Development of life cycle assessment for residue-based bioenergy
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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.

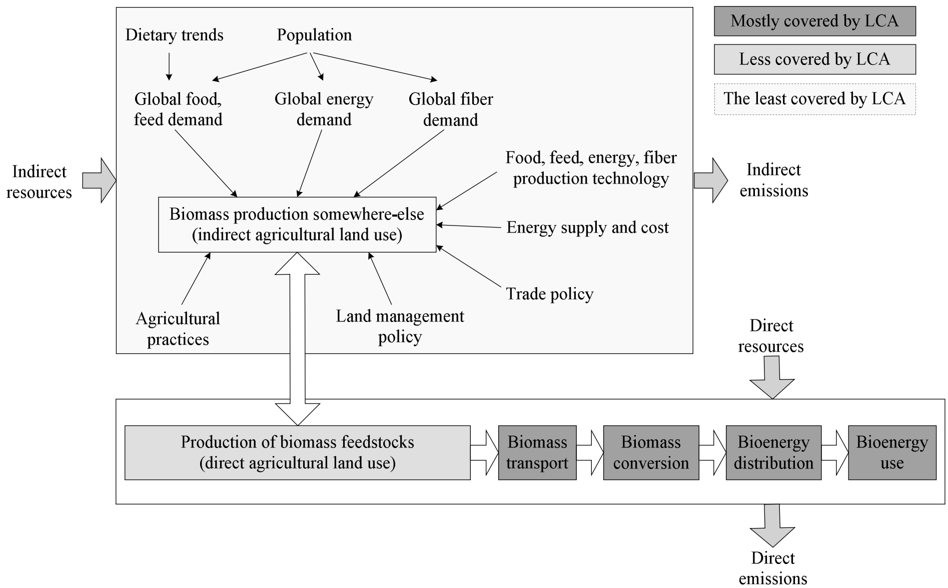

Figure 2.1. Direct and indirect effects of a generic bioenergy system (modified from Sheehan, 2009). Different shading intensity indicates present coverage in LCA studies.

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.

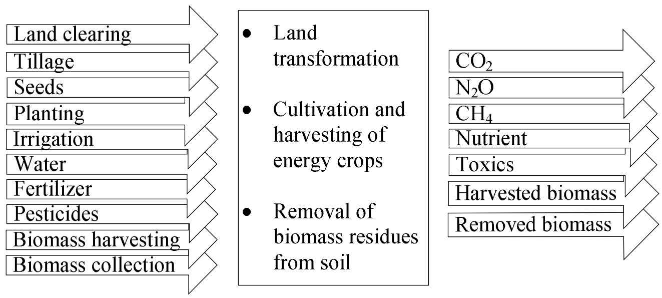

Figure 2.2. Inventory of activities, resource use, and emissions in the agricultural chain of biomass feedstock.

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.

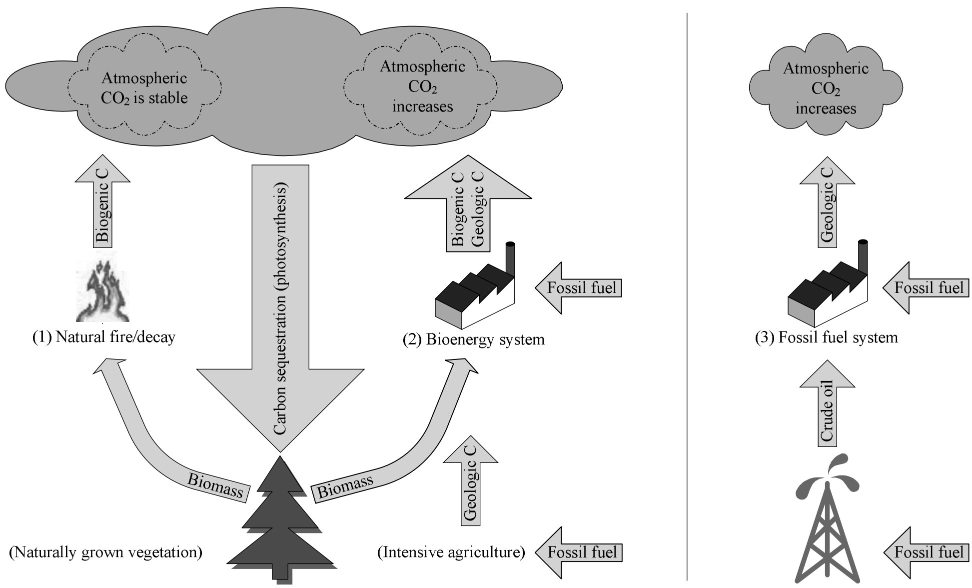

Figure 2.3. Biogenic-carbon cycle versus 'irreversible' geologic-carbon emission.

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).

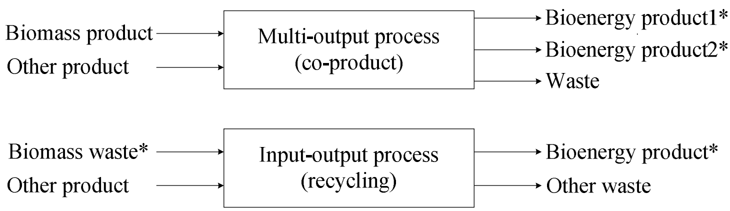

Figure 2.4. Relevant multi-functional processes in bioenergy systems (*= functional flows).

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).

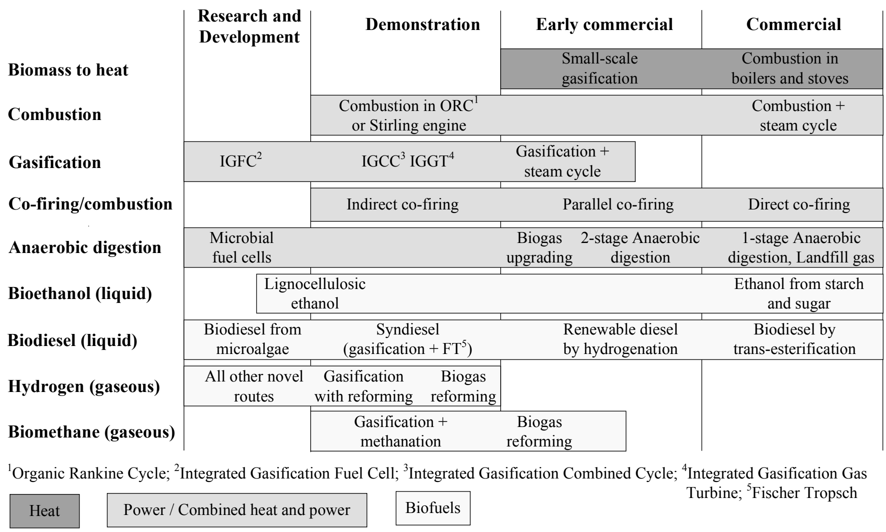

Figure 2.5. State of the art of the conversion technologies for bioenergy (modified from IEA-Bioenergy, 2009).

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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2.8 References

Bare JC, Norris GA, Pennington DW, McKone TE. 2003. TRACI, the tool for the reduction and assessment of chemical and other environmental impacts. J Indust Ecol 6(3–4): 49–78.

Basset-Mens C, van der Werf HMG, Durand P, Leterme P. 2006. Implications of uncertainty and variability in the life cycle assessment of pig production systems. Int J Life Cycle Assess 11(5): 298–304.

Baumann H, Tillman AM. 2004. The Hitch Hiker's guide to LCA: an orientation in life cycle assessment methodology and application. Studentlitteratur, Sweden.

Borrion AL, McManus MC, Hammond GP. 2012. Environmental life cycle assessment of lignocellulosic conversion to ethanol: a review. Renew Sust Energy Rev 16: 4638–4650.
Brander M, Tipper R, Hutchison C, Davis G. 2009. Consequential and attributional approaches to LCA: a guide to policy makers with specific reference to greenhouse gas LCA of biofuels. Technical paper TP-090403-A, Ecometrica April 2009. http://ecometrica.com/assets/approachesto_LCA3_technical.pdf. Accessed 12 Oct 2012.
BSI. 2011. PAS 2050: specification for the assessment of the life cycle greenhouse gas emissions of goods and services. British Standards Institution, London.
Cherubini F, Strømman AH. 2011. Life cycle assessment of bioenergy systems: state of the art and future challenges. Bioresour Technol 102: 437–451.
EU Directive. 2009. Directive 2009/28/EC of the European Parliament and of the Council of 23 April 2009 on the promotion of the use of energy from renewable sources. Official J EurUnion 52(L140): 16–62.
EC-JRC-IES. 2010. International reference life cycle data system (ILCD) handbook—general guide for life cycle assessment—detailed guidance, 1st edn. European commission—joint research centre—institute for environment and sustainability. Publications Office of the European Union. Luxembourg Ecoinvent. 2010. Ecoinvent data v2.2. Swiss center for life cycle inventories.
Finnveden G, Hauschild MZ, Ekvall T, Guinée JB, Heijungs R, Hellweg S, Koehler A, Pennington D, Suh S. 2009. Recent developments in life cycle assessment. J Environ Manag 91: 1–21.
Guinée JB (ed). 2002. Handbook on life cycle assessment: operational guide to the ISO standards. Kluwer Academic Publisher, Dordrecht.
Guinée JB, Heijungs R, van der Voet E. 2009. A greenhouse gas indicator for bioenergy: some theoretical issues with practical implications. Int J Life Cycle Assess 14: 328–339.
Guinée JB, Heijungs R, Huppes G, Zamagni A, Masoni P, Buonamici R, Ekvall T, Rydberg T. 2011. Life cycle assessment: past, present, and future. Environ Sci Technol 45: 90–96.
Harvey M, Pilgrim S. 2011. The new competition for land: food, energy, and climate change. Food Policy 36: S40–S51.
Hedegaard K, Thyo KA, Wenzel H. 2008. Life cycle assessment of an advanced bioethanol technology in the perspective of constrained biomass availability. Environ Sci Technol 42(21): 7992–7999.
Heijungs R, Guinée JB. 2007. Allocation and ''what-if'' scenarios in life cycle assessment of waste management systems. Waste Manag 27(8): 997–1005.
Heijungs R, Wiloso EI. 2012. LCA of second generation bioenergy and biomaterials: progress and problems. In: Proceedings of the the 2nd Korea-Indonesia workshop and international symposium on bioenergy from biomass, Serpong, Tangerang, Indonesia.
Heijungs R, Guinée JB, Huppes G, Lankreijer RM, Udo de Haes HA, Wegener Sleeswijk A, Ansems AMM, Eggels PG, van Duin R, de Goede HP. 1992. Environmental life cycle assessment of products: guide and backgrounds. CML Leiden University, Leiden.
IEA-Bioenergy. 2009. Bioenergy—a sustainable and reliable energy source. International Energy Agency. http://www.ieabioenergy.com/LibItem.aspx?id=6479. Accessed 11 April 2012.
IPCC. 2001. Special report on land use, land-use change and forestry. IPCC special reports on Climate change. http://www.grida.no/publications/other/ipcc_sr/?src=/climate/ipcc/land_use/044.htm#s2-2-1. Accessed 4 Sept 2012.
IPCC. 2011. Bioenergy. In: IPCC special report on renewable energy sources and climate change mitigation. Cambridge University Press, Cambridge.

ISO. 2006. Environmental management—life cycle assessment—requirements and guidelines (ISO 14044). International Organization for Standardization, Geneva.

Johnson E. 2009. Goodbye to carbon neutral: getting biomass footprints right. Environ Impact Assess Rev 29: 165–168.

Koellner T, de Baan L, Beck T, Brandão M, Civit B, Goedkoop M, Margni M, Milài Canals L, Müller-Wenk R, Weidema B, Wittstock B. 2012. Principles for life cycle inventories of land use on a global scale. Int J Life Cycle Assess. doi:10.1007/s11367-012-0392-0.

Levasseur A, Lesage P, Margni M, Samson R. 2012a. Biogenic carbon and temporary storage addressed with dynamic life cycle assessment. J Indust Ecol. doi:10.1111/j.1530-9290.2012.00503.x.

Levasseur A, Lesage P, Margni M, Brandão M, Samson R. 2012b. Assessing temporary carbon sequestration and storage projects through land use, land-use change and forestry: comparison of dynamic life cycle assessment with ton-year approaches. Climatic Change. doi:10.1007/s10584-012-0473-x.

Lim S, Lee KT. 2011. Parallel production of biodiesel and bioethanol in palm-oil- based biorefineries: life cycle assessment on the energy and greenhouse gases emissions. Biofuels Bioprod Bioref 5: 132–150.

Liska AJ, Perrin RK. 2009. Indirect land use emissions in the life cycle of biofuels: regulations vs science. Biofuels Bioprod Bioref 3: 318–328.

McKone TE, Nazaroff WW, Berck P, Auffhammer M, Lipman T, Torn MS, Masanet E, Lobscheid A, Santero N, Mishra U, Barrett A, Bomberg M, Fingerman K, Scown C, Strogen B, Horvath A. 2011. Grand challenges for life-cycle assessment of biofuels. Environ Sci Technol 45: 1751–1756.

McLauchlan K. 2006. The nature and longevity of agricultural impacts on soil carbon and nutrients: a review. Ecosystems 9: 1364–1382.

Milà i Canals L, Bauer C, Depestele J, Dubreuil A, Knuchel RF, Gaillard G, Michelsen O, Wenk RM, Rydgren B. 2007. Key elements in a framework for land use impact assessment within LCA. Int J Life Cycle Assess 12(1): 5–15.

Potting J, Hauschild MZ. 1997. Predicted environmental impact and expected occurrence of actual environmental impact. Part 2: spatial differentiation in life-cycle assessment via the site-dependent characterisation of environmental impact from emissions. Int J Life Cycle Assess 2: 209–216.

Potting J, Schöpp W, Blok K, Hauschild MZ. 1998. Site-dependent life-cycle assessment of acidification. J Indust Ecol 2: 63–87.

Schmer MR, Vogel KP, Mitchell RB, Perrin RK. 2008. Net energy of cellulosic ethanol from switchgrass. PNAS 105(2): 464–469.

Searchinger T, Heimlich R, Houghton RA, Dong F, Elobeid A, Fabiosa J, Tokgoz S, Hayes D, Yu TH. 2008. Use of U.S. croplands for biofuels increases greenhouse gases through emissions from land-use change. Science 319: 1238–1240.

Seppälä J, Posch M, Johansson M, Hettelingh J. 2006. Country-dependent characterization factors for acidification and terrestrial eutrophication based on accumulated exceedence as an impact category indicator. Int J Life Cycle Assess 11: 403–416.

Sheehan JJ. 2009. Biofuels and the conundrum of sustainability. Curr Opin Biotechnol 20: 318–324.

Singh A, Pant D, Korres NE, Nizami AS, Prasad S, Murphy JD. 2010. Key issues in life cycle assessment of ethanol production from lignocellulosic biomass: challenges and perspectives. Bioresour Technol 101: 5003–5012.

Smyth BM, Murphy JD. 2011. The indirect effects of biofuels and what to do about them: the case of grass biomethane and its impact on livestock. Biofuels Bioprod Bioref 5(2): 165–184.

Sørensen B. 2002. Renewable energy, 2nd edn. Academic, San Diego.

Thomassen MA, Dalgaard R, Heijungs R, de Boer I. 2008. Attributional and consequential LCA of milk production. Int J Life Cycle Assess 13(4): 339–349.

Tillman AM, Ekvall T, Baumann H, Rydberg T. 1994. Choice of system boundaries in life cycle assessment. J Clean Prod 2(1): 21–29.

Tilman D, Socolow R, Foley JA, Hill J, Larson E, Lynd L, Pacala S, Reilly J, Searchinger T, Somerville C, Williams R. 2009. Beneficial biofuels—the food, energy, and environment trilemma. Science 325: 270–271.

van Dam J, Junginger M, Faaij APC. 2010. From the global efforts on certification of bioenergy towards an integrated approach based on sustainable land use planning. Renew Sust Energy Rev 14(9): 2445–2472.

van der Voet E, Lifset RJ, Luo L. 2010. Life-cycle assessment of biofuels, convergence and divergence. Biofuels 1: 435–449.

Wegener-Sleeswijk A, Kleijn R, Meeusen-van Onna MJG, Leneman H, Sengers HHWJM, van Zeijts H, Reus JAWA. 1996. Application of LCA to agricultural products; 1. core methodological issues; 2. supplement to the LCA guide; 3. methodological background. Centre of Environ Sci. Leiden.

Weidema BP. 2003. Market information in life cycle assessment. Environmental project no. 863. Danish Environmental Protection Agency, Copenhagen.

Williams PRD, Inman D, Aden A, Heath GA. 2009. Environmental and sustainability factors associated with next-generation biofuels in the US: what do we really know. Environ Sci Technol 43: 4763–4775.

Wiloso EI, Heijungs R, de Snoo G. 2012. LCA of second generation bioethanol: a review and some issues to be resolved for good LCA practice. Renew Sust Energy Rev 16: 5295–5308.

WRI. 2005. Navigating the numbers: greenhouse gas data and international climate change policy. World Resources Institute, Washington DC.

Zamagni A, Guinée J, Heijungs R, Masoni P, Raggi A. 2012. Lights and shadows in consequential LCA. Int J Life Cycle Assess 17: 904–918.