Chapter 6
Life Cycle Assessment of Renewable Energy Production from Biomass

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Abstract Among the different alternatives to conventional fossil fuels, the production of renewable energy from biomass (i.e. bioenergy) is regarded as an interesting option since it involves the valorisation of waste streams, residues and non-food crop biomass. Although a standardised framework regulates the Life Cycle Assessment (LCA) methodology, its application in practice poses some methodological difficulties. This chapter reviews the main methodological issues that a LCA practitioner has to face when it comes to the environmental assessment of bioenergy systems. Despite its complexity, consequential LCA is considered an interesting approach for informing policy-makers and decision-makers about the indirect effect of a specific strategy. In this sense, indirect environmental burdens such as indirect land use change should be included in the study. Moreover, the selection of the system function and system boundaries are other methodological issues that directly affect the results obtained and, therefore, the comparability of LCA studies, intensified in particular in the case of bioenergy systems due to their complexity. In more detail, some bioenergy systems co-produce multiple products, increasing the variability of

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the functions provided by the system, as well as of the system boundaries chosen to overcome multifunctionality (subdivision, system expansion or allocation). The selection of the appropriate methodology and impact categories, as well as the gaps in characterisation factors, is other methodological drawbacks.

Keywords Biomass · Life cycle assessment · Methodological issues
Bioenergy systems

6.1 Introduction

6.1.1 Interest in Renewable Energy Production and Use

Energy is a potential indicator of economic and social development and improved quality of life (Ahiduzzaman and Sadrul Islam 2011). Currently, about 85% of the world's energy requirements are supplied by conventional fossil fuels (Srirangan et al. 2012). However, there are important issues regarding the sustainability of their use, including (i) depletion of fossil reserves, (ii) significant environmental impacts and (iii) large price fluctuations. Society's concerns about environmental and health issues arising from the use of fossil fuels has increased due to the increasing concentration of greenhouse gases (GHGs) such as CO_2, CH_4, CFCs, ozone, N_2O and halons in the atmosphere. The release of these gases derives mainly from human activities, which threaten not only environmental sustainability but also the socio economic situation, favouring global climate change and its related societal consequences (Berners-Lee et al. 2012). In this sense, there is abundant scientific evidence that changes in global climate are caused by anthropogenic activity. The effects of climate change are manifested even on a daily basis through the multiplication of extreme weather events such as heat waves, floods and droughts, the distribution of vector-borne diseases and their impact on disaster risk and malnutrition (Panwar et al. 2011). However, facing global climate change represents a great challenge (Ahiduzzaman and Sadrul Islam 2011). Effective measures to counteract the drivers of ongoing climate change and improve public response to its consequences are essential (Adamo 2015). Moreover, the concept of Green Economy has received increasing support from researchers and policy-makers. In this context, the development of renewable energy can reduce GHG emissions into the atmosphere, while contributing to solve other crucial challenges, such as improving the reliability of energy supply, saving fossil energy sources, securing local energy supply, creating 'green jobs' opportunities and ensuring sustainable development in rural areas (Panwar et al. 2011; Gasparatos et al. 2017).

With this aim, the European Commission published in 1997 'Energy for the future: Renewable sources of energy', a White Paper for a Community Strategy and Action Plan laying the foundations of the European Union (EU) policy on renewable energy (European Commission 1997). This document proposed to increase the share of renewable energy in the European gross energy consumption to 12% by 2010. Thereafter, the EU promoted the production of electricity from renewable energy sources under Directive 2001/77/EC (European Parliament 2001). In 2007, the European Commission proposed an integrated Energy and Climate Change programme, which included the commitment to achieve a reduction of at least 20% of GHG emissions by 2020 compared to 1990 levels. Subsequently, EU Directive 2009/28/EC set the target of achieving a 20% share of renewable energy in gross energy consumption and 10% of renewable energy in transport by 2020 (European Parliament 2009). To this purpose, each Member State has its own target for the share of energy from renewable sources and should have implemented a set of policies to achieve this objective. Therefore, the Member States had to prepare National Renewable Energy Action Plans with detailed roadmaps and measures to reach the 2020 renewable energy targets (Scarlat et al. 2015). Despite these activities, renewable energy currently accounts for a relatively small proportion of global final energy consumption (~19% of global primary energy); however, it has the potential to supply all human energy needs (Edenhofer et al. 2011; Ahiduzzaman and Sadrul Islam 2011). Recently, new targets for 2020–2030 have been introduced through the 2030 Framework for Climate and Energy. The targets are to achieve a 40% cut in GHG emissions compared to 1990 levels, at least 27% of renewable energy consumption and at least 27% energy savings compared to the business-as-usual scenario. The European Commission has therefore proposed specific policies to support the achievement of these targets, mainly by means of trading schemes and indicators for competitiveness and security of the energy system.

6.1.2 Renewable Energy Sources: The Potential of Biomass

According to the literature (Ellabban et al. 2014), renewable energy sources have the potential to supply the total present global energy needs. Among the different alternatives, the use of biomass for bioenergy production is considered one of the most promising sources (Cherubini and Strømman 2011) and its potential adds up to 20 times the current global energy requirements, being superior to hydroelectric, marine and geothermal energy.

The term biomass includes all organic material (containing residues) derived from crops, plants and trees and the biomass-based energy implies its conversion into heat, electricity and biofuels (Ellabban et al. 2014). Therefore, sources for bioenergy production can be very different and may include several different production processes, such as wood for thermal energy production or oilseeds production for oil extraction to produce biofuels. All biomass sources are regarded as an alternative capable of replacing fossil resources by producing different fuels and chemicals due to its carbon content. Biomass is synthesised through the photosynthetic process that converts atmospheric carbon dioxide and water into sugars, which are used by plants to produce complex materials, generically known as biomass. In bioenergy systems, it is important to ensure the supply of renewable, consistent and regular feedstock. In particular, feedstock can be sourced from different parts of plants, distinguishing the biomass-to-energy production in first, second and third generation. First-generation biofuels are those produced from dedicated crops that compete with food and feed production, resulting in multiple ethical, political and environmental concerns (Cherubini 2010). Second-generation biofuels come from raw materials based on waste, residues or biomass from non-food crops. They are considered a sustainable alternative to fossil fuels and to first-generation biofuels as well (Cherubini 2010). Third generation biofuels are derived from algae and microalgae cultivation.

Ensuring the environmental sustainability of biomass production is a crucial issue for the sustainable production of biofuels (Cherubini 2010). The Life Cycle Assessment (LCA) methodology has been widely used to compare the environmental impacts produced by fossil and renewable energy sources. With regard to climate change, the LCA studies available in the literature showed that bioenergy entails, in most cases, a reduction of GHG emissions. However, these benefits are not unambiguously obtained in all the environmental impact categories studied. Biomass production has been identified as an important source of other environmental impacts, affecting impact categories such as land use, acidification, eutrophication and ecotoxicity, among others. Additionally, large-scale cultivation of dedicated biomass (i.e. energy crops) could affect bioenergy potential, global food prices and water scarcity. To ensure the sustainable development of bioenergy, integrated policies for energy, land use and water management are needed, along with international cooperation, regulations, certification mechanisms and sustainability criteria (Popp et al. 2014). In this sense, a strategy has been proposed based on the development of biorefinery and biotransformation technologies to transform biomass feedstock into clean forms of energy (An et al. 2011; Kamm and Kamm 2004; Srirangan et al. 2012; Volsky and Smithhart 2011).

6.1.3 Environmental Aspects Linked to Bioenergy

It is generally believed that the use of renewable energies contributes to mitigating the environmental impacts associated with the use of fossil fuels. When biomass is burned or used after conversion into other biofuels (e.g. biodiesel, ethanol, biogas), its carbon content is released into the atmosphere as CO_2, which had been previously captured by the plant in the photosynthetic process. Therefore, biomass-based energy is considered carbon neutral. In addition, the use of biomass reduces NO_X and SO_X emissions into the environment (compared to the use of conventional fossil fuels) as it contains less nitrogen and sulphur than, for example, coal (Herbert and Krishnan 2016).

However, numerous studies indicate that biomass energy is not entirely clean (Field et al. 2008; Rahman et al. 2013; Herbert and Krishnan 2016). Increased production of biomass for renewable energy has the potential to offset fossil fuels requirements, but negative aspects can also be identified that threaten ecosystem conservation and diminish food/feed security (Field et al. 2008).

The production of biomass-based energy always involves the indirect use of fossil energy for the cultivation, transport or manufacturing phases of the process. Cultivation of bioenergy crops could hypothetically damage the environment due to agricultural practices and land and water degradation (Bindraban et al. 2009; Herbert and Krishnan 2016). Biodiversity loss, water harvesting, reduced soil productivity, introduction of invasive energy crops, use of agrochemicals (e.g. pesticides, herbicides and fertilisers) and their derived effects on the aquatic environment, as well as air emissions associated with NO_X, SO_2, NH_3, N_2O should not be omitted. In this sense, the use of non-food biomass produced on marginal land is considered as a potential sustainable option (Bindraban et al. 2009). Significant amounts of biomass could be produced in the short term without displacing food crops. It is therefore justified that we need to address the environmental impacts associated with biomass production, including background processes.

In addition, other negative aspects are also linked to the production and use of biomass. Managing biomass for energy production requires a large amount of storage space, as well as land and water. Soil erosion reduces soil productivity due to agriculture activities (Pimentel 2001), which contributes to water and nutrients run off and subsequently, eutrophication. In addition, changes in soil carbon content resulting from some agricultural activities that can lead to deforestation problems are well known. In this sense, the type of crop may behave differently. According to the literature (Field et al. 2008), management of agricultural land declassified with perennial grassland can increase carbon content, mainly due to the inputs to the soil, including roots and leaf litter.

Therefore, promoting biomass-based renewable energy requires knowledge of the risks (e.g. food security, soil degradation) and opportunities (increasing energy independence, improving rural economies and offsetting climate change) in the area. The modernisation of biomass conversion technologies, together with more efficient biomass production and conversion routes, are challenges to be undertaken. In addition, it is essential to promote standards, practices and regulations to protect the environment.

6.2 Key Methodological Aspects in LCA of Biomass-Based Energy System

LCA has evolved from its origins in the early 1970s into a complex tool that is now being widely applied in research, industry, policy and standards and continues to expand as it is able to determine the environmental impact of products or systems (McManus and Taylor 2015). Renewable energy policies are increasingly considering LCA as the driving tool for selecting the most adequate bioenergy pathways and guiding decision-makers (European Commission 2014). As part of the EU sustainability framework for biofuels and bioliquids, the EU Renewable Energy Directive (RED) (2009/28/EC) and the Fuel Quality Directive (FQD) (2009/30/EC) contain minimum GHG emission requirements that biofuels must meet on a mandatory basis in order to obtain public funding (Edwards et al. 2017). In more detail, GHGs must be reduced by 35% compared with fossil fuels in installations built before October 2015, while the threshold is raised to 50% for installations working from 2017 (Edwards et al. 2017). In order to standardise the quantification of these environmental impacts, the International Organisation for Standardisation (ISO) reissued a regulatory framework for LCA studies during the period 1997–2000. Updates to these documents were completed in 2006, so that the previous standards were combined in ISO 14040 (2006) and ISO 14044 (2006). However, in some aspects, these standards are rather generic, leading to some difficulties in the practical application of this tool, especially when associated with the assessment of complex processes such as bioenergy systems. Among the studies on bioenergy available in the literature, there are often important differences in results, not only due to different approaches to biomass production, conversion technologies and end-use options, but also due to the definition of system boundaries, functional units, allocation methods, assumptions in the building of life cycle inventory (LCI), the fossil energy reference system, etc. (Cherubini et al. 2009), which makes it difficult to compare the different studies.

6.2.1 Life Cycle Model

Traditionally, the life cycle model of LCA studies included all the processes that are identified to make a significant contribution to the supply chain of the system, known as attributional LCA (aLCA). aLCA describes the potential environmental impacts that can be directly attributed to a process or product throughout its life cycle, assuming that it is embedded into a static technosphere (Wolf et al. 2010).

With regard to biomass-based energy production, the current trend is to move to life cycle assessments in more complex decision-making contexts that describe how environmental impacts could change in response to possible policy decisions. This approach, named consequential LCA (cLCA), integrates the supply chain as theoretically expected from the consequence of the decision taken, including the changes resulting from the interaction between the system and markets (Wolf et al. 2010). Hence, this model does not reflect the actual or estimated supply chain, but rather models a hypothetical generic supply chain through market-mechanisms and potentially includes policy interactions and changes in consumer behaviour.

Compared to conventional aLCA, the consequential approach has proven to be particularly interesting in informing policy and decision-makers about the indirect effects of a specific strategy (Vázquez-Rowe et al. 2014). However, this perspective adds great complexity to the LCA study, as it often means that it should include additional economic aspects such as marginal costs and market effects (McManus and Taylor 2015). Moreover, a consistent approach for cLCA has not yet been established by LCA practitioners. The coexistence of these two different approaches with markedly different perspectives is one of the greatest challenges faced by the LCA community (Zamagni et al. 2012; McManus and Taylor 2015). The potential application of cLCA to the different processes for the production of renewable energy from biomass is particularly interesting, even if biomass waste is used as feedstock, due to

- The multifunctionality of these systems since the production of co-products and by-products (e.g. heat and digestate in the biogas-to-electricity production process; press cake or glycerol in the production of first-generation biofuels) adds complexity to the analysis;
- The presence of a framework of public subsidies for the energy produced, since the environmental consequences due to changes in subsidies could be evaluated;
- The reduction of the carbon storage in soil derived from the removal of biomass residues from agricultural or forestry land to produce bioenergy (Cherubini et al. 2009);
- The expansion of land use due to the creation of a market for biomass residues or by-products (Cherubini et al. 2009).

According to Ahlgren et al. (2015), the choice of the LCA approach (aLCA or cLCA) is closely related to the question of research. Despite the assertions made in the ILCD Handbook (JRC 2010), the choice of aLCA or cLCA is not always straightforward. In particular, careful considerations should be given to every study whether the methodological choices that have been made supply meaningful answers to the research questions.

6.2.2 Function of the System and Functional Unit

In every LCA study, the selection of the function of the system is an important methodological step, as it is directly related to the identification of the functional unit (FU) and to the delimitation of the system boundaries. The FU is a key aspect when comparing biofuels with the appropriate fossil reference, since it should guarantee that both systems provide the same service (Cherubini and Strømman 2011).

The selection of the FU in LCA studies focusing on renewable energy systems is not trivial; it is important to note that it is closely related to the objective of the study. Among the different LCA studies of bioenergy systems, there is a remarkable variability in the FU used, making it difficult to compare the results obtained in these studies. In fact, depending on the objective, more than one FU may be appropriate (Cherubini and Strømman 2011). According to the literature (Ahlgren et al. 2015; Cherubini and Strømman 2011), it can be identified:

- The input-based FU such as ‘feedstock use’, suitable for determining the best use of land or biomass, which allows comparison between different uses for a given feedstock. This approach is particularly interesting for first-generation crops since it allows the quantification of land use efficiency (Cherubini 2010). For example, González-García et al. (2013) considered 1 tonne of dry biomass for the environmental comparison of the production of three different energy crops (maize, triticale and wheat) cultivated for biogas production in Italy;

- The output-based FU, such as energy production, which identifies the best way to supply a product such as biogas or electricity from different feedstocks. For instance, for the comparison of two different feedstocks for biodiesel production in Italy, Bacenetti et al. (2017) considered 1 GJ of energy contained in each biofuel, while Carneiro et al. (2017) suggest using 1 MJ of energy contained in ethanol;
- The function-based FU, which represents the best choice according to the ISO standard. For example, in biofuels studies, if the aim is to compare different fuels, travelled distance may be a good option, as fuels have different engine conversion efficiencies. A simple comparison with 1 MJ of fuels would not reflect the diversity of the fuels. As an example, González-García et al. (2012) considered a distance of 1 km driven by a mid-size flexi-fuel vehicle that uses an 85% blend of ethanol and gasoline for the environmental comparison of ethanol production from different fast-growing wood crops.

6.2.3 Managing Multifunctionality

The problem becomes more complex as some bioenergy systems have more than one function and provide more than one product or service. In this sense, the one selected to define the FU depends on the goal and scope of the study. As regards the production of biodiesel from oil crops such as soybean, sunflower and rapeseed, the extraction of crude vegetable oil also includes the production of press cake, while during transesterification, in addition to biofuels, glycerol is also produced. In addition, the conversion of wood biomass into electricity by means of Organic Rankine Cycle (ORC) or by gasification and pyrolysis coupled to a Combined Heat and Power (CHP) plant involves the co-production of heat (Wolf et al. 2016). Another example in renewable energy systems is anaerobic digestion of biomass for biogas production. These systems involve the co-production of different products, both from the anaerobic digestion process and the end-use of the biogas produced. Moreover, this is especially relevant when anaerobic digestion is selected as a treatment option for organic waste management, as resource recovery from waste results in cost-effective multifunctional systems (Heimersson et al. 2017). In more detail, in these systems the main function may be the treatment of organic waste, while other secondary functions can be identified, such as: (i) the production of biogas, (ii) the production of electricity, heat and/or biomethane from biogas and (iii) the production of digestate to be used as organic fertiliser. Therefore, the way these multiple products are considered in LCA studies is becoming increasingly important (Heimersson et al. 2017). Different approaches to solving multifunctionality have been proposed. The choice of the most appropriate one depends, among others, on the goal of the study, available data and information, and the characteristics of the multifunctional process or product (Wolf et al. 2010). In detail:

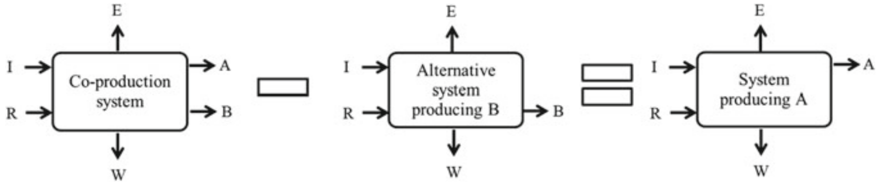

Fig. 6.1 System expansion. Acronyms: I—inputs, R—resources, E—emissions, W—wastes, A—product A, B—product B

- Subdivision refers to the collection of data individually for several mono-functional processes that are components of the multifunctional process and give rise to the production of the product under study;
- System expansion includes two options: (i) adding another function to make the system comparable (i.e. system expansion in the stricter sense) or (ii) extracting the non-required function of the system by subtracting the processes that provide an equivalent function (i.e. substitution by system expansion). System expansion should only be applied if a direct substitution effect can be robustly modelled. The Renewable Energy Directive (European Commission 2009) requires allocation by partitioning, based on the lower heating value (LHV) of the products, with the exception of excess electricity, which is addressed by system expansion;
- Allocation solves multifunctionality by partitioning the flows of individual inputs and outputs between the co-products according to certain criteria. According to ISO 14044, allocation should be avoided whenever possible by applying subdivision or system expansion (ISO 14044 2006). When unavoidable, the inputs and outputs of the system should be partitioned between its different products or functions in such a way that reflects a relationship between them, either physical, economic, energetic or exergetic.

In these complex systems, subdivision cannot be conducted since it is not possible to inventory the system in such detail that it allows each flow to be linked to each product (Heimersson et al. 2017). The use of substitution to avoid allocation is consistent with the recommendations of ISO 14044 and the International Reference Life Cycle Data (ILCD) Handbook (ISO 14044 2006; Wolf et al. 2010). This can be done by giving to the system a credit for secondary functions, awarding the system with the avoided negative impacts of the avoided product or service that the secondary functions replace, as shown in Fig. 6.1.

This perspective was followed, for example, in the study performed by Lijó et al. (2017) to manage the multifunctionality of a biogas system. In this case, anaerobic digestion is a co-production system in which biogas (product A) and digestate (product B) are co-produced. As shown in Fig. 6.1, given that biogas was identified as the main product, the burdens of an alternative system providing the same function as the digestate (i.e. fertilisation of agricultural soil) were subtracted. However, this alternative adds uncertainty to the results obtained due to the lack of primary data for the replaced processes. Moreover, this approach should be considered carefully

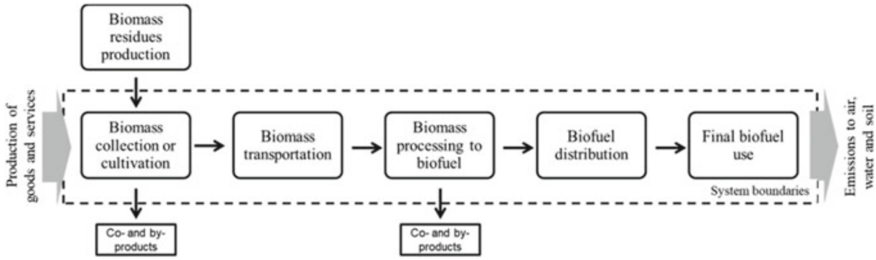

Fig. 6.2 Life cycle chain and system boundaries of bioenergy systems

since biogas systems using agricultural feedstock (e.g., cereal silages), digestate can be used as an organic fertiliser for the production of the energy crop; therefore, it does not leave the system boundaries. Moreover, if biogas is produced from animal manure, the digestate may play the same role as animal waste if it has not been diverted to anaerobic digestion. Fernández-Tirado et al. (2016) first performed a system expansion to address multifunctionality due to biodiesel and press cake co-production; however, subsequently, economic allocation between oils (for biodiesel) and meals co-produced was also considered. Similarly, Bacenetti et al. (2017) performed an economic allocation between crude vegetable oil and press cake, as well as between biodiesel and glycerol in the LCA of two biodiesel systems. Moreover, Wardenaar et al. (2012) conducted an analysis considering different allocation methods (economic, physical and substitution).

6.2.4 System Boundaries

The system boundaries define which unit processes belong to the analysed system, since they are necessary to provide the function to the system. Therefore, the system boundaries separate the analysed system from the rest of the technosphere and nature, defining where the system exchanges elementary flows with nature, and, therefore, produces the environmental impacts (Wolf et al. 2010).

Figure 6.2 depicts the processes included within the system boundaries when the overall life cycle chain of bioenergy system is under study. However, depending on the goal of each LCA study, the system boundaries of the bioenergy chain may change. For example, for the comparison of different feedstock for biodiesel production, a cradle-to-gate approach may be considered, i.e. considering the background and foreground processes up to the gate of the biodiesel factory (Fernández-Tirado et al. 2016; Bacenetti et al. 2017).

By maintaining the example of biogas production, the definition of the system boundaries of some systems for the anaerobic digestion of organic waste streams may be conflicting. In more detail, the distribution of the burdens related to waste production between the production system and the treatment system may be problematic (Doka 2007). The question is whether waste can be regarded as a valuable product, as it can produce biogas or a waste material that needs to be managed. Organic waste is considered as a zero-value product because biogas plants do not usually have to pay for it. In these cases, it is common practice in LCA studies to consider that the production of organic waste such as manure, food and industrial wastes are excluded from the system boundaries of the biogas system, as they are considered to be waste streams from other production systems (i.e. livestock and food sectors).

Broadly speaking, although its use in biogas plants is an option for its management, the production of this waste would not be influenced by a change in the biogas management scheme. A similar analysis can be applied to the digestate fraction from anaerobic digestion. Farmers who use the produced digestate as an organic fertiliser do not usually have to pay for it; therefore, it can be considered as a waste and the environmental impacts of its handling should be allocated to the anaerobic digestion process. This poses a major problem when applying digestate to agricultural land; while emissions of mineral fertilisers or animal manure are fully attributed to the agricultural production, emissions from the digestate would be allocated to the biogas system. This would lead to questionable conclusions from the LCA studies. According to Doka (2007), there are good reasons to include the application of digestate as a waste in LCA of biogas systems, there are also equally justified reasons to set the cut-off limit that includes the application of digestate in agriculture as a recycled material.

Unlike animal manure in the biogas process, for the wood-to-energy production chain, the use of lignocellulosic matrices as feedstock for energy purposes cannot be managed with the zero-value approach. When biomass such as pruning residues, leaves or branches from forestry utilisation are used as feedstock for energy plants, their production is included in the system boundary and, usually, allocation is performed (Muench and Guenther 2013; Patel et al. 2016).

In order to quantify the environmental gains of bioenergy production, a fossil-based reference system is required. Therefore, its definition can also play a key role in the outcomes obtained for a specific study. The reference system should reflect the most representative conventional way of providing the same function as the bioenergy system under study for a specific geographical location. When analysing the production of electricity from biogas, the reference fossil-based electricity system can be produced from coal, oil or natural gas, which entails different potential environmental burdens used as reference (Cherubini and Strømman 2011). For example, Van Stappen et al. (2016) displaced electricity from oil and natural gas, while Lijó et al. (2017) considered the average electricity production in the country mix of the site under study.

Following the issue of system boundaries, the consideration (or not) of biogenic carbon within the boundaries of LCA studies also deserves special mention. This biogenic carbon is temporarily stored in vegetation, litter, dead wood and soil (Cherubini and Strømman 2011). Therefore, this consideration is particularly important when dealing with bioenergy systems, since they use biomass as feedstock, whether energy crops, by-products or organic waste, which can be considered a temporary carbon storage. Biogenic carbon is defined as the carbon contained or derived from biomass that accumulates during plant growth as a result of photosynthesis (Wiloso et al. 2012). Conventionally, LCA studies do not assign any environmental burden to carbon dioxide emissions from biogenic sources (Brandão et al. 2013). In these cases, carbon neutrality is considered on the basis that the expected uptake of carbon dioxide from biomass growth equals the expected carbon emitted over the full life cycle, whether it is naturally decomposed or burned (Wiloso et al. 2012). Therefore, it is considered that there is no net increase in atmospheric carbon dioxide content and the benefits of temporarily removing it from the atmosphere and the impacts related to its subsequent emission are excluded from many LCA studies (Brandão et al. 2013; Wiloso et al. 2012).

However, with the aim of validating this assumption, the previously harvested biomass should be replaced by a new biomass growing in the short term. In this sense, the use of annual crops may not increase the amount of atmospheric carbon due to compensation by the relatively undelayed photosynthesis (Wiloso et al. 2012). Many authors disagree with this statement. Carbon sequestration during biomass growth can be considered as a negative emission in LCA. The argument to support this approach is that during the time between biomass harvesting and its decomposition or combustion, the concentration of carbon dioxide in the atmosphere decreases temporarily and radiative forcing is partially avoided.

Other authors support the idea that temporary storage of biogenic carbon may have a negative effect due to the change in the concentration gradient between atmosphere and oceans, causing oceans to absorb less carbon dioxide (Wiloso et al. 2012). The consideration of temporary carbon storage and delayed emissions within the system boundaries of LCA studies is discouraged by the ILCD Handbook, unless the goal of the study clearly includes it (Wolf et al. 2010). In any case, the inclusion of biogenic carbon within the system boundaries in LCA studies is still under discussion. Timing of emissions is usually not included in LCA of renewable energy. However, according to Ahlgren et al. (2015), when there are significant differences in time between CO_2 uptake and emissions from the system under study, this should not be ignored and discussed in the study, promoting efforts to quantify the impact.

However, biogenic carbon in biomass is not the only one to be considered; changes in biogenic carbon contained in the soil should be taken into consideration. By changing the way land is used, named as land use change (LUC), these storage pools may change until they reach a new equilibrium (

Cherubini 2010). The consideration of these carbon changes is directly related with cLCA studies (Carneiro et al. 2017) and has an important impact on the carbon balance of bioenergy systems due to the large quantities of carbon in soil. Therefore, emissions due to land use change may reduce GHG savings from bioenergy systems when comparing to fossil-based alternatives, especially when considering dedicated energy crops or agricultural and forestry residues as raw material (Cherubini 2010). Direct land use change (dLUC) occurs when the use of land is changed to produce energy crops for bioenergy purposes, displacing previous land use. Depending on the earlier use of the land and the energy crop to be established, carbon stock in soil can increase or decrease (Cherubini 2010). For example, if a forest land is converted into palm plantations, there would be a loss of carbon stocks; while, when the abandoned land is converted into sustainable maize cultivation, carbon stock may increase (Cherubini and Strømman 2011). Indirect land use change (iLUC) (or leakage) occurs when land currently used for feed or food crops is transformed into the production of feedstock for bioenergy and the demand for the previous land use (i.e. feed, food) remains, the displaced agricultural production will be shifted to other places where unfavourable land use change may occur. For instance, Buchspies and Kaltschmitt (2018) analysed different first and second-generation ethanol production, considering the mechanisms of LUC linked to straw removal in Germany, transformation from scrubland to soybean cultivation in Brazil and decrease in forest cover due to oil palm cultivation in Indonesia and Malaysia.

6.2.5 Building the Life Cycle Inventory

In LCA studies, the LCI is built by collecting data for each unit process defined at the system boundary and it is expressed on the basis of the FU selected. Collected data would include energy inputs, raw material inputs, ancillary inputs, other physical inputs, products, co-products and waste, emissions to air, discharges to water and soil and other environmental aspects. Two types of data can be distinguished: i) foreground data that refer to the process data required to produce the product under study and ii) background data that include data from processes required to produce generic materials, energy, transport and waste management. In accordance with ISO 14044, data quality requirements shall be specified to ensure compliance with the goal and scope of the LCA. It should include time-related coverage (data age and the minimum period of time for data collection), geographical coverage (area from which data should be collected for each unit process), technology coverage, accuracy (measurement of variability in data values for each process), completeness (percentage of flow that is measured or estimated), consistency (assessment of whether the methodology for data collection is uniformly applied to the data collection process), reproducibility (assessment of the extent to which information on the methodology and data values would allow an independent practitioner to reproduce the study), data sources and uncertainty of the information.

Specifically, regarding LCA studies of bioenergy systems, the calculation of different foreground data is required in different unit processes and at different stages of the life cycle, especially data related to direct emissions, such as those from storage or application of digestate, since they are not usually measured due to their difficulty. Estimating this type of data is a crucial issue in biogas LCA studies because they play an important role in the environmental outcomes. Therefore, to consider these emissions within the system, they are usually estimated using the methodologies available in the literature. Numerous studies have shown that the most important hotspots in biogas systems are associated with emissions, especially in the eutrophication and acidification impact categories (Lijó et al. 2017). However, there are several different methodologies and there is no a general consensus on which of them should be selected. And, therefore, in the literature, these emissions are generally estimated using different methodologies (De Vries et al. 2012; Fantin et al. 2015). These differences translate into disparities in LCA studies. For example, Bacenetti et al. (2017) used the Estimation of Fertilisers Emissions-Software (EFE-So) model to calculate emissions from the fertilisation of two non-conventional oilseed crops for biodiesel production in Italy. Moreover, Lijó et al. (2017) and Fantin et al. (2015) conducted a sensitivity analysis to quantify the effect on the results obtained on different methodologies.

Following the topic of data quality, uncertainty analyses are also recommended. In LCA literature, Monte Carlo simulations are the most commonly used methodologies, for example in Van Stappen et al. (2016) or Fantin et al. (2015); however, this assessment is not performed in most cases.

6.2.6 Selecting the Life Cycle Impact Assessment Methodologies and Impact Categories

According to the ILCD Handbook, the selection of the impact categories and characterisation models should be internationally accepted (Hauschild et al. 2011). In addition, the category indicators shall include those relevant for the specific study performed in accordance with the goal and scope, as well as to the results of the LCI. The characterisation model for each category indicator shall be scientifically and technically valid. Moreover, all the characterisation factors should not have significant gaps in the coverage of the impact category to which they relate. For the selection of relevant impact categories, initial knowledge based on experience gained from studies of similar systems may help to identify which impact categories are of significant global relevance and which may seem irrelevant to a specific system (Hauschild et al. 2011). In previous reviews focused on renewable energy production from biomass (Von Blottnitz and Currain 2007; Hijazi et al. 2015; Bacenetti et al. 2016), a great variation in the number of impacts considered and the Life Cycle Impact Assessment (LCIA) methods used to estimate them was found. In a review focused on agricultural biogas production carried out by Bacenetti et al. (2016) considering 105 studies, the number of impact categories ranged from 1 to 18 and LCIA method used included almost all known LCIA methods, such as EcoIndicator 99, CML 2001, Impact 2002+, ReCiPe and ILCD methods. Undoubtedly, the carbon footprint (also called global warming potential or climate change) was the most widely used environmental indicator. Nevertheless, different assessment methods were also used for this impact category: IPCC (2007), RED (European Union 2009) as well as the standard ISO/TS 14067:2013 (ISO, 2013). Moreover, tackling the carbon footprint alone offers a very limited version of the overall environmental performance of a bioenergy system. Regarding this issue, Venkatesh and Elmi (2013) criticised the importance of focusing on climate change and noted the importance of avoiding problem shifting; i.e. reducing the environmental impacts produced in climate change by increasing them to other impact categories, including acidification or eutrophication categories.

6.3 Conclusions

The LCA has been widely applied to assess the environmental impacts of renewable energy production systems. Although LCA has proven to be a valid methodology for environmental assessments of bioenergy supply chains, some methodological choices remain critical and challenging, and still lead to inconsistent conclusions.

This chapter analyses these unsolved issues and methodological choices to provide a solid basis for further harmonisation of the LCA evaluation activities. In particular, the aim is to improve the robustness of LCA results and the awareness about the methodological choices, as well as to make the outcomes of different studies comparable. Critical methodological factors such as goal definition, selection of FU, system boundaries and allocation have been discussed.

This chapter contributes to the current debate on harmonisation of environmental impact assessment using LCA for the different and more common renewable energy processes.

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