil companies that have a less hierarchical structure for decision-making (McKone et al., 2011).

Data gaps and uncertainties are typical to agricultural processes because field measurements are difficult to obtain. Different feedstocks, types of soil, agricultural practice, and climate conditions result in various emission levels so that it is difficult to generalize the environmental performance of biofuels. For example, in the debate around palm oil biodiesel, the emissions from soil related to the agricultural process depend heavily on local circumstances, while the GHG benefits over fossil fuels are global in nature. These emissions vary from very positive to very negative. Such differences are problematic in the sense that they would offer an uncertain basis for policy making (van der Voet et al., 2010).

As previously mentioned, there are three time periods examined to determine the long-term consequences of agricultural activities. The period before agriculture is highly uncertain since the history of when the transformation was taking place is usually unknown. Similarly, the restoration period after the cessation of agriculture activities is highly dynamic. In relation to restoration time, McLauchlan (2006) mentioned that some systems may reach the condition of pre-agricultural time after decades to millennia. From the above description, it is clear that periods before and after agriculture are not easy to adopt in the assessment of land-use impact, mainly due to lack of data availability to follow such a long-term soil quality dynamic. Furthermore, topography, soil, and climate variability within a region prevent direct scaling of LCA balances to geographical scales (Schmer et al., 2008).

2.3.5.2 Conversion process variability

Data gaps and uncertainties related to bioenergy technological routes, particularly on an industrial scale, are not fully resolved. Many advance bioenergy processes are still in a stage of development, and data will become more informative as technologies are deployed. This fact makes LCA methodology difficult to apply during the early phases of a major technology shift (McKone et al., 2011). This is especially true for immature technologies where validation is presently not possible. In the case of second-generation bioethanol, for example, most of the LCA studies use advanced process configurations that are still in developing stages and no existing commercial scale can be referred to for validation. In this regard, there is a risk of under- or over-estimating the real impacts of the current production technology; Therefore, sensitivity analysis is necessary (Wiloso et al., 2012).

There are many technological routes which can be used to convert raw biomass feedstock into bioenergy products. These different technologies all have a different development status as illustrated in Figure 2.5. For example, the production of heat by direct combustion of biomass is historically practiced and still the leading bioenergy application throughout the world. For a more energy efficient use, modern and large-scale heat applications are often combined with electricity production (combined heat and power) systems. The use of biomass residues for second generation biofuels production would significantly decrease the potential pressure on land use and improve GHG emission reductions when compared to some first generation biofuels, leading to lower environmental risk. These second-generation technologies mainly use lignocellulosic feedstocks for the production of ethanol, synthetic diesel, or aviation fuels. In this regard, they are still immature and require further development to demonstrate reliable operation on a commercial scale (IEA-Bioenergy, 2009).

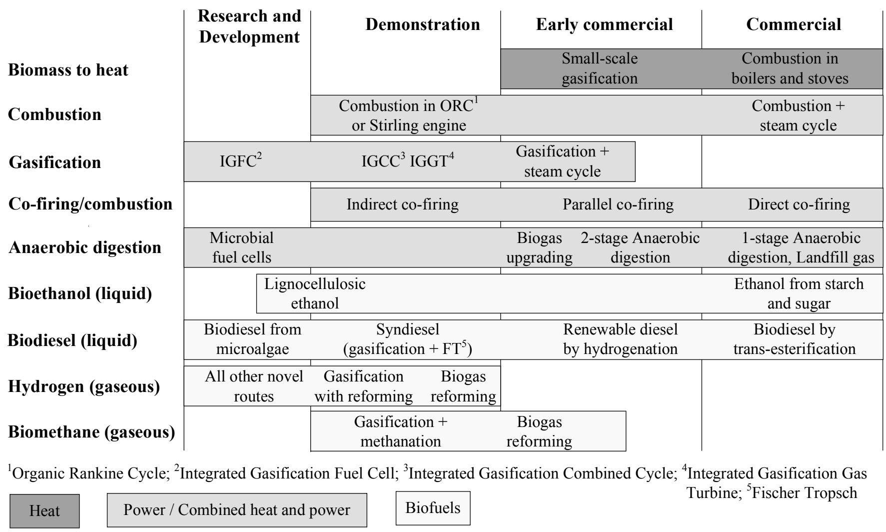

Figure 2.5. State of the art of the conversion technologies for bioenergy (modified from IEA-Bioenergy, 2009).

2.3.5.3 Regional variability

Data gaps in bioenergy LCA are also present with respect to coverage of feedstock types and of geographical areas with an over-representation of Europe and North America (Cherubini and Strømman, 2011). Economic and political interactions that influence land use can cause more variation as the system boundary expands across ecosystems and political borders (Singh et al., 2010). Many studies also show that water consumption varies significantly, depending on regional irrigation requirement and practices (Borrion et al., 2012).

2.4 Impact assessment

In general, environmental impact assessment can be regarded as either potential impact or real impact. But in LCA, only potential impact or maximum possible impact is considered (Baumann and Tillman, 2004). In addition, impact category should be mutually independent in order to avoid double counting of environmental burden. Life cycle impact assessment (LCIA) consists of seven activities, i.e. selection of impact category, classification, characterization, normalization, grouping, weighting, and data quality analysis. According to the ISO standard for LCA (ISO, 2006), the first three are mandatory, while the rest are optional. Two aspects of LCIA that need further elaboration with regard to bioenergy systems are impact categories and regionalized impact assessment.

2.4.1 Impact categories

It is important to properly select the set of relevant impact categories in the bioenergy systems under study. Areas of protection in environmental impact assessment include ecosystem health, human health, resource availability, and man-created environment. Assessment of bioenergy production from specific biomass is suggested to be based on a complete set of impact categories, including climate change, ozone depletion, human and ecotoxicity, photo-oxidant formation, acidification, eutrophication, land-use impacts, and depletion of abiotic resources. But McKone et al. (2011) suggested a balance between being comprehensive and being parsimonious. Failure to address a key impact can lead to incomplete or unreliable information, creating biased decisions. Clearly, the set of chosen impact categories need to be fixed accordingly in the formulation of goal and scopes of the study, but a default minimum would restrict the risk of biased decisions.

Early LCA studies were often limited to net energy output and global warming. The net energy output is an important parameter because, in many cases, the process of producing fuels from the feedstock is energy intensive and, therefore, limits the overall benefit. This parameter (net energy output), however, only determines the technical feasibility of the bioenergy systems rather than being an impact itself. For global warming, the result of the LCI is a list of GHG emissions of all processes in the chain, which are then added up and translated into CO_2 equivalents (so-called carbon footprints). According to the recent review by Cherubini and Strømman (2011), approximately 90 % of bioenergy LCAs include global warming in their evaluation while primary energy demand rates second (71 %). Other impact categories, mainly acidification and eutrophication, are estimated by 20–40 % of the studies. Only 9 % included the land-use category in their impact assessment. The reason for including global warming in most of the studies is because climate policy dominates the scene, while other impacts are not considered as important. In addition, some of them are site specific, which may limit the generalization of the result. Also, there is significantly less agreement in the quantification methods of some impact categories. Particularly notorious are the impacts related to land use, water use, biodiversity, and genetically modified organisms (Heijungs and Wiloso, 2012). With the increasing pressure of a growing population, water use is now also considered as increasingly relevant. Water footprints specify water requirements on a cradle-to-gate basis, and their studies in bioenergy systems are now emerging. It is concluded that energy from biomass has, by far, the largest water footprint compared with other energy sources (van der Voet et al. 2010).

According to van Dam et al. (2010) in IPCC (2011), environmental impacts of bioenergy systems can be distinguished by two classifications based on the coverage of impacted areas. The first is global or regional in nature, including GHGs, acidification, eutrophication, water availability, and air quality. The second is local coverage, including soil quality, biodiversity, water availability, and air quality. Other important classifications related to bioenergy systems are genetically modified organisms and food security (replacement of staple crops and safeguarding local food security). Recent LCA studies typically include a wider scope of impacts supported by sufficient databases and characterization models. Standard life cycle impact assessment methods are available, namely ReCiPe, EDIP, TRACI, LIME, and CML-IA. These methods include selected set of impact categories.

2.4.2 Regionalized impact assessment

Regionalized impact assessment is important in bioenergy system as the boundary also includes agricultural systems. Therefore, assessment criteria should reflect the regional or local conditions of the specific bioenergy system under study. For truly global impact categories like climate change and stratospheric ozone depletion, this is not a problem since the impact is independent of where the emission occurs. For the other impacts, however, they are often regional or even local in nature. In this case, a global set of standard conditions can disregard large and unknown variations in the actual exposure of the sensitive parts of the environment. Sometimes, differences in sensitivities of the receiving environment can have a stronger influence on the resulting impact than differences in inherent properties of the substance (Potting and Hauschild, 1997; Bare et al., 2003). In general, these spatial differentiations relate to the characteristics of both the emitting source and the receiving environment (Finnveden et al., 2009). LCA can address net changes across large geographical areas, but it must also address how the impacts will be experienced on local or regional scales. Accurate assessments must not only capture spatial variation in appropriate scales (from global to farm level) but also provide a process to aggregate spatial variability that can be applied on all geographical scales (McKone et al., 2011).

Several groups have worked on developing site-dependent characterization for LCIA. Recently, methods supporting site-dependent characterization of a range of non-global impact categories were published for processes in Europe, the USA, and within some countries (Finnveden et al., 2009). There are some differences between these data sets partly related to the different definitions of the characterization factors (Seppälä et al., 2006). For example, the variation in acidification impact can be as high as three orders of magnitude between different countries within Europe (Potting et al., 1998). For eutrophication, the uncertainty associated with field emissions contributes more than the uncertainty associated with emissions from the other system components (Basset-Mens et al., 2006).

Inherent differences associated with variability in soil types and complex interactions with local climates must be considered in order to obtain a more representative value in relation to location-dependent aspects. Other types of influencing variability are different soil management and vegetation types. Similarly, dryer climates will rely increasingly on irrigation placing pressure on groundwater supplies. In this regard, the impacts of biofuels on water are highly regional (Sheehan, 2009). This issue is of concern for LCA methods in general as well as a challenge specific to biofuel development.

2.5 Future trends

Most of the assumptions and data used in LCA studies of existing bioenergy systems are related to conditions and practices in Europe and North America, but more studies are now becoming available for other regions such as Brazil, China, and Southeast Asia (Cherubini and Strømman, 2011). First-generation biofuel options based on sugar or starch feedstock are currently available commercially, but lignocellulosic biofuels are expected to be deployed over the year 2020 (IPCC, 2011). In this regard, LCA studies of prospective bioenergy options are more uncertain than LCA studies on current bioenergy feedstocks. The way that uncertainties and parameter sensitivities are handled is an important aspect to be developed. Another important aspect to be resolved in the LCA of bio-based renewable energy systems is the proper way to define system boundaries, particularly in relation to direct and indirect effects of land use and land-use change. Further, consensus on the treatment of biogenic carbon should also be prioritized.

2.6 Conclusion

Bio-based renewable energy sources are presently the largest global contributor to renewable energy as alternative sources of heat, electricity, and biofuel. From the perspective of LCA, they pose more methodological challenges than other renewable energy systems. One of the main reasons is that biomass feedstocks are produced through agricultural systems that are a notorious case to LCA. Agricultural land use has been indicated as the major contributor of GHG emissions in the bioenergy life cycle chain. However, this is not conclusive since quantification methods in terms of functional unit, system boundary, the treatment of biogenic carbon and multi-functional processes, and regionalized impact assessment are not agreed upon. In addition, the inherent variability in the agricultural data and immature production technology increase the uncertainty of the result of LCA studies. There is homework to do for harmonizing LCA framework for bioenergy systems from the point of view of LCA methodology development and demand for policy consideration.

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Chapter 3

Life cycle assessment of second generation bioethanol

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Abstract

This paper aims at reviewing the life cycle assessment (LCA) literature on second generation bioethanol based on lignocellulosic biomass and at identifying issues to be resolved for good LCA practice. Reviews are carried out on respective LCA studies published over the last six years. We use the classification of lignocellulosic biomass to define system boundaries, so that the comparison among LCA results can be thoroughly assessed based on identified system components. A basis for attributing environmental burden for different biomass feedstocks is also suggested. Despite the non-homogeneous systems, we conclude that second generation bioethanol performs better than fossil fuel at least for the two most studied impact categories, net energy output and global warming. For the latter category, carbon sequestration at the biomass generation stage can even consistently offset the GHG emissions from all parts of the life cycle chains at high ethanol percentage (Z85%). The aspect of biogenic carbon and agrochemical input for energy crops and biomass residues, and the effect of removal of the latter from soil have not been treated consistently. In contrast, the exclusion of upstream chain of biomass waste feedstocks is observed in practice. The bioethanol conversion process is mostly based on simultaneous saccharification and co-fermentation, characterized by high yield and low energy input. In this regard, the LCA results tend to under estimate the real impacts of the current technology. The choice of allocation methods strongly influences the final results, particularly when economic value is used as a reference. Substitution of avoided burden seems to be the most popular allocation method in practice, followed by partition based on mass, energy, and economic values.

Keywords
Agriculture; fermentation; life cycle assessment; lignocellulosic biomass; second generation bioethanol; transportation biofuel.

3.1 Introduction

Over the last ten years, there has been a dramatic increase in bioethanol production from 16.9 billion liters in 2000 to 72 billion liters in 2009 (Sorda et. al., 2010). The production is dominated by the USA and Brazil and based on corn grain and sugarcane syrup, respectively. The bioethanol product is mostly used as an alternative transportation biofuel in response to escalating prices of fossil oil, due to limited supplies, and global warming. This development however has been retarded by the growing concerns of competition with food availability, actual net energy output, and consequences related to land-use change (de Oliviera et al., 2005). The competition with food in the USA is reflected by reallocation of 20% US corn to ethanol production in 2006, and this allocation has been partially blamed for the increase of food prices between 2003 and 2008 (Sorda et. al., 2010). In Brazil, the high production cost for sugarcane ethanol is governed by the price of raw materials that account for 70% of the total production cost (Soccol et al. 2010). Given this background, there is a need to explore alternative feedstocks such as non-edible lignocellulosic biomass. This type of feedstock is available in abundance in many countries/regions, and its utilization only competes with food resources to a limited extent. In many cases, this kind of feedstock does not need fertile land or extensive maintenance for its generation, so that the potential environmental and social impacts of the biofuel system are expected to reduce to a great extent. Another motivation to explore such lignocellulosic based biofuels is to improve the emission balance of greenhouse gases (GHGs). In this paper, we follow a terminology of the bioethanol derived from non-edible lignocellulosic biomass as second generation bioethanol (Bright et al., 2009; Garcia et al., 2009a; Slade et al., 2009).

Although there is much potential for lignocellulosic feedtocks, the realization for bioethanol production target in many countries has been so far discouraging. In the US for example, high production costs make bioethanol as a transportation biofuel still prohibitively expensive (Schnoor et al., 2011). The reason for these high production costs is partly related to the characteristic of lignocellulosic feedstock that need advanced processing technologies including pretreatment, hydrolysis of cellulose and hemicelluloses, and co-fermentation of the resulted sugars (Hamelinck et al., 2005) As a result, recently, the US Government has reduced the cellulosic ethanol mandate from 250 to only 6 million gallon per year (Schnoor et al., 2011). This suggests that the conversion technology for bioethanol production at the commercial stage remains insufficient. More time is still needed for developing advanced and efficient technology. In conclusion, Phalan (2009) mentions that the promise of replacing fossil oils with biofuels may still not be applicable to a great extent in some countries, but can overall help to diversify supply and reduce our dependence on fossil fuels.

Life cycle assessment (LCA) is a method to evaluate the inputs, outputs, and potential environmental impacts of a product system throughout its life cycle (ISO, 2006). Despite the fact that there is an ISO standard (ISO 14040, 2006), the application of LCA in practice is not always straight forward and indeed LCA studies on similar products yield diverging results. This is particularly true when studying agricultural systems for which the parameters vary depending on their specific conditions. Important aspects pertaining to the LCA methodology of biofuel system are the definition of the system boundary, the choice of functional unit, the choice of allocation methods, the treatment of biogenic carbon, the selection of impact categories, the choice of reference system, and the effect of biomass removal from soils (Singh et al., 2010; van der Voet et al., 2010; Cherubini and Strømman, 2011). Recent review papers have shown some conflicting LCA outcomes when using different allocation methods, in particular the use of economic values as opposed to physical properties as a basis to allocate burdens from multi-output processes (van der Voet et al., 2010). The same authors also pointed out the importance of including or excluding biogenic carbon in the outcome of the LCA results. In addition, many LCA studies of cellulosic bioethanol system have not included important system components, particularly in the agricultural chain where the type of feedstocks vary (energy crops, biomass residues, and biomass wastes) and in bioethanol production where different levels of technology (current and future scenarios) are adopted (Cherubini and Strømman, 2011; Spatari et al., 2010). This creates problem when comparing different bioethanol systems.

The above-mentioned reviews on LCA of biofuel systems (van der Voet et al., 2010; Cherubini and Strømman, 2011) and the earlier ones (von Blottnitz et al., 2007; Cherubini and Strømman, 2009) often cannot easily conclude if there are benefits with biofuel over fossil fuel. It is even not possible to clearly differentiate the performance between first and second generation biofuels, or among different energy crops, biomass residues and wastes (van der Voet et al., 2010). These studies cover a wide range of variables, such as different feedstock types (edible and non-edible resources) and biofuel types (biodiesel and bioethanol). In this context, variation in the assessment results may be caused by not only the classical problem related to the freedom to choose specific LCA methodology, but also by differences in the real world situation (van der Voet et al., 2010). These reasons are mixed and not easily differentiated from each other, creating complication in the comparison and interpretation of LCA results. The current review therefore limits its coverage only to LCA studies describing second generation bioethanol based on lignocellulosic feedstock, so that a comparison can be made on a rather homogeneous system. To date, at least three review papers on bioethanol LCAs specifically based on lignocellulosic biomass have been published (Singh et al., 2010; Fleming et al., 2006; Williams et al., 2009), but with a rather limited number of cases. Our review conveys a more comprehensive number of papers on the LCAs of lignocellulosic bioethanol system published over the last six years. Also, our study identifies the coverage of system components within the life cycle chains in relation to biomass feedstock specifications, so that the comparison among LCA results can be thoroughly assessed based on identified system components.

Ultimately, this paper aims at reviewing the LCA literature on second generation bioethanol based on lignocellulosic biomass and at identifying issues to be resolved for good LCA practice, particularly with respect to discrepancies in methodological and practical approaches. Emphasis has been put on system definitions in relation to feedstock specifications (energy crops, biomass residues, and biomass wastes), levels of bioethanol conversion technology (current and future scenarios), bioethanol use as transportation fuel, functional units, allocation methods, and included impact categories. The outcome aims to provide decision makers with an increased understanding of the status of second generation bioethanol based on most studied impact categories. It may also aid researchers to develop a LCA framework for the bioethanol system with correct parameters considering typical problems encountered in the agricultural and bioethanol production chains.

3.2 Methods

This study reviewed 22 papers published between 2005 and 2011, covering 14 different lignocellulosic biomass feedstocks. The selection of the papers was based on previous bioethanol LCA reviews (van der Voet et al., 2010; Cherubini and Strømman, 2011) and more recent publications using Boolean search of Web of Sciences database and Google Scholars. Only peer-reviewed journal articles are included in this current study. They are all LCA studies or claimed to use a life cycle approach to assess environmental impacts, described bioethanol as the system product and used lignocellulosic biomass as the raw materials.

Data from the reviewed papers is presented from Figures 3.2 until Figure 3.6. The references to these survey results can be found in Tables 3A.1, 3A.2, 3A.3, and 3A.4 of Appendix 3A.

The analysis is initiated by presenting a general framework of the expected system components of a complete (cradle to grave) bioethanol system. This framework is then used as a basis for a critical assessment, particularly to seek an explanation as why an LCA study of the same feedstock can end up with different outcomes. Considering data availability, the comparison of the LCA results in terms of environmental impacts of the bioethanol system is limited to only two impact categories that are mostly studied by the reviewed papers, global warming and net energy output. Other important aspects discussed in detail are carbon sequestration, agrochemical input, the effect of biomass removal, biomass transport, enzyme production, bioethanol conversion processes, and bioethanol use as transportation fuel.

3.3 LCA framework for the bioethanol system

The system boundary of the bioethanol system is defined so that at least the agricultural chain and the bioethanol production are included as a cradle to gate boundary, with additional bioethanol use in the case of a cradle to grave boundary. The set of included processes need to be defined precisely, so that there is a firm basis to properly describe the bioethanol system for the different biomass feedstocks. The importance of functional units, allocation methods, and choice of impact categories is discussed at the end of this section. All of these factors, in turn, will cause variation in the overall impact assessment results of the second generation bioethanol. It is found that the main conversion technology to produce ethanol in the 22 reviewed papers is through fermentation routes, mostly preceded by pre-treatment and hydrolysis steps, except two papers (Bright and Strømman, 2009; Stichnothe and Azapagic, 2009) preceded by gasification to produce Syngas followed by direct fermentation or catalytic conversion. A number of separation and purification steps are provided to bring the ethanol concentration to 99.5% (dry ethanol), after which it is then blended with gasoline before being finally used as a transportation fuel (see Table 3A.2 of Appendix 3A). Therefore, the main element of the bioethanol system is as illustrated in Figure 3.1.

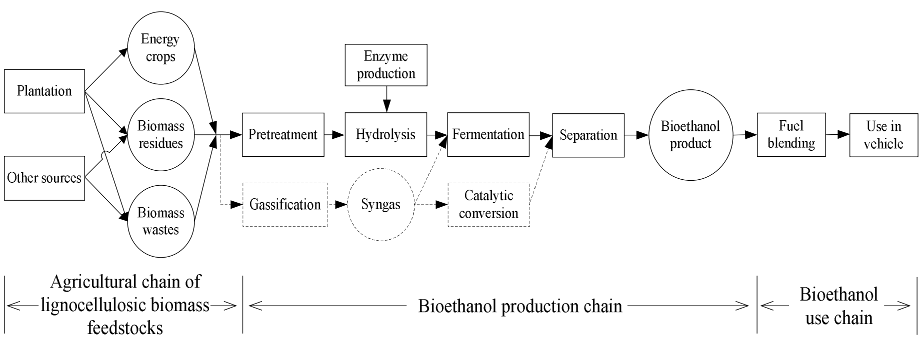

Figure 3.1. The system components of the life cycle of bioethanol system. Dashed areas denote minor ethanol conversion routes.

3.3.1 Feedstock classification

An important feature of the 2nd generation bioethanol as compared to the 1st generation is in the type of feedstock use. The 1st generation bioethanol uses edible sugar and starch as the raw materials, while the 2nd generation bioethanol uses non-edible lignocellulosic biomass. Lignocellulosic biomass refers to plant biomass consisting of cellulose, hemicellulose, and lignin. The term biomass is often used in a broader meaning. For example, Lal (2005) adopts a definition of biomass as renewable organic matter including plant materials, animal product and manure, food processing and forestry by-products, and urban wastes. In this review we will be concerned only with the biomass containing cellulosic materials as potential bioethanol feedstock that may originate from agriculture, plantation, forestry, grasses, or wastes. At the level of biomass generation chains (agriculture, post-harvest processing, or other industrial activities), feedstock classification will bring specific consequences on the way environmental burden to be attributed. From the LCA methodology point of view, it is important to set a criterion as how the burden will be attributed to different feedstocks. In this current study, the following guideline for three different feedstock types is proposed.

Energy crops are crops grown primarily to provide a feedstock for energy production (Spatari et al., 2005), included in this category are those generated from agricultural activities and forest log (Cherubini and Strømman, 2011). In this case, all or most parts of the crops are used as the feedstock for bioethanol production. To avoid excess environmental burden related to agricultural chain, the types of energy crops grown are suggested to use high yielding species (Cherubini and Strømman, 2011) and require minimal maintenance (Luo, et al., 2009) so that it can survive in marginal or degraded lands. In the case of dedicated energy crops grown on productive soil, there will certainly be direct land-use impact. Also, if there is the possibility of competition for land use with other crops, indirect land-use consequences should be included.

Biomass residues are lignocellulosic biomass generated in the plantation, from post-harvest processing, or from other activities. The first group is known as crop residues; these are parts of plants usually left in the agricultural fields after harvest (Lal, 2005; Spatari et al., 2005), including forest residues (wood pieces leftover after timber extraction). As a physical buffer, crop residues protect the soil from direct impacts of rain, wind, and sunlight, leading to improved soil structure, reduced water runoff and soil erosion. Crop residues also contribute to soil organic matter (carbon and other nutrients), soil microbial biodiversity, and soil carbon sequestration (Lal, 2005). In addition to their function to maintain in-situ soil quality, off-site competing uses of the biomass residues include animal feed, fiber, industrial raw materials, and energy feedstocks. In conclusion, biomass residues are not the main products, but may have economic value due to several potential uses. Indiscriminate removal of these residues from the soil may give significant environmental consequences that reduce the above mentioned functions (Lal, 2005). In relation to the development of bioethanol LCAs based on biomass residues, therefore, it is important to consider the above mentioned competing uses or functionality as a reference to the proposed bioethanol system.

Biomass waste is lignocellulosic biomass generated in the plantation, in post-harvest processing or other activities which has no economic value or functional uses. In addition, it is available in excess and in need of treatment or disposal. Biomass wastes result from an activity of which the main product has already been attributed most of the burden. In practice, it is observed that biomass wastes were not attributed with any environmental burden from the upstream chain (Stichnothe and Azapagic, 2009; Kalogo et al., 2007; Zhang et al., 2010; Kemppainen and Shonnard, 2005). Although, in theory, how these wastes were generated need to be further studied. Moreover, its conversion into bioethanol can even be credited for avoiding the need for waste treatment chain.

3.3.2 System components of the bioethanol system

3.3.2.1 Agricultural chain

Important aspects of the agricultural chain in relation to LCA studies are land use, carbon sequestration, addition of agrochemical input, the effects of removing biomass residues from soil, and biomass transport to ethanol conversion facilities. Carbon sequestration, agrochemical input, and biomass removal are related to the land use aspect, but they are treated separately due to their specific relevance in the bioethanol system, as latter shown in the survey results.

With regard to the land use component, direct land use change (dLUC) occurs when new agricultural land for producing bioethanol feedstock displaces prior land use, for example the conversion of forest land into corn plantation (Cherubini and Strømman, 2011), while indirect land use change (iLUC) can be illustrated as an increase in demand for forest log and forest residues in one place due to increased logging activities or deforestation in another place (Phalan, 2009). For the purpose of producing bioethanol feedstock, the latter authors suggested neither using carbon-rich land covers nor displacing an existing agricultural activity. In the case where these preferences cannot be realized, an iLUC may be induced to a certain extent and this indirect impact should be included. In this regard, some frameworks on indirect land use modeling have been proposed (Kloverpris et al., 2008; Schmidt, 2008).

Energy crops are responsible for environmental burden in relation to the use of land area and water, provision of seeds, and agrochemical input such as fertilizers and pesticides needed to grow the plant. Cultivation of agricultural land does not only give benefits in terms of an increased carbon sequestration by the vegetation and soil microorganisms, but can also be a source for atmospheric GHGs, depending on the land use and management options (Lal, 2005). These are emission of N_2O due to the application of nitrogen based fertilizer and organic decomposition, increased emissions of CH_4 due to a decreased rate of CH_4 oxidation, and increased emissions of CO_2 due to organic decomposition (Cherubini and Jungmeier, 2010; Cherubini and Ulgiati, 2010). The same authors also stated that organic carbon is stored in three different pools: vegetation, litter, and soil. Soil carbon sequestration is enhanced when biomass residues are kept in the soil due to an increase in the biodiversity and activities of soil microorganisms (Lal, 2005). The carbon sequestration of the energy crops is usually calculated as CO_2 uptake of the photosynthetic activity of the main vegetation.

Agrochemicals introduced into agricultural soil are mainly fertilizers to enhance designated plant growth and pesticides to minimize pest. The use of nitrogen-based fertilizer may increase the risk of eutrophication and acidification (Cherubini and Jungmeier, 2010) when its application fails to consider migration possibility of the excess (un-adsorbed) fertilizer into surface or ground water. A similar mechanism applies to the use of excess pesticides that increases potential impacts of toxicity to human or animals. Such impacts have been demonstrated by Bai et al. (2010) who concluded that in a bioethanol system based on switchgrass, the agricultural chain is the main contributor to eutrophication, acidification, and toxicity.

Removal of biomass residues may lead to a decline of soil quality and agronomic productivity that further leads to a reduced carbon sequestration capacity of the soil (Lal, 2005). The benefit of reduction of nitrogen-related emissions (N_2O and NO_x, emissions, NO_3^- leaching) due to biomass removal may be offset by reduction in carbon, nitrogen, and other nutrients of the soil (Kim et al., 2009). The level of carbon reduction due to biomass removal is equal to the biogenic carbon of the biomass residues (fraction of the crop) removed from the soil. Soil with a low carbon level contributes to increased levels of GHG in the atmosphere; on the other hand, soils to which crop residues are returned tend to store more soil organic carbon (and nitrogen) than plots where residues are taken away (Reijnders, 2008). In this regard, fertilizer supplement may be necessary to maintain the nutrient level of a healthy soil (Spatari et al., 2005). The nature of these effects however is not consistent from one place to another, depending on local conditions such as climate, soil type, and crop management (Cherubini and Strømman, 2011). Other important inventory items related to the biomass removal from soil are energy use for collection of the biomass (Spatari et al., 2005). For biomass residues generated from a post-harvest processing unit (non-agricultural soil), the consequences of its removal from the site should be considered by referring to its competing uses such as heat feedstock, animal feed, fiber, fertilizer, or compost. If the biomass is available in excess, it may also be treated as wastes with no burden attribution.

Biomass transport from the plantation to the bioethanol production plant is also an important aspect of the bioethanol system. The bioethanol production site is supposedly not too far away from the agricultural field, so that environmental burden due to transportation of the biomass feedstock is not really a problem. This consideration also applies to transportation activities at different chains within the life cycle. The sensitivity of different transport distances to the overall LCA outcomes has been demonstrated by Bai et al. (2010). Table 3A.1 shows that the range of distances for biomass transportation in the reviewed papers is between 20 and 180 km. Consideration of transportation distances is related to the potential energy yield of the feedstock relative to energy used for transportation. The manageable transportation distance also depends on water content of the biomass. It will control energy density and quality of the feed- stock at the bioethanol production gate since 'wet' biomass will deteriorate faster during long-distance delivery. According to International Energy Agency (IEA-Bioenergy), maximum economic transport distance of biomass for bioenergy is limited to 100 km (Bauen et al., 2012).

3.3.2.2 Bioethanol production chain

Lignocellulosic biomass conversion technologies are also the source of variation in the outcome of the overall impact assessment. The conversion of the biomass into bioethanol consists of several processing steps, such as pretreatment to remove lignin from the fiber matrix or hydrolysis of hemicelluloses to C5-sugar (pentose), hydrolysis of cellulose to produce C6-sugar (glucose), and fermentation to convert both sugars into bioethanol. Yeast is conventionally used to convert glucose only, but recently some micro-organisms are known to be able to consume both C5- and C6-sugars, giving a higher bioethanol yield. Process configurations can be arranged as SHF (separate hydrolysis and fermentation), SSF (simultaneous saccharification and fermentation), SSCF (simultaneous saccharification and co-fermentation), or the most advance process CBP (consolidated bioprocess). These different process configurations reflect an increasing level of technology. Co-fermentation of C5- and C6-sugars and CBP process in particular will give a higher yield and require less energy input. However, these advanced processes are still in developing stages and classified as near-term and long-term technologies, depending on the maturity of the technology. The drawback of using such future technologies is that no validation can be made since no commercial processes yet exist. LCA analysis focused on the uncertainty aspects of the application of this emerging production technology has been studied thoroughly by Spatari et al. (2010).

Cellulase enzyme used to hydrolyze cellulosic polymer into sugar monomer is known to be a dominant factor in the overall cost of bioethanol production. The production of this enzyme is expensive and energy intensive. It costs about \$0.50 per gallon of cellulases from

Novozymes (Schnoor, 2011). Its coverage in the inventory of bioethanol system is influential on the overall outcome of the LCA studies. Another important feature in the bioethanol production chain is the distillation of fermentation broth at rather low (8%) ethanol concentration to produce 95% ethanol, and the following de-hydration process to bring the ethanol up to 99.5% purity. These processes obviously require large amount of energy to remove water, but the final impact on net-energy output would clearly depend on energy mixtures of each country. The same bioethanol system can end up with different LCA results due to energy variability by country.

3.3.2.3 Bioethanol use chain

Bioethanol use as transportation fuel refers to the combustion of the fuel mix (5%, 10%, 85% ethanol in gasoline) in a vehicle internal engine. Tail pipe emission in this case is the most important aspect to be considered since ethanol blending and pure gasoline will certainly produce different emissions. Up to 10% ethanol in the fuel blend can be used in a conventional vehicle, while an 85% ethanol fuel needs a modified engine (flexible fueled vehicle). According to Festel (2008) and Balat et al. (2008), the energy density of bioethanol (21.14 MJ/L) is around 33– 34% less than that of gasoline (32 MJ/L). But the combustion efficiency of a fuel blend in terms of km distance traveled is determined also by the type of vehicles, type of roads, fuel composition, and the speed of the vehicle. High ethanol percentage in the fuel blend seems to be more helpful to clearly see the environmental performances of using bioethanol in comparison to conventional gasoline (Bai et al., 2010). In this relation, 100% ethanol blend is also used in the study, but only as a reference for fuel use comparison (see Table 3.3).

3.3.3 Aspects of LCA methodology

3.3.3.1 Functional unit

According to ISO 14040 (2006), the functional unit is defined as the quantification of the identified functions (performance characteristics) of the products. Its main role is to be used as a reference to quantitatively connect inputs and outputs of a life cycle inventory. In this way, LCA results of the same functional unit could be compared from one another, provided that the system boundary is also similar. Proper functional unit that reflects the reality well is very important in the LCA study since different choice of functional units from the same system may result in different outcomes (van der Voet et al., 2010). In bioethanol systems, the functional unit can take many forms depending on specific conditions of the system which is formulated in goal and scope of an LCA study. There are two main concerns related to this parameter: which function to choose and what unit that reflects reality well. Typical functional units of a bioethanol system in the reviewed papers are input land area, volume or mass of input biomass, volume or mass of ethanol product, caloric value of ethanol product, and driving distance of a car. These choices of functional units are driven by the main questions or goals of the LCA study. For example, to compare the benefit of gasoline and bioethanol systems as transportation fuels will lead to a functional unit in terms of 1-km driving distance. A functional unit in the form of 1 MJ would be more appropriate to compare the best use of biomass as bioenergy (bioethanol, heat, or electricity). Therefore, it may be difficult to interpret results of different functional units. In the case that comparison is still to be made, van der Voet et al. (2010) recommend to re-calculate two LCA studies of different functional units by first making their units the same. To be able to perform this procedure, the boundary of the compared systems should be well defined. As defined above, usually a functional unit refers to the characteristics of a product. However, in the case of biorefinary or parallel processes which result in multiple products, but with no clear criteria to choose the main product, an input reference flow is often used, such as in the case of Uihlein and Schebeck (2009). Detailed data are given in the following section (see Figure 3.5).

3.3.3.2 Allocation methods

Allocation is a procedure to attribute environmental burden of multi-functional processes to their input or output flows of the products under study. With respect to second generation bio-fuel, specific problems that often appear are in relation to different types of feedstock that may give specific consequences to burden allocation. In relation to the above feedstock classification, we suggest the following allocation criteria. Processes related to the energy crops will receive most of the burden, mainly from the burden of growing the plant. In this regard, energy crops are treated as a main product in agricultural chain. In contrast, wastes are not attributed with any environmental burden; in fact, their conversion into bioethanol can be credited for avoiding the use of waste treatment chain. For biomass residues, the attribution of the burden is not straight forward as they will be treated as co-product or by-product. Referring to Clean Development Mechanisms, Singh et al. (2010) defined co-products, by-products, and wastes according to their economic values. Co-products have similar revenues as the main-product, by-products have a much lower value than co-products, and wastes have no or even a negative value. Based on these definitions, theoretically, the weight of environmental burden attributed to biomass residues should be between energy crop (fully attributed) and waste (no attribution), depending on the degree of economic values or functional uses (soil conditioner, compost, heat feedstock) of the biomass residues. However, we still need further elaboration as to how this kind of approach will be specifically applied in relation to different feedstock classification for bioethanol systems. The problem of allocation methods is more pronounced in the second generation bioethanol than the first generation bioethanol (van der Voet et al., 2010; Luo et al., 2009a). This unique problem of allocation is obviously driven by different types of feedstock generated in the agricultural chain.

ISO 14040 suggests the system expansion in the first place, but allows an allocation approach to deal with co-products as well. This points to obvious methodological choices. Deviating from ISO, the EU Directives on biofuel suggest allocation based on energy content in preference to the system expansion (van der Voet et al., 2010). We deliberately elaborate more on the allocation approach to demonstrate how these three different feedstocks should be treated, although it may not be as important in the system expansion approach.

3.3.3.3 Impact categories

Impact categories of concern for biofuel LCAs are mainly whether or not the systems give surplus energy, followed by concern on global warming. Besides these two, there are many other categories of high relevance, although these have not been considered in sufficient detail. These include land and water use in relation to increasing pressure of growing population (Sheehan, 2009), toxicity and biodiversity (Phalan, 2009). In this relation the latter authors emphasize that producing biofuel from land other than degraded land is likely to increase GHG emission, damage biodiversity, and affect food security. Agriculture for the feedstock generation is not the only part of the chains that required water. Koh and Ghazoul (2008) reported an estimate that bioethanol production required 4 gallons of water per gallon bioethanol produced, while a fuel oil refinery only needs 1.5 gallons of water per gallon fuel oil produced. The importance of other relevant impact categories in relation to bioethanol system, such as eutrophication and acidification, has been described in Section 3.3.2.1.

3.4 Result and discussion

3.4.1 Feedstock classification

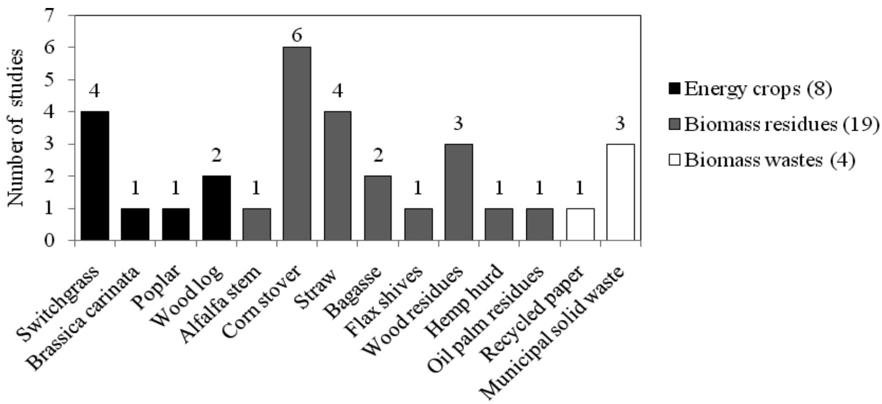

Figure 3.2. Types of lignocellulosic biomass feedstock in the LCA studies of the reviewed papers. Numbers within bracket in the series legend denotes total number of studies on each feedstock classification.

Based on our feedstock classification described in Section 3.3.1 and data in the tables of Appendix 3A, energy crops include switchgrass, *Brassica carinata*, poplar, and wood log; biomass residues include alfalfa stem, corn stover, straw, bagasse, flax shives, wood residues, hemp hurd, and oil palm biomass residues; and biomass wastes include recycled paper and municipal solid wastes. We included a biomass feedstock as energy crops if it is the main product and used most parts of the plant for example as in the case of wood log (Slade et al., 2009; Kemppainen and Shonnard, 2005), while wood residues as biomass residues (Bright and Strømman, 2009; Zhang et al., 2010). Detail on the feedstock type of the reviewed papers can be seen in Figure 3.2. It is clearly seen that the most studied feedstocks are biomass residues (19 studies), followed by energy crops (8 studies), and biomass wastes (4 studies).

3.4.2 Agricultural chain

Figure 3.3 shows that no study included land-use in the inventory calculation, both at the transformation or occupation levels (Canals et al, 2007). A possible reason for this is that the energy crops are mainly grown on marginal or degraded soils. Such land is believed not to compete with other utilization such as for food, feed or fiber. Therefore, its environmental impact is minimal; in fact, utilization of these lands for cropping can even increase the capacity of carbon sequestration (Cherubini et al., 2009).

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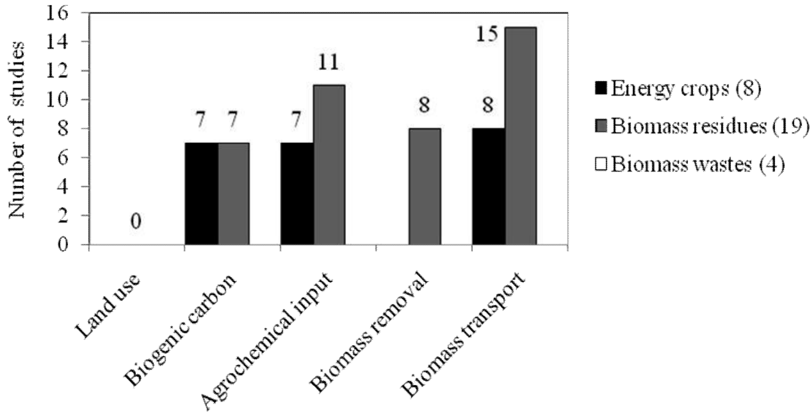

Figure 3.3. System components of the agricultural chain included in the LCA studies of the reviewed papers. Numbers within bracket in the series legend denote total number of studies on each feedstock classification. No environmental burden was attributed to the biomass wastes, i.e. the values of all parameters = 0.

In Figure 3.3, LCA studies that considered carbon sequestration are grouped as biogenic carbon on the horizontal axis. A rather high percentage (88%) of the LCA studies based on energy crops includes this aspect in their system boundary, while for biomass residues, only 37% of the studies considered carbon sequestration. Data in Table 3.3 will demonstrate that the aspect of biogenic carbon hold a very important feature in the agricultural chain of the overall bioethanol system.

A large number of studies based on energy crops (88%) and biomass residues (58%) included agrochemical input in the analysis. These agrochemical inputs are in the form of fertilizers and pesticides for energy crops and fertilizer as nutrient replace- ment in the case of biomass residues. The fertilizer is added only when the rate of biomass removal interferes with its function to maintain top soil quality. To avoid such environmental burden, the removal of biomass residues therefore is done only at a rate (percentage) that does not bring any consequences to the reference systems. Data in Figure 3.3 shows that only 42% of the studies based on biomass residues considered the consequences of biomass removal. This category includes those that removed only the surplus biomass in the amount that does not affect the environment and those that compensated the biomass removal by inorganic fertilizer as nutrient replacement. The rest of the studies (58%) did not mention or might not consider biogenic carbon in the inventory. No environmental burden attributed to the biomass wastes as a feedstock category means that these studies excluded the agricultural chain from the life cycle inventory. The aspect of biomass transport is self-explained and the impacts will depend partly on the distance between agricultural site and the bioethanol processing facilities. In this case the distances are between 20 km and 180 km. A larger distance means that a higher amount of fuel is needed and more pollution from the tail pipe is emitted.

The above conditions show a very high variation in the practice of defining the system boundary in the agricultural chain, particularly with respect to land-use, biogenic carbon, agrochemical input, biomass removal consequences, and biomass transport. These facts bring a strong message that a clearer guidance to properly include or exclude certain parts of system components based on feedstock classification is needed. The main motivation is in order to be able to allocate environmental burden accordingly.

3.4.3 Bioethanol production and use chains

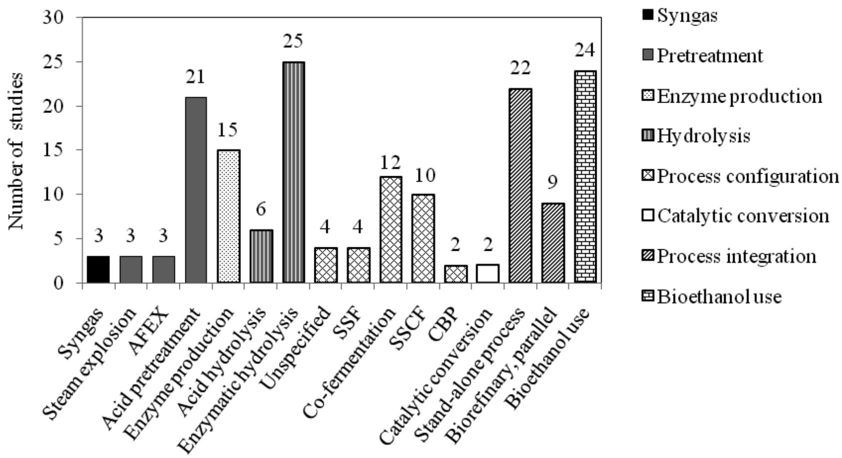

Figure 3.4. System components of the bioethanol production chain included in the LCA studies of the reviewed papers. AFEX=ammonia fiber explosion, SSF=simultaneous saccharification and fermentation, SSCF=simultaneous saccharification and co-fermentation, CBP=consolidated bioprocess.

Figure 3.4 shows important system components in the bioethanol production and use chains that include pretreatment, enzyme production, hydrolysis, fermentation, and bioethanol use. The pretreatment technology used in the bioethanol production chain is dominated by acid pretreatment (21 studies), followed by steam explosion (3 studies), and AFEX process (3 studies). The conversion of cellulose into sugar is dominated by enzymatic hydrolysis (25 studies), while only 6 studies dealt with acid hydrolysis. Although quite some studies included enzymatic hydrolysis, only 15 studies incorporated enzyme production in their inventory analysis.

With regard to the process configuration in terms of hydrolysis and fermentation processes, none of the studies was based on a relatively simple technology such as SHF. Most of them used advanced technology involving co-fermentation of C5- and C6-sugars (12 studies), SSCF (10 studies), and CBP (2 studies). The syngas process and the following catalytic conversion or fermentation was found as a minor technological route in the LCA studies, possibly because this technology is less established and therefore less data are available as compared to the fermentation route.

Figure 3.4 shows that only 88% of the studies based on fermentation reported their specific process configurations, 75% of which used advanced processes such as co-fermentation, SSCF, and CBP. In this review, we differentiate co-fermentation from SSCF, and refer to the first one if the authors mentioned co-fermentation only without detail explanation on how the hydrolysis was done. The above trend has been previously reported by Sheehan et al. (Sheehan et al., 2004) who stated that most of the LCA studies are based on projected future technology that is not yet commercially proven. In this regard, there is a risk to underestimate the real impacts of the current production technology (SHF or SSF of glucose only) which is typically lower in bioethanol yield. Fluctuation in the LCA results due to different technological levels has been demonstrated based on simulation of different technological scenarios: near-term (SSCF technology in 2010) and mid-term (CBP technology in 2020) (Spatari et al., 2010). Therefore, sensitivity analysis at different levels of technology becomes necessary, so that the conclusion can be understood within the context of the study.

With regard to process integration, 22 studies covered the inventory at the level of bioethanol production only (stand-alone process), isolated from the other system processes. Meanwhile, 9 other studies expanded the inventory of the bioethanol systems to include their respective biorefinery systems or related processes. The examples to these are the inclusion of biodiesel or sugarcane production system in the system boundary in case where palm oil biomass (Lim and Lee, 2011) or bagasse residues (Melamu and von Blottnitz, 2011) were used as feedstocks, respectively.

Table 3.1. Data on sugar recovery and fermentation efficiency in bioethanol production that were referred to by most of the reviewed papers.

Authors	Sugar recovery after pretreatment and hydrolysis (%)		Fermentation efficiency of different sugars (%)		Cited by
	Glucose	Xylose	Glucose	Xylose	
Hamelinck *et al.* (2005)	90	85	92.5	85	(Cherubini and Jungmeier, 2010); (Cherubini and Ulgiati, 2010)
Sheehan *et al.* (2004)	63.5	67.5	95	90.2	(Spatari et al., 2005)
Wooley *et al.* (1999)*	80	85	92	85	(Bright and Strømman, 2009); (Kemppainen and Shonnard, 2005)
Aden *et al.* (2002)**	90	90	95	85	(Garcia et al., 2009); (Luo et al., 2009); (Zhang et al., 2010); (Garcia et al., 2009); (Garcia et al., 2010b); (Garcia et al., 2010a); (Garcia et al., 2010c)

* Near term, best of industry, page 60; ** Process parameters, Appendix E.

The potential amount of sugar available for fermentation is governed by the cellulose and hemicellulose content of the biomass feedstocks, and by the effectiveness of the pretreatment and hydrolysis steps. The higher cellulose and hemicelluloses content of a biomass, the higher sugar yield can be achieved as long as the pretreatment and hydrolysis can be done easily. In turn, these resulting sugars will be fermented to yield the final product, bioethanol. Table 3.1 illustrates variation in sugar recovery and fermentation efficiency that were used by most of the reviewed papers in developing a LCA of bioethanol system, reflecting advance level of technology. These data are based on the work done directly (Sheehan et al., 2004; Wooley, et al., 1999; Aden et al., 2002) or indirectly (Hamelinck et al., 2005) at the National Renewable Energy Laboratory (NREL) of the USA. In addition, Table

3.2 shows the gaps between the values observed at the bench scale results (in 2004) and those used in the LCA study based on future projections (Sheehan et al., 2004). The data spread is quite large, between 63.5% and 90% for sugar recovery, and between 0% and 85% for fermentation efficiency of arabinose, mannose, and galactose. It is therefore important to justify clearly the choices of assumed values for process parameters at the pretreatment, hydrolysis, and fermentation stages. These different process scenarios in terms of sugar recovery and fermentation efficiency, at the end, will bring consequences to different amount of required energy input and yield of bioethanol as the final product.

Table 3.2. Sugar recovery and fermentation efficiency at bench-scale and projection used in the LCA study (Sheehan et al., 2004).

Conversion reaction	Yield (%)					
	Pretreatment		Hydrolysis		Fermentation	
	Obser-ved bench scale*	Future project-ion used in LCA	Obser-ved bench scalex	Future project-ion used in LCA	Obser-ved bench scale	Future project-ion used in LCA
Xylan to xylose	67.5	90	-	-	-	-
Arabinan to arabinose	67.5	90	-	-	-	-
Mannan to mannose	67.5	90	-	-	-	-
Galactan to galactose	67.5	90	-	-	-	-
Cellulose to glucose	-	-	63.5	90		
Xylose to ethanol	-	-	-	-	90.2	**
Arabinose to ethanol	-	-	-	-	0	85
Mannose to ethanol	-	-	-	-	0	85
Galactose to ethanol	-	-	-	-	0	85
Glucose to ethanol	-	-	-	-	95	95

* Observed bench scale data refers to the year of 2004 (Sheehan et al., 2004); ** The original paper notes a lower value (85%) than that of the observed bench scale.

3.4.4 Aspects of LCA methodology

Figure 3.5 shows that the bioethanol system in most cases comprises a cradle to grave boundary by including bioethanol-use as transportation biofuel (24 studies), while only in 7 studies the inventory up to the gate of bioethanol production chain was taken into account.

The functional units of the reviewed papers are quite diverse. They are based on land area use for certain period (2 studies), volume or mass of input biomass feedstock (8 studies), volume or mass of bioethanol product (6 studies), caloric value of bioethanol product (8 studies), and driving distance of ethanol blends as transportation biofuel (10 studies).

Likewise, the allocation methods of the reviewed papers are quite varied. They are based on system expansion (1 study), substitution of avoided burden (15 studies), mass partitioning (7 studies), energy partitioning (4 studies), and economic partitioning (3 studies).

The remaining studies either do not mention allocation (6) or consider allocation (4) in their studies. In this paper, we deliberately differentiate between system expansion and substitution with the aim to clearly show different kinds of system expansions really applied by LCA studies in practice. Allocation based on substitution of avoided burden was the most popular allocation method (50%), followed by partition methods (mass, energy, economic based, 47% in total). The allocation method based on system expansion is only 3%; it involves a rather tedious calculation since a lot of information is needed. It seems that the preference to use substitution methods is motivated mainly by data availability and practical considerations. This conclusion is similar to general biofuel LCAs based on more heterogeneous feedstocks (van der Voet et al., 2010).

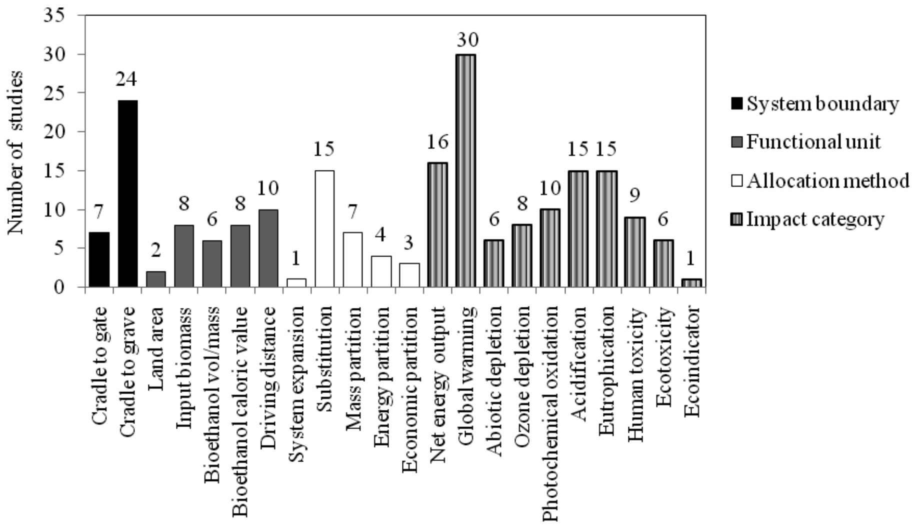

Figure 3.5. Different aspects of LCA methodology in the reviewed papers.

Global warming is the most studied impact category (29 studies) followed by net energy output (18 studies), acidification and eutrophication (15 studies each). Although the bioethanol systems reviewed use rather diverse approaches in terms of LCA methodology, system definition, and level of technology, the conclusion to favor the second generation bioethanol is quite robust for net energy output and global warming. Not all studies declare in detail the methods used to assess the impact of global warming. Only 7 studies mentioned a time horizon of 100 years, and 4 studies explicitly listed the equivalency factors relative to CO_2, i.e. between 21 and 25 for CH_4, and between 296 and 310 for N_2O.

We conjecture, however, that environmental sustainability based on a more complete set of impact categories may give different outcomes. The extent to which they really do so is not verifiable since most of the studies are based on different system boundaries and consider different impact categories. Despite this, bioethanol systems that include productive-land use in the case of energy crops or coal as energy sources are believed to be consistently unsustainable. Relevant impact categories which have not been dealt with by many of these LCA studies are eutrophication, acidification, toxicity, land use, biodiversity, and water use. With regard to the last four categories, they are often not included since the required parameters are not well developed in the LCA methodology and consequently not readily available in the commercial LCA software. Ecoindicator is a single-score impact category based on end-point assessment (Goedkoop and Spriensma, 1999).

3.4.5 Impact assessment results with reference to conventional fossil oil system

Referring to Figure 3.6, the second generation bioethanol provides a more unified direction in favor of its implementation based on two major criteria: net energy output and global warming. This convergence of LCA results cannot be observed in a study with a broader spectrum of feedstocks (van der Voet et al., 2010; von Blottnitz and Curran, 2007; Cherubini et al., 2009). The impact assessment results revealed that 14 studies concluded a positive net-energy output (between 64% and 86% compared to the gasoline system), while only 2 studies (Lim and Lee, 2011; Melamu and von Blottnitz, 2011) reported the opposite. Similarly, 28 studies reported an increase in GHG saving (between 11% and 145% compared to the gasoline system), while only 3 studies (Luo et al., 2009a; Lim and Lee, 2011; Melamu and von Blottnitz, 2011) reported worse cases. We noticed that the divergence took place only in the case of biomass residues. It may suggest that the system components of this type of feedstock are rather difficult to define, as compared to energy crops or biomass wastes. In addition, there are also other technical reasons that explain this divergence as illustrated in the following.

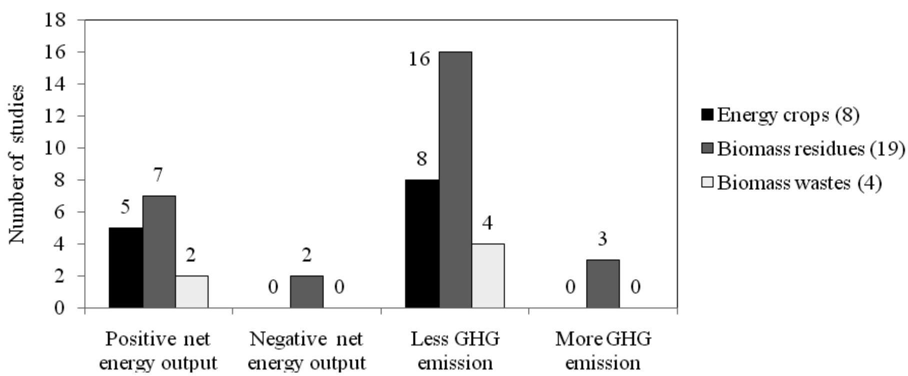

Figure 3.6. Summary of impact assessment results for net energy output and global warming from the LCA studies of the reviewed papers. Numbers within bracket in the series legend denote total number of studies on each feedstock classification.

Luo et al. (2009a) revealed that using corn stover as the feedstock and fossil oil as the reference, the impact scores on global warming are contradicting one another when using different allocation methods. Partitioning based on physical properties (mass or energy content) resulted in smaller global warming, while the same system emitted more GHG when using economic allocation. The partitioning ratios between corn grain and corn stover based on physical properties and economic value are so big that the ratio shifted from 1.7 to 7.5. The ratio based on physical allocation is rather stable, while that based on economic value will vary following market forces that depend on time and location. In this relation, it would be nice if the prices of the compared products are fixed, but in fact price ratios may be so volatile that their usefulness is limited in practice. On the other hand, choosing a physical basis for allocating burden does not always describe the reality well (Guine´e et al., 2004).

Lim and Lee (2011) reported that the inclusion of bioethanol production parallel to a biodiesel system will decrease the environmental performances in terms of output energy and global warming. They assumed a very low bioethanol yield based on 26.5% cellulosic materials contained in the palm oil biomass residues. However, a sensitivity analysis with a higher bioethanol yield (cellulosic materials above 60%) reveals opposite results, a lower energy input and GHG emission, meaning a better environmental performance. Referring to the provided composition of cellulose and hemicellulose of the EFB (empty fruit bunch) in this paper (Lim and Lee, 2011), it seems that the low cellulosic content (26.5%) refers to the use of C6-sugar originating from cellulose only, while the high cellulosic content (60%) suggests co-fermentation of both C5- and C6-sugars originating from cellulose (26.5%) and hemicelluloses (34.4%), respectively. This analysis demonstrates that a difference in technology level of bioethanol conversion (fermentation of C6-sugar only and co-fermentation) also plays an important role in determining the outcome of the LCAs.

Melamu and von Blottnitz (2011) reported that the conversion of sugarcane bagasse into bioethanol in a sugar milling industry will give negative results for net energy output and global warming. They refer to the baseline case, a sugar industry with self-sufficient energy system by burning the bagasse as a heat feedstock to generate boiler steam. In the proposed bioethanol scenario, the heat generated from bagasse is replaced by coal, a typical energy source available in the area (South Africa). Comparison between these two energy scenarios in a sugar industry indicated that coal becomes the dominant contributor to global warming, governing the bad performance of the overall system.

Table 3.3. Contribution analysis on global warming at different ethanol percentages.

Feedstock	% Ethanol in a fuel blend	GHG emission intensity				References
		Agricultural chain		Production of fuel blends[a]	Use of fuel blends	
		CO_2	N_2O			
Brassica carinata	10	--	+	++	+++++++	(Garcia et al., 2009)
Brassica carinata	85	-------	+	+++	++	(Garcia et al., 2009)
Switchgrass	10	--	+	++	+++++++	(Bai et al., 2010)
Switchgrass	100	-------	+	+++	++	(Bai et al., 2010)
Switchgrass	100	-------	+	+++	++	(Spatari et al., 2005)
Corn stover	100	-------	+	+++	++	(Spatari et al., 2005)
Corn stover[b]	100	----	+	+++++++	+++	(Luo et al., 2009)
Flax shives	100	-------	+	+++	++	(Garcia et al., 2009)

\+ denotes GHG emission; – denotes GHG saving; Number of + or – indicates relative emission or saving intensity, respectively, interpreted from the figures of the respective papers; More + than – means net GHG emission; More – than + means net GHG saving;

[a] Environmental burden for the production of fuel blends include both for the gasoline and bioethanol fractions;

[b] Based on economic allocation.

Pimentel and Patzek (2005) worked on an energy analysis of bioethanol production based on corn, switchgrass, and wood, and found negative energy outputs as high as 29%, 50%, and 57% respectively. Since this is not an LCA study, the data are not included in Figures 3.2 till Figure 3.6, but discussed here because of its relevance. These negative energy outputs, however, seem to be related in different ways in treating co-products between energy analysis and LCA studies. For example, in the case of the corn system, the authors did not incorporate co-products (corn stover) in the calculation as a typical approach in an LCA study. An attempt to recalculate Pimentel's energy assessment by considering also the co-products in the agricultural (feedstock) and biorefinery (energy product) chains has been made (Luo et al., 2009b). It was found that the net energy output of the overall feedstock system (corn grain and corn stover) shifted from negative (–5.8 MJ/L) to positive (+22 MJ/L) values. This case nicely demonstrates the consequence of removing some of the relevant components from the system, leading to contradictory results.

Up to this point, we have shown that many LCA studies of cellulosic bioethanol system have not consistently included important system components, particularly in the agricultural chain where the type of feedstocks (energy crops, biomass residues, and biomass wastes) varies and where different levels of bioethanol production technology (current and future scenarios) are adopted. In this regard, there is a need to develop common rules for applying LCA to agricultural systems, so as to increase the comparability of studies based on the identified system components. If the difference is large, the environmental impact of certain parts of life cycle chain may fluctuate depending on the coverage of system components and the technology levels at that specific part. In such cases, a contribution analysis is therefore necessary. It is a method to sort out hot spots, the parts of the system components in a life cycle that make up most of the environmental burden or impact. The changes of the hot spot can be seen as a shift in dominant contributors (agricultural chain, production of fuel blends, or use of fuel blends) relative to the overall environmental burden, as illustrated in the following.

The data in Table 3.3 demonstrates that agricultural chains of energy crops and biomass residues are consistently found as the dominating factor in term of GHG saving to determine the overall impact on global warming at high ethanol percentage (≥ 85%). On the other hand, the use of fuel blends dominated by GHG emission controls the direction of overall impact at low ethanol percentage (10%). There are two important aspects in the agricultural chain related to the impact on global warming pointing at different directions: carbon sequestration and emission of nitrous oxide and methane. In the case of high ethanol percentage, the carbon sequestration quantified as carbon uptake from the atmosphere offsets the emission from all parts of the life cycle chains (N_2O and CH_4 emissions, production of fuel blend, and use of fuel blend). When the ethanol fraction in the fuel blend is reduced to only 10%, such as in the case of *Brassica carinata* and switchgrass, the capacity of carbon sequestration decreases significantly, leading to a net GHG emission. Table 3.3 also shows that the hotspots of the life cycle chain shift from the use of fuel at low ethanol percentage to the fuel production at high ethanol percentage.

The above analysis shows the importance of indicating the blending percentage of the ethanol in the fuel blends when comparing environmental performance of a bioethanol system with reference to conventional fossil oil. It is also recommended to use a high ethanol percentage (high gasoline displacement) in the LCA study, so that its effect on the overall GHG emission relative to the reference system can be seen even clearer. One exception at high ethanol percentage is the corn stover case of Luo et al. (2009a) in which the dominant burden was found to be the bioethanol production chain. As discussed previously, this appears to be influenced by the choice of economic allocation in this multi-product system.

3.5 Conclusions

We suggest that a classification of lignocellulosic biomass can be used as a guidance to include or exclude certain parts of system components and to attribute environmental burden accordingly. With reference to the feedstock generating history, energy crops are treated as a main product in the agricultural chain and receive most of the burden of growing the plantation. In practice, biomass wastes are not attributed with any environmental burden. Theoretically, however, how these wastes were generated need to be considered. In fact, waste conversion into bioethanol can be credited for avoiding the need for waste treatment chain. Biomass residues can be treated as co- or by-products, and the share of environmental burden attributed to them is between energy crops (fully attributed) and biomass wastes (no attribution), depending on their economic values or functional uses.

Although the bioethanol studies reviewed use rather varied approaches in terms of LCA methodology, definition of bioethanol system, and level of technology, the conclusion to favor the second generation bioethanol is quite robust at least for the two most studied impact categories, net energy output and global warming. For the latter category, carbon sequestration at the biomass generation stage can even consistently offset the GHG emissions from all parts of the life cycle chains at high ethanol percentage ($\geq$ 85%). Studies based on a more complete set of impact categories such as eutrophication, acidification and toxicity are likely to lead to different outcomes, particularly when productive land or coal is included in the bioethanol system. Other relevant impact categories which have not been sufficiently dealt with are land use, biodiversity, and water use. In the agricultural part of the chain, the aspect of biogenic carbon and agrochemical input for energy crops and biomass residues, and the effect of removal of the latter from agricultural soil have not been treated consistently.

Most of the LCA studies use advanced process configurations (such as SSCF and CBP) that are still in developing stages and no existing commercial scale can be referred to for validation. In this regard, there is a risk of under-estimating the real impacts of the current production technology which are typically lower in bioethanol yield and consumed more energy. Sensitivity analysis is necessary, so that the conclusion can be understood within the context of a specific technology level.

The choice of allocation methods has a strong influence on the overall LCA results. This may lead to contradictory outcomes, particularly when economic value (which depends on market forces) is used as opposed to physical properties (mass or energy content) which are relatively more stable. Allocation based on substitution of avoided burden seems to be the most popular allocation method in practice, followed by partition based on mass, energy, and economic values. The preference to use such allocation methods is motivated mainly by practical considerations and data availability.

Overall, we conclude that a clearer guidance in terms of best practice for LCA of second generation bioethanol based on lignocellulosic biomass is needed, with a focus on methodological aspects (functional unit, allocation, system boundaries, impacts included) as well as data aspects (technological level, feedstock specification).

3.6 Acknowledgement

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Appendix 3A. Overview of the studies

This section contains detailed information of data presented in Figures 3.2-3.6. The references to the survey results can be found in Tables 3A.1-3A.4.

Table 3A.1.
System components of the agricultural chain.

Class	References	Feedstock[a]	Biogenic carbon[b]	Agrochemical input[c]	Consideration on the effect of biomass residues removal[d]	Biomass transport[e]
Energy crops	(Bai et al., 2010)	Switchgrass	C-sequestration	Fertilizer, Herbicide	Not applicable	20 km
	(Garcia et al., 2009)	*Brassica carinata*	C-sequestration	Fertilizer, Pesticide		25 km
	(Kemppainen and Shonnard, 2005)	Wood log	Excluded	Excluded		Included
	(Slade et al., 2009)	Soft wood	C-sequestration	Fertilizer		Included
	(Spatari et al., 2010)	Switchgrass	C-sequestration	Fertilizer, Herbicide		65 km
	(Spatari et al., 2005)	Switchgrass	C-sequestration	Fertilizer, Herbicide		90 km
	(Cherubini and Jungmeier, 2010)	Switchgrass	C-sequestration	Fertilizer, Herbicide		120 km
	(Garcia et al., 2010a)	Poplar	C-sequestration	Fertilizer, Pesticide		25 km
Biomass residues	(Botha and von Blottnitz, 2006)	Bagasse	–	Fertilizer	–	50 km
	(Bright and Strømman, 2009)	Wood residues	–	–	Surplus residues only	50-120 km
	(Garcia et al., 2009)	Flax shives	C-sequestration	Fertilizer, Pesticide	–	180 km
	(Luo et al., 2009)	Corn stover	C-sequestration	Fertilizer, Pesticide	60% removal	56 km
	(Searcy and Flynn, 2008)	Straw	Excluded	–	Assumed no effect	Included
	(Searcy and Flynn, 2008)	Corn stover	Excluded	–	Assumed no effect	Included

Class	References	Feedstock[a]	Biogenic carbon[b]	Agrochemical input[c]	Consideration on the effect of biomass residues removal[d]	Biomass transport[e]
	(Slade et al., 2009)	Straw	Excluded	Nutrient replacement	Considered	Included
	(Spatari et al., 2010)	Corn stover	C-sequestration	Nutrient replacement	50% removal	65 km
	(Spatari et al., 2005)	Corn stover	C-sequestration	Nutrient replacement	62% removal	90 km
	(Zhang et al., 2010)	Hard wood residues	–	–	–	–
	(Zhang et al., 2010)	Soft wood residues	–	–	–	–
	(Zhang et al., 2010)	Corn stover	–	–	–	–
	(Uihlein and Schebek, 2009)	Straw	–	–	–	100 km
	(Cherubini and Ulgiati, 2010)	Corn stover	–	Nutrient replacement	Considered	120 km
	(Cherubini and Ulgiati, 2010)	Wheat straw	–	Nutrient replacement	Considered	120 km
	(Lim and Lee, 2011)	Oil palm residues	C-sequestration	Fertilizer, Pesticide	Considered	100 km
	(Melamu and von Blottnitz, 2011)	Bagasse	–	–	–	–
	(Garcia et al., 2010b)	Hemp hurd	C-sequestration	Fertilizer	–	Included
	(Garcia et al., 2010c)	Alfalfa stem	C-sequestration	Fertilizer, Pesticide	–	100 km
Bio-mass was-tes	(Kalogo et al., 2007)	Biodegradable MSW	Excluded	Excluded	Not applicable	Excluded
	(Kemppainen and Shonnard, 2005)	Recycled paper	Excluded	Excluded		Included
	(Stichnothe and Azapagic, 2009)	Biodegradable MSW	Excluded	Excluded		Excluded
	(Zhang et al., 2010)	Biodegradable MSW	Excluded	Excluded		–

MSW=municipal solid waste, – = not mentioned. No land-use aspect is considered in the agricultural chain.

[a] Wood log (Slade et al., 2009; Kemppainen and Shonnard, 2005) as the main biomass product is treated as an energy crop since most parts of the crop are used as feedstock, while wood residues (Bright and Strømman, 2009; Zhang et al., 2010) is classified as biomass residues. *Brassica*

carinata is also treated as an energy crop, and assumed to yield only biomass, not oil seeds (Garcia et al., 2009). Alfalfa stem (50% of the biomass) is categorized as biomass residues since the main product is the leaf used as forage (Garcia et al., 2010c). Flax shives are woody parts of the stems (about 25-30%) that are left over from fiber processing (Garcia et al., 2009). The mass ratio between corn grain and corn stover is 1:1 (Luo et al., 2009).

[b] Biogenic carbon in the case of energy crops is CO_2-sequestered during the plant growth, and in the case of biomass residues as carbon sequestered in the biomass fraction removed from the soil.

[c] Fertilizer is added for energy crops to enhance the plant growth, for biomass residues to replace nutrient removed from the soil.

[d] Whether or not the effect of biomass residues soil removal (in term of soil quality: soil organic matter, soil biodiversity, soil carbon sequestration, soil structure) is considered.

[e] Biomass transport from the plantation to bioethanol production plant.

Table 3A.2.
System components of the bioethanol production and use chains.

Class	References	Feedstock	Pretreat-ment[a]	Enzyme production	Hydrolysis	Process configuration[b]	Process integration	Bioethanol use[c]
Energy crops	(Bai et al., 2010)	Switchgrass	AFEX	Included	Enzyme	SSCF	Stand alone	FFV: E10, E85
	(Garcia et al., 2009)	*Brassica carinata*	Acid	Included	Enzyme	Co-fermentation (Aden et al., 2002)	Stand alone	FFV: E10, E85
	(Kemppainen and Shonnard, 2005)	Wood log	Acid	Included	Enzyme	SSCF (Wooley et al., 1999)	Stand alone	Excluded
	(Slade et al., 2009)	Soft wood	Acid	Included	Acid, enzyme	SSF (Sassner, 2008)	Stand alone	Excluded
	(Spatari et al., 2010)	Switch-grass	Acid, AFEX	Included	Enzyme	SSCF, CBP	Stand alone	Excluded
	(Spatari et al., 2005)	Switch-grass	Acid	Included	Enzyme	Co-fermentation (Sheehan et al., 2004)	Stand alone	FFV: E85
	(Cherubini and Jungmeier, 2010)	Switch-grass	Steam explosion	–	Enzyme	SSCF (Hamelinck et al., 2005)	Bio-refinary	Car: 2.45 MJ/km
	(Garcia et al., 2010a)	Poplar	Acid	Included	Enzyme	Co-fermentation (Aden et al., 2002)	Stand alone	FFV: E10, E85
Biomass residues	(Botha and von Blottnitz, 2006)	Bagasse	Acid	Excluded	Acid	Co-fermentation (Kadam, 2002)	Parallel, sugar production	Included
	(Bright and Strømman, 2009)	Wood residues	Acid	–	Enzyme	SSCF (Wooley et al., 1999)	Stand alone	FFV: E85
			Syngas			Catalytic conversion		

Class	References	Feedstock	Pretreat-ment[a]	Enzyme production	Hydrolysis	Process configuration[b]	Process integration	Bioethanol use[c]
	(Garcia et al., 2009)	Flax shives	Acid	Included	Enzyme	Co-fermentation (Aden et al., 2002)	Stand alone	FFV: E10, E85
	(Luo et al., 2009)	Corn stover	Acid	Included	Enzyme	Co-fermentation (Aden et al., 2002)	Stand alone	Car: E10 E85
	(Searcy and Flynn, 2008)	Straw	–	–	Enzyme	Fermentation	Stand alone	Included
	(Searcy and Flynn, 2008)	Corn stover	–	–	Enzyme	Fermentation	Stand alone	Included
	(Slade et al., 2009)	Straw	Acid	Included	Acid, enzyme	SSF (Sassner, 2008)	Stand alone	Excluded
	(Spatari et al., 2010)	Corn stover	Acid, AFEX	Included	Enzyme	SSCF, CBP	Stand alone	Excluded
	(Spatari et al., 2005)	Corn stover	Acid	Included	Enzyme	Co-fermentation (Sheehan et al., 2004)	Stand alone	FFV: E85
	(Zhang et al., 2010)	Hard wood residues	Acid	–	Enzyme	Co-fermentation (Aden et al., 2002)	Stand alone	FFV: E85
	(Zhang et al., 2010)	Soft wood residues	Syngas			Catalytic conversion (Phillips, 2007)	Stand alone	FFV: E85
	(Zhang et al., 2010)	Corn stover	Acid	–	Enzyme	Co-fermentation (Aden et al., 2002)	Stand alone	FFV: E85
	(Uihlein and Schebek, 2009)	Straw	Acid	Excluded	Acid	Co-fermentation* (Arkenol, 1999)	Biorefinary	Car: E5
	(Cherubini and Ulgiati, 2010)	Corn stover	Steam explosion	–	Enzyme	SSCF (Hamelinck et al., 2005)	Biorefinary	Car: 2.45 MJ/km
	(Cherubini and Ulgiati, 2010)	Wheat straw	Steam explosion	–	Enzyme	SSCF (Hamelinck et al., 2005)	Biorefinary	Car: 2.45 MJ/km
	(Lim and Lee, 2011)	Oil palm residues	Acid	–	Enzyme	Fermentation	Parallel, biodiesel production	Car: E10

Class	References	Feedstock	Pretreat-ment[a]	Enzyme production	Hydrolysis	Process configuration[b]	Process integration	Bioethanol use[c]
	(Melamu and von Blottnitz, 2011)	Sugarcane bagasse	Acid	–	Enzyme	SSCF (Piccolo and Bezzo, 2009)	Parallel, sugar production	Excluded
	(Garcia et al., 2010b)	Hemp hurd	Acid	Included	Enzyme	Co-fermentation (Aden et al., 2002)	Stand alone	FFV: E10, E85
	(Garcia et al., 2010c)	Alfalfa stem	Acid	Included	Enzyme	Co-fermentation (Aden et al., 2002)	Stand alone	FFV: E10, E85
Biomass wastes	(Kalogo et al., 2007)	Biodegradable MSW	–	–	Acid	SSF (GeneSyst, 2011)	Parallel, waste treatment	LDV: E85
	(Kemppainen and Shonnard, 2005)	Recycled paper	Acid	Included	Enzyme	SSCF (Wooley et al., 1999)	Stand alone	Excluded
	(Stichnothe and Azapagic, 2009)	Biodegradable MSW	Syngas			Fermentation	Parallel, waste treatment	LGV
	(Zhang et al., 2010)	Biodegradable MSW	–	–	Acid	SSF (GeneSyst, 2011)	Stand alone	FFV: E85

MSW=municipal solid waste, – = not mentioned.

[a] AFEX=ammonia fiber explosion.

[b] SSF=simultaneous saccharification and fermentation, SSCF=simultaneous saccharification and co-fermentation, CBP=consolidated bioprocess. The Arkenol process is a technology converting cellulosic materials into sugars through concentrated acid hydrolysis, followed by co-fermentation of C5- and C6-sugars to produce ethanol. If detailed fermentation methods are not given in the text of the reviewed papers, further information is retrieved from the cited references.

[c] FFV=flexible fuelled vehicle, LDV=light duty vehicle, LGV=light good vehicle

Table 3A.3.
Aspects of LCA methodology.

Class	References	Feedstock	System boundary[a]	Functional unit	Allocation method	Impact category[b]
Energy crops	(Bai et al., 2010)	Switchgrass	Well to wheels	1 km driving	Energy, Economy	GW, AD, OD, PO, A, E, HT, ET
	(Garcia et al., 2009)	*Brassica carinata*	Cradle to grave	1 km driving, 1 kg Ethanol	No allocation	GW, PO, A, E
	(Kemppainen and Shonnard, 2005)	Wood log	Cradle to gate	83 T biomass/hr	No allocation	GW, OD, A, E, HT
	(Slade et al., 2009)	Soft wood	Cradle to gate	1 GJ Ethanol	Energy, Substitution	GW
	(Spatari et al., 2010)	Switchgrass	Cradle to gate	1 L Ethanol	Substitution	GW, Air Pollutant
	(Spatari et al., 2005)	Switchgrass	Cradle to grave	1 km driving	Mass, Substitution	GW, Air Pollutant
	(Cherubini and Jungmeier, 2010)	Switchgrass	Cradle to grave	477 kT biomass/year	–	GW, AD, OD, PO, A, E, HT, ET
	(Garcia et al., 2010a)	Poplar	Cradle to grave	1 km driving	Mass	GW, PO, A, E
Biomass residues	(Botha and von Blottnitz, 2006)	Bagasse	Cradle to grave	Sugar/Ha.year	Substitution	GW, AD, A, E, HT, ET
	(Bright and Strømman, 2009)	Wood residues	Well to wheels	1 km driving	–	GW, A, E, HT
	(Garcia et al., 2009)	Flax shives	Cradle to grave	1 km driving	Mass, Economy	GW, OD, PO, A, E
	(Luo et al., 2009)	Corn stover	Well to wheels	1 km driving	Economy, Mass, Energy, System expansion	GW, AD, OD, PO, A, E, HT, ET
	(Searcy and Flynn, 2008)	Straw	Well to wheels	1 L Ethanol	Substitution	GW

Class	References	Feedstock	System boundary[a]	Functional unit	Allocation method	Impact category[b]
	(Searcy and Flynn, 2008)	Corn stover	Well to wheels	1 L Ethanol	Substitution	GW
	(Slade et al., 2009)	Straw	Cradle to gate	1 GJ Ethanol	Energy, Substitution	GW
	(Spatari et al., 2010)	Corn stover	Cradle to gate	1 L Ethanol	Substitution	GW, Air Pollutant
	(Spatari et al., 2005)	Corn stover	Cradle to grave	1 km driving	Mass, Substitution	GW, Air Pollutant
	(Zhang et al., 2010)	Hard wood residues	Well to wheels	1 MJ Fuel	Substitution	GW
	(Zhang et al., 2010)	Soft wood residues	Well to wheels	1 MJ Fuel	Substitution	GW
	(Zhang et al., 2010)	Corn stover	Well to wheels	1 MJ Fuel	Substitution	GW
	(Uihlein and Schebek, 2009)	Straw	Cradle to grave	1 T Straw	Substitution	EcoIndicator
	(Cherubini and Ulgiati, 2010)	Corn stover	Cradle to grave	477 kT biomass/year	–	GW, AD, OD, PO, A, E, HT, ET
	(Cherubini and Ulgiati, 2010)	Wheat straw	Cradle to grave	477 kT biomass/year	–	GW, AD, OD, PO, A, E, HT, ET
	(Lim and Lee, 2011)	Oil palm residues	Seed to wheels	1 Ha land/100 year	Substitution	GW
	(Melamu and von Blottnitz, 2011)	Sugarcane bagasse	Gradle to gate	1 MJ Ethanol	–	GW, A, E
	(Garcia et al., 2010b)	Hemp hurd	Cradle to grave	1 km driving	Mass	GW, PO, A, E
	(Garcia et al., 2010c)	Alfalfa stem	Cradle to grave	1 km driving	Mass	GW, PO, A, E
Biomass wastes	(Kalogo et al., 2007)	Biodegradable MSW	Well to wheels	1 kg Ethanol, 24 MT MSW/hour	No allocation	GW, Air Pollutant
	(Kemppainen and Shonnard, 2005)	Recycled paper	Cradle to gate	83 T biomass/hr	No allocation	GW, OD, A, E, HT

Class	References	Feedstock	System boundary[a]	Functional unit	Allocation method	Impact category[b]
	(Stichnothe and Azapagic, 2009)	Biodegradable MSW	Cradle to grave	1 MJ Ethanol, 190,000 T MSW/year	–	GW
	(Zhang et al., 2010)	Biodegradable MSW	Well to wheels	1 MJ Fuel	Substitution	GW

MSW=municipal solid waste, – = not mentioned.

[a] For simplicity, system boundaries presented in Figure 5 and Table 3A.3 do not differentiate among 'cradle to grave', 'well to wheels', and 'seed to wheels'. They are all represented as 'cradle to grave'.

[b] GW=Global warming, AD=Abiotic depletion, OD=Ozone depletion, PO=Photochemical oxidation, A=Acidification, E=Eutrophication, HT=Human toxicity, ET=Ecotoxicity.

Table 3A.4.
Impact assessment results with reference to conventional fossil oil system.

Class	References	Feedstock	Net energy output[*]	GHG emission[a]	Overall environmental impact[b]
Energy crops	(Bai et al., 2010)	Switchgrass	–	65% Less, E85	–
	(Garcia et al., 2009)	*Brassica carinata*	Positive, 64%	145% Less, E85	Better
	(Kemppainen and Shonnard, 2005)	Wood log	Positive, 86%	Less	–
	(Slade et al., 2009)	Soft wood	–	Less	–
	(Spatari, et al., 2010)	Switchgrass	Positive	Less	–
	(Spatari et al., 2005)	Switchgrass	–	57% Less, E85	–
	(Cherubini and Jungmeier, 2010)	Switchgrass	Positive, 80%	79% Less	–
	(Garcia et al., 2010a)	Poplar	Positive	51% Less, E85	Better
Biomass residues	(Botha and von Blottnitz, 2006)	Bagasse	Positive	Less	Better
	(Bright and Strømman, 2009)	Wood residues	----	46-68% Less, E85	Better
	(Garcia et al., 2009)	Flax shives	Positive	37%Less, E85	Better
	(Luo et al., 2009)	Corn stover	–	Less (mass allocation) More (economy allocation)	–
	(Searcy and Flynn, 2008)	Straw		Less	–
	(Searcy and Flynn, 2008)	Corn stover		Less	–
	(Slade et al., 2009)	Straw	–	Less	–
	(Spatari, et al., 2010)	Corn stover	Positive	Less	–
	(Spatari et al., 2005)	Corn stoverh	–	65% Less E85	–
	(Zhang et al., 2010)	Hard wood residues	–	Less, E85	–
	(Zhang et al., 2010)	Soft wood residues	–	Less, E85	–
	(Zhang et al., 2010)	Corn stover	–	Less, E85	–
	(Uihlein and Schebek, 2009)	Straw	–	–	Better
	(Cherubini and Ulgiati, 2010)	Corn stover	Positive, 80%	50% Less	–
	(Cherubini and Ulgiati, 2010)	Wheat straw	Positive, 80%	50% Less	–
	(Lim and Lee, 2011)	Oil palm residues	Negative	More	–

Class	References	Feedstock	Net energy output*	GHG emission[a]	Overall environmental impact[b]
	(Melamu and von Blottnitz, 2011)	Sugarcane bagasse	Negative	More	–
	(Garcia et al., 2010b)	Hemp hurd	Positive	11% Less, E85	Better
	(Garcia et al., 2010c)	Alfalfa stem	Positive	88% Less, E85	Better
Biomass wastes	(Kalogo et al., 2007)	Biodegradable MSW	Positive	65% Less	Better
	(Kemppainen and Shonnard, 2005)	Recycled paper	Positive, 73%	Less	–
	(Stichnothe and Azapagic, 2009)	Biodegradable MSW	–	92% less	–
	(Zhang et al., 2010)	Biodegradable MSW	–	Less, E85	–

MSW=municipal solid waste, – = not mentioned.

[a] Relative to the gasoline reference system at cradle to gate or cradle to grave boundaries, GHG=greenhouse gas.

[b] Based on the included impact categories.

3

3.7 References

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Chapter 4

Effect of biogenic carbon inventory on the life cycle assessment of bioenergy

Submitted as “Effect of biogenic carbon inventory on the life cycle assessment of bioenergy: challenges to the neutrality assumption” (manuscript number: JCLEPRO-D-15-01966) to Journal of Cleaner Production, 2 July 2015
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Abstract

Biogenic carbon is defined as carbon contained in biomass that is accumulated during plant growth. In spite of considerable progress towards the inventory of biogenic carbon in the life cycle assessment (LCA) of bioenergy in policy guidelines, many scientific articles tend to give no consideration to biogenic carbon, due to the neutrality assumption, rather than employing a complete inventory according to the LCA principles. Meanwhile, the assumption of biogenic carbon neutrality has been previously challenged on the basis of changes in soil carbon stock due to land use change and carbon storage capacities of long-rotation trees or wood products. Supporting this argument, we investigate three other inventory aspects, namely, differences in framing system boundaries (cradle to grave vs. cradle to gate), forms of carbon emissions (carbon dioxide vs. methane), and valuation of biogenic carbon (biomass products vs. biomass residues). Referring to a generic bioenergy system, our analysis is focused on eight scenarios of various carbon flows encompassing biomass decomposition in fields and its alternative utilization as bioenergy feedstocks. These scenarios are applicable to both biomass products and biomass residues, which impacts proportionally depend on the chosen allocation criteria between the two. Further, a framework to quantify the performances of the various carbon flows on global warming impacts is formulated. The operation of the framework demonstrates that the assumption of biogenic carbon neutrality introduces a bias to the 'true' values based on a complete inventory. This can make the values of global warming impacts substantially higher or lower when different system boundaries, forms of carbon emissions, and biomass valuation are taken into account. The results of this study are expected to contribute to the harmonization of future bioenergy LCA by directing further research to adopt more the concept of utilizing a complete inventory rather than the neutrality assumption.

Keywords
Biogenic carbon neutrality, Biomass residues, Biomass valuation, Global warming impact, Life cycle inventory, System boundary.

4.1 Introduction

Biogenic carbon is defined as carbon contained in biomass that is accumulated during plant growth involving photosynthetic processes. Variations in the inventory of biogenic carbon can be easily ascertained in life cycle assessment (LCA) practices. Downie et al. (2014), for example, has recently discussed various approaches to greenhouse gas (GHG) accounting for biogenic carbon. These different inventory approaches often incite confusion in interpretation due to their divergent final results. In addition, many LCA studies concerning bioenergy tend to give no consideration to biogenic carbon, due to carbon neutrality assumptions, rather than employing a complete inventory. The reason behind the neutrality assumption is that biogenic carbon sequestered during growth is believed to be released back in the same amount and forms, either naturally decomposed or burned, so that there is no net increase in the atmospheric GHGs. One recent example is an article published by Liska et al. (2014) regarding the estimation of CO_2 emissions from crop residue-derived biofuels. The article has stimulated scientific debates and thus has received several reactions, one of which is the critique of omitting biogenic carbon from the inventory (Bentsen et al., 2014). In this regard, it is important to identify which approaches are more accurate than others for a certain situation. Theoretically, any deviations when performing an inventory from the LCA principles, i.e., treating all relevant input-outputs as genuine flows including biogenic carbon, would move the final scores away from the true values. Moreover, it would be biased if such deviating results are for example exploited for policy making such as in taxes or incentives applied to bioenergy products.

The objective of this study is to examine the consequences of applying carbon neutrality assumptions (excluding biogenic carbon from an inventory) in the LCA of bioenergy systems whereby the analysis is focused on the global warming impact at different system boundaries (cradle to grave vs. cradle to gate), forms of carbon emissions (CO_2 vs. CH_4), and valuation of biogenic carbon (biomass products vs. biomass residues). In doing so, this study proposes a method to quantify global warming performances of various possible carbon flows in a generic bioenergy system based on a life cycle approach. These carbon flows encompass decomposition of biomass in fields and its alternative utilization as bioenergy feedstocks.

The remainder of the article is structured in the following manner. Section 2 describes the problems of inconsistencies in biogenic carbon inventories in policy guidelines and scientific literature, challenges to the biogenic carbon neutrality assumption, and inventory aspects in terms of system boundaries, forms of carbon emissions, and biomass valuation. Section 3 describes a step-by-step method for assessing global warming impacts of various carbon inventory scenarios. Finally, Section 4 discusses the results and their implications on the assumption of biogenic carbon neutrality.

4.2 In-depth description of the problems

4.2.1 Biogenic carbon inventory in policy guidelines and scientific literature

There is still debate regarding the manner to treat biogenic carbon in bioenergy systems in a policy context. At the global level, the Intergovernmental Panel on Climate Change (IPCC) adopts a carbon stock method rather than an input-output flow approach in accounting for biogenic carbon (Levasseur et al., 2013). A similar approach is also employed by the United Nations Framework Convention on Climate Change (UNFCCC) in order to report country's emissions resulting from land use and energy as separate sectors (Haberl et al., 2012). Encompassed within these frameworks, if trees are cleared from forest and, later in the life cycle, the biomass is combusted as bioenergy, the carbon that is lost from combustion can alternatively be expressed as a land use change (LUC). Consequently, in this case, the CO_2 emissions from biomass combustion will no longer be considered to avoid double counting, i.e., the emissions from biomass combustion are considered as zero. This latter argument is often utilized inaccurately as a basis to assume biogenic carbon neutrality. In contrast, Haberl et al. (2012) clearly stated that the above accounting practices do not justify biogenic carbon neutrality but, instead, provide an option that emissions generated from biomass combustion can optionally be inventoried as LUC. If done properly, either ways (claims as combustion or land-use emissions) would lead to the same results. More precisely, the authors stated that "the assumption that all biomass is carbon-neutral results from a misapplication of the original guidance provided for the national-level carbon accounting under the UNFCCC". We think that such an observation could be derived easily since Haberl and coworkers observed the entire bioenergy system from the life cycle perspective. This approach is based on the recognition that a complete inventory will provide a more accurate estimate of carbon balances. The current article adopts this view to establish criteria to examine the assumption of biogenic carbon neutrality in bioenergy systems.

The carbon accounting based on the carbon stock method would essentially result in the correct results if applied consistently, i.e., carbon emissions are all expressed either as land use or as energy use. Considering this, not-inventorying the emissions from biomass-use as bioenergy do not imply that the biomass is automatically 'carbon neutral'. The following is an example of possible inconsistency when applying the carbon stock method. The coverage of the UNFCCC accounting system for global warming is applicable to all countries worldwide. Meanwhile, Kyoto Protocol caps emissions from land use and from energy sectors differently for Annex 1 and non-Annex 1 parties, primarily consist of developed and developing countries, respectively (Haberl et al., 2012; UNFCCC, 2014). The first authors further stated that the protocol potentially caused errors as a consequence of the non-homogenous implementation of the accounting rules in different countries. In this regard, incomplete information regarding the inventory during an initial phase (LUC) could result in double counting or GHG emissions never being accounted for at all.

In spite of some progress towards the inventory of biogenic carbon in the life cycle assessment (LCA) of bioenergy, variations can still be ascertained between policy guidelines. As discussed in Johnson (2009), a life-cycle based method such as the British PAS 2050 initially did not consider biogenic carbon uptakes and emissions. Subsequently, its revised version clearly states that all biogenic carbon flows must be considered (BSI, 2011). The same applies for the GHG protocol developed by World Resources Institute and World Business Council for Sustainable Development (WRI-WBCSD, 2011). Similarly, the International Reference Life Cycle Data System (EC-JRC-IES, 2010) that was developed by the European Commission recommended the use of LCA principles with a complete input-output flow approach. Meanwhile, EU (2009) expressed carbon neutrality in a slightly different manner, where the capture of CO_2 in the cultivation of biomass and emissions from biofuel use were valued as zero.

Divergence in ways to develop biogenic carbon inventories is more extensively found in the scientific literature of bioenergy LCA. Based on a survey that exceeded 100 publications, most solid bioenergy studies disregarded biogenic carbon emissions in the combustion of biomass (Johnson, 2009). More specifically, out of the 25 researchers working on the GHG emissions of wood fuel, only one group did not assume biomass to be carbon neutral. Similarly, nearly half of the liquid biofuel studies did not include biogenic carbon in their inventory (Wiloso et al., 2012). In this survey, 13 out of 27 LCA studies of second generation bioethanol based on energy crops and biomass residues did not consider carbon sequestration through photosynthesis. This indicates that the attitude towards adopting the concept of biogenic carbon neutrality varies between LCA studies, i.e., 96% for solid bioenergy and 48 % for second-generation liquid biofuel. This is quite a surprising observation that many scientific articles tend to give no consideration to biogenic carbon considering the neutrality assumption, which admittance varies depending on the forms of bioenergy (solid vs. liquid). In accordance with the trend in the policy guidelines, perhaps, additional scientific publications in the future would adopt more the concept of utilizing complete inventory rather than assuming carbon neutrality. This paper intends to contribute to the harmonization of the LCA of bioenergy in favor of the above perspective.

4.2.2 Challenges to the assumption of biogenic carbon neutrality

According to the LCA principles, carbon uptake from the atmosphere should be incorporated into the inventory like all other relevant emission flows. Therefore, biogenic carbon can be regarded as carbon neutral when the net global warming scores are zero, which is an ideal condition where flows between carbon capture (photosynthesis) and carbon emissions (natural decomposition) are balanced. Based on the neutrality assumption, these two natural processes are traditionally excluded from the inventory with the belief that this would not affect the final results. However, there is an additional requirement in order for this scenario to be valid. The new growing trees must replace the biomass that was previously harvested in relatively short term, i.e., fast growing trees, as opposed to slow-growing trees like forests (Cherubini et al., 2011; Guest et al., 2013). Annual crops for example are encompassed within the category of fast-growing trees. In this case, the decayed woody biomass could simply be a component of the short-lived carbon cycle which does not increase the amount of atmospheric carbon due to compensation by, relatively, the un-delayed photosynthetic processes.

In contrast to the natural process of biomass decomposition, bioenergy systems typically involve an increasingly complex, greater number of input-output flows. Along the life cycle, fossil fuel is incorporated in substantial amounts in both the agricultural and bioenergy conversion phases. The subsequent disturbed carbon balance will typically result in a non-zero global warming score. From an LCA perspective, the bioenergy systems can be perceived as not carbon-neutral due to the net increase in GHG emissions. Such a conclusion however cannot be observed easily when employing the previously discussed carbon stock method. This indicates that a life cycle approach is more suitable as a tool for examining the carbon neutrality assumption of a bioenergy system.

Certain other factors have been recognized as potentially disturbing the above carbon balance even more, serving as an argument often used to question the concept of carbon neutrality (Rabl et al., 2007; Johnson, 2009). The most important one is the effect of land use and LUC on soil carbon stock (Searchinger et al., 2008). This is often the case since biomass or bioenergy systems typically involve agricultural activities entailing intensive carbon exchange among trees, soil, and the atmosphere. It has been concluded that the production of food-crop-based biofuels may create a carbon debt, i.e., a situation where GHGs released from soil are much more substantial than the annual GHG savings from biofuels substituting fossil fuels.

Another influencing factor concerns the temporal aspects that delay carbon emissions into the atmosphere which relates to the temporary carbon-storage capacity of wood products. To address this time frame issue, Levasseur et al. (2013) proposed treating biogenic carbon with dynamic LCA. It is considerably appropriate for describing the environmental performance of, for example, long-storage wood-based products such as furniture. The argument in support of this approach is that, during the use-phase of a product, the concentration of CO_2 in the atmosphere is temporarily decreased and some radiative forcing is avoided (Levasseur et al., 2012). The primary features of this dynamic LCA are the utilization of dynamic inventory and dynamic characterization factors. A similar contribution was also initiated by Cherubini et al. (2011) who developed a characterization factor of biogenic global warming potential (GWPbio) for bioenergy based on various types of energy crops. The GWPbio ranging from 0 to 1 characterizes the climate change effects of biogenic carbon relative to fossil carbon. This concept is appropriate for describing the bioenergy systems based on biomass feedstock of different degrees of renewability, for example, annual crops versus boreal-forest of a more than 100 year rotation period. The GWPbio considers two related parameters including carbon sequestration in biomass and CO_2 decay in the atmosphere.

To support the idea of applying complete inventory in the LCA of bioenergy, including biogenic carbon, the current paper evaluates three relevant inventory aspects. They are differences in framing system boundaries (cradle to grave vs. cradle to gate), forms of carbon emissions (CO_2 vs. CH_4), and valuation of biogenic carbon (biomass products vs. biomass residues). Integrative discussion regarding these three aspects with regard to the assumption of biogenic carbon neutrality in the literature has, thus far, been lacking, but will be elaborated further in the following sections.

4.2.3 Coverage of system boundaries

To acquire the true value according to the LCA principles, ideally, an analysis ought to be carried out on a complete life cycle system boundary. However, in practice, many processes are left out for different reasons, for example, constraint in terms of time and money. In this regard, a cradle-to-gate approach often can be sufficient for comparing options to make the same bioenergy from different feedstocks (biomass products vs. residues; biomass from LUC vs. land use phases). It is performed by excluding the use and final-disposal stages, but it is, of course, admissible only when there is no difference within these stages between the compared options. However, the presence of biogenic carbon at the agricultural phase and at end of life cycle in some cases adds layer of complication. Options to include or exclude the biogenic carbon could affect the quantification of global warming impacts, particularly, if the calculation is made at different system boundaries. The varying final results require a clear interpretation. To further examine this effect, this study quantifies the global warming impacts of bioenergy on the basis of different system boundaries (cradle-to-gate vs. cradle-to-grave).

4.2.4 Forms of carbon emissions

Theoretically, the most appropriate way to treat carbon cycles is to consider them as genuine cycles. All possible carbon pathways of captured and emitted carbon species such as CO_2, CH_4, and CO should be, ideally, considered in the inventory. For example, during tree growth, a certain amount of atmospheric CO_2 is fixed and ultimately released as CO_2. However, it is not always the case. Some of the carbon may be released back to the atmosphere not in the same form but, rather, as CH_4 or CO. The latter is a by-product of an incomplete combustion. Having GWP100 in the range of 2 to 5, CO is relatively short-lived in the atmosphere and easily transformed into other forms (IPCC, 2013). For this reason, CO is not discussed any further here. Meanwhile, CH_4 is a well known, important GHG having a much more substantial GWP100, 34 times higher than CO_2 (IPCC, 2013). The formation of CH_4 in bioenergy systems may occur, for instance, when the biogenic carbon is subject to anaerobic decomposition with leakages occurring along the way. The analysis on the effect of CH_4 emissions on global warming has recently been reported by Whitman and Lehmann

(2011). The current paper attempts to address the same issues in connection with other relevant aspects in bioenergy systems such as biomass valuation and system boundary.

This paper focuses on the discussion on the inventory of carbon emissions only. It is more a carbon accounting at the inventory level rather than a complete global warming assessment which is typically to include also non-carbon emissions such as nitrous oxide (N_2O) or hydro fluorocarbons (HFCs). However, since it is presupposed that these non-carbon emissions are constant for all developed scenarios, omitting the non-carbon GHG emissions in the inventory would not affect the outcome of the comparative analysis (including or excluding biogenic carbon), which is the main objective of the study. Subsequently, we will restrict the inventory to a narrow definition of GHG emissions comprising only CO_2 and CH_4. This principle is further adopted in the methodology section to demonstrate how different forms of carbon emissions affect final results. Accordingly, the results of this study should be viewed from the perspective of accounting carbon flows only. For complete global warming assessment, the other relevant GHG species should be evaluated as well.

4.2.5 Valuation of biomass

A large number of studies on the LCA of bioenergy have not clearly defined the economic status of biomass residues. This is an important issue since bioenergy feedstocks in general can be derived from biomass products (food products, energy crops) or biomass residues. In this paper, biomass residues are referred to as non-edible portions of plants generated in fields. To avoid complexity, those generated at post-harvest processing plants are not discussed here. Biomass residues are not intentionally produced for certain purposes, instead, as co-products of a specific product system (for example, food, feed, or fibers). While biomass products are obviously goods, biomass residues can be regarded either as goods or as waste depending on the nature of the situation. Unless specifically mentioned, in this paper, the term 'biomass' refers to either biomass products or residues. The bioenergy system is formulated in generic forms, thus, applicable to both biomass products and biomass residues.

It is particularly important to elucidate how the valuation schemes for biomass would actually affect final LCA results. In LCA, economic flows travel between two unit processes whereby each economic flow must be the output of one process or the input of another process (Heijungs and Frischknecht, 1998). Additionally, the flow can be regarded as goods if it possesses a positive economic value, whereas waste features a negative economic value (Guinée et al., 2009). These criteria can be further utilized as a basis to appropriately allocate environmental burdens to flows of different economic status. A waste stream in LCA is conventionally assumed to be free of environmental burden (Wiloso et al., 2015). The impact of this stream is allocated entirely to the sub-system preceding the waste stream. In contrast, if a flow is goods, the preceding streams will be automatically included in the inventory of the entire product system. Such a procedure is of particular importance considering the increasing trend of utilizing biomass residues as bioenergy feedstock. These principles are further adopted in the methodology section in order to examine how biomass valuation can affect final LCA results. Accordingly, the impacts of bioenergy derived from biomass products and residues are considered based on partitioning methods (allocation criteria), i.e., relative valuation between the two biomass. This aspect has been discussed thoroughly in the previous paper (Guinée et al., 2009). To avoid complexity, other ways of treating multi-functional issues in LCA, such as system expansion, is not considered in this current paper.

4.3 Methods

To illustrate how a global warming impact at various biogenic carbon inventories are calculated, a generic bioenergy system was created as exhibited in Figure 4.1. Within the system boundary, different sources of biomass feedstocks (biomass products or residues, from LUC or land use phases) and related carbon flows were identified. Biomass residues are either naturally decomposed when left in fields or removed and converted into bioenergy. Meanwhile biomass products will be used only as bioenergy feedstocks. Several biogenic carbon inventory scenarios were developed encompassing the most relevant biomass utilization routes, i.e., naturally decayed or converted into bioenergy. The inventory considered relevant carbon flows of the entire life cycle of the bioenergy system. The models were developed in such a way that it can simulate the above processes (decomposed or converted into bioenergy) as comprehensively as possible. Each types of carbon flows (soil, fossil, and biogenic carbons) were explicitly presented to allow for proper accounting. The analysis was focused on three major inventory issues, i.e., coverage of system boundaries (cradle to gate or cradle to grave), forms of carbon emissions (CO_2 or CH_4), and valuation of biogenic carbon (biomass products or residues).

Within the system boundary, bioenergy feedstocks were differentiated as biomass products or biomass residues, which contributions to the overall impacts depend on the chosen allocation criteria between the two. Accordingly, all biomass products are valued as goods while biomass residues are valued as either goods or waste. In this regard, biomass residues were not only generated in fields but also at post-harvest processing plants. For simplicity, the latter cases (biomass residues as waste and generated at post-harvest processing plants) are not discussed any further here.

The study was designed to compare the global warming performances in cases where biomass residues were naturally decomposed in fields (in hypothetically perfect aerobic or perfect anaerobic conditions) or where biomass products or residues was removed from fields, converted into bioenergy, and finally used (combusted). To that end, relevant inventories were consequently developed as exhibited in Table 4.1. In cases where biomass residues are naturally decomposed, it was assumed that all carbons emitted from biomass residues were either in the form of CO_2 (as a result of perfect aerobic decomposition or perfect burning) or all in the form of CH_4 (as a result of perfect anaerobic decomposition such as being disposed of into an enormous pile or land filled). Table 4.1 also lists the various carbon inventories of cases where biomass products or residues are removed from the fields, converted into bioenergy, and subsequently used. Different from the decomposition cases, all carbons emitted within the bioenergy system were assumed in the form of CO_2.

As indicated in the last two columns in Table 4.1, the procedure to quantify global warming impacts is divided into two categories, i.e., including biogenic carbon (the preferred approaches) and excluding biogenic carbon (the biogenic carbon neutral assumption). Results of these different approaches will be compared in order to examine how the neutrality assumptions would move results away from the complete inventory.

The hypothetical inventory data comprising relevant carbon capture and emissions are provided in Table 4.2. The explanation for utilizing hypothetical data was due to the fact that inventory datasets for actual situations are not typically available with such levels of comprehension, i.e., differentiating carbon flows (soil, fossil, and biogenic carbons), modes of decomposition (aerobic and anaerobic), and biomass valuation (products and residues). Except for soil emissions and biomass decomposition, all other positive emissions indicated in the table are introduced by input fossil fuels within the system boundary (LUC, LU, bioenergy production and use phases).

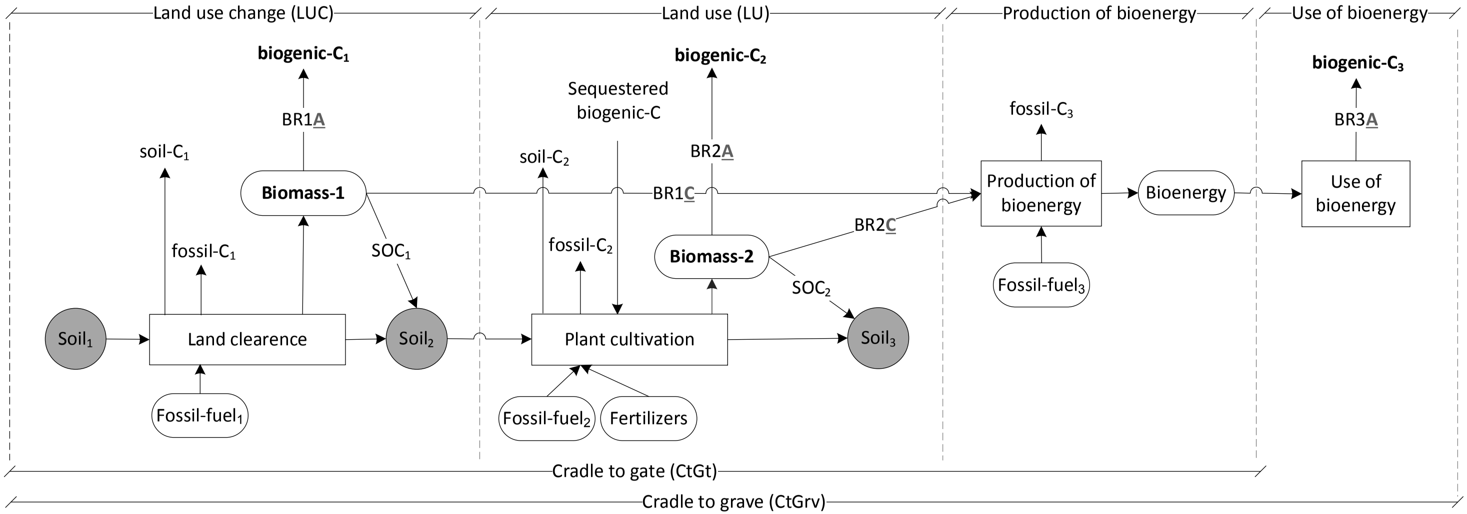

Figure 4.1. System definition of biomass-residue decomposition and alternative carbon flows as bioenergy feedstocks.

This is a generic bioenergy system, applicable to biomass products and residues. Biomass residues are either naturally decomposed in fields or removed and converted into bioenergy, while biomass products will only be used as bioenergy feedstocks. Not all of the carbon from the decomposed biomass is emitted as biogenic carbon (BR1A and BR2A), some fractions would be recycled back as soil organic carbon (SOC_1 and SOC_2). Bioenergy feedstocks can be derived from biomass products or residues generated at LUC (BR1C) or LU (BR2C) phases. ▭ = anthropogenic processes; ⬭ = economic flows (biomass residues, fossil fuels, fertilizers, and bioenergy); text without structures = environmental flows (soil-C, fossil-C, and biogenic-C). CtGt is a boundary for bioenergy at the gate of production facilities, while CtGrv is a complete bioenergy life cycle boundary, including the use phase.

Table 4.1. Procedures to quantify global warming performances of various carbon flows in bioenergy systems [a].

Biomass residues decomposed in fields	Codes	CO_2-equivalent (kg)			
		Excluding biogenic carbon (carbon neutral)		Including biogenic carbon (preferred approaches)	
At a LUC phase, emitted as CO_2	BR1A-CO_2	soil-C_1+ fossil-C_1		soil-C_1+ fossil-C_1+ *biogenic-C_1 as CO_2+* *biogenic-C_1 as SOC_1*	
At a LUC phase, emitted as CH_4	BR1A-CH_4	soil-C_1+ fossil-C_1		soil-C_1+ fossil-C_1+ *biogenic-C_1 as CH_4+* *biogenic-C_1 as SOC_1*	
At a LU phase, emitted as CO_2	BR2A-CO_2	soil-C_2+ fossil-C_2		soil-C_2+ fossil-C_2+ *sequestered biogenic-C+* *biogenic-C_2 as CO_2+* *biogenic-C_2 as SOC_2*	
At a LU phase, emitted as CH_4	BR2A-CH_4	soil-C_2+ fossil-C_2		soil-C_2+ fossil-C_2+ *sequestered biogenic-C+* *biogenic-C_2 as CH_4+* *biogenic-C_2 as SOC_2*	
Biomass products or residues converted into bioenergy		CtGt	CtGrv	CtGt	CtGrv
From a LUC phase	BR1C	soil-C_1+ fossil-C_1+ fossil-C_3	soil-C_1+ fossil-C_1+ fossil-C_3	soil-C_1+ fossil-C_1+ fossil-C_3	soil-C_1+ fossil-C_1+ fossil-C_3+ *biogenic-C_3, CO_2*
From a LU phase	BR2C	soil-C_2+ fossil-C_2+ fossil-C_3	soil-C_2+ fossil-C_2+ fossil-C_3	soil-C_2+ fossil-C_2+ fossil-C_3+ *sequestered* *biogenic-C*	soil-C_2+ fossil-C_2+ fossil-C_3+ *sequestered* *biogenic-C+* *biogenic-C_3, CO_2*

[a] LU is land use, while LUC is land use change. CtGt is a boundary for bioenergy at the gate of production facilities, while CtGrv is a complete life cycle boundary. Flows in italic represent biogenic carbon added to complete the inventories. System boundaries, modes of biomass decomposition, and biomass valuation generate different effects on the inventories.

The data in Table 4.2 were employed to illustrate the application of the procedure to quantify global warming impacts exhibited in Table 4.1. The impacts were calculated for the cases where biomass was naturally decomposed or converted into bioenergy. In this regard, total carbon emissions are tentatively determined either as 20 kg CO_2 or 7.27 kg CH_4. This equivalency is based on the fact that the mass of 1 mol CH_4 = 0.3635 mol CO_2. The emissions for biomass products and residues in the last two columns vary, depending on the chosen allocation procedure, which in this case is tentatively determined as 90% for the biomass products and 10% for the biomass residues. The impact assessment method was in accordance with the IPCC guidelines for climate change, where the GWP of CH_4 is equal to 34 CO_2, based on a 100 year time horizon (IPCC, 2013).

Table 4.2. Hypothetical inventories of various carbon flows in bioenergy systems.

Types of carbon flows	Carbon emissions (kg)			
			Allocation factors	
	Total [a] (100%)		Biomass products (90%)	Biomass residues (10%)
	CO_2	CH_4	CO_2-eq [b]	
Soil-C_1	20	.	18	2
Soil-C_2	20	.	18	2
Fossil-C_1	20	.	18	2
Fossil-C_2	20	.	18	2
Fossil-C_3	20	.	18	2
Biogenic-C_1	20	.	18	2
Biogenic-C_1, emitted as CH_4	.	7.27	222.3	24.7
Biogenic-C_2	20	.	18	2
Biogenic-C_2, emitted as CH_4	.	7.27	222.3	24.7
Biogenic-C_3	20	.	18	2
Sequestered biogenic-C [c]	-20	.	-18	-2
Biogenic-C, emitted as SOC [c]	0	0	0	0

[a] Based on equal molar, total carbon emissions are tentatively determined either as 20 kg CO_2 or 7.27 kg CH_4. The emissions from biomass products and residues in the last two columns vary, depending on the chosen allocation factors between biomass products and residues, which in this case are tentatively determined as equal to 90% for the biomass products and 10% for the biomass residues.
b Based on the recent IPCC report, global warming potential for 100-year time horizon of 1 kg CH_4 is equal to 34 kg CO_2-eq (IPCC, 2013).
c Considering the directions of the flows in Figure 4.1 which are opposite to the other parameters, the values of *sequestered biogenic-C* and SOC are negative. However, SOC_1 and SOC_2 are in the forms of organic carbon, thus, cannot be expressed in terms of CO_2-eq. To avoid complexity in the analysis, while still achieving the objectives of the paper, the decomposed biomass carbon are all assumed emitted into the atmosphere, thus, SOC_1 and SOC_2 in terms of CO_2-eq are equal to zero.

The carbon neutrality assumption in various inventory situations was examined using a carbon balance based on a life cycle approach. The criterion to include biogenic carbon neutrality assumption could be valid only if the final scores based on excluding biogenic carbon from an inventory were the same as those based on a complete inventory (including biogenic carbon). The latter is further referred to as the 'true' values. Deviations of assuming biogenic carbon neutrality from the 'true' values are expressed in percentages, which values = ([excluding biogenic-C - including biogenic-C]) / (including biogenic-C).

4

4.4 Results and discussion

4.4.1 Results

The results regarding global warming performances of various biogenic carbon scenarios are listed in Table 4.3. The percentages in the last column indicate the extent that the results become biased, i.e. the deviation of assuming biogenic-carbon neutrality from the ‘true’ value based on a complete inventory. These results are further analyzed against the criterion that biogenic carbon neutrality assumption is valid only if no distinction occurs between the final scores based on excluding or including biogenic carbon, thus, the percentages listed in the last column should be zero (see the criterion in Section 3). However, this is not the case; the extent of the bias ranges between -86% to +50%. These values depend on the respective scenarios encompassing specific system boundaries, forms of carbon emissions, and biomass valuation.

Table 4.3. Global warming performances of various carbon flows in bioenergy systems[a].

Sce-nario	Biomass residues decomposed in fields	CO_2-equivalent (kg)				Deviations from the ‘true’ values (%)	
		Excluding biogenic carbon (carbon neutral)		Including biogenic carbon (‘true’ values)			
1	At a LUC phase, emitted as CO_2	4		6		-33	
2	At a LUC phase, emitted as CH_4	4		28.7		-86	
3	At a LU phase, emitted as CO_2	4		4		0	
4	At a LU phase, emitted as CH_4	4		26.7		-85	
Biomass products converted into bioenergy		CtGt	CtGrv	CtGt	CtGr	CtGt	CtGrv
5	From a LUC phase	54	54	54	72	0	-25
6	From a LU phase	54	54	36	54	50	0
Biomass residues converted into bioenergy		CtGt	CtGrv	CtGt	CtGr	CtGt	CtGrv
7	From a LUC phase	6	6	6	8	0	-25
8	From a LU phase	6	6	4	6	50	0

[a] This table illustrates results of the operation of the criteria presented in Table 4.1 using data in Table 4.2. LU is land use, while LUC is land use change. CtGt is a boundary for bioenergy at the gate of production facilities, while CtGrv is a complete life cycle boundary. Percentages in the last column = ([excluding biogenic-C - including biogenic-C]) / (including biogenic-C) at the respective system boundaries. They indicate the extent of deviations, which can be positive or negative, from the ‘true’ values based on a complete inventory (including biogenic-C).

Zero percent means that the biogenic carbon neutrality assumption does not move the global warming impacts away from the values based on complete inventories. These are cases of Scenarios 3, 5-CtGt, 6-CtGrv, 7-CtGt, and 8-CtGrv where the neutrality assumption holds true. In the other scenarios, however, the neutrality assumption clearly introduces substantial biases to the 'true' values. These biases make the final results higher (Scenarios 6-CtGt and 8-CtGt) or lower (Scenarios 1, 2, 4, 5-CtGrv, and 7-CtGrv) from the values based on complete inventories. Higher global warming impacts can be found in Scenarios 6 and 8 that assumed carbon neutrality. These are the effect of excluding the sequestered carbon during plant growth at the land use phase from the inventory.

4.4.2 Discussion

The LCA results listed in Table 4.3 have demonstrated that dissimilar inventory situations (modes of biomass decomposition and biomass valuation) and methodological choices (system boundaries and allocation factors) based on complete inventories can generate various final scores, which are often different from that assuming biogenic carbon neutrality. On the basis of the scope of system boundary, the deviations in Scenarios 5-CtGrv, 6-CtGt, 7-CtGrv, and 8-CtGt are non-zero. These deviations are due to the emissions of biogenic carbon during combustion of bioenergy taking place at the use phase. In addition, the extent of the deviations is different between cradle-to-gate and cradle-to-grave boundaries. In this respect, to acquire the true value, the analysis is ideally carried out on a complete life cycle system boundary (cradle-to-grave) rather than at a cradle-to-gate boundary. It is based on the recognition that a complete inventory will provide a more accurate estimate of carbon balance.

Processes involving anaerobic decomposition (Scenarios 2 and 4) provided substantially greater global warming impacts than aerobic decomposition (Scenarios 1 and 3). These facts demonstrate the effect of decomposition modes under anaerobic conditions which is much more influential on the final results relative to aerobic conditions. This is due to the fact that the GWP of CH_4 from anaerobic decomposition is 34 times, much higher, than that of CO_2 (IPCC, 2013). It is also important to note that not all mineralized carbon in field decomposition would be emitted into the atmosphere; some would be recycled as organic carbon into the soil matrix (see SOC_1 and SOC_2 in Figure 4.1). Such a split of carbon flows between the atmosphere and soil is often neglected in the study of bioenergy LCA, likely due to incomplete knowledge of the inventory of such processes. We speculate that a more refined modeling of the decomposition process may have substantial improvement on the outcomes of bioenergy LCA studies. Further discussion on the impact assessment of SOC can be found in Wiloso et al. (2014).

Biomass valuation plays an important role in determining global warming impacts. For example, the scores of product-based bioenergy (Scenarios 5 and 6) are consistently higher than those from biomass residues (Scenarios 7 and 8) accordingly to the allocation factors, i.e. relative valuation between biomass products and residues. It is also observed that biomass residues have more inventory variations (options to be naturally decomposed in aerobic/anaerobic conditions or converted into bioenergy) than those based on biomass products (converted into bioenergy). Similarly, global warming scores of biomass residues valued as goods (having positive economic values) would generally be higher than those valued as wastes (having negative economic values). This is an important observation since biomass residues can be valued either as goods or as wastes, depending on regulations or market prices.

As indicated earlier, the assumption on biogenic carbon neutrality has been questioned primarily on the basis of changes in soil carbon stock due to LUC (Searchinger et al., 2008) and carbon storage capacities of long-rotation trees (Cherubini et al., 2011) or wood products (Levasseur et al., 2012, 2013). In this regard, the current study provides additional arguments that, indeed, the assumption on biogenic carbon neutrality introduces a bias to the 'true' values. Also, this analysis establishes the advantages of a life cycle approach to examine carbon neutrality assumption whereby the variation in final scores due to different system boundary, forms of carbon emissions, and biomass valuation cannot be easily revealed using a carbon stock method, an approach typically used by IPCC (Levasseur et al., 2013).

The results presented here require proper interpretation since they are based on certain hypothetical inventory data. Further analysis exploiting actual inventory data, for example, those available in literature, may be necessary to further verify if the conclusion for the non-neutrality assumption of biogenic carbon is also valid for other inventory situations. In this regard, the absolute values of the variations (-86% to +50%) may not be of particular importance since they would fluctuate accordingly, depending on the inputted inventory data. However, the predominant message of the study has convincingly reported substantial deviations in the global warming results if biogenic carbon is excluded from the inventory.

The literature study also reveals that divergence in ways to develop carbon inventories is in general more extensively found in scientific literatures of bioenergy LCA than in policy guidelines. This is quite a surprising observation that [1] many scientific articles tend to give no consideration to biogenic carbon because of the neutrality assumption and [2] attitude towards the adoption of this assumption varies depending on the forms of bioenergy. The latter can be distinguished clearly between LCA studies on solid bioenergy and second-generation (liquid) biofuel. We speculate that in the future such divergence in the area of bioenergy might adjust towards a more harmonized LCA approach. This is in accordance with the tendency in policy guidelines that additional scientific publications would adopt more the concept of utilizing a complete inventory rather than assuming carbon neutrality.

4.5 Conclusions

This study demonstrates that the global warming impacts of scenarios based on the LCA principles, treating all relevant input-outputs as genuine flows (including biogenic carbon), provides various results which are in many cases different from those assuming biogenic carbon neutrality (excluding biogenic carbon). In addition to the existing arguments of carbon stock change (Searchinger et al., 2008) and carbon storage capacities of long-rotation trees (Cherubini et al., 2011) or wood products (Levasseur et al., 2012, 2013), the current study supports the assertion that the assumption on biogenic carbon neutrality introduces biases to the 'true' values based on a complete inventory. The extent of such a bias, which can make the values of global warming impacts substantially higher or lower, depends on specific situations. Three factors are identified to substantially propagate biases in the evaluation of global warming performances of a bioenergy system. They are differences in framing system boundaries (cradle to grave vs. cradle to gate), forms of carbon emissions (CO_2 vs. CH_4), and valuation of biogenic carbon (biomass products vs. biomass residues). The results of this study are expected to contribute to the harmonization of future bioenergy LCA by directing further research to adopt more the concept of utilizing a complete inventory rather than the neutrality assumption.

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Chapter 5

A novel life cycle impact assessment method on biomass residue harvesting

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Abstract

Second generation bioenergy such as cellulosic bioethanol is expected to become commercially available in the near future. Large scale production of this bioenergy will require secure and continuous supplies of raw materials. One promising source of materials is biomass residues that currently remain on the fields following harvest, a feedstock that does not compete with food. However, unsustainable removal of these residues may adversely affect soil quality and hamper future harvests. In order to assess this effect, an impact assessment method was developed within the life cycle assessment framework based on a specific system definition, i.e. decomposed biomass residues above soil surfaces resulted in net carbon flow into soil compartment. This soil organic carbon functions as an elementary flow to complete the overall carbon balance. The assessment method considers effects of soil organic carbon on soil biomass productivity. This impact is expressed as loss of net primary production, a midpoint indicator. The impact assessment method follows the ISO-standard format, comprising a characterization factor and an input term representing changes in elementary flows. The operation of the proposed method was illustrated with a small case study. At 10% biomass removal, the impact is 7.16 g-carbon/m^2.year loss in biomass productivity as compared to the undisturbed reference. We believe that the method has a good potential to be adopted and employed as metrics for assessing the sustainability of bioenergy systems, although data for spatially detailed implementation within bioenergy systems in various regions are to be supplemented.

Keywords
Life cycle impact assessment, Bioenergy, Biomass residue, Biomass productivity, Soil organic carbon, Global warming.

5.1 Introduction

When considering bio-based energy production, current LCA linked methods have a number of shortcomings: namely, they do not relate to biotic processes and they are not systematically linked to climate impacts. Methods to determine the effect of removing biomass residues from soil for example are now increasingly recognized as an important component in LCA studies. However, these effects are often ignored because they are difficult to quantify. We develop a number of improvements to better characterize the impact of using these residues as feedstock to generate cleaner biobased energy.

Feedstocks for bioenergy systems are principally based on woody biomass of forestry or agricultural residues, while only a minimal portion is derived from sugar, starch, and vegetable oil (IEA, 2009). Concerns regarding the competition with the food sector on a global level have encouraged the utilization of non-food resources as bioenergy feedstocks (IEA, 2009). A potential source for this purpose is biomass residues. Lal (2005) defined them as the non-edible portion of plants that remain on fields following harvest. These residues are differentiated from biomass products (e.g. energy crops) as they are in most cases not intentionally produced for energy resources (Cherubini et al., 2009). This biomass, therefore, could be exploited to increase the share of renewable energy mix that is targeted by many countries (Sorda et al., 2010) with a possibility of minimal environmental risks. Despite this optimism, there is an ongoing debate regarding the feasibility of removing crop residues from soil, particularly when complying with the allowable rates that continue to sustain agricultural systems (Cherubini et al., 2009; Lal, 2005; Wilhelm et al., 2004). The main concern is the potential impacts on degradation of soil quality that may further influence future crop yields. As for the technological status of bioenergy conversion, certain advanced processes, such as cellulosic bioethanol production, are relatively immature and continue to require further development to demonstrate profitable operations at commercial scales (Dale, 2011; IEA, 2009; Schnoor, 2011). However, some countries plan a substantial increase in the contribution of cellulosic ethanol in their energy mix. The United States for example aims at producing 16 billion gallon cellulosic ethanol in 2020 (Schnoor, 2011). When such new biofuel processes become technically and commercially feasible in the near future, a sudden increase in the demand for raw materials will certainly occur. Such situation may encourage import of biomass residues from other parts of the world, and farmers may become more willing to sell their residues for economic benefit.

Significant quantities of lignocellulosic biomass in crop residues are available in many regions of the world and, therefore, possesses great potential as a feedstock for bioenergy (IEA, 2009). Global production of crop residues in 2001 was estimated at 3.8 billion metric ton annually with a total energy value of 70 EJ (Lal, 2005). This potential share of lignocellulosic biomass as a feedstock for second generation bioenergy is obviously of strategic importance. The International Energy Agency (2009) recommends the biomass residues as one of the pivotal resources for sustainable bioenergy. However, the availability of these biomass residues is limited due to other competing uses such as feed, organic fertilizer, orfiber. For example, in the United States, corn stover is customarily exploited as a fodder. In addition, biomass residues fulfill an important role for agricultural lands. Their stock of nutrients provides a basic recycling mechanism that keeps soils fertile, and they improve the structure of the soil for improved aeration and water management. Excessive removal of this inflow of biomass, therefore, creates a risk of the degradation of soil quality. This can be a serious threat to long-term sustainability of second generation bioenergy systems and should, therefore, be considered as part of the sustainability assessment in terms of the harvested fraction in agriculture and forestry. Additionally, the biomass potential of crop residues is not fully available for bioenergy production due to the other competing uses. The fundamental questions in this research are: (1) what fraction of biomass is readily available as an additional feedstock for bioenergy? and (2) what would be the environmental impact if such amount of biomass is to be harvested from soil?

5.1.1 LCA studies on the removal of biomass residues

Life cycle assessment (LCA) studies of bioenergy systems were initially focused on the environmental impact assessment of feedstock cultivation originating from food-related products and energy crops (Cherubini and Strømman, 2011). Successively, more attention has been directed toward biomass residues left on fields (Cherubini and Ulgiati, 2010; Cherubini and Strømman, 2011). However, a recent review on LCA studies of second generation bioethanol concluded that the residues were treated inconsistently and that the divergence in LCA outcomes with this type of feedstock is much more significant than in LCAs on crop products or wastes (Wiloso et al., 2012). The surveyed LCA studies were often assumed to employ only the surplus biomass or to consider a greater fraction of residues but with fertilizer compensation to maintain soil fertility. In certain forest and crop management practices in which residues are burned following harvest, it was assumed that removal of the biomass residues will not significantly alter the carbon entering the soil carbon pool (Cowie et al., 2006). Considering these variations in residue treatment and significant divergence in LCA outcomes, development a comprehensive LCA method for evaluating impacts on soil quality is warranted.

Most of the modeling studies in literature presented results on forestry cases with the exception of the crop residue cases presented by Cherubini and Ulgiati (2010) and Liska et al. (2014). The first authors proposed a lump parameter, the ‘land use change’, as the summation of reduction in soil organic carbon (SOC), methane emissions, and nitrous oxide emissions, among others. This parameter was determined to be significantly important, contributing to approximately half of the overall greenhouse gas (GHG) emissions. Liska et al. (2014) reported that removing corn residue from soil can reduce soil carbon and increase CO_2 emissions. Mckechnie et al. (2011) integrated emissions from forest carbon stocks and downstream bioenergy systems into the analysis. They concluded that ethanol produced from standing trees increased overall GHG emissions within a 100 year time frame while that from biomass residues reduced GHG emissions after a 74 year delay. Repo et al. (2011, 2012) described the impact of bioenergy systems in terms of decreasing carbon stock of the forest floor due to removal of biomass residues. They concluded that an improved global warming mitigation technique such as faster exploitation of decomposing residues (e.g. branches) would result in fewer ‘indirect emissions’ relative to slower decomposing residues (e.g. stumps). Further, Guest et al. (2013) modeled global warming impact as a function of the growth rates of plant biomass. In a typical long rotation forest ecosystem, the biomass residues could contribute between 44% and 62% of the overall CO_2 emissions (Guest et al., 2013). The results of the above mentioned studies on global warming impact have already integrated the impact on SOC stock. In contrast, Milà i Canals et al. (2007b) and Brandão and Milà i Canals (2013) have addressed the relationship of soil organic carbon and biomass productivity. In this regard, the current research models the effect of removing biomass residues on the flow of organic carbon into the soil matrix, which changes the SOC content, leading to the loss of biomass productivity.

5.1.2 Impact indicators for soil biomass-productivity

Ecosystems provide society with goods (e.g. food, fiber, and energy) and services (e.g. climate regulation, water and nutrient cycles, soil fertility, and photosynthesis) (MEA, 2005). In this regard, the loss of ecosystem functions may be represented in terms of various impact indicators such as the loss of soil fertility, biodiversity, or Net Primary Production (NPP) (Milà i Canals et al., 2007a). The last indicator is defined as net photosynthesis, or gross primary production (GPP) minus plant or autotrophic respiration, which is equivalent to above- and below-ground living plant biomass (IPCC, 2003). The NPP can be physically measured in the field or estimated by geographic information system (GIS) and remote sensing methods (Crabtree et al., 2009; Daughtry et al., 2005; Lu, 2006). GIS and remote sensing methods can estimate the amount of above-ground biomass over relatively extensive agricultural areas while direct field-measurement may only be practical in limited areas.

Previously proposed LCA frameworks regarding the aspects of ecosystem functions of land use have principally concentrated on three indicators: biodiversity (Koellner and Scholz, 2007; Milà i Canals et al., 2007a), soil quality (Milà i Canals et al., 2007b), and NPP (Lindeijer, 2000). Van der Voet (2002) and Lindeijer (2000) have suggested employing the NPP indicator to evaluate the ecosystem functions of soil to provide biomass. Milà i Canals et al. (2007a) employed Biotic Production Potential (BPP) which is defined as the ability or capacity of an ecosystem to sustain future biomass production depending on particular land use and sensitivity of the ecosystem. This concept is slightly different from the NPP which is more related to present biomass production capacity as a result of particular land use (Brandão and Milà i Canals, 2013). An additional relevant parameter is Net Ecosystem Productivity (NEP) which is largely adopted in forest-related studies (Cherubini et al., 2012; Luyssaert et al., 2007, 2008). It is defined as the difference between NPP and carbon released through soil or heterotrophic respiration (decomposition of ’dead’ biomass) (Luyssaert et al., 2007, 2008). This study employed NPP rather than NEP since the effect of biomass residues removal was described in term of SOC as an intermediate indicator. Both parameters appear in the equations and are required in order to operate the impact assessment method.

5.1.3 Objective and structure of the paper

This research focuses on developing an impact assessment method to estimate the impacts of removing biomass residues from soil on the loss of biomass productivity, a midpoint indicator, employing SOC as an elementary flow input. The harvested biomass becomes a co-product of the specific bioenergy LCA.

The remainder of the article is presented in the following manner. In the methods section, the basic assumptions and development of impact assessment methods are presented. In the results and discussion section, the implementation of the proposed method is subsequently illustrated exploiting available data from the literature. Finally, sources of uncertainties, removal scenarios, and further implementation are discussed.

5.2 Methods

In this section, possible scenarios on the removal of biomass residues from soil and the effects on soil quality are described. Further, relevant impact indicators leading to the loss of biomass productivity are presented as an environmental pathway. Finally, impact assessment methods are developed based on system definition and assumptions.

5.2.1 Removal scenarios

This paper utilized IPCC (2003) in defining carbon pools of a soil system. This includes above- and below-ground living biomass, above-ground dead organic matter (wood and litter), and soil (SOC). In this context, SOC refers to organic carbon contained within the mineral horizons of soil while those above the soil surface are considered as the separate carbon pool of dead organic matter. Biomass residues were referred to those originating from agriculture, forestry, grasses, or post-harvest processing units. Excessive removal of these residues may cause a decline in SOC and consequently, biomass productivity.

In order to maintain sustainable agriculture, nutrient supplement in the form of additional fertilizer may be required depending upon soil type, humidity, and other local conditions. In this regard, a sustainable supply of the feedstock is defined as a condition where biomass is sufficiently produced to meet the present demand while soil fertility is maintained to secure future harvests. From an LCA perspective, there are three possible scenarios as to the level of impacts of removing biomass residues from soil:

1) removing only the actual surplus amount with negligible soil carbon degradation;
2) removing a larger fraction with fertilizer compensation; and
3) removing a larger fraction without fertilizer compensation, reducing soil carbon to levels with lower biomass productivity.

The first scenario is unambiguous as the surplus biomass-residues can be perceived as free goods bearing no significant environmental burden (levels of SOC and biomass productivity are maintained). The second scenario is more a shift of impacts on biomass productivity to an inventory issue, i.e. the production of additional fertilizer consequently leading to conventional impacts such as global warming, eutrophication, or acidification. The inorganic fertilizer is added to supplement the limiting nutrients (such as nitrogen and phosphor) contained in the removed biomass-residues while carbon is captured from the atmosphere through photosynthetic processes during plant growth. The third scenario would likely lead to a loss in SOC and biomass productivity due to unsustainable removal practices.The actual amount of the harvested residues is greater than the allowable removal rates with no balanced fertilizer compensation.

The terms “surplus” and “larger fraction” are used only to qualitatively define different scenarios. It is the proposed impact assessment method as an analytical tool to quantitatively determine the extent of biomass removal that affects SOC and subsequently biomass productivity.

5.2.2 Inventory aspects

In LCA, land use related processes, in general, lack a clear flow character as input or output components of a typical production system (Udo de Haes, 2006). The same problem is encountered in describing processes related to the degradation of soil quality. This parameter is also inadequate in terms of representative aggregate indicators, characterization factors, and data availability (Milà i Canals et al., 2007a). These are the main challenges in describing inventory aspects of biomass residues left on fields. However, if data on stabilized soil indicators of a certain region (such as SOC and NPP) could be made available, such information could be utilized to assess the consequences of removing parts or all biomass residues from the soil.

Various types of bioenergy feedstocks (biomass products or biomass residues) are characterized as having different environmental burden attribution schemes and impact categories. Land use related processes, in general, can lead to a number of impact categories such as global warming, eutrophication, acidification, toxicity, and others. These conventional impact categories, however, will not be the subject of further discussion in the current research. Instead, focus will be placed on impacts related to the states of SOC and biomass productivity. In general, land use impacts have been perceived as not only depleting physical surface areas (m^2.years of land use) but also altering land quality (Milà i Canals et al., 2007b). In this latter case, land is perceived as a distinct environment compartment, and elementary flows into the soil matrix can be considered as an environmental intervention. In this research, we are more interested in the aspect of quality degradation due to environmental intervention, so that depletion of SOC can be systematically linked to GHG emissions. The expanded system boundary (fields and bioenergy plants) and relevant elementaryflows to air and soil compartments are indicated in Figure 5.1.

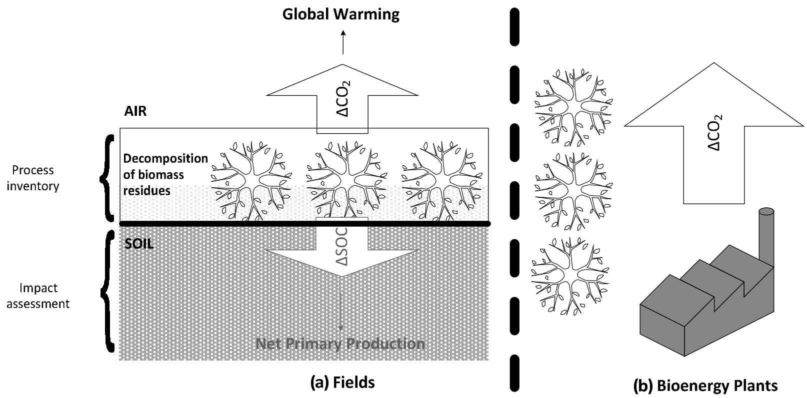


Figure 5.1. Effects of removing biomass residues on SOC and CO_2 flows at two contrasting situations: all residues are (a) left on fields and (b) removed for bioenergy production. Reduction in ΔSOC input leads to loss in biomass productivity. Carbon balance of an expanded system: $(\Delta CO_2)_{fields}+(\Delta SOC)_{fields}=(\Delta CO_2)_{bioenergy}$. The focus of this study is the assessment of the elementary flow taking place in soil compartment, indicated in red color. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article).

Under the IPCC accounting rules (IPCC, 2006), countries separately report their emissions from energy use and from land use change. For example, if a hectare of forest is cleared, the carbon depletion from the forest is considered as a land use emission. If the wood is then exploited for bioenergy, the regulations allow countries to ignore the same carbon when it is released after combustion in order to avoid double-counting. In other guidelines or databases such as EU-Directive (2009) or Ecoinvent database (2010), for example, both emissions in the field and in the bioenergy system are considered, but their values are somehow similar, $(\Delta CO_2)_{fields} \approx (\Delta CO_2)_{bioenergy}$, resulted in zero net emissions. Referring to Figure 5.1, these guidelines and database considered GHG emissions at different system boundaries but failed to recognize the significant elementary flow of organic carbon into the soil matrix. These two distinct elementary flows to two different environmental compartments (air and soil) are, in fact, taking place simultaneously, as illustrated in Figure 5.1. From an LCA perspective, this SOC flux must also be taken into consideration during the inventory process. Therefore, in the current research, this 'missing' elementary flow (SOC) was included to complete the carbon balance of relevant processes occurring within the pile of biomass residues on soil surfaces. The complete carbon balance of the GHG emissions and organic carbon flows at the expanded system boundary (field and bioenergy plants), therefore, can be expressed as $(\Delta CO_2)_{fields}+(\Delta SOC)_{fields}=(\Delta CO_2)_{bioenergy}$. This additional elementary flow was further employed to evaluate the impact that removing biomass residues has on biomass productivity of the affected soil.

Biomass residues left on fields experience certain processes that are governed by natural and human factors. The main natural process is decomposition of the biomass in which intensity is affected by human intervention. The intervention is in the form of extracting biomass residues at a certain rate from the fields to a bioenergy plant (see Figure 5.1). A larger fraction of biomass removed from the field will certainly reduce the amount of biomass experiencing the decomposition process and, consequently, the organic carbon flows into the soil matrix. To translate these processes into an LCA framework, the system boundary can be defined as piles of biomass residues situated on top of soil. Within these piles, a natural decomposition process occurs, which converts organic biomass into simpler forms of organic carbon and further into CO_2 or CH_4. These degradation products are then simultaneously released as organic carbon flows and emitted as GHGs at different environmental interfaces, biomass-soil and biomass-air, respectively (see Figure 5.1). The effect of such processes on carbon pools has recently been modeled by Guest et al. (2013) for changes in CO_2 emissions and Repo et al. (2011, 2012) for changes in carbon stocks. The decomposed biomass residues deliver organic carbon into the soil matrix, providing nutrients and other qualities that improve soil quality. This elementary flow results in a different impact category, i.e., soil biomass-productivity represented by NPP, a midpoint indicator. This research, therefore, complements and improves previous studies of similar systems by using a different impact category.

5.2.3 Environmental pathways of impact indicators

Biomass production capacity is affected not only by local climates and soil types but also by processes that occur in living vegetation (McBride et al., 2011). In this research, the parameters having the greatest influence on NPP are integrated into a cause-effect chain of an impact pathway as demonstrated in Figure 5.2. Endpoint impacts on human and ecosystem health are included only to illustrate their relationship with NPP, a midpoint indicator. In order to comprehend the complex nature of NPP as an impact indicator, this section describes some of the factors that are influential to SOC and biomass-productivity. For example, in second generation bioenergy systems, raw material in the form of lignocellulosic biomass is supplied mainly from an agricultural chain. Therefore, the fertility of the soil where the biomass is produced plays a key role. According to Sleeswijk et al. (1996), one manner to evaluate soil resources is by monitoring its degradation process. This can be evaluated in terms of dynamic properties of the soil over time (Anton et al., 2007). Soil conditions are affected not only by the natural condition of the ecosystem but also by the activities subjected by it (Mattsson et al., 2000; Sleeswijk et al., 1996). Those induced by human activities include land management, agricultural inputs, and biomass harvest including the extraction of biomass residues from the soil. Factors of natural origin include soil types (sandy, clay, peat), local climates (arctic, temperate, tropic), and processes occurring in living vegetation such as photosynthesis and respiration.

A single aggregate indicator for soil quality is not easily established due to numerous soil characteristics and their complex interactions with agricultural practices (Garrigues et al., 2012). Disregarding this difficulty, Milà i Canals et al. (2007b) have initiated exploiting soil organic matter (SOM) as the sole indicator of soil quality, which is represented as SOC. The current research is in accordance with this approach and employs SOC to represent the state of soil quality. This is supported by McLauchlan (2006) who stated that SOC, in many ways, dominantly regulates soil moisture, structure, nutrient, and microbial activity. Otherwise stated, the SOC performs a central function in determining the state of soil quality. Based on this perspective, the current research assumed SOC as the dominant factor influencing biomass productivity.

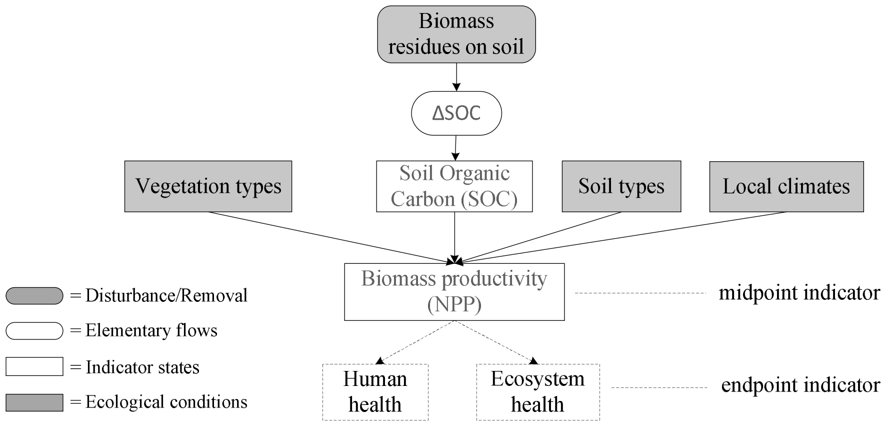


Figure 5.2. Environmental pathways of impact indicators with ΔSOC as an elementary flow input. (Endpoint impacts on human and ecosystem health –dashed arrows and boxes– are included only to show their relation with the NPP as a midpoint impact indicator).

However, there are some exceptions to the above rule. For example, in the case of peat soil consisting mainly of organic materials, the SOC is likely to be less dependent on the amount of above-ground biomass that is extracted from the soil. In contrast, the impact would be much more pronounced when the removal occurred in clay soils where the SOC has a strong correlation with external carbon inputs. In addition to the above-ground biomass as the source of carbon inputs, roots can also contribute to more than half of the SOC depending on the types of vegetation (Wilhelm et al., 2004). In this last case, the state of the SOC is also likely to be less sensitive to different biomass removal rates. These different conditions may explain why most studies could not agree on the allowable removal rates that do not affect soil quality and biomass productivity (Lal, 2005; Wilhelm et al., 2004). Considering the above deviations from the main assumptions, the proposed model can be regarded as a simplification of the actual situation. The impact assessment method can be beneficial for practical reasons and regarded as an initial proposal until an improved alternative becomes available.

Ecological processes affecting soil quality (represented as SOC) and soil biomass-productivity are described in the following. Soil organic matter comprises residues of plants and animals decomposing by microorganisms at any stage (Cowie et al., 2006). The process of organic mineralization results in carbon (CO_2, CH_4) and nitrogen emissions to the atmosphere and simultaneously provides nutrients for plants to grow. Such a decomposition process is enhanced in warm and moist climates (tropical) and reduced in low temperature and limited moisture regions (temperate and arctic) where microbial activities are inhibited. Therefore, soil carbon stocks are typically lower in the wet tropics where organic matter is quickly cycled and lowest in dry environments where plant growth is limited (Cowie et al., 2006). The rate of mineralization is also influenced by the chemical composition of residues. Residues with low nitrogen content or a high portion of the recalcitrant component such as lignin has the lowest rate. In addition, soil with significant clay content better preserves nutrients than that of sandy soil due to improved absorption performances. For example, soil carbon stocks are high in soils with high-activity clay (e.g. Andisols, Vertisols, Mollisols), followed by low-activity clay soils (Oxisols, Ultisols) and low in all sandy soils (Cowie et al., 2006). Other important aspects of soil quality are soil salinization, soil compaction, soil erosion, and biodiversity (Garrigues et al., 2012). These natural states can affect soil organic matter in various ways as increases or losses (Garrigues et al., 2012; McLauchlan, 2006), but were not specifically included in the characterization factor of the current study.

5.2.4 Development of the impact assessment method

The changes in organic carbon flows due to biomass removal at different rates will lead to a reduction in SOC and a loss of biomass productivity at different levels. As demonstrated in the environmental pathways (Figure 5.2), organic carbon flows into the soil matrix are categorized as elementary flows while the concentration of SOC and level of NPP are categorized as indicator states. These parameters (organic carbon flows, concentration of SOC, and level of NPP) were employed as the basis to develop the impact assessment method. The generic expression of an impact assessment method for a midpoint indicator is expressed as:

$$\text{Impact} = \text{CF} \times \text{Input} \qquad (5.1)$$

Equation (5.1) comprises a characterization factor (CF) and an input term representing changes in elementary flows. The CF depicts the impact assessment step while the term input represents the results of a process inventory. The CF is correlated with the upstream activity, extraction of biomass residues, and, with the downstream indicator, soil biomass-productivity. This type of expression is in accordance with the formulation of an impact assessment method suggested by the ISO standard (ISO, 2006) as the linear product of inventory results and characterization factors. A similar expression of impact assessment models has been utilized by Van Zelm et al. (2007) for various impact categories, for example, loss in biodiversity due to acidification processes. More specifically, the characterization factor is formulated as the following:

$$\text{CF} = \frac{\text{dNPP}}{\text{dSOC}} \qquad (5.2)$$

where NPP is the net primary production in g-carbon/(m^2.year). The dNPP is the change in the amount of carbon contained in living biomass as a consequence of removing biomass residues from soil. The dNPP is illustrated later in Figure 5.3 (b) as the difference between two points within the fitted line. SOC represents the concentration of organic carbon in a soil layer of certain depth. Even though the concentration of SOC in a unit volume of soil is expressed in g-carbon/m^3, it is established practice to choose a fixed top layer depth, and express the SOC per unit area, in g-carbon/m^2 (Milà i Canals et al., 2007b; Brandão and Milà i Canals, 2013; IPCC, 2006). We follow this practice, thus expressing our characterization factor in year^{-1}. The soil layers in the study conducted by Wang et al. (2008) for example were between 20 cm and 30 cm, while IPCC (2006) used 30 cm deep as the default value.

The two differential terms (dNPP and dSOC) in the CF describe the changes in SOC and NPP due to biomass removal from fields. This suggests that the data required are only the instant/immediate changes as a response to human disturbance with the removal of biomass residues from soil. Therefore, the area of application of the method can be rather general. For example, it could be applied for both forest and agricultural systems. The characterization factor is formulated in terms of differential changes, a steady state approximation of a dynamic process. These changes represent responses in terms of magnitude of the impact to disturbances. A larger magnitude of impacts would be indicated with a steeper slope.

Biomass productivity not only depends on the level of SOC but also on variability in local climates, vegetation species, and soil types. These ecological factors must also be considered in order to obtain a more representative value in relationship to location-dependent aspects. In the current proposed model, local climates, for example, are categorized as arctic, temperate, and tropical while soil types are classified into sandy, clay, and silt. It is generally accepted that precipitation and temperatures are the main factors governing local climates which subsequently cause considerable spatial variation in SOC and NPP (McLauchlan, 2006). Vegetation types could be categorized as fast or slow growing species. Also, the type of roots (coarse or fine roots) and the ratio of roots (belowground biomass) to shoot (above-ground biomass) may affect the variability of the above parameters. Considering the above arguments, Equation (5.2) must be modified into:

$$\mathrm{CF} = \left(\frac{\mathrm{dNPP}}{\mathrm{dSOC}}\right)_{\mathrm{Climate,\,Vegetation\,,Soil}} \qquad (5.3).$$

Geographical variability regarding agricultural data is a common issue in an LCA of agricultural systems. This issue has been systematically incorporated into the characterization factor as ecological conditions in terms of climate, vegetation, and soil types. This expression of CF can serve as a generic platform to describe how local conditions may affect indicator states. Therefore, the values of the CF must be updated in accordance with the data regarding the effect of biomass removal on SOC and NPP in various regions.

The term input in Equation (5.1) is the difference in concentration of SOC as a function of flows of organic carbon entering the soil matrix. It can conveniently be expressed as $(\Delta\mathrm{SOC})_{\Delta\mathrm{Organic\text{-}Carbon}}$. The magnitude of this elementary flow is determined by the amount of biomass residues remaining on the soil following extraction for bioenergy or other purposes. This is in accordance with the environmental pathways illustrated in Figure 5.2. However, the organic carbon flow is not an easy parameter to measure, and such data are hardly available. To solve this problem, the changes of SOC concentration was expressed in terms of the percentage of removed biomass, i.e. amount of harvested biomass over initial amount of biomass, $(\Delta\mathrm{SOC})_{\%\mathrm{Removal}}$. Therefore, the working equation becomes:

$$\mathrm{Input} = (\Delta\mathrm{SOC})_{\%\mathrm{Removal}} = \frac{\mathrm{dSOC}}{\mathrm{d\%Removal}} \times \%\mathrm{Removal} \qquad (5.4)$$

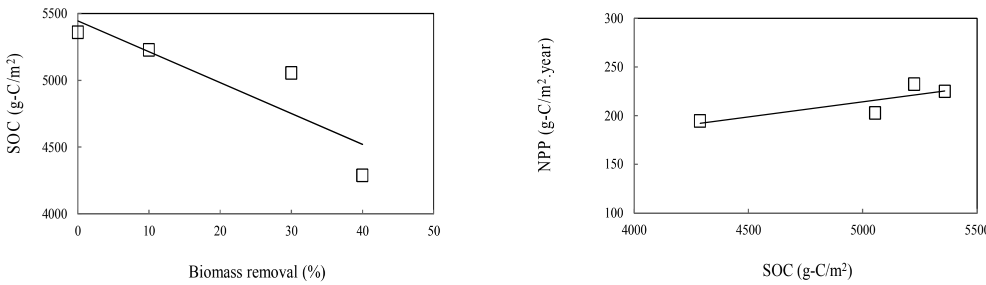
Considering all of the above equations, the proposed impact assessment method can be re-written as the following:

$$(\Delta\mathrm{NPP})_{\%\mathrm{Removal}} = \left(\frac{\mathrm{dNPP}}{\mathrm{dSOC}}\right)_{\mathrm{Climate,\,Vegetation\,,Soil}} \times \frac{\mathrm{dSOC}}{\mathrm{d\%Removal}} \times \%\mathrm{Removal} \qquad (5.5).$$

5.3 Results and discussion

5.3.1 Operation of the proposed method

The following example illustrates the operation of the proposed impact assessment method to assess the effect of removing biomass residues from soil. Figure 5.3 (a) and (b) were drawn based on simulation data of Wang et al. (2008) regarding the grazing system for livestock farming, meadow steppe dominated by *Leymus chinensis* in Songnen grasslands, Northeast China. The SOC and NPP responses to various grazing intensities were recorded over duration of 50 years of observation. The data point in the X-axis represents a 10 year average of 0%, 10%, 30%, and 40% grazing intensities. In this illustration, we did not include data at higher grazing intensity (60% and 80%) since the effect of biomass removal around these rates resulted in a very sharp decrease in NPP, indicating a non-sustainable agricultural practice. In this regard, Wang et al. (2008) suggested an optimal grazing intensity up to 40% in order to maintain the sustainability of *L. chinensis* grassland. Zero percent biomass removal signifies that all of the grass was left undisturbed in the field. Under this condition, the values of intact NPP and SOC were 225 g-carbon/(m^2.year) and 5444 g-carbon/m^2, respectively.



(a) (b)

Figure 5.3. (a) Effect of biomass removal on SOC, where SOC = (-2311 x biomass removal) + 5444, data points indicate the measured relationship between the extent of biomass removal and reduction in SOC; and (b) Effect of changes in SOC on NPP, where NPP = (0.031 x SOC) + 59.34, data points indicate the measured relationship between SOC and NPP. (based on data of Wang et al., 2008).

The grazing region comprises meadow Chernozem soil (35% clay, 45% silt, 20% sand), a semiarid temperate monsoon climate, and mean annual precipitation of 564 mm. Conditions of a pasture area, in many aspects, are similar to the case of biomass residues removal described in this current paper. This is mainly due to management operations of the pasture system being less demanding when compared to typical crop production systems. Therefore, agricultural inputs are minimal. If the mature grass is not harvested, it will, naturally, become litter and decompose, following a closed-loop nutrient cycle.

Assuming that the grassing scenarios presented by Wang et al. (2008) are equivalent to the removal of biomass residues for bioenergy systems and changes in the slopes indicate magnitude of impact to biomass residues removal (strong correlation), the data can be exploited as a basis to illustrate the derivatives of SOC and NPP at various rates of biomass removal. The effect of grazing intensities between 0% and 40% on SOC and NPP are depicted as slopes ((dSOC/d%Removal) and $(dNPP/dSOC)_{Climate,Vegetation,Soil\text{-}types}$ in Figure 5.3 (a) and (b), respectively. Following the parameters expressed in Equation (5.1), CF (dNPP/dSOC) represents the term for the impact assessment step while input (dSOC/d%Removal) represents the results of process inventory. Assuming that an effect of 0%–40% biomass removal on SOC and NPP are linear, as exhibited as straight lines in Figure 5.3 (a) and (b), the value of (dSOC/d%Removal) is –2311 g-carbon/m^2, and (dNPP/dSOC) is 0.031/year. Therefore, at 10% biomass removal, Equation (5.2) becomes:

$$(\Delta \mathrm{NPP})_{10\%} = \left(\frac{0.031}{\mathrm{year}}\right)_{\text{Temperate monsoon, Leymus chinensis, Chernozen soil}} \times -2311 \frac{\mathrm{g \cdot carbon}}{\mathrm{m}^2} \times 10\% = -7.16 \frac{\mathrm{g \cdot carbon}}{\mathrm{m}^2.\mathrm{year}}.$$

It signifies that, at 10% biomass removal, the impact in terms of loss in biomass productivity is 7.16 g-carbon/m^2.year compared to the undisturbed reference. Otherwise stated, the biomass production capacity of the soil associated with the biomass removal at approximately 10% will reduce the carbon stock by 7.16 g for an area of one m^2 annually. Similarly, assessment at 20%, 30%, and 40% removal rates will result in biomass productivity loss as much as 14.33 g-carbon/m^2.year, 21.49 g-carbon/m^2.year, and 28.66 g-carbon/m^2.year, respectively.

The operation of the proposed impact assessment method has been clearly illustrated based on data published in the literature (Wang et al., 2008). The operation of the method is fairly straightforward. It indicates the types of the required data. Therefore, the example in pasture areas under various grazing patterns should be sufficient to illustrate the operation of the proposed method. It is expected that other studies in other regions with similar ecological conditions would show similar changes in carbon stocks.

5.3.2 Sources of uncertainty

There are at least three sources of uncertainties in the proposed impact assessment method. These are related to the model, spatial variability, and data variability. The model uncertainty refers to correlations among parameters, such as between removal rates and SOC (dSOC/dRemoval) or between SOC and NPP (dNPP/dSOC). As shown in Figure 5.3 (a), for example, the dSOC/dRemoval correlation shows non-linear responses at higher biomass removal rates thus creating a higher uncertainty. This kind of responses reflect actual processes taking places within soil systems, i.e. carbon flows into the soil matrix as a result of the decomposition of biomass residues. In this regard, a linear approximation by local derivatives was made over a sufficiently small interval. This approach may be seen by some as major limitation for modeling soil processes. However, in LCA, where small functional units are usual, such linear approximations almost always form the basis of characterization factors (Hauschild, 2005). For example, most toxicity processes (dose-response relationship) are non-linear, but default linearized derivatives were typically used (Crettaz et al., 2002). This approach is more often in agreement with the international standards by ISO (ISO, 2006) and ILCD (EC-JRC-IES, 2010) which use characterization factors as a linearized approximation. This is justified in case of small changes compared to background data, which is true for most functional-unit-based LCA studies.

Spatial distributions are described in terms of local climates, vegetation, and types of soil. It suggests that regional variability of the characterization factor is represented by only 3 ecological conditions. This is obviously a simplification of real situations. However, this classification is by no means strict. Rather, it could be expanded to accommodate more detailed geographical variations. The characterization factor serves as a generic platform to describe how ecological conditions may affect indicator states. They could be implemented at various spatial levels (local or regional), depending on data availability, also in relation to the spatial resolution of the inventory data. The characterization factor could also be employed to estimate spatial variability of different regions depending on data availability. Additionally, the proposed method is intended to describe only a static model, temporal variability is outside the scope of the study.

Uncertainty related to data is a typical issue in agricultural systems. In this regard, soil quality and biomass productivity depend on various ecosystem properties in complex dynamics. From a modeling perspective, not all of the other factors could be included in the method. For example, it is possible that, in certain vegetation types, a considerable fraction of NPP is in the fast-growing form of fine roots entering the soil carbon pool (Cowie et al., 2006). In this case, the contribution of underground biomass may be significant. Therefore, the effect of above-ground biomass removal on SOC may not be as sensitive as when the root to shoot ratio and turnover rates are minimal. Included in this category is variability in biomass decomposition rates which depends heavily on the physical chemistry of the biomass material such as content of less degradable lignin in stumps versus branches (Repo et al., 2011, 2012).

5.3.3 Removal scenarios

In this paper, three possible removal scenarios which require proper impact assessment were described. The focus is on scenario 3 (removing larger fractions without fertilizer compensation) which affects biomass productivity. The option of keeping NPP constant by removing only the surplus amount (scenario 1) or removing larger fractions with fertilizer compensation (scenario 2) would, theoretically, not result in a loss in biomass productivity. This is not an impact assessment issue but a different inventory specification. In the case of constant NPP, the scenarios 1 and 2 would still indicate a loss of SOC as a climate emission. However, this paper is not intended to suggest which option among the three scenarios is better. From an LCA standpoint, it is always an open question to determine the more acceptable option between scenario 2 and scenario 3, which one is more acceptable depending on a specific case of bioenergy systems. Without prior judgments, the proposed impact assessment method could provide additionally quantitative information on the choices of the best option among scenarios. Perhaps, other assessment approaches will cohabitate side-by-side to serve different paradigms.

5.3.4 Further implementation

The operation of the impact assessment method requires data on SOC and NPP. Databases on these parameters are available for certain regions. For example, some databases listed over 700 estimates of NPP worldwide for selected zones and vegetation types (Alexandrov et al., 1999). Recently, Koellner et al. (2012) standardized classification of land use types and regionalization of land use inventories. For other areas, further exploration could be performed since methods for measuring or estimating SOC and NPP are well established. Currently, the primary issues in the implementation of the proposed method on bioenergy systems are locating data demonstrating the relationship between biomass removal at different rates on the states of SOC and NPP (dSOC/dRemoval and dNPP/dSOC). Further, due to data availability, the illustrated example used data from a grazing system, not a bioenergy system. A validation step using proper data is still needed. This latter problem exists because advanced bioenergy technology such as 2nd generation bioethanol is still emerging. In the near future, when the technology becomes feasible on a commercial scale, additional research should be directed to address this issue so that relevant data would become more available.

Discussion on the proposed method is focused on bioenergy application, but the method is also distinctly appropriate for other product systems involving the removal of biomass residues from soil. This is the case since the characterization method provides a general framework converting elementary flows (SOC) into an impact (NPP). Therefore, if data on bioenergy systems are not available, the proposed model could still be useful for other product systems.

5.4 Conclusions

Biomass residues left on the fields following harvest is a promising source of bioenergy feedstock. Unsustainable removal of these residues may adversely affect soil quality and hamper future harvests. Methods to determine such impacts are now increasingly recognized as an important component in LCA studies. We developed a number of improvements to better characterize bioenergy systems. The proposed impact assessment method relates the removal of biomass residues with the soil organic carbon and net primary production. The method does not cover a cradle-to-grave life cycle of bioenergy systems, but one stage, the provision of biomass feedstocks.

Considering the straightforward operation of the method and data availability in the future, we believe that the proposed method has a good potential to be adopted and employed as metrics for assessing the sustainability of bioenergy systems. The method can be regarded as an initial proposal and an assistance in moving research frontiers on the characterization of bioenergy systems.

5.5 Acknowledgement

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Chapter 6

Methodological issues in comparative life cycle assessment

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Abstract

Palm oil systems generate substantial amounts of biomass residues which are, according to best agricultural practices, preferably returned back to plantation in order to maintain soil fertility. However, there are often variations in this practice. Differences in economic status and possible treatment options for biomass residues determine the preferences to perform life cycle assessment (LCA), leading to a divergence in results. Difficulties when comparing LCA results based on literature are not unusual. The objectives of this paper are to provide guidelines for methodological choices that enable a systematic comparison of diverse scenarios for the treatment and valuation of empty fruit bunches (EFBs) and to explore effects of the scenarios on the environmental performances of a palm oil system.

Eleven scenarios were selected to address the possible EFB valuation and expanded boundaries with reference to the main palm oil system (EFBs applied as mulch, converted to compost or ethanol, treated in an incinerator, and sold as coproducts). The life cycle inventories were modeled based upon an Ecoinvent database. Solutions to multifunctional problems were suggested, including the application of system expansion, substitution, and partitioning, depending upon the nature of the scenarios.

Comparison among LCA results based on the same multifunctional units (crude palm oil + palm kernel oil + palm kernel cake) can be accomplished only in cases where additional coproducts were utilized internally. Based on the global warming impact, the mulch option was preferred. The effect of the avoided process of producing synthetic fertilizers and the assumption that all parts of mulch are available as soil nutrient dominantly determined the final result. These need further verification. This study also demonstrates that the status of EFB as waste or goods is influential on the final results if the EFB is employed externally but has no effect if it is utilized internally.

The proposed guidelines provide methodological choices in terms of system boundary, functional unit, and solutions to multifunctional problems. The methods can be used to systematically compare LCA results of different treatment options and valuation of EFB. The preferred alternative for managing this biomass residue could improve environmental performances and orient toward best practices, such as those suggested by the Roundtable on Sustainable Palm Oil (RSPO). Further studies incorporating a site-specific case of palm oil systems would better illustrate the usefulness of the proposed guidelines.

Keywords
Allocation methods . Bioethanol . Biomass residues . Compost . Global warming . Mulch . Multifunctionality . System boundary.

6.1 Introduction

6.1.1 Palm oil and sustainability

Elaeis guineensisis a tropical forest palm that is native to West and Central Africa. It produces three to eight times more oil for a specified area than any other tropical or temperate oil crops (Sheil et al., 2009). Palm oil is an extremely productive business on a large scale and is commercially profitable due to the increasing global demand for edible oils and biofuels (Sheil et al., 2009). Indonesia has become the world's largest palm oil producer, with approximately 21 million metric ton produced in 2009. Indonesia and Malaysia collectively produced around 87 % of the global palm oil (Stichnothe and Schuchardt, 2011). However, the sustainability of the oil palm cultivation and production of palm oil have come under increasing scrutiny, particularly concerning the impacts on global warming as a consequence of massive land use changes (Koh and Ghazoul, 2010). To address these issues, the Roundtable on Sustainable Palm Oil (RSPO) was established in 2003 (legally registered in 2004) in order to promote the use of sustainable palm oil through a voluntary certification scheme and to identify methods that would lead to environmental improvement (Laurance et al., 2010). Among the promoted good practices, a potential instrument to improve sustainability in the life cycle of palm oil systems is proper management of biomass residues (Hansen et al., 2012).

6.1.2 Potential of solid biomass residues and treatment options

Oil palm biomass comprises fronds, leaves, trunks, root, fruit bunches, and inflorescences, of which approximately only about 10 % yields palm oil and palm kernel oil (Lee and Ofori-Boateng, 2013). Fronds and trunks are generated in plantation areas from periodic harvesting of fresh fruit bunches (FFBs) and periodic replanting of old palm trees, respectively. The cumulative amount of fronds for the 23 years of the productive period of a palm tree is about 1.8 t on a dry weight basis, and the total biomass that is cut down during replanting is about 0.71 t of trunk and fronds per palm (Yusoff, 2006). The exact amount will vary significantly depending upon planting material and field management. In 2011 alone, Indonesia and Malaysia generated nearly 182 million metric ton of dry solid palm biomass which is projected to increase to almost 230 million metric ton by 2020 (MPOB, 2012). Palm oil mills also generate substantial amounts of biomass residues. For example, 1 t of FFB on wet basis results in 0.220 t of empty fruit bunch (EFB), 0.135 t of mesocarp fiber, and 0.055 t of palm kernel shell (Yusoff, 2006).

Press fiber and shell are commonly exploited as solid fuels for steam boilers in order to generate electricity and to meet the internal energy demand for the operation of the palm oil mill, which are often located in remote areas far from national grids (Stichnothe and Schuchardt, 2011). From the perspective of best agricultural practices, fresh EFBs are preferably returned to plantation as mulch to maintain soil fertility (Salétes et al., 2004). This closed loop nutrient cycle can reduce the need for external fertilizers, which subsequently results in an efficient palm oil system. However, the extensive distance between oil mills and plantations may develop into a limiting factor for the feasibility of land application. Indeed, fresh EFBs, which are wet, bulky, and voluminous, are undesirable for handling and transportation. Consequently, there are variations in practice. Some of the EFBs may be further processed into bioenergy, converted to compost, directly sold as coproducts, or incinerated with or without energy recovery. These various treatment options are more likely to occur in oil mills with limited or no plantation areas, which typically process FFBs from other plantations.

The interest in converting biomass residues into other valuable products is also increasing (Stichnothe and Schuchardt, 2010; Hansen et al., 2012; Chiew and Shimada, 2013; Tuck et al., 2012). Some of these developments are directed toward bioenergy development (Lim and Lee, 2011; Wiloso et al., 2012; Chiew and Shimada, 2013). In Malaysia, for instance, the Small Renewable Energy Power Program (SREP) was launched in 2001 to encourage utilization of agriculture residues for generating electricity that would be connected to the national grid. This policy has attracted investments for developing combined heat and power plants (CHPs) exploiting palm oil biomass residues, including EFB. Some CHPs were installed at the palm oil mills, and others were independent power plants connected to the grid. Thus far, there are three CHPs operating from 1 to 14 MW as reported under the SREP program (Chiew and Shimada, 2013). In Indonesia, the government has also recently issued new regulations concerning the price of electricity for bioenergy-based power plants (Kusdiana, 2013). Within the last 10 years, ten on-grid power plants based on palm oil residues were constructed, with a contracted capacity of 2 to 10 MW. However, not all of these plants are continuously in operation. The primary issues are the increasing price and the lack of continuous supply of biomass feedstock (Kusdiana, 2013).

Considering the significant amounts and the diversity of palm biomass residues, potential use and manners of valuation are numerous. Certain options may offer better economic and environmental benefits than others. However, most of the palm oil producers have not yet received a specific directive for selecting which options are most environmentally appropriate. As a consequence, some of these companies are continuing to practice old disposal methods, such as dump and burn (Chiew and Shimada, 2013), thus wasting economic opportunities and adding carbon emissions to the atmosphere.

6.1.3 Comparison of previous LCA studies on EFB

Recent life cycle assessment (LCA) studies on palm oil systems involving further treatment of EFB are illustrated in Table 6.1. In addition to the primary products (palm oil or biodiesel), the system also produced coproducts such as compost, bioethanol, biochar, biooil, and/or syngas. The tabulated LCA studies were limited to those investigating the impact on global warming, representing the most studied impact category. For that purpose, quantitative data were extracted from the papers as depicted in the last row of Table 6.1. The LCA results show that the global warming impacts ranged broadly from positive values (greenhouse gas (GHG) emissions) to negative values (GHG savings). From the point of the LCA procedure, these results are not practically comparable since the scores were not based on the same functional units. This is the primary difficulty when utilizing literature data to compare LCA results. The use of different functional units is not unusual since each study is developed for a specific goal and scope, depending on the objective of the study.

Comparing and interpreting results among independent LCA studies are not a straightforward task. The ISO 14044 requires comparison between product systems to be made on the basis of the same functional unit, which provides a reference to relate the inputs and the outputs (ISO 2006). With this reference, comparison among different product systems could be made on a common basis. In contrast, comparison based on different functional units would be of no values. To properly compare different EFB treatments, therefore, a dedicated LCA study must be conducted specifically for the purpose of that comparison.

6.1.4 Valuation of biomass residues

The common criteria in the valuation of biomass residues are that coproducts provide relatively similar proceeds as the main product, while by-products have lesser value than coproducts, and waste has a negative value, i.e., treatment costs that are not offset by further valuation (Singh et al.2010). However, in the LCA community, by-products are not typically differentiated from coproducts. Rather, all economic outputs other than the main product are considered coproducts with different values. These coproducts are encompassed within a generic term that comprises all potential outputs from a process. When adopting this view, the system boundary of a palm oil system must include all generated biomass residues throughout the process chains. Therefore, in addition to trunks, fronds, and inflorescences from the plantation, the life cycle inventory (LCI) must also incorporate POME, shell, fiber, and EFB from the oil mills.

Table 6.1. Comparison of LCA results on global warming involving different treatments for EFB.

LCA Parameters	Stichnothe & Schuchardt (2010)	Lim & Lee (2011)	Hansen *et al* (2012)
Product systems	Palm oil	Biodiesel	Biodiesel
Expanded product systems	Palm oil + compost	Biodiesel + bioethanol	Biodiesel + pyrolysis products (biochar, biooil, syngas)
Goals	To evaluate environmental impacts of treating EFB (and POME*) in a palm oil system	To maximize the output from a limited amount of land by integrating bioethanol processes in a biodiesel system	To compare GHG balances of different treatments of EFB in a bioediesel system
Functional units	1 metric ton of FFB	Use of 1 ha of land in 100 years	1 metric ton of biodiesel
GHG emissions (+) GHG savings (–)	+5.1 up to +7.4 kg CO_2-eq/ metric ton FFB (explanation of Figure 2[b])	+100 up to +900 t CO_2-eq/ ha land (estimated from Figure 4[a])	–440 kg CO_2-eq/ metric ton biodiesel (Table 6.3[b])

POME = palm oil mill effluent.

[a]Scenario 1 = 200+800+200+0+0–1100 = 100; Scenario 2 = 200+800+200+0+1100–1400 = 900; Scenario 3 = 200+950+200+0+350–1200 = 500 (for detail see Figure 4 of Lim & Lee (2011)).

[b]Stichnothe and Schudhardt (2010) assumed that biogas was used to replace the fuel for starting a boiler (internal use), while Hansen *et al* (2012) assumed biogas was used for electricity production to supply the national grid (external use).

Economic flows in LCA travel between two unit processes; therefore, each economic flow must be the output of one process or the input of another process (Heijungs and Frischknecht, 1998). The economic value of flows can be employed as a criterion to determine the status of biomass residues. Guinée et al. (2009) defined products as possessing a positive economic value, whereas waste featured a negative economic value. More specifically, products in the LCA terminology include goods, energy, or services (Guinée et al., 2009). In the current paper, we considered EFB as either waste or goods, depending on the specific conditions of the scenarios.

The process following a waste flow can be either a treatment unit to reduce the pollution strength of the waste or a conversion unit to create a certain product. The latter process provides both a waste treatment function and a function intending to produce a certain product (Bellon-Maurel et al., 2013). In the context of defining a system boundary, a waste stream is conventionally assumed to be free of environmental burden. The impact is directed entirely at the products and coproducts preceding the waste stream. This signifies that actors in the upstream chain must compensate for the treatment or elimination of the waste stream.

There are numerous cases where it is uncertain whether the price of an agricultural residue is positive or negative. Due to technological developments, fluctuations in markets, and governmental policy, waste may rapidly become goods or vice versa. Depletion of natural resources has encouraged the recycling of waste into useful products. These developments may profoundly affect the valuation of biomass residues in a palm oil system. For the moment, the EFB may not yet have an actual market value; however, in the future, it may become valuable. In this context, there has been increasing interest in utilizing EFB as a potential feedstock for bioenergy (Lim and Lee, 2011; Wiloso et al., 2012; Chiew and Shimada, 2013) and other biorefinery products such as biochar, biooil, and syngas (Hansen et al., 2012), but LCA studies addressing biomass residues within different valuation schemes are, thus far, lacking. This paper intends to fill the gap.

6.1.5 Multifunctionality and burden allocation

A multifunctional process is a unit process yielding more than one functional flow. One way to solve a multifunctional problem is by partitioning methods which artificially split the multifunctional process into a number of independently operating monofunctional processes (Heijungs and Guinée, 2007). With this approach, the emissions will decrease; however, the functional unit is not modified. There are different types of multifunctional processes depending on specific situations, i.e., coproduction, recycling, and combined waste processing (Guinée et al., 2004). Coproduction features more than one functional outflow and no functional inflow. Recycling comprises one or more functional outflows and one or more functional inflows. It reduces potentially harmful emissions from waste while simultaneously creating a useful product. Combined waste processing comprises no functional outflow but more than one functional inflow. The illustrated application of the above concept on handling biomass residues in an agricultural system is shown in Figure 6.1 (based on Wiloso and Heijungs, 2013). If the biomass residues are valued as goods or waste (cases a and c), the environmental burden is partitioned between product1 and product2 or waste1 and waste2, respectively. If the biomass residues valued as waste are converted to products (case b), the environmental burden is to be partitioned between the upstream (waste input) and downstream (product output) links. In cases b and c, some and all burdens, respectively, will be attributed to the upstream product system. However, for simplicity, these upstream links are not shown in Figure 6.1. The partitioning factors can be based on different principles: physical properties or economic values of the functional flows. The physical properties can be based on the relative mass, carbon content, or energy content, whereas economic values are based on the relative market value of the functional flows.

The ISO standard (ISO 2006) prefers to avoid the above allocation methods when addressing multifunctional problems. The priority is to divide processes into subprocesses or expand the boundary of the product system. System expansion includes a coproduct as an additional function to a product system. The resulting expanded system, therefore, consists of more than one functional flow. It modifies the original functional unit into a new functional unit with two or more products with no change in emissions. The ISO standard mentions system expansion and partitioning but does not mention substitution, also referred to as subtraction or avoided burdens (Heijungs, 2014). However, almost all guidelines mention substitution.

The term system expansion is often mixed up with the substitution method. Both approaches address multifunctional problems but manifest quite differently. Substitution adds an avoided process to the system that exactly cancels out the coproduct. The production of a coproduct by the system under study circumvents another production process in another system. This avoided production process results in avoided emissions that should be subtracted from the studied product system (Wardenaar et al., 2012).

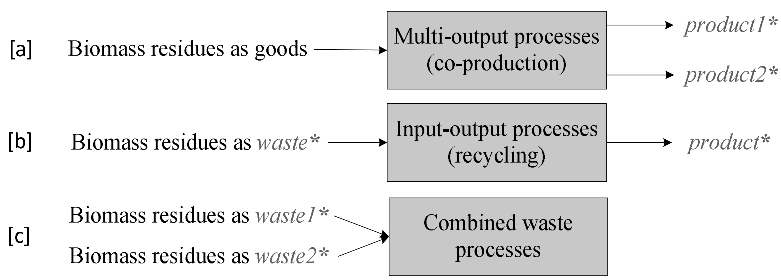

Figure 6.1. Status of biomass residues and possible multifunctional processes. The last case (combined waste processes) does not yield products, but emissions. (*in *italic* = functional flow). For simplicity, the upstream links producing biomass residues are not shown.

6.1.6 Objective of the paper

There is an increasing interest in utilizing EFB in palm oil systems as feedstock for useful products. The pace of LCA research in the area of coproduct valuation is also accelerating. However, these developments are not without issues. The ISO 14044 leaves too much room in terms of methodological choices to perform an LCA (Heijungs and Guinée, 2007). In addition, the overall complexity is potentially increased by different valuation of biomass residues as goods or waste. Diversity in treatment options for biomass residues, which is particularly prevalent in the case of palm oil system, may also cause variations in the preferences to perform LCA, leading to divergence in results. Meanwhile, in order to select suitable options, valid and consistent methodology is required. The above discussion leads to an important research question of how to properly assess and compare the effect of different treatment options and valuation of EFB on the performance of a palm oil system. The objectives of this paper are to provide guidelines for methodological choices that enable a systematic comparison of diverse scenarios for the treatment and valuation of EFB and to explore effects of the scenarios on the environmental performances of a palm oil system. Methodological choices in terms of system boundary, functional units, and solutions to multifunctional problems are suggested, and their implementations on assessing various scenarios are illustrated.

6.2 Methods

The LCI models were developed to represent a palm oil system integrated with various options in handling EFB. Eleven scenarios were selected to cover possible EFB valuation (as goods or waste) and expanded boundaries with reference to the main palm oil system (application as mulch, conversion to compost or ethanol, treatment in an incinerator, and EFBs directly sold as coproducts). Illustration on these scenarios can be seen in Fig. 2a and 2b. Ecoinvent assumes that, in the palm oil system, the trunks, fiber, and shell are internally (closed loop) recycled (Jungbluth et al., 2007). More specifically, the biomass residues in the plantation (trunks) were recycled with no significant additional inputs or net emissions. Fronds cut down for harvesting the FFB were not mentioned in the report; however, we assumed that besides trunks, fronds were also internally recycled. Meanwhile, fiber, shell, and EFB were cogenerated to produce heat and electricity to be used internally in the oil mills. Our current study assumed the same as above (Ecoinvent) but excluded the EFB from the cogeneration process and treated it further in various ways.

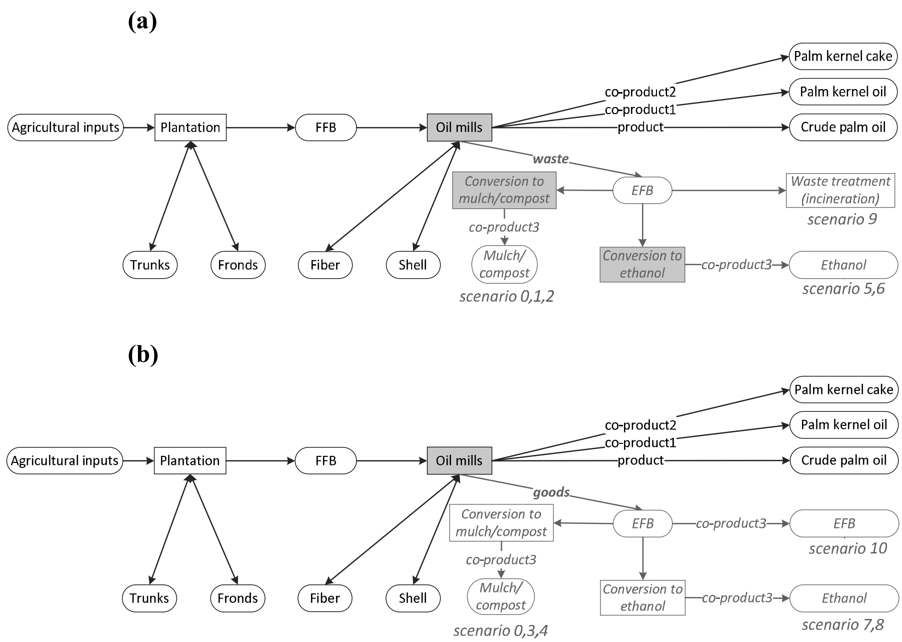

Figure 6.2. (a) System boundary of possible treatment options for EFB when valued as *waste*: applied as mulch or converted to compost (Scenarios 0, 1, and 2), converted to ethanol (Scenarios 5 and 6), and treated in an incinerator (Scenario 9). EFB sub-systems are in *italic*. (= goods or waste; = unit process; = multifunctional process; ↔ = biomass is used internally). (b) System boundary of possible treatment options for EFB when valued as *goods*: applied as mulch or converted to compost (Scenarios 0, 3, and 4), converted to ethanol (Scenarios 7 and 8), sold as a co-product (Scenario 10). EFB sub-systems are in *italic*. (= goods or waste; = unit process; = multifunctional process; ↔ = biomass is used internally).

The application of EFB as mulch or conversion of EFB into compost and ethanol was seen as a way to manage biomass residues leading to environmental improvement.

Incineration was used to represent treatment of EFB in a waste processing unit. EFB can also be regarded as a direct coproduct when it has market values. Processing of these additional coproducts was assumed to take place within the oil mill area so that no transportation was required for the EFB feedstock. The mulch, compost, and ethanol can be employed internally or externally. Internal uses indicate that the mulch or compost is applied to the plantation field as a substitution for inorganic fertilizer or the ethanol is used as biofuel to substitute gasoline for the oil mill operation. External uses mean that these coproducts will become a component of another product system that is external to the palm oil system.

Table 6.2 summarizes the guidelines for the methodological choices to assess environmental impact for the 11 scenarios reflecting different decision situations. The approaches to solve multifunctional problems are a combination of system expansion, substitution, and partitioning depending upon the nature of the scenario. For example, scenarios 0–8 employ a combination of system expansion and substitution or system expansion and partitioning approaches. These scenarios are considered expanded systems since they included additional coproducts (mulch, compost, or ethanol). Scenario 10 uses only one method to solve multifunctional problems, i.e., partitioning. Substitution refers to the use of the resulting coproducts within the main palm oil system (scenarios 0, 1, 3, 5, and 7) which consequently avoided the use of other products of similar functions. In this regard, inorganic fertilizer and gasoline were selected to substitute the mulch or compost and the ethanol, respectively. Currently, diesel oil is dominantly used in a palm oil system. The possible change from the current practice (diesel oil) to the future scenario (ethanol) could be evaluated in terms of their environmental performances.

Table 6.2. Guidelines for methodological choices for comparison of different treatment options and valuation for EFB.

Sce-nario	System boundary of different treatment options with reference to the main palm oil system	EFB valuation	Approaches in dealing with multifunctional issues		
			Expanding the product system with additional coproducts related to EFB	Partitioning of multifunc-tional processes	Substituting with avoided processes
0–M	Direct application of fresh EFB as mulch, internal or external uses[a]	Waste Goods	Mulch	Production of mulch	Production of inorganic fertilizer
1–WCI	Conversion of EFB to compost, internal use	Waste	Compost	·	Production of inorganic fertilizer
2–WCE	Conversion of EFB to compost, external use	Waste	Compost	Production of compost	·
3–GCI	Conversion of EFB to compost, internal use	Goods	Compost	·	Production of inorganic fertilizer

Sce-nario	System boundary of different treatment options with reference to the main palm oil system	EFB valuation	Approaches in dealing with multifunctional issues		
			Expanding the product system with additional coproducts related to EFB	Partitioning of multifunc-tional processes	Substituting with avoided processes
4–GCE	Conversion of EFB to compost, external use	Goods	Compost	·	·
5–WEI	Conversion of EFB to ethanol, internal use	Waste	Ethanol	·	Production of gasoline
6–WEE	Conversion of EFB to ethanol, external use	Waste	Ethanol	Production of ethanol	·
7–GEI	Conversion of EFB to ethanol, internal use	Goods	Ethanol	·	Production of gasoline
8–GEE	Conversion of EFB to ethanol, external use	Goods	Ethanol	·	·
9–WI	Treatment of EFB in an incinerator, internal treatment	Waste	·	·	·
10–GcoP	Coproduction (EFB is direct coproducts), external use	Goods	·	Production of CPO, PKO, PKC, and EFB	·

CPO = Crude Palm Oil, *PKO* = Palm Kernel Oil, *PKC* = Palm Kernel Cake.
[a]The effect of the preparation of EFB as mulch on field sites (apart from transportation from oil mills to plantation fields) was so small that it did not change the base line value (see detail in Table 6.3). Therefore, it does not make any different either EFB was valued as waste or goods, or either used internally or externally. For convenient, therefore, all of these variations are combined as one scenario.

Comparison among scenarios was performed based on the multi-functional unit, CPO+PKO+PKC. It was employed as a baseline without including EFB in the inventory. The reason for selecting these three products rather than a mono-functional unit (CPO) is to better represent the environmental burden of the overall system. Further processes on EFB (Scenarios 0-8 and 10 in Figure 6.2) result in additional co-products, i.e. mulch, compost, ethanol, or EFB. When these co-products are introduced in the inventory, the expanded product systems become CPO+PKO+PKC+mulch, CPO+PKO+PKC+compost, CPO+PKO+PKC+ethanol, or CPO+PKO+PKC+EFB, respectively. Meanwhile, the incineration option (Scenario 9) is a simple waste treatment case with no additional co-product.

In addition to producing mulch, compost and ethanol, Scenarios 0, 2, and 6 were also recycling cases since the input EFB was valued as waste. In this case, the environmental burden would need to be partitioned between the upstream and downstream flows. This partitioning reflects burden attribution between the function to reduce the pollution strength of the waste (treatment) and the function to create new products (production). Scenario 10 is a co-production case with EFB as a direct co-product exhibiting certain market values. In this regard, EFB as a co-product is sold to external parties whereby there is no control over their final uses. It could be used, for example, for compost, fibers, or energy.

The models were developed with the LCA software CMLCA v5.2 (2012) and based on inventories of an Ecoinvent database v2.2 (2010). An impact indicator on global warming was selected as the primary criterion to compare the LCA results. The impact assessment referred to the CML 2001 method for climate change (GWP 100 year average, global). The following section describes the inventories of the main palm oil system and additional EFB processes in more detail. All processes were described by indicating the ID-number, region, and year of the Ecoinvent database. Also, assumptions that were used in every process are indicated so that confirmation for the final LCA results could be made. Some modification from the default inventories was made, particularly for EFB availability (initially co-generated to produce energy) and ethanol processes (initially including feedstock transportation). In addition to Sections 6.2.1-6.2.6, a more complete description of the product systems is located in the supplementary material, Table SM1.

6.2.1 Palm oil

The LCI model consisted of the production of FFB at a farm (ID#199: Malaysia, 2002–2006) and palm oil in oil mills (ID#150MO: Malaysia, 1995–2006). The first inventory assumed that land provision included conversion of tropical rain forest to agricultural area. Plantation operation included seedling preparation; field emissions; and transportation of FFB, pesticides, and fertilizers.

Most palm oil mills produce palm kernels, which are then transported to specialized kernel oil extraction facilities. For simplicity, in this study, we assumed that the palm oil mills processed all potential coproducts, i.e., CPO, PKO, and PKC. Therefore, the total burden could be distributed properly among these coproducts. If the kernels are to be sent to other mills, we need to introduce transportation factor, which may add layers of uncertainty.

The second inventory included a 100-km transport of FFB from farm to oil mill gates. The oil production was based on mechanical processes including extraction of oil by screw press and removal of impurities in a settling tank with a centrifuge and evaporator. Every kilogram of processed FFB resulted in 0.2156 kg CPO, 0.0266 kg PKO, and 0.0317 PKC. Economic values of these products were CPO=Ringgit Malaysia (RM) 1.490/kg, PKO=RM 2.565/kg, and PKC=RM 0.175/kg, in which RM denotes Malaysian currency. Based on these data, economic partitioning coefficients were determined as CPO=81.3 %, PKO=17.3 %, and PKC=1.4 %. Environmental performances of the palm oil system were based on a multifunctional unit of 1,000 kg CPO+123 kg PKO+147 kg PKC or 1,270 kg CPO + PKO + PKC in short. In addition, the system also coproduced 1,051 kg fresh EFB at 40 % dry matter. All of the above data are based on Ecoinvent report No. 17 (Jungbluth et al. 2007). A modification was made to the default inventory by excluding the contribution of EFB in energy production, a cogeneration process (ID#79MO).

6.2.2 Mulch

The LCI model consisted of the application of mulch (ID#171). Production of inorganic fertilizers such as ammonium nitrate as N (ID#40<006484-52-2>), single superphosphate as P_2O_5 (ID#54), and potassium chloride as K_2O (ID#50<007447-40-7>) was also considered to account for the effect of mulch substitution with inorganic fertilizers

(Nemecek and Kägi 2007). Transportation of mulch from oil mills to plantation fields included lorry transport (ID#1941) and tractor transport (ID#188). Inorganic fertilizers were provided by utilizing additional rail transport (ID#1983). The transportation distances were based on 100 km between oil mills and farm gates (lorry), 25 km to reach plantation fields (tractor) for mulch, and an additional 600 km of rail transport for substituted fertilizers (Jungbluth et al. 2007).

In the inventory, 1,051 kg fresh EFB was applied directly as mulch. Land application as mulch would require approximately 30 t EFB per hectare (Haron, 2013). Therefore, the economic outputs of the expanded system were 1,270 kg CPO + PKO + PKC + 0.035 ha of plantation area. The fertilizing values of EFB mulch were adopted from Haron (2013), i.e., 0.8 % N, 0.22 % P_2O_5, and 2.9 % K_2O fertilizer on a dry basis. Similar values were also provided by Caliman et al. (2013). Based on the above unit processes, the mulch was equivalent to 9.61 kg ammonium nitrate, 4.40 kg superphosphate, and 20.32 kg potassium chloride. The production of the above amount of inorganic fertilizers emitted 103.9 kg CO_2-eq. The fertilizing value of the mulch is credited if it is internally employed as fertilizer (scenario 0).

6.2.3 Compost

The LCI model consisted of the production of compost (ID#58). The technology was based on open windrow composting as described in Ecoinvent report No. 15 (Nemecek and Kägi, 2007). Unit processes for the production and transportation of inorganic fertilizers were identical to those of the mulch. Chiew and Shimada (2013) suggested that 2,600 kg of fresh EFB resulted in 1,000-kg compost with fertilizing values of 2.2 % N, 1.28 % P, and 2.79 % K on a dry basis. Based on that, in the inventory, 1,051 kg fresh EFB was converted to 404.2-kg compost of 50 % dry matter. As a result, the economic outputs of the expanded system were 1,270 kg CPO + PKO + PKC + 404.2-kg compost. Based on the above unit processes, the compost was equivalent to 12.70 kg ammonium nitrate, 28.21 kg superphosphate, and 11.32 kg potassium chloride. The production of the above amount of inorganic fertilizers emitted 188.3 kg CO_2-eq. The fertilizing value of the compost is credited if it is internally employed as fertilizer (scenarios 1 and 3).

6.2.4 Ethanol

The LCI models consisted of the production of 95 % ethanol (ID#161MO) and further dehydration to 99.7 % ethanol (ID#11795). The first inventory included the production of ethanol and electricity from hardwood chips. Process stages included pretreatment to isolate cellulose from wood matrix, simultaneous saccharification and cofermentation, and distillation to recover ethanol. Economic partitioning coefficients of the resulted ethanol and electricity were 99.7 and 0.3 %, respectively. A further description can be found in Jungbluth et al. (2007). A modification was made to the default inventory by excluding the transportation of wood chips from forest to distillery (ID#161MO). Further, wood chip feedstock was replaced by fresh EFB based on equivalent dry weight. Production of gasoline (ID#1570) was considered to account for the effect of ethanol substitution.

In the inventory, 0.55448-kg dry mass of EFB, equivalent to 0.00232-m^3 hardwood chips, was converted to 0.144-kg 99.7 % ethanol. All inputs and emissions for the same dry mass of EFB were assumed equal to those for dry mass of hardwood chips. As a result, the economic outputs of the expanded system were 1,270 kg CPO + PKO + PKC + 109.3 kg ethanol. The energy content of ethanol and gasoline is 31 and 46 MJ/kg, respectively (Chiew and Shimada, 2013) Therefore, 109.3 kg ethanol is equivalent to 73.66 kg gasoline. The production of this amount of gasoline emitted 50.1 kg CO_2-eq. The energy content of the bioethanol is credited if it is internally utilized as biofuel (scenarios 5 and 7). The comparison between ethanol and gasoline was done at the production gates of ethanol and gasoline. This is quite a reasonable approximation since the difference in emissions from the combustion of these fuels is negligible compared to the difference in the upstream processes (fuel production). If such use phase will be calculated, the combustion of biogenic carbon (ethanol) should be considered as well because carbon capture during plant growth was included in the inventory (Electronic Supplementary Material, Table SM1).

6.2.5 Incineration

The LCI model consisted of the controlled burning of wood in a municipal solid waste incinerator (D#2130). A controlled incineration was chosen since open burning is prohibited in a palm oil system. The incinerator produced electricity and heat; however, no burden allocation was assigned to these coproducts. The generated solid residues were landfilled. A further description can be found in Ecoinvent report No. 13 (Doka, 2003). Prior to being fed into an incinerator, drying is required to bring the water content of the EFB from 60 to 20 %. The unit process employed for this purpose was grass drying (ID#160). Overall, based on 1,051-kg EFB input, two processes were involved, i.e., evaporation of 525.5 kg water and incineration of 525.5 kg EFB of 20 % water content.

6.2.6 EFBs as direct coproducts

The free on board (FOB) prices of EFB at the oil mills ranged between Indonesian Rupiah (IDR) 20/kg EFB and IDR 50/kg EFB, but it was often available at no cost (anonymous field survey in Northern Sumatera, July 2011). The FOB price of palm oil at oil mills was IDR 9,000/kg CPO (GAPKI, 2013). These data were used to determine the partial environmental burden attributed to EFB as a direct coproduct. For another currency, the following conversion rates can be used: US$1=IDR 9,070 in December 2011 and US$1=IDR 12,250 in December 2013 (www.freecurrencyrates.com).

6.3 Results and discussion

The global warming performance at the cradle-to-gate boundary (the plantation and oil mill phases) was 2,068 kg CO_2-eq. and at the gate-to-gate boundary (the oil mill phase) was 144.7 kg CO_2-eq. These results were based on the Ecoinvent assumption that EFBs together with shell and fiber were burned in a cogeneration process. In the current paper, we modified this assumption that EFB was available for other purposes while the energy produced by fiber and shell was sufficient for the entire mill operation. In fact, this is often the case in practice. Therefore, we excluded the EFB contribution to the cogeneration process, which was ascertained to be 21.1 kg CO_2-eq. Subtracting this from the default values, the global warming performances of the above systems change to 2,047 kg CO_2-eq. and 123.6 kg CO_2-eq., respectively. Detailed calculation presented in this section is included in the Electronic Supplementary Material, Tables SM2 and SM3.

Contribution of the upstream operations to the farm gate amounted to 94 % of the total emissions (2,047 kg CO_2-eq.). Transport of FFB from the farm gate to the oil mill and the oil mill operations, hence, only accounted for the remaining 6 % or 123.6 kg CO_2-eq./1,270 kg CPO + PKO + PKC. The contribution of the plantation phase was so dominant that the effects of different treatments on EFB in the final LCA results could hardly be observed at the cradle-to-gate boundary. We further examined changes due to different treatments to EFB only within the oil mill boundary. Therefore, the process of producing FFB in the plantation was cut off. This was meant to zoom in the quantitative figures to be able to see the effect of different treatments. In the case of mulch and compost, coproducts which are internally recycled, the physical substitution with mineral fertilizers would of course be taken place in the plantation phase. This substitution should satisfy two general requirements: (1) the options deliver the same function and (2) the function has the same unit. In the fields, mulch and compost function as nutrient provider to soil. Therefore, these organics and their substituted synthetic fertilizers can be compared to each other on the basis of their fertilizing values. Additionally, the substitution of synthetic fertilizers with mulch or compost requires that all the emissions up to the point of substitution (for example, the compost process and field emissions) are assigned to the main product system. Furthermore, in order to have meaningful comparisons, all quantitative results presented in Table 6.3 were calculated based on the same, gate-to-gate, system boundary. This is quite a common practice in comparative LCA.

The implementation of the proposed guidelines on methodological choices to compare 11 possible scenarios is presented in Table 6.3. It illustrates a step-by-step calculation of the final results. More detailed calculation is included in the Electronic Supplementary Material, Tables SM4 to SM7. The global warming impacts were adjusted considering multifunctional problems in terms of expanding the product system with additional coproducts, substitution with equivalent products, or burden partitioning.

Table 6.3. Global warming performances of a palm oil system reckoning with different treatment options and valuation for EFB (kg CO_2-eq/1270 kg CPO+PKO+PKC).

Sce-nario	System boundary of different treatment options with reference to the main palm oil system	Initial value	Adjustment on LCA scores considering multifunctional issues			Final value	
		CPO+ PKO+ PKC[a]	Expanding the product system with additional coproducts	Partition-ing of multi-functional processes	Substitu-ting with avoided processes[b]	CPO+ PKO + PKC	Mulch, compost, ethanol, EFB for external uses
0−M	Wastes or Goods, Mulch, Internal or External[c]	123.6	+0.7	negligible	−103.9	20.4	negligibl e
1−WCI	Wastes, Compost, Internal	123.6	+146.4	·	−188.3	81.7	·
2a−WCE	Wastes, Compost, External (treatment:product ion=2:1)	123.6	·	+97.6[d]	·	221.2	48.8[d]
2b−WCE	Wastes, Compost, External (treatment:product ion=1:2)	123.6	·	+48.8[d]	·	172.4	97.6[d]
3−GCI	Goods, Compost, Internal	123.6	+146.4	·	−188.3	81.7	·
4−GCE	Goods, Compost, External	123.6	·	·	·	123.6	146.4
5−	Wastes, Ethanol,	123.6	+42.2[e]	·	−50.1	115.7	·

Sce-nario	System boundary of different treatment options with reference to the main palm oil system	Initial value	Adjustment on LCA scores considering multifunctional issues			Final value	
		CPO+ PKO+ PKC[a]	Expanding the product system with additional coproducts	Partition-ing of multi-functional processes	Substitu-ting with avoided processes[b]	CPO+ PKO + PKC	Mulch, compost, ethanol, EFB for external uses
WEI	Internal						
6a–WEE	Wastes, Ethanol, External (treatment:production=2:1)	123.6	·	+28.1[d]	·	151.7	14.1[d]
6b–WEE	Wastes, Ethanol, External (treatment:production=1:2)	123.6	·	+14.1[d]	·	137.7	28.1[d]
7–GEI	Goods, Ethanol, Internal	123.6	+42.2[e]	·	−50.1	115.7	·
8–GEE	Goods, Ethanol, External	123.6	·	·	·	123.6	42.2[e]
9 WI–[f]	Wastes, Incinerator	123.6	·	·	·	366.8	·
10a–GcoP	Goods, co-Production (EFB price = 0.0022*CPO)	123.6	·	−0.3	·	123.3	0.3
10b–GcoP	Goods, co-Production (EFB price = 0.0056*CPO)	123.6	·	−0.8	·	122.8	0.8

Some figures do not add up due to round off. All data presented in this table can be traced back to Tables SM1-SM7 of the Electronic Supplementary Material (Online Resource).

[a]Corrected values, i.e. 144.7 (default) – 21.1 (EFB contribution in co-generation process) = 123.6 kg CO_2-eq.

[b]Substitution with NPK fertilizer (9.61 kg ammonium nitrate + 4.40 kg superphosphate + 20.32 kg potassium chloride = 1051 kg or 0.035 ha of EFB mulch), (12.70 kg ammonium nitrate + 28.21 kg superphosphate + 11.32 kg potassium chloride = 404.2 kg of EFB compost), or with fossil fuel (73.66 kg gasoline = 109.3 kg 99.7% ethanol).

[c]The effect of the application of EFB as mulch was so small (0.7 kg CO_2-eq) that it practically became negligible when partitioned.

[d]Partitioning ratio of 2:1 indicates that Scenarios 2a and 6a allocated twice heavier burden for reducing the pollution strength of EFB than for producing compost or ethanol. In contrast, Scenarios 2b and 6b (1:2) allocated twice heavier burden for producing compost or ethanol than reducing the pollution strength of EFB.

[e]Corrected values, i.e. 57.1 (default) – 14.9 (transportation of wood chips from forest to distillery) = 42.2 kg CO_2-eq.

[f]Consisted of two processes: drying (237.1 kg CO_2-eq) and incineration (6.2 kg CO_2-eq).

Based on the last two columns in Table 6.3, the global warming impacts of the 11 scenarios are visualized in Figure 6.3. The white bars represent the impact of the additional coproducts (mulch, compost, ethanol, or EFB) when employed externally, while the black bars represent the final impacts of the primary palm oil products (CPO + PKO + PKC). These results are point value data with no uncertainty estimates. LCA results are compared based on these point values since additional assumptions and data, other than those from Ecoinvent, were not completed with uncertainty estimates. However, these data are sufficient to illustrate how comparison between different scenarios was performed.

The final results are presented based on how products of the EFB processes are exploited with reference to the palm oil system: internal uses (scenarios 0, 1, 3, 5, 7, and 9) or external uses (scenarios 2, 4, 6, 8, and 10). Comparison based on the same multifunctional units CPO + PKO + PKC is possible only for the internal use cases. These are the cases where the mulch, compost, and ethanol were used internally to substitute inorganic fertilizers and gasoline, respectively. It is assumed that the inorganic fertilizer processes were the avoided processes, producing coproducts with functioning equivalent to that of mulch or compost. Similarly, the gasoline processes were the avoided processes, producing coproducts with functioning equivalent to that of ethanol. Therefore, the functional units of these scenarios after the inclusion of coproducts and substitution with equivalent products are the following:

- Scenario 0: (CPO+PKO+PKC) + (mulch) - (fertilizer) ≈ (CPO+PKO+PKC)’
- Scenarios 1 and 3: (CPO+PKO+PKC) + (compost) - (fertilizer) ≈ (CPO+PKO+PKC)’’
- Scenarios 5 and 7: (CPO+PKO+PKC) + (ethanol) - (gasoline) ≈ (CPO+PKO+PKC)’’’
- Scenario 9: (CPO+PKO+PKC) ≈ (CPO+PKO+PKC)’’’’.

These multifunctional flows have different emission values that can be utilized as a basis for comparison since they have the same functional unit (CPO + PKO + PKC) and the same unit (kg CO_2-eq.). Referring to the baseline value of 123.6 kg CO_2-eq./1,270 kg CPO + PKO + PKC, the mulch option (20.4 kg CO_2-eq.) was the best choice as compared to compost (81.7 kg CO_2-eq.), ethanol (115.7 kg CO_2-eq.), or incineration (366.8 kg CO_2 -eq.) options.

Incorporation of transportation of processed EFB (125 km) and the avoided substituted fertilizers (725 km) increased the impact by 33.2 kg CO_2-eq. for the mulch and 10.6 kg CO_2-eq. for the compost options. These transportation-related burdens are presented in Figure 6.3 as dashed boxes placed on top of the black boxes. The effect of the avoided process of producing substituted fertilizers (103.9 kg CO_2-eq. and 188.3 kg CO_2-eq. for mulch and compost, respectively) was more dominant than transportation. A sensitivity analysis for different processes of substituted fertilizers and different transport distances appears to be necessary in these types of closed loop applications. Such analysis, however, was not included in the current study.

The conclusion on mulch as the best option needs further verification since we assumed that all parts of the EFB were available as soil nutrient. In fact, due to the nature of EFB which is wet and bulky, some parts would undergo anaerobic degradation which emits methane, a strong GHG. Naturally, aerobic oxidation would also take place. In mulch application with one EFB layer, an anaerobic process may be negligible, but in thicker piles, the methane emission could be significant. These aerobic and anaerobic emissions would obviously reduce the amount of nutrients entering the soil matrix and thus reduce the amount of the substituted synthetic fertilizers. In general, the impacts of mulch and compost on soil fertility and field emissions involve complex processes which are not well characterized. Additionally, the processes depend on a number of site-specific conditions. All of these factors potentially increase uncertainty of the final results.

In practice, there are other more influential factors determining the decisions. For example, a company that we visited in Sumatera informed us that, when applying EFB on commercial plantation fields, the total distance is usually within 10 km. This criterion to limit transport distances for EFB field application was primarily based on economic consideration rather than environmental assessment. However, this situation could serve as a basis for the company to define which portion of EFB may be available for ethanol conversion.

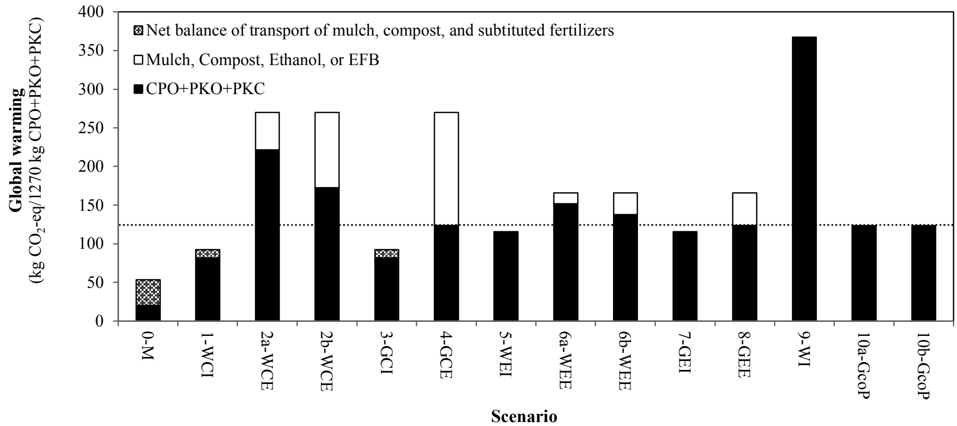

Figure 6.3. Global warming performances of different scenarios. Dashed line is the reference case (EFB treatments were not included in the inventory) with an impact score of 123.6 kg CO_2-eq/1270 kg CPO+PKO+PKC. Emissions from the transportation of the mulch (0–M) and the compost (1–WCI or 3–GCI) are 33.2 and 10.6 kg CO_2-eq/1270 kg CPO+PKO+PKC, respectively. All others are based on data in Table 6.3.

The process of producing compost (146.4 kg CO_2-eq.) had a much greater impact than producing ethanol (42.2 kg CO_2-eq.). The explanation is related to the choice of using an open windrow process which emitted GHG from composting piles directly to the atmosphere. However, this highly burdened process of producing compost was compensated by the avoided process of producing substituted fertilizers. As a result, the overall performance of the compost was better than the ethanol options. The incineration scenario was the worst case because fresh EFB contained excessive amount (60 %) of moisture which is required to be first evaporated to only 20 %. This prior drying step was discovered to be the major contributor (237.1 kg CO_2-eq.) to the incineration option. In practice, EFB is normally not dried beforehand. Prior drying was modeled only for the purpose of estimating the emissions of incinerating such wet EFB. Theoretically, this approach would give less emission than direct incineration (without drying). In this closed loop system (scenarios 0, 1, 3, 5, and 7 for mulch, compost, and ethanol), the status of EFB, as waste or goods, had no effect on the final results.

Besides functional units, technological choices and assumptions related to the inventory could as well strongly influence the final results. Functional units are parts of methodological choices, while technological choices and other assumptions are rather arbitrary, depending on the scope of the study. Difference in final results is possible if the same comparison studies used different methodological choices, technological choices, or assumptions. For example, the conclusion on mulch as the best option in this paper is different from Hansen et al. (2012) who suggested pyrolysis products as a better option. Since all aspects in our study have been transparently presented, we believe that the conclusion is valid within the context of LCA methodology. The relative importance of functional unit, technological choices, and assumptions to the final results could be explored further by performing sensitivity analysis. However, such analysis is outside the scope of the current study.

Comparison of LCA results cannot be made for scenarios 2, 4, 6, 8, and 10. The expanded functional units of these scenarios are CPO + PKO + PKC + additional coproducts (compost, ethanol, or EFB). These coproducts are employed externally, and any knowledge regarding their specific utilization by other parties is unknown. Therefore, substitution mechanism, as in the case of internal uses, could not be performed. Instead, these coproducts with their embedded emissions entered other product systems that are external to palm oil systems. Selling the EFB as coproducts to an external ethanol plant or converting the EFB internally, for example, would exhibit the same impact provided that the same technology is used.

In scenarios 2 and 6, the status of EFB as waste strongly influences the final LCA results. This is because the environmental burden was divided between the upstream and downstream links. Partitioning also applied to the coproduction cases (scenario 10), but the effect of EFB as coproducts was so minimal that it cannot be ascertained in Fig. 3. This is because the values of EFB were much less than the prices of the main palm oil products (CPO, PKO, and PKC). If in the future the price of EFB increases, the effect of this coproduct to the palm oil system will increase accordingly.

The above comparative analysis was by no means complete. For example, the inventory did not include transportation of ethanol from a distillery to gas station and its emissions on use. Also, the plantation phase might use imported fertilizers thereby increasing transport distances. The mulch and compost substituted synthetic fertilizers based on equivalent fertilizing values, which is quite a simplistic approach. It might not accurately consider carbon- and nitrogen-based GHG emissions on field, the difference in nitrogen emissions between organic and mineral fertilizers, the role of organic fertilizers on soil structure, biodiversity, and long-term soil fertility. However, the fertilizer equivalent may be the only easily implementable approach available at the present time. In the context of time and location, the palm oil inventory represented Malaysian averages for 2002–2006, while the EFB processes were primarily European cases. Further studies utilizing a more site-specific data would reduce some uncertainty and better illustrate the applicability of the proposed guidelines. However, we think that the presented analysis is sufficient to illustrate how comparison among different scenarios was performed.

6.4 Conclusions

Comparison between LCA results based on the same multifunctional units can be conducted only in the cases where additional coproducts were employed internally. In this closed loop system, the status of EFB as waste or goods has no effect on the final results. Based on the global warming impact, the mulch option was preferred as compared to the compost, ethanol, or incineration options. This preference, however, needs further verification since we assumed that all parts of the EFB were available as soil nutrient; in fact, some parts would undergo aerobic and anaerobic degradation, emitting GHGs to the atmosphere. The effect of the avoided process of producing synthetic fertilizers also dominated the final result. If used externally, the coproducts with known burden characteristics will become a component of another product system that is external to the palm oil system. In this regard, the status of EFB as waste strongly influences the final LCA results due to burden partitioning between the function to reduce the pollution strength of waste and the function to create products. Comparison among external use scenarios requires further analysis incorporating additional information on specific uses of the coproducts by external parties.

The proposed guidelines provide methodological choices in terms of system boundary, functional unit, and solutions to multifunctional problems. The methods can be used to systematically compare LCA results of different treatment options and valuation of EFB. The preferred alternative for managing this biomass residue could improve environmental performances and orient toward best practices, such as those suggested by RSPO. Further studies incorporating a sitespecific case of palm oil systems would better illustrate the usefulness of the proposed guidelines.

6.5 Acknowledgements

Financial support from the Netherlands Fellowship Programs (NFP) for EIW is gratefully acknowledged. Appreciation is given to Indonesian Institute of Sciences (LIPI) for providing an opportunity for EIW to pursue a PhD study at Leiden University. The authors would like to thank the French National Research Agency (ANR) for the field trip support to Sumatera, Indonesia, within the frame of the SPOP project (http://spop.cirad.fr/) Agrobiosphere program. We also would like to thank Lauran van Oers of CML, Leiden University, for assisting with the CMLCA software and Bayuaji Kencana for useful information on palm oil biomass residues. The comments and suggestions from two anonymous reviewers are highly appreciated.

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Electronic Supplementary Material

Methodological issues in comparative life cycle assessment: treatment options for empty fruit bunches in a palm oil system

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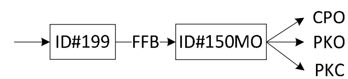
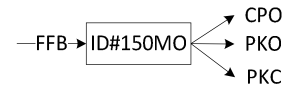
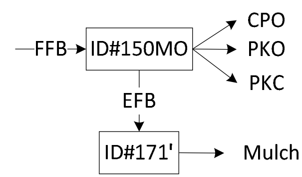
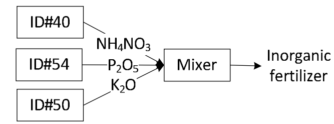
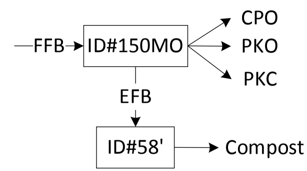
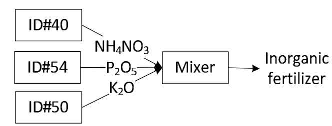
This electronic supplementary material (Online Resource) contains more information on system definition, assumptions, and detail calculation of the results presented in the manuscript.

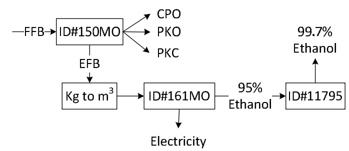
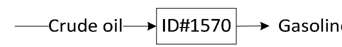
Table SM1. System definition[*)].

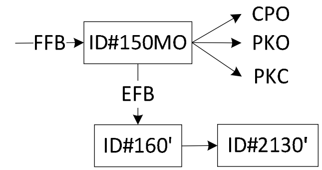
System component	Foreground processes (default UPR of Ecoinvent v2.2, 2010)			This study	
	Main processes	**Substituted processes**	**Allocation**	**Data from literature, assumptions, and calculation**	**Fragments of process flow diagram**
Palm oil	• Palm fruit bunches, at farm (ID#199: Malaysia, 2002-2006). Economic outflow = 1 kg fresh fruit bunches. (Trunk was internally recycled; CO_2 capture was modelled as an environmental inflow). • Palm fruit bunches, in oil mill (ID#150MO:	-	ID#150MO: Economic partitioning coefficients: CPO = 81.3%, PKO = 17.3%, PKC = 1.4%. (Economic values in 2006: CPO = RM 1.490/kg,	• Fresh fruit bunches of 1 kg resulted in 0.1488 kg fibers of 60% DM, 0.0696 kg shell of 90% DM, and 0.2266 kg EFB of 40% DM (Jungbluth *et al*, 2007). • At the production of 1000 kg CPO, the resulted fibers, shell, and EFB are 690 kg, 323 kg, and 1051 kg on wet basis, respectively; or 414 kg, 290.7 kg, and 420.4 kg on dry basis, respectively. • The impact of the default palm oil	• Cradle-to-gate boundary (plantation and oil mills)  • Gate-to-gate boundary (oil mills only)

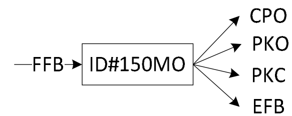
System component	Foreground processes (default UPR of Ecoinvent v2.2, 2010)			This study	
	Main processes	Substituted processes	Allocation	Data from literature, assumptions, and calculation	Fragments of process flow diagram
	Malaysia, 1995-2006). Economic outflows = 0.2156 kg CPO, 0.0266 kg PKO, and 0.0317 kg PKC. (Fiber, shell, and EFB were internally recycled; Treatment of POME was modelled as an economic inflow).		PKO = RM 2.565/kg, and PKC = RM 0.175/kg).	systems was corrected by excluding the contribution of EFB in energy production, a co-generation process as much as 21.1 kg CO_2-eq. • See Tables SM2 and SM3 for further detail.	
Mulch	• Mulching (ID#171: Switzerland, 1991-2002). (Services only, no material input for compost).	• Ammonium nitrate, as N, at regional storehouse (ID#40<006 484-52-2>: Europe, 1999). Economic outflow = 1 kg ammonium nitrate. • Single superphosphate, as P_2O_5,		• Land application as mulch would require approximately 30 metric ton fresh EFB per hectare (Haron, 2013). Therefore, 1051 kg EFB resulted in 0.035 ha mulch. The economic outputs of the expanded system were 1270 kg CPO+PKO+PKC and 1051 kg or 0.035 ha EFB mulch. • The fertilizing values of EFB mulch were adopted from Haron (2013), i.e. 0.8% N, 0.22% P_2O_5, and 2.9% K_2O fertilizer on dry basis.	ID#171' is a modified process with fresh EFB as an inflow and EFB mulch as an outflow.  Substituted processes (1051 kg EFB mulch to replace 34.32 kg inorganic fertilizers)

System component	Foreground processes (default UPR of Ecoinvent v2.2, 2010)			This study	
	Main processes	Substituted processes	Allocation	Data from literature, assumptions, and calculation	Fragments of process flow diagram
		at regional storehouse (ID# 54: Europe, 1999). Economic outflow = 1 kg superphosphate. • Potassium chloride, as K_2O, at regional storehouse (ID#50<007 447-40-7>: Europe, 2000). Economic outflow = 1 kg potassium chloride.		• Based on the above unit processes, the mulch was equivalent to 9.61 kg ammonium nitrate, 4.40 kg superphosphate, and 20.32 kg potassium chloride. The production of the above amount of inorganic fertilizers emitted 103.9 kg CO_2-eq. Therefore, internal utilization of 1051 kg or 0.035 ha EFB mulch will avoid global warming impact as much as 103.9 kg CO_2-eq. • See Tables SM6a and SM7 for further detail.	

System component	Foreground processes (default UPR of Ecoinvent v2.2, 2010)			This study	
	Main processes	Substituted processes	Allocation	Data from literature, assumptions, and calculation	Fragments of process flow diagram
Com-post	• Compost, at plant (ID#58: Switzerland, 1999). Economic outflow = 1 kg compost. (Services only, no material input).	• (ID#40<006 484-52-2>: Europe, 1999). • (ID# 54: Europe, 1999). • (ID#50<007 447-40-7>: Europe, 2000). The same as above	-	• Chiew and Shimada (2013) suggested that 2600 kg of fresh EFB resulted in 1000 kg compost with fertilizing values of 2.2% N, 1.28% P, 2.79% K on dry basis. • In the inventory, 1051 kg EFB of 40% dry matter was converted to 404.2 kg compost of 50% dry matter. As a result, the economic outputs of the expanded system were 1270 kg CPO+PKO+PKC+ 404.2 kg compost. • Based on the above unit processes, the compost was equivalent to 12.70 kg ammonium nitrate, 28.21 kg superphosphate, and 11.33 kg potassium chloride. • The production of the above amount of inorganic fertilizers emitted 188.3 kg CO_2-eq. Therefore, internal utilization of 404.2 kg EFB compost will avoid global warming impact as much as 188.3 kg CO_2-eq. • See Tables SM6b and SM7 for further detail.	ID#58' is a modified process with fresh EFB as an inflow and EFB compost as an outflow.  Substituted processes (404.2 kg EFB compost to replace 52.24 kg inorganic fertilizers): 
Etha-nol	• Wood, in distillery (ID#161MO:	• Petrol, unleaded, at	ID#161MO: Economic	• Bulk density of hardwood chips (u=80% or 44.4% water content)	Hardwood chips as default feedstock for ethanol production

System component	Foreground processes (default UPR of Ecoinvent v2.2, 2010)			This study	
	Main processes	Substituted processes	Allocation	Data from literature, assumptions, and calculation	Fragments of process flow diagram
	Switzerland, 1999-2006). Economic outflows = 0.144 kg 95% ethanol, electricity = 0.00649 kWh. (Input material was 0.00232 m^3 hardwood chips u=80% or 55.6% DM). • Ethanol, 99.7% in H_2O, from wood, at distillation (ID#11795: Sweden, 2000-2008). Economic outflow = 1 kg 99.7% ethanol. (Input material was 1 kg 95% ethanol).	refinery (ID#1570: Switzerland, 1980-2000). Economic outflow = 1 kg petrol. (Petrol = gasoline)	partitioning coefficients: 95% ethanol = 99.7%; electricity = 0.3%.	was 556 kg DM/ton or 2.325 m^3 chips (Jungbluth et al, 2007). • Solid content of 0.00232 m^3 hardwood chips was equal to 0.55448 kg dry EFB. This amount of solid was converted to 0.144 kg of 95% ethanol, and further to 0.144 kg of 99.7% ethanol. Or, 420.4 kg dry EFB resulted in 109.3 kg of 99.7% ethanol. • As a result, the economic outputs of the expanded system were 1270 kg CPO+PKO+PKC+ 109.3 kg of 99.7% ethanol • Assumed that inputs, ethanol product, and emissions resulted from 1 kg dry EFB were equal to those from 1 kg dry softwood chips. • The impact of the default ethanol process (57.1 kg CO_2-eq) was corrected by subtracting the impact of transporting feedstock as much as 14.9 kg CO_2-eq. • Energy content of ethanol and gasoline were 31 MJ/kg and 46 MJ/kg, respectively (Chiew and Shimada, 2013). Energy content	were replaced by EFB.  Substituted processes (109.3 kg 99.7% ethanol from EFB to replace 73.66 kg gasoline): 

System component	Foreground processes (default UPR of Ecoinvent v2.2, 2010)			This study	
	Main processes	Substituted processes	Allocation	Data from literature, assumptions, and calculation	Fragments of process flow diagram
				of 109.3 kg of 99.7% ethanol was the same as that of 73.66 kg gasoline. • The production of the above amount of gasoline emitted 50.1 kg CO_2-eq. Therefore, internal utilization of 109.3 kg 99.7% ethanol will avoid global warming impact as much as 50.1 kg CO_2-eq. • See Table SM4 for further detail.	
Incinerator	• Grass drying (ID#160: Switzerland, 1985-2002). Economic outflow = 1 kg water evaporated. (Services only, no grass input). • Disposal, wood untreated, 20% water, to municipal incineration (D#2130: Switzerland, 1994-2000). Economic outflow = disposal of 1 kg wood waste.	-	<u>ID#2130</u>: No allocation applied for electricity and heat. All burdens were charged to the waste input.	• Solid content of 0.2266 kg EFB of 40% DM was equal to 0.1133 kg wood of 80% DM, i.e. 0.09064 kg. • Assumed that inputs and emissions resulted from 1 kg dry EFB were equal to those of 1 kg dry wood. • The EFB process consisted of ID#160’ (evaporation of 525.5 kg water) and ID#2130’ (incineration of 525.5 kg wood of 80% DM). • See Table SM5 for further detail.	 ID#160’ is a modified process with EFB of 40% DM as an inflow and EFB of 80% DM as an outflow. ID#2130’ is a modified process with EFB of 80% DM as an inflow.

System component	Foreground processes (default UPR of Ecoinvent v2.2, 2010)			This study	
	Main processes	Substituted processes	Allocation	Data from literature, assumptions, and calculation	Fragments of process flow diagram
	(Services only, no wood input).				
EFB as co-products	• Palm fruit bunches, in oil mill (ID#150MO: Malaysia, 1995-2006). Economic outflow = 0.2156 kg CPO, 0.0266 kg PKO, 0.0317 kg PKC, and 0.2266 kg EFB at 40% DM.	-	ID#150MO: The same as above.	Market values of CPO and EFB as co-products: • FOB price of CPO at oil mills was IDR 9000/kg (GAPKI website, 2013). • FOB prices of EFB at oil mills were IDR 20/kg and IDR 50/kg. Often, it was free (Anonymous, field survey in Northern Sumatera, July 2011).	

Notes:
1) Ecoinvent processes were based on single-output and multi-output UPR (unit process raw) with infra-structure databases.
2) Multi-output processes (ID#MO150 and ID#MO161) were added to the single-output process database, and the corresponding single-output processes were removed.
3) ID#MO150 (palm oil) and ID#MO161 (ethanol 95%) are ID numbers of the multi-output UPR database, while the rest are ID numbers of the single-output UPR database.
4) ID#171' (mulch), ID#58' (compost), ID#160' (drying), and ID#2130' (incineration) are modified version of the Ecoinvent processes in terms of input-output flows for the purpose of easy to understand presentation of the process diagram.
5) IDR = Indonesian Rupiah; RM = Malaysian Ringgit; DM = dry matter; POME = palm oil mill effluent.

Table SM2. Contribution analysis on the global warming performances of a palm oil system[a].

Processes	System boundary	
	Cradle to gate[b]	Gate to gate[c]
Provision of land	43%	0
Plantation operation	36%	0
Lorry operation	4%	54%
Oil mill operation	< 0.5%	7%

[a]EFB together with fibers and shell were burned in a co-generation unit, and the resulted electricity and heat were used internally.
[b]Production of FFB (plantation) and CPO+PKO+PKC (oil mills). For the production of 1000 kg CPO, 123 kg PKO, and 147 kg PKC, the global warming performances were 1681.9 kg CO_2-eq, 357.9 kg CO_2-eq, and 29.0 kg CO_2-eq, respectively. (2067.8 kg CO_2-eq/1270 kg CPO+PKO+PKC). These figures do not add up due to round off.
[c]Production of CPO+PKO+PKC only (plantation stage was cut off). For the production of 1000 kg CPO, 123 kg PKO, and 147 kg PKC, the global warming performances were 117.7 kg CO_2-eq, 25.0 kg CO_2-eq, and 2.0 kg CO_2-eq, respectively. (144.7 kg CO_2-eq/1270 kg CPO+PKO+PKC). These figures do not add up due to round off.

Table SM 3. Co-generation[a] of fiber, shell, and EFB to produce heat and electricity.

Raw materials (at 1000 kg produced CPO)	Wet weight (kg)	Dry weight (%)	Dry weight (kg)	Dry weight Ratio (%)	Global warming impact[b] (kg CO_2-eq)
Fibers	690	60	414.0	36.8	20.8
Shell	323	90	290.5	25.8	14.6
EFB	1051	40	420.4	37.4	21.1
Total				100	56.4

[a]Unit process = wood chips, burned in cogen 6400kWth, allocation energy (ID#79MO: Switzerland, 2000-2001). In this inventory, partitioning coefficients based on energy content for electricity and heat were 9.7% and 90.3%, respectively. These figures do not add up due to round off.
[b]Bulk density of wood chips = 188.6 kg dry matter/m^3 (wood chips, mixed, from industry, u=40%; ID# 2353: Europe, 2002); Emissions were assumed proportional to dry matter. The corrected global warming impact of the default palm oil system by excluding the contribution of EFB. Cradle-to-gate boundary = 2067.8 – 21.1 = 2046.7 kg CO_2-eq; Gate-to-gate boundary = 144.7 – 21.1 = 123.6 kg CO_2-eq.

Table SM4. Transportation of hardwood chips from forest to distillery.

Transported materials	Weight (kg)	Transport[a] (km)		Global warming impact (kg CO_2-eq)
		Tractor and trailer	Lorry	
Hardwood chips	1051	5	65	14.9

[a]Transport, tractor and trailer (ID#188: Switzerland, 1991-2002); Transport, lorry 20-28 metric ton, fleet average (ID#1942: Switzerland, 2005). The corrected global warming impact of the default ethanol process (excluding the contribution of feedstock transportation) = 57.1 – 14.9 = 42.2 kg CO_2-eq.

Table SM5 Incineration of EFB

Unit processes	EFB input	Evaporated water	Global warming impact
Grass drying	1051 kg at 60% water	525.5 kg	237.1 kg CO_2-eq
Incineration	525.5 kg at 20% water	-	6.2 kg CO_2-eq

Table SM6. Substituted inorganic fertilizers

(a) for mulch

Fertilizer substitutes for mulch	Active compoundsbased on Ecoinvent LCIs[a] (%)			Weight (kg)[b]	
	N	P_2O_5	K_2O	Substituted mulch (dry matter)	Inorganic fertilizers
Ammonium nitrate	35			420.4	9.61
Superphosphate		21		420.4	4.40
Potassium chloride			60	420.4	20.32

Total weight, kg	34.32
Impact of producing fertilizers, kg CO_2-eq	103.9

[a]The fertilizing value of EFB mulch was equivalent to 0.8% N, 0.22% P_2O_5, and 2.9% K_2O fertilizer (Haron, 2013); 0.79% N, 0.23% P_2O_5, and 2.80% K_2O fertilizer based on dry basis (Caliman et al, 2013).

[b]Based on molecular weight, N in NH_4NO_3 is 35%, P in P_2O_5 is 43.66%, and K in K_2O is 82.98%. 420.4 kg dry mulch = 1051 kg fresh EFB x 40% dry matter.

(b) for compost

Fertilizer substitutes for compost	Active compounds based on Ecoinvent LCIs (%)			Weight (kg)	
	N	P_2O_5	K_2O	Substituted compost[c] (dry matter)	Inorganic fertilizers
Ammonium nitrate	35			202.1	12.70
Superphosphate		21		202.1	28.21
Potassium chloride			60	202.1	11.33

Total weight, kg	52.24
Impact of producing fertilizers, kg CO_2-eq	188.3

[c]Fresh EFB contained 40% DM (Jungbluth et al, 2007); Compost product had 50% DM (Nemecek and Kägi, 2007); Chiew and Shimada (2013): 2600 kg EFB was converted to 1000 kg EFB compost; EFB compost has fertilizing values equivalent to 2.2% N, 1.28% P, and 2.79% K on dry basis.

Table SM7. Transportation of compost, mulch, and the substituted inorganic fertilizers[a].

Transported materials	Weight (kg)	Transport (km)			Global warming impact (kg CO_2-eq)
		Tractor and trailer	Lorry	Rail	
Compost	404.2	25	100	0	13.6
Mulch	1051	25	100	0	35.2
Inorganic fertilizers (compost substitute)	34.3	25	100	600	2.0
Inorganic fertilizers (mulch substitute)	52.2	25	100	600	3.0

[a]Transport, tractor and trailer (ID#188: Switzerland, 1991-2002); Transport, lorry 3.5-16 metric ton, fleet average (ID#1941: Europe, 2005); Transport, freight, rail (ID#1983: Europe, 2000). Fresh EFB contained 40% DM (Jungbluth et al, 2007); Compost product contained 50% DM (Nemecek and Kägi, 2007).

Additional transportation factors for the compost option = 13.6 – 3.0 = 10.6 kg CO_2-eq.

Additional transportation factors for the mulch option = 35.2 – 2.0 = 33.2 kg CO_2-eq.

Chapter 7

Answers to the research questions

7.1 Answers to the research questions

As indicated in Chapter 1, the objectives of this thesis are to identify the key issues when conducting an LCA of residue-based bioenergy and to propose improvement in LCA procedures, specifically in the areas of LCI, LCIA, and methodological choices in the comparative LCA. It was discovered that the key issues associated with residue-based bioenergy relate to four primary characteristics with respect to its raw materials (biomass residues), i.e., excessive removal from plantation fields which can affect soil fertility; valuation (relative to biomass products); competing uses (bioenergy, feed, fiber, fertilizers); and treatment of biogenic carbon (assumptions of carbon neutrality vs. complete inventory). These unique features require specific LCA approaches which vary from those of conventional product-based bioenergy.

The thesis is structured in the following manner. The five research questions are discussed and answered in the preceding Chapters 2, 3, 4, 5, and 6, respectively, and are further summarized in this section. Chapters 2 and 3 are not discussed further since the issues on the LCA of bioenergy, in general, and second generation bioethanol have been thoroughly presented in the respective chapters. Meanwhile, the proposed approaches that are analyzed in Chapters 4, 5, and 6 are further discussed in terms of potential application and additional required research in Sections 7.2 and 7.3. Overall, the thesis proposes the following approaches: solutions to the existing dissimilar practices in the LCI of biogenic carbon (Chapter 4); an LCIA method of removing biomass residues from soil on biomass productivity (Chapter 5); and methodological choices in comparative LCA of biomass residues utilization (Chapter 6). The answers to the research questions are listed below.

7.1.1 What are the key issues in conducting an LCA of bioenergy systems?

Key issues in conducting an LCA of bioenergy systems include the followings. Goal and scope definition is an important initial step since the choice of methodology used depends on the purpose of an individual study. These methodological choices include system boundary, functional unit, inventory data, and treatment of multi-functional processes (allocation). Within this category, there is also a debate between attributional and consequential LCA which, for bioenergy, is even more crucial than for ordinary products.

Bioenergy poses more methodological challenges than other types of renewable energy since the system boundary also incorporates agricultural phases to produce biomass feedstock. In this aspect, there is a question of how to properly treat biogenic carbon in terms of capture and emission flows in the inventory. In addition, agricultural processes lie in the intersection between environmental and economic systems. Meanwhile, LCA methodology has not been particularly unambiguous with regard to direct and indirect land-use changes, regional variability leading to regionalized impact assessment, and the choice of impact categories. These are challenges for further development of an LCA in order to improve analyzing a bioenergy system.

7.1.2 What is the environmental sustainability status of second generation bioethanol?

Despite the non-homogeneous system definition of the reviewed LCA studies, it was concluded that second generation bioethanol exhibits improved performances over fossil fuel for, at least, the two most studied sustainability aspects, i.e., net energy output and global warming. For the latter parameter, carbon sequestration at the biomass generation stage can consistently offset total GHG emissions from all elements of the life cycle chains at high ethanol percentage (≥85%).

The aspects of biogenic carbon and agrochemical input for energy crops and biomass residues and the effect of the removal of the latter from soil have not been addressed consistently. The bioethanol conversion process is primarily based on simultaneous saccharification and co-fermentation which is characterized by high yield and low energy input. In this regard, the LCA results tend to under estimate the actual impacts of current technology. The choice of allocation methods strongly influences the final results, particularly when economic value is employed as a reference. Substitution of avoided burden appears to be the most popular allocation method in practice, followed by partition based on mass, energy, and economic values.

It is believed that residue-based bioenergy will play more important roles in the global energy mix. Future exploitation of biomass residues is speculated to be driven mainly by emerging technology for second-generation bioethanol. This is a natural consequence of strive competition for the same raw materials (sugar, starch, and vegetable oil) between food and energy sectors.

7.1.3 How much would the final results of biogenic carbon neutrality assumptions deviate from the true values based on a complete inventory?

This study demonstrates that the global warming impacts of scenarios based on the LCA principles, treating all relevant input-outputs as genuine flows (including biogenic carbon), provides various results which are in many cases different from those assuming biogenic carbon neutrality (excluding biogenic carbon). In addition to the existing arguments of carbon stock change and carbon storage capacities of long-rotation trees or wood products, the current study supports the assertion that the assumption on biogenic carbon neutrality introduces biases to the ‘true’ values based on a complete inventory. The extent of such a bias, which can make the values of global warming impacts substantially higher or lower, depends on specific situations. Three factors are identified to substantially propagate biases in the evaluation of global warming performances of a bioenergy system. They are differences in framing system boundaries (cradle to grave vs. cradle to gate), forms of carbon emissions (CO_2 vs. CH_4), and valuation of biogenic carbon (biomass products vs. biomass residues). The results of this study are expected to contribute to the harmonization of future bioenergy LCA by directing further research to adopt more the concept of utilizing a complete inventory rather than the neutrality assumption.

7.1.4 How to assess the impact of removing biomass residues from soil on biomass productivity?

An impact assessment method relating the removal of biomass residues with soil organic carbon and net primary production has been developed. The assessment method was developed within the LCA framework based on a specific system definition, i.e., decomposed biomass residues above soil surfaces resulted in net carbon flow into soil compartments. In addition to GHG emissions into the atmosphere, the soil organic carbon functions as an additional elementary flow to complete the overall carbon balance. The assessment method considers the effects of soil organic carbon on soil biomass productivity as a midpoint indicator, which is expressed as a loss of net primary production. The impact assessment method is in accordance with the ISO-standard format, comprising a characterization factor and an input term representing changes in elementary flows:

$$\left(\Delta \mathrm{NPP}\right)_{\%\mathrm{Removal}} = \left(\frac{\mathrm{dNPP}}{\mathrm{dSOC}}\right)_{\mathrm{Climate,\,Vegetation,\,Soil}} \times \frac{\mathrm{dSOC}}{\mathrm{d\%Removal}} \times \%\mathrm{Removal}.$$

The operation of the proposed method was illustrated with a moderate case study. It demonstrated that the method has genuine potential to be developed as metrics for assessing the sustainability of bioenergy systems. However, to operate the assessment model for a specific region, data for spatially detailed implementation within bioenergy systems in various regions need to be completed.

7.1.5 How to compare the impact of various treatment options on biomass residues in a palm oil system?

Guidelines for methodological choices in terms of system boundary, functional unit, and solutions to multi-functional problems have been proposed. The methods can be employed to systematically compare the LCA results of different treatment options and valuation of EFB in a palm oil system. Comparison based on the same multi-functional units can be accomplished only in cases where additional co-products were utilized internally. Based on a global warming impact, the mulch option was preferred among other treatments (compost, ethanol, treated in incinerator, and sold as coproducts). This conclusion requires further verification since results are extensively dependent on local inventory data. The effect of the avoided process of producing synthetic fertilizers and the assumption that all parts of mulch are available as soil nutrient dominantly determined the final result. The study also demonstrates that the status of EFB as waste or goods is influential on the final results if the EFB is employed externally but has no effect if it is utilized internally.

Knowledge regarding preferred alternatives for managing biomass residues like EFB could improve environmental performances and orient towards best practices. Further studies incorporating a site-specific case of palm oil systems would better illustrate the usefulness of the proposed guidelines. Besides palm oil systems, the approach is also readily applicable to biomass residues in other agro-based industrial systems.

7.2 Potential applications of the proposed approaches

Although LCA methods in general have already significantly advanced for describing the sustainability of bioenergy systems, there are still additional questions to be answered for residue-based bioenergy. The overall framework of the thesis was designed to develop an approach to compare the impacts between leaving biomass residues in fields or utilizing them for bioenergy. It also offers an approach to distinguish methods of handling biomass residues differently from biomass products in bioenergy systems. The potential applications of the proposed approaches encompass certain aspects of environmental sustainability assessment as discussed below.

The primary contribution of Chapter 4 is the re-evaluation of the widely applied biogenic carbon neutrality assumption in scientific literature. In addition to the existing arguments of carbon stock change (Searchinger et al., 2008) and carbon storage capacities of long-rotation trees (Cherubini et al., 2011) or wood products (Levasseur et al., 2012), the current study supports the assertion that the assumption on biogenic carbon neutrality introduces biases to the ‘true’ values based on a complete inventory. Therefore, the recommendation to the existing dissimilar practices with biogenics is to treat carbon flows as genuine flows, i.e., not assuming carbon neutrality. In this respect, all input-output flows based on the entire life cycle of a product system are to be included in the inventory. The results of this study are expected to contribute to a more harmonized bioenergy LCA in the future. Considering its broad spectrum of implementation in the field of bioenergy, the potential users of this proposed approach include LCA researchers, practitioners, and policy makers.

Previous studies regarding the removal of biomass residues have focused only on the impact of GHG emissions into the atmosphere. To complete the carbon balances, more recent development also considered the effect of carbon flows into the soil matrix, i.e., an integrated assessment of impacts on global warming and SOC (Liska et al., 2014; Mckechnie et al., 2011; Repo et al., 2011, 2012; Guest et al., 2013). Additionally, Milà i Canals et al. (2007) and Brandão and Milà i Canals (2013) have further addressed the relationship of SOC and biomass productivity. Likewise, this thesis also models the effect of removing biomass residues on the flow of organic carbon into the soil matrix which alters the SOC content and subsequently leads to the loss of biomass productivity. Compared to the previous studies, the novelty of the approach is that the impact is expressed in terms of a characterization factor and an input term representing changes in elementary flows. Although the method in Chapter 5 has been developed for bioenergy systems, it is also appropriate for other product systems involving the removal of biomass residues from soil since the characterization method provides a general framework of converting elementary flows (soil organic carbon, SOC) into an impact (net primary production, NPP). Considering that the method is formulated in a somewhat generic form, perhaps it can provide direction to LCA researchers and developers for further development.

To be able to properly compare and interpret the results of different studies, an assessment framework to evaluate biomass residues in a specific agro-based industrial system is necessary. For example, previous LCA studies in a palm oil system indicate a broad range of global warming impacts from net positive (GHG emissions) to net negative values (GHG savings) (Stichnothe and Schuchardt, 2010; Lim and Lee, 2011; Hansen et al., 2012). These results, however, are not practically comparable since the scores were not based on the same functional units. In this regard, the proposal in Chapter 6 contributes to harmonizing the approaches for conducting LCAs of biomass residues in a palm oil system so that the results could be compared. The proposed methodological choices encompass system boundary, functional unit, and solutions to multi-functional problems. Although the approaches were illustrated for a palm oil system, they are also readily applicable for managing biomass residues in other agro-based industrial systems since biomass residues generated in agricultural phases or post-harvest processing units are considered as co-products which are all included in the inventory. These various co-products are considered as an economic output with different functions or values. Considering the practical nature of the proposal, the potential users of this approach are LCA practitioners and researchers.

7.3 Areas of future research

In spite of the potential application of the proposed approaches, additional research is still required to elaborate a number of embedded problems. One of the major issues is in regard to the primary function of biomass residues in agricultural fields. Naturally, their stock of nutrients provides a basic recycling mechanism that maintains soil fertility. They can also improve the structure of soil for enhanced aeration and water management. Excessive removal of biomass residues may degrade soil quality and also potentially influence future crop yields (Wilhelm et al., 2004; Lal, 2005; Cherubini et al., 2009). This could be a serious detriment to the long-term sustainability of residue-based bioenergy and should, therefore, be considered as a component of the sustainability assessment. One of the key solutions is to compensate the substantial nutrient export with inorganic fertilizers. However, this option may create further consequences on environmental and financial cost that must be carefully evaluated.

Bioenergy poses an increasing number of methodological challenges more so than other types of renewable energy since the system boundary also incorporates agricultural phases to produce biomass feedstock. The problem is that agricultural processes lie in the intersection between environmental and economic systems. Meanwhile, the existing LCA methodology has not been particularly unambiguous with regard to direct and indirect land-use changes, local variability leading to regionalized impact assessment, and the choice of impact categories. These are challenges for further development of the LCA in order to better analyze a bioenergy system.

Required research related to additional development of the proposed approaches is described below. The results presented in Chapter 4 require proper interpretation since they are based on certain hypothetical inventory data. Further analysis exploiting actual inventory data, for example, those available in literature, may be necessary to further verify if the assertion for the non-neutrality assumption of biogenic carbon is also valid for other inventory situations.

The primary issue in the implementation of the proposed method in Chapter 5 is locating specific data on bioenergy systems that demonstrate the relationship between biomass removal at different rates on the states of SOC and NPP. Data unavailability is the main reason why the illustrated example did not utilize data from a bioenergy system but, instead, from a grazing system. Additional extensive research is necessary to accordingly determine the effect on future biomass productivity from removing biomass residues from various regions. These data are necessary for developing specific characterization factors for impact assessment in certain regions. This effort would be even more relevant in the near future considering the substantial advancement in the research and development on second generation bioethanol.

The conclusion in Chapter 6 regarding mulch being the most appropriate treatment option requires further verification since the study was based on the assumption that all parts of the EFB were available as soil nutrient. In reality, some elements would experience aerobic and anaerobic degradation and emit GHGs into the atmosphere. Therefore, proper mass balances on biomass decomposition processes in fields which include carbon flows into the atmosphere and soil matrix are required. In addition, the inventory models were based on processes available in the Ecoinvent database. Further studies incorporating a site-specific case of palm oil systems would better illustrate the usefulness of the proposed guidelines.

7.4 Concluding remarks

An optimistic estimate suggested that, by 2050, bioenergy could sustainably contribute up to one third of the global primary energy supply from the current state of approximately only ten percent (IEA, 2009). More specifically, residue-based bioenergy is speculated to increase its key role in the future energy mix as a consequence of the harsh competition between food and energy sectors over the same raw materials (sugar, starch, and vegetable oil) (IPCC, 2011). This competition has encouraged the exploitation of non-food resources which are so far still dominated by fuel wood for traditional cooking (IPCC, 2011). This fact demonstrates that there continues to be significant opportunities for improved biomass utilization for bioenergy that is more efficient and environmentally friendly. This trend would likely be enhanced by the advancement in research and development on emerging conversion technology such as second-generation bioethanol. In line with this trend, potential biomass residues as bioenergy feedstock are abundant and available in many different regions worldwide. The global production of just crop residue can reach 3.8 billion metric tons annually, which is equivalent to a total energy of 70 EJ (Lal, 2005). These resources, however, must be diligently managed since, in addition to bioenergy, biomass residues also play many other competing roles. In particular, residues generated in fields naturally contribute to maintaining soil fertility and future harvests. Meanwhile, those generated in post-harvest facilities may also be used as fertilizers, feed, fibers, or other biorefinery feedstocks. In this regard, the contribution of the thesis is to propose guidelines to be able to determine, among the various potential uses of biomass residues, which options would have the least environmental impacts.

An LCA can influence the future of residue-based bioenergy provided that the method is formulated accordingly, referring to the specificity of the feedstock. It is an environmental assessment tool for a product system which differentiates the status of an economic flow associated with a unit process as goods or waste. The problem is that many studies on the LCA of residue-based bioenergy have not clearly defined the economic status of the biomass feedstock. Perhaps this is due to the difficulty in deciding whether the price of a residue is positive or negative. Meanwhile, in theory, proper identification of an economic status of input flows to and output flows from a unit process is required as it will consequently influence how an LCA should be conducted. This thesis offers an approach for residue-based bioenergy to be managed accordingly that is different from that for product-based bioenergy.

Although a number of improvements have been suggested, sustainability of bioenergy cannot be comprehensively evaluated by only utilizing LCA. It requires other complimentary analyses because sustainability of bioenergy systems encompasses a wide variety of parameters including environmental, economic, and social aspects. Under these three pillars, GBEP (2011), for example, established twenty four sustainability indicators for bioenergy which are more than the number of impact indicators typically employed in LCA.

Another relevant development on the sustainability assessment of bioenergy is an effort to assess production and consumption aspects of a product system in an integrated manner. In that regard, Heijungs et al. (2014) proposed incorporating the concept of planetary boundaries (Rockström et al., 2009) in an LCA framework to appraise sustainability. This innovative approach, so called backcasting LCA (Guinée and Heijungs, 2011), assesses not only environmental impacts, as in a conventional LCA, but also specifies consumption levels of a product that are appropriate within the limits of environmental sustainability.

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Summary

An optimistic estimate suggested that, by 2050, bioenergy could sustainably contribute up to one third of the global primary energy supply from the current state of approximately only ten percent. More specifically, residue-based bioenergy is speculated to increase its key role in the future energy mix as a consequence of the harsh competition between food and energy sectors over the same raw materials (sugar, starch, and vegetable oil). This competition has encouraged the exploitation of non-food resources which are so far still dominated by fuel wood for traditional cooking. This fact demonstrates that there continues to be significant opportunities for improved utilization for bioenergy that is more efficient and environmentally friendly. This trend would likely be enhanced by the advancement in research and development on emerging conversion technology such as second-generation bioethanol. In line with this trend, potential biomass residues as bioenergy feedstock are abundant and available in many different regions worldwide. The global production of just crop residue can reach 3.8 billion metric tons annually, which is equivalent to a total energy of 70 EJ. These resources, however, must be diligently managed since, in addition to bioenergy, biomass residues also play many other competing roles. In particular, residues generated in fields naturally contribute to maintaining soil fertility and future harvests. Meanwhile, those generated in post-harvest facilities may also be used as fertilizers, feed, fibers, or other biorefinery feedstocks.

The overall framework of this thesis is to develop an approach to compare the impacts between leaving biomass residues in fields and utilizing them for bioenergy. In that regard, the objectives of this thesis are to identify the key issues when conducting a life cycle assessment (LCA) of residue-based bioenergy and to propose improvement in LCA procedures, specifically in the areas of life cycle inventory (LCI), life cycle impact assessment (LCIA), and methodological choices in the comparative LCA. It was discovered that the key issues associated with residue-based bioenergy relate to four primary characteristics with respect to its raw materials (biomass residues), i.e., excessive removal from plantation fields which can affect soil fertility; valuation (relative to biomass products); competing uses (bioenergy, feed, fiber, fertilizers); and treatment of biogenic carbon (assumptions of carbon neutrality vs. complete inventory). These unique features require specific LCA approaches which vary from those of conventional product-based bioenergy. These approaches include: [1] solutions to the existing dissimilar practices in the LCI of biogenic carbon; [2] an LCIA method of removing biomass residues from soil on biomass productivity; and [3] methodological choices in comparative LCA of biomass residues utilization.

To accomplish the above objectives, five research questions are discussed and answered in Chapters 2 to 6. In Chapter 2, the key issues in conducting an LCA of bioenergy systems are identified. Goal and scope definition is an important initial step since the choice of methodology used depends on the purpose of an individual study. These methodological choices include system boundary, functional unit, inventory data, and treatment of multi-functional processes (allocation). Within this category, there is also a debate between attributional and consequential LCA which, for bioenergy, is even more crucial than for ordinary products.

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categories. These are challenges for further development of an LCA in order to improve analyzing a bioenergy system.

In Chapter 3, the environmental sustainability status of second generation bioethanol is evaluated. Despite the non-homogeneous system definition of the reviewed LCA studies, it was concluded that second generation bioethanol exhibits improved performances over fossil fuel for, at least, the two most studied sustainability aspects, i.e., net energy output and global warming. For the latter parameter, carbon sequestration at the biomass generation stage can consistently offset total GHG emissions from all elements of the life cycle chains at high ethanol percentage (≥85%).

The aspects of biogenic carbon and agrochemical input for energy crops and biomass residues and the effect of the removal of the latter from soil have not been addressed consistently. The bioethanol conversion process is primarily based on simultaneous saccharification and co-fermentation which is characterized by high yield and low energy input. In this regard, the LCA results tend to under estimate the actual impacts of current technology. The choice of allocation methods strongly influences the final results, particularly when economic value is employed as a reference. Substitution of avoided burden appears to be the most popular allocation method in practice, followed by partition based on mass, energy, and economic values.

In Chapter 4, the assumption of biogenic carbon neutrality is evaluated. This study demonstrates that the global warming impacts of scenarios based on the LCA principles, treating all relevant input-outputs as genuine flows (including biogenic carbon), provides various results which are in many cases different from those assuming biogenic carbon neutrality (excluding biogenic carbon). In addition to the existing arguments of carbon stock change and carbon storage capacities of long-rotation trees or wood products, the current study supports the assertion that the assumption on biogenic carbon neutrality introduces biases to the 'true' values based on a complete inventory. The extent of such a bias, which can make the values of global warming impacts substantially higher or lower, depends on specific situations. Three factors are identified to substantially propagate biases in the evaluation of global warming performances of a bioenergy system. They are differences in framing system boundaries (cradle to grave vs. cradle to gate), forms of carbon emissions (CO_2 vs. CH_4), and valuation of biogenic carbon (biomass products vs. biomass residues). The results of this study are expected to contribute to the harmonization of future bioenergy LCA by directing further research to adopt more the concept of utilizing a complete inventory rather than the neutrality assumption.

In Chapter 5, the impact of removing biomass residues from soil on biomass productivity is modeled. An impact assessment method relating the removal of biomass residues with soil organic carbon and net primary production has been developed. The assessment method was developed within the LCA framework based on a specific system definition, i.e., decomposed biomass residues above soil surfaces resulted in net carbon flow into soil compartments. In addition to GHG emissions into the atmosphere, the soil organic carbon functions as an additional elementary flow to complete the overall carbon balance. The assessment method considers the effects of soil organic carbon on soil biomass productivity as a midpoint indicator, which is expressed as a loss of net primary production. The impact assessment method is in accordance with the ISO-standard format, comprising a characterization factor and an input term representing changes in elementary flows:

$$(\Delta \mathrm{NPP})_{\%\mathrm{Removal}} = \left(\frac{\mathrm{dNPP}}{\mathrm{dSOC}}\right)_{\mathrm{Climate,\,Vegetation,\,Soil}} \times \frac{\mathrm{dSOC}}{\mathrm{d\%Removal}} \times \%\mathrm{Removal}$$

where NPP is the net primary production in g-carbon/(m^2.year) and SOC is the soil organic carbon in g-carbon/m^2.

The operation of the proposed method was illustrated with a moderate case study. It demonstrated that the method has genuine potential to be developed as metrics for assessing the sustainability of bioenergy systems. However, to operate the assessment model for a specific region, data for spatially detailed implementation within bioenergy systems in various regions need to be completed.

In Chapter 6, the methodological choices on how to compare the impact of various treatment options on biomass residues in a palm oil system are proposed. Guidelines for methodological choices in terms of system boundary, functional unit, and solutions to multi-functional problems have been proposed. The methods can be employed to systematically compare the LCA results of different treatment options and valuation of empty fruit bunches (EFB) in a palm oil system. Comparison based on the same multi-functional units can be accomplished only in cases where additional co-products were utilized internally. Based on a global warming impact, the mulch option was preferred among other treatments (compost, ethanol, treated in incinerator, and sold as coproducts). This conclusion requires further verification since results are extensively dependent on local inventory data. The effect of the avoided process of producing synthetic fertilizers and the assumption that all parts of mulch are available as soil nutrient dominantly determined the final result. The study also demonstrates that the status of EFB as waste or goods is influential on the final results if the EFB is employed externally but has no effect if it is utilized internally.

Knowledge regarding preferred alternatives for managing biomass residues like EFB could improve environmental performances and orient towards best practices. Further studies incorporating a site-specific case of palm oil systems would better illustrate the usefulness of the proposed guidelines. Besides palm oil systems, the approach is also readily applicable to biomass residues in other agro-based industrial systems.

Samenvatting

Volgens optimistische schattingen zou rond 2050 bio-energie op duurzame wijze eenderde van de mondiale energieproductie kunnen verzorgen, tegenover de tien procent van dit moment. In het bijzonder wordt verwacht dat de rol van op residuen gebaseerde bio-energie zal toenemen, omdat voedsel en energie elkaar beconcurreren wat betreft grondstoffen (suiker, zetmeel en plantaardige oliën). Ten gevolge van deze competitie is het gebruik van niet voor voedsel geschikte grondstoffen, zoals stookhout voor de bereiding van eten, toegenomen. Deze ontwikkeling toont aan dat er nog steeds belangrijke redenen zijn om het gebruik van bio-energie op een efficiënte en milieuvriendelijke wijze verder te ontwikkelen. De genoemde trend wordt verder versterkt door de resultaten van onderzoek naar opkomende omzettingstechnologieën, zoals tweedegeneratie bio-ethanol. De potenties voor biomassaresiduen zijn groot omdat dit materiaal in overvloedaanwezig is in veel verschillende landen. De mondiale productie van alleen al gewasresiduen kan 3,8 miljard ton per jaar bedragen, wat energetisch neerkomt op 70 EJ. Deze grondstoffen moeten echter met zorg beheerd worden omdat ze naast een energetische rol ook andere rollen kunnen vervullen. In het bijzonder dragen dergelijke residuen op plantages bij aan de instandhouding van de bodemvruchtbaarheid en oogsten in latere jaren. Verder kunnen ze verwerkt worden tot kunstmest, voedsel, vezels, of andere bronnen voor bioraffinaderijen.

Het overkoepelende idee van deze dissertatie is de ontwikkeling van een methode ter vergelijking van de effecten van enerzijds het achterlaten van biomassaresiduen en anderzijds het gebruik ervan ten behoeve van bio-energie. Dientengevolge zijn de doelstellingen van dit proefschrift het voerkennen van de aandachtspunten bij het gebruik van levenscyclusanalyse (LCA) van op residuen gebaeerde bio-energie, en het voorstellen van verbeteringen in de procedures voor LCA, in het bijzonder van de onderdelen levenscyclusinventarisatie (LCI), de levenscycluseffectbeoordeling (*life cycle impact assessment*, LCIA), en de methodische keuzes in een vergelijkende LCA. Er blijken vier elementaire onderdelen van belang te zijn, te weten excessieve verwijdering van residuen van het land waardoor de bodemvruchtbaarheid aangetast wordt, de waardering van de residuen ten opzichte van de biomassaproducten, de concurrerende aanwending (bio-energie, voedsel, vezels, bemesting), en de behandeling van kooldoxide van biogene oorsprong (de aanname van koolstofneutraliteit versus een volledige koolstofboekhouding). Deze kenmerken vereisen een specifieke behandeling in LCA-methode die verschilt van de LCA-methoden voor gewone producten. De methodische aspecten betreffen: [1] een oplossing voor de diversiteit in methode om met biogeen koolstof om te gaan; [2] een LCIA-methode voor de effecten van het weghalen van biomassaresiduen op biomassaproductiviteit; en [3] methodische keuzen wat betreft de vergelijkende LCA van biomassagebruik.

Om deze doelen te bereiken zijn vijf onderzoeksvragen geformuleerd en beantwoord in de hoofdstukken 2 tot en met 6. In hoofdstuk 2 worden de kernaspecten geïdentificeerd. De doelbepaling is een belangrijke eerste stap, omdat de keuze van de te hanteren methode afhangt van de doelstelling van het onderzoek. De methodische keuzen betreffen de systeemgrens, de functionele eenheid, de inventarisatiegegevens, en de oplossing van het multifunctionaliteitsprobleem met behulp van toerekening. Onder deze noemer valt daarnaast de discussie over attributionele en consequentiële LCA, die voor bio-energie nog belangrijker is dan voor gewone producten.

Bio-energie stelt meer methodische uitdagingen dan andere energiebronnen omdat de systeemgrens bij landbouwprocessen een grote rol speelt. Hierbij speelt onder meer de vraag hoe om te gaan met biogeen koolstof wat betreft het vastleggen van kooldioxide en de uitstoot daarvan. Daarnaast bevinden landbouwprocessen zich op de grens van economie en milieu. De LCA-methode is verder nogal onduidelijk in het onderscheid tusen directe en indirecte veranderingen van het gebruik van land, de regionale variabiliteit die tot een

geregionaliseerde effectanalyse leidt, en de keuze van de effectcategorieën. Dit zijn allemaal uitdagingen voor de verdere ontwikkeling van LCA voor een betere analyse van bio-energiesystemen.

Hoofdstuk 3 onderzoekt de milieuduurzaamheid van tweedegeneratiebiobrandstoffen. Ondanks de verschillen in de systeemafbakening in de genalyseerde LCA-studies kan worden geconcludeerd dat tweedegeneratiebiobrandstoffen een milieuverbetering opleveren, in ieder geval wat betreft de twee meest onderzochte criteria, te weten de netto energieopbrengst en het broeikaseffect. Wat betreft dit laatste criterium blijkt dat koolstofvastlegging door biomassaprodictie de uitsoot van broeikasgassen kan reduceren bij hoge ethanolpercentages (≥85%).

De gevolgen van de berekeningswijze van biogeen koolstof en van de agrochemische grondstoffen zijn in de literatuur niet op consistente wijze geanalyseerd. De conversie van bio-ethanol is voornamelijk gebaseerd op simultane saccharificatie en co-fermentatie, die worden gekenmerkt door een hoge opbrengst en een laag energiegebruik. De LCA-resultaten geven hier een onderschatting van de echte milieueffecten. De keuze van de toerekeningsmethode bepaalt in sterke mate de eindresultaten, vooral wanneer de economische waarde gebruikt wordt. De substitutiemethode blijkt de meest gebruikte methode te zijn, gevolgd door de verdelingsmethode op basis van massa, energie en economische waarde.

In hoofdstuk 4 wordt de veronderstelling van koolstofneutraliteit onderzocht. Het blijkt dat de broeikaseffectscores bij inachtname van de LCA-principes waarbij een volledige boekhouding van alle in- en uitstromen wordt gevolgd resultaten geeft die sterk afwijken van die waarbij koolstofneutraliteit wordt aangenomen. In aanvulling op de gebruikelijke argumenten betreffende de koolstofvoorraadverandering en de koolstofopslag in langdurige bosbouw of houtproducten vindt deze studie aanwijzingen dat de aanname van neutraliteit van biogeen koolstof tot afwijkingen leidt van de ‘juiste’ waarde bij een complete analyse. De grootte van deze afwijkingen, die de resultaten aanzienlijk omhoog of omlaag laten gaan, verschilt van geval tot geval. Drie factoren spelen wat dat betreft een rol. Dit zijn de verschillen in de systeemgrenzen (wieg-tot-graf vs. wieg-tot-poort), de vorm van de koolstofemissie (CO_2 vs. CH_4), en de waardering van biogeen koolstof (product vs. residu). Verwacht wordt dat de resultaten van deze studie bij zullen dragen aan een nadere harmonisatie van toekomstige LCA-methodes voor bio-energie; dit door in toekomstige onderzoeken meer gebruik te maken van een volledige analyse in plaats van de aanname van koolstofneutraliteit.

In hoofdstuk 5 wordt het effect van de verwijdering van biomassaresiduen op biomassaproductiviteit gemodelleerd. Dit gebeurt door de ontwikkeling van een methode voor effectbepaling, die de verwijdering van biomassaresiduen relateert aan het organisch koolstof in de bodem en de netto primaire productie. Deze methode is ontwikkeld voor een specifieke LCA-systeemafbakening, namelijk een waarbij afgebroken biomassaresiduen boven het bodemoppervlak leiden tot een nettostroom van koolstof naar het bodemcompartiment. Het organische koolstof in de bodem is ook een element in de balans waarbij verder broeikasgasemissies naar de atmosfeer een rol spelen. De beoordelingsmethode beschouwt de effecten op organisch koolstof in de bodem als een zogeheten midpoint indicator, uitgedrukt in het verlies aan netto primaire productie. De methode is in overeenstemming met de ISO-normen, en bestaat uit een karakteriseringsfactor en een term die de milieu-ingreep vertegenwoordigt:

$$(\Delta \mathrm{NPP})_{\%\mathrm{Removal}} = \left(\frac{\mathrm{dNPP}}{\mathrm{dSOC}}\right)_{\mathrm{Climate,\,Vegetation\,,Soil}} \times \frac{\mathrm{dSOC}}{\mathrm{d\%Removal}} \times \%\mathrm{Removal}.$$

Hierbij stelt NPP de netto primaire productie voor in g-koolstof/(m^2.jaar), en SOC het organische koolstofgehalte in g-koolstof/m^2.

Het gebruik van de voorgestelde methode is geïllustreerd aan de hand van een beperkte case study. Hierbij bleek dat de methode goede mogelijkheden biedt om een indicator voor de beoordeling van de duurzaamheid bio-energiesystemen te ontwikkelen. Voor een algemeen gebruik in andere situaties zijn nadere regio-specifieke gegevens nodig.

In hoofdstuk 6 worden de effecten van een aantal verwerkingsmethodes van biomassaresiduen tot palmolie vergeleken, met de nadruk op de rol van de methodische keuzes. Richtlijnen voor dergelijke keuzes wat betreft systeemafbakening, functionele eenheid en de behandeling van het multifunctionaliteitsprobleem worden voorgesteld. Deze methodes kunnen worden gebruikt om op systematische wijze de LCA-resultaten van verschillende behandelingsmethodes en waarderingsscenarios van lege palmtrossen (*empty fruit bunches*, EFB) in een palmoliesysteem te onderzoeken. Een vergelijking op basis van dezelfde functionele eenheid is alleen mogelijk als de overige co-producten intern gebruikt worden. Op basis van het broeikaseffect is de verwerking tot bodembedekking te verkiezen boven de andere opties (verwerking tot compost, productie van ethanol, verwijdering in een afvalverbrander, verkoop als co-product). Omdat de resultaten sterk afhangen van de lokale gegevens is een nadere controle van dit resultaat echter noodzakelijk. Het eindresultaat wordt in sterke mate bepaald door het effect van het vermeden proces van kunstmestproductie en de aanname dat de bodembedekking geheel beschikbaar is als bemester. Het onderzoek laat ook zien dat de status van EFB als zijnde een goed of afval invloed heeft op de eindresultaten, maar alleen wanneer EFB extern gebruikt wordt en niet wanneer het gebruik intern is.

Kennis aangaande de beste alternatieven voor het beheer van biomassaresiduen zoals EFB kan de milieuprestaties verbeteren en de richtlijnen voor verantwoord gebruik aanscherpen. Een nadere studie waarbij plaatsspecifieke details wat betreft palmoliesystemen worden meegenomen kunnen de bruikbaarheid van de voorgestelde richtlijnen verder illustreren. Deze benadering is behalve voor palmoliesystemen ook bruikbaar voor biomassaresiduen in andere op landbouw gebaseerde industriële systemen.

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Curriculum vitae

Edi Iswanto Wiloso was born in Malang, Indonesia in 1962. He studied at the Senior High School 42 in Jakarta, Indonesia between 1976 and 1980. At the university levels, he got a bachelor degree from the Agroindustrial Technology Department, Bogor Agricultural University, Indonesia in 1984. Following his graduation, he has been employed as a researcher at the Research Center for Chemistry, Indonesian Institute of Sciences (Pusat Penelitian Kimia - LIPI) since 1985. During his employment, he pursued a master degree at the Chemical Engineering Department, University of Waterloo, Canada between 1989 and 1991. His research was about ‘anaerobic biodegradation of phenol by mixed cultures in an immobilized fixed bed reactor.’

Since early 2011, he has been working on his PhD at the Institute of Environmental Sciences (CML: Centrum voor Milieuwetenschappen), Leiden University, Netherlands. He undertook a research project on the ‘development of life cycle assessment of residue-based bioenergy’ under the supervision of Prof. Geert R. de Snoo and Dr. Reinout Heijungs. The plan is that, after completing his PhD, he will continue the career as a senior researcher at LIPI in Indonesia. His current project at this institute is to study the ‘life cycle assessment of second-generation bioethanol of oil palm empty fruit bunches based on a pilot-scale production inventory.’ Additionally, he has initiated the establishment of Indonesian Life Cycle Asssessment Network (www.ilcan.or.id) at the end of 2014. It is a voluntary initiative with the objectives to promote the application of life cycle thinking and life cycle assessment in Indonesia.

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