Bioresource Technology 102 (2011) 437–451

Contents lists available at ScienceDirect

Bioresource Technology

journal homepage: www.elsevier.com/locate/biortech

Review

Life cycle assessment of bioenergy systems: State of the art and future challenges

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ARTICLE INFO

Article history:
Received 14 May 2010
Received in revised form 30 July 2010
Accepted 2 August 2010
Available online 6 August 2010

Keywords:
Bioenergy
LCA
Biofuels
Bioelectricity
Bioheat

ABSTRACT

The use of different input data, functional units, allocation methods, reference systems and other assumptions complicates comparisons of LCA bioenergy studies. In addition, uncertainties and use of specific local factors for indirect effects (like land-use change and N-based soil emissions) may give rise to wide ranges of final results. In order to investigate how these key issues have been addressed so far, this work performs a review of the recent bioenergy LCA literature. The abundance of studies dealing with the different biomass resources, conversion technologies, products and environmental impact categories is summarized and discussed. Afterwards, a qualitative interpretation of the LCA results is depicted, focusing on energy balance, GHG balance and other impact categories. With the exception of a few studies, most LCAs found a significant net reduction in GHG emissions and fossil energy consumption when bioenergy replaces fossil energy.

1. Introduction and background

With the current energy policies and management, world market energy consumption is projected to increase by 44% from 2006 (497 EJ) to 2030 (715 EJ) (IEO, 2009). As highlighted by the Fourth Assessment Report of The Intergovernmental Panel on Climate Change (IPCC), this growing fossil fuel consumption, in conjunction with the world's growing population, is leading to the rapid increase in greenhouse gas (GHG) emissions (IPCC, 2007). CO_2 emissions are projected to rise from 29 billion tons in 2006 to 33.1 billion tons in 2015 and 40.4 billion tons in 2030 (corresponding to an increase of 39%) (IEO, 2009).

In addition to the sustainability aspects related to fossil fuel use, this background raising fossil energy demand will face issues of supply, because of the progressive depletion of fossil resources, which makes the availability of conventional oil and natural gas geographically restricted (Bentley et al., 2007; Hanlon and McCartney, 2008). Alternative options able to simultaneously mitigate climate change and reduce the dependence on fossil sources are already in development. The use of biomass for energy (i.e. bioenergy) is deemed to be one of the most promising renewable energy alternatives. In particular, modern biomass applications are becoming increasingly important to countries as a low-carbon, distributed, renewable component of national energy sources. There is a growing interest in bioenergy at a national and global level, as proven by recent policy documents approved by the US congress (see for instance the American Clean Energy and Security Act, either called the Waxman–Markey Bill) and by the European Parliament (Directive 2009/28/EC on the promotion of the use of energy from renewable sources). Despite these regulations promoting biofuels, questions about sustainability of bioenergy pathways were raised (Dickie 2007; Petrou and Pappis, 2009; Sheehan 2009). The conversion of biomass to bioenergy has input and output flows which may affect its overall environmental performances. In addition, indirect effects like land-use change and N-based soil emissions may contribute to complicate the overall picture. This paper elaborates on these topics by performing a thorough review of LCA bioenergy studies, followed by a specific assessment of the key methodological issues and indirect effects, according to the aims and objectives explained in the following section.

2. Aim and scope

This paper performs a review of a large portion of the existing scientific literature that explicitly used life cycle assessment (LCA) methodology, or a life-cycle approach, to estimate the environmental impacts of biomass energy uses. Authors of this paper assume that the reader already has a basic knowledge of LCA and bioenergy production chains, so that general information on these aspects is not provided here.

The main purpose of this work is to discuss and synthesize the key issues and striking features emerged from a review process of the wide scientific literature available, and analyzing the approaches used by the different authors to face these issues, thus

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doi:10.1016/j.biortech.2010.08.010

reporting the current state of the art. Contrarily to other bioenergy review studies (Gnansounou et al., 2009; Larson, 2006; von Blottnitz and Curran, 2007), in this paper there is not an attempt to harmonize results across studies and report them in bars with wide ranges (usually estimated by gathering data from papers located in different areas and based on different data sources and assumptions), but qualitative results will be rather discussed. Qualitative results for energy balance, GHG balance and other environmental impact categories are each described in specific sections. References to studies showing quantitative results are given in the text and specific figures and examples from reliable studies are sometimes reported across this paper to reinforce and enrich the results and the following discussion. Then, existing methodological constraints and bottlenecks are described and discussed in relation with policy maker's requirements and normative frameworks, so identifying existing shortcomings and future research challenges.

This review covers a time period of approximately fifteen years, in which a large numbers of studies have been published. In order to narrow down the number of studies and focus the discussion on the recent and future trends in LCA of bioenergy systems, but at same time without disregarding older contributions, the literature search was mainly based on the following criteria: before 2006, only review papers and relevant case studies were included; from 2006 till date, both reviews and original research papers were considered.

Only studies with a clear claim to be based on a life-cycle approach to estimate environmental impacts are included. In addition, only papers written in English and with good and reliable supporting data and references were selected. Cost analysis and economic assessments are out of the scope of this paper. The total number of reviewed studies is 94, most of which (74) are papers published in scientific journals and the remaining (20) are grey literature. This set of studies does not include the complete literature on LCA of bioenergy, but it does represent a thorough cross section of public available papers. In the Appendix, detailed information for each of the reviewed studies can be found in Tables A1 and A2. All the results described in the following section are derived from an interpretation and critical assessment of these tables.

3. Results

3.1. Outcomes of the review process

3.1.1. Location and scope

The geographical distribution of the papers is shown in Fig. 1, along with the scope of the studies. More than half of the studies were undertaken in a European or North American context, covering a wide range of bioenergy products and biomass raw materials. In 2006, in his excellent review paper, Larson (2006) noticed that the number of studies set in developing countries was just three, and palm oil biodiesel, despite its increasing interest, was not the subject of any comprehensive LCA. In the recent years, as reported in Fig. 1, an increasing number of studies located in developing countries (mainly in South-Eastern Asia) can be observed, with some studies directly evaluating the production of biodiesel from palm oil in Malaysia and Thailand. By contrast, a limited increase of studies can be observed in Africa and South America, while none of them is undertaken in Russia, despite the large biomass resources available in these regions.

Concerning the scope of the study, half of the papers (47) limited the assessment to GHG and energy balances without considering any possible contribution of bioenergy to other impact categories. This approach is usually supported by the evidence that mitigation of climate change and reduction of fossil fuel consumption are the main driving factors for worldwide bioenergy development. The remaining half of the papers performs an analysis which goes beyond GHG and energy balances, providing information on other impact categories or airborne emissions. In Fig. 1, these studies are labelled as "LCA".

3.1.2. Feedstocks and products

Fig. 2 shows the type of bioenergy products and biomass raw materials which are assessed by the reviewed studies. Among transportation biofuels, there is a similar number of studies evaluating 1st and 2nd generation biofuels, even if these latter are mainly at a pre-commercial stage (and are predicted to enter the market within the next 5–10 years). The majority of papers focuses on bioethanol and biodiesel production, which are the

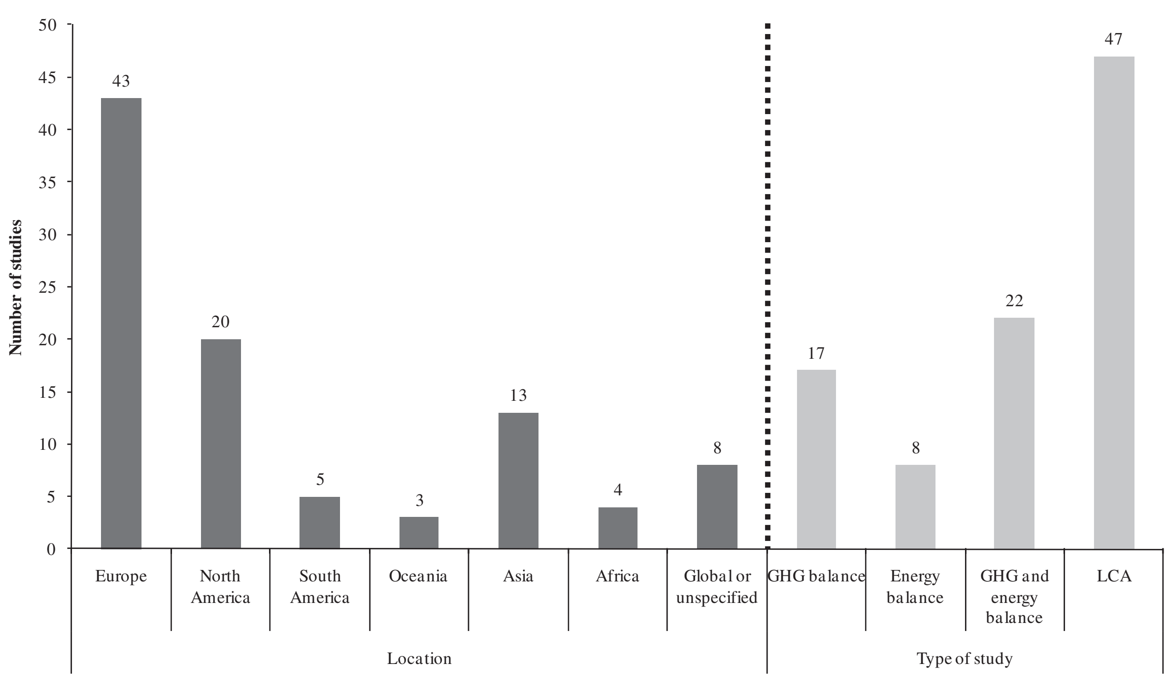

Fig. 1. Location and type of study.

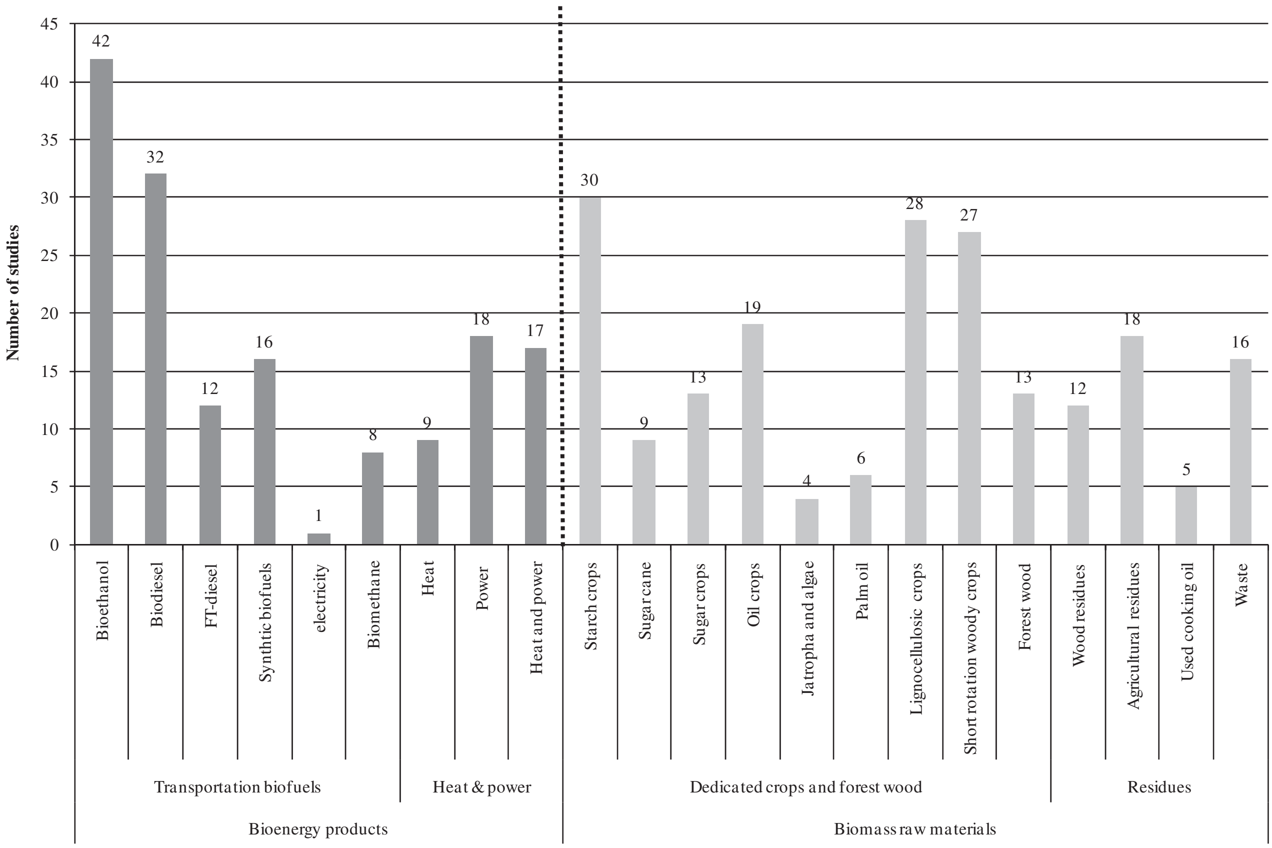

Fig. 2. Type of bioenergy products and biomass raw materials covered by the reviewed studies.

most common transportation biofuels produced today. There is also a relevant number of papers which undertakes an assessment of synthetic biofuels like biomethane, FT-diesel and others.

Concerning studies evaluating the environmental performances of biomass for heat and power production, Fig. 2 shows that their number is slightly lower than that for transportation biofuels. While electricity and heat can be produced by a variety of renewable sources (wind, solar, hydro, biomass, etc.), the only alternative to fossil resources for production of fuels and chemicals is biomass, which is the only C-rich material source available on the Earth besides fossils. This aspect, coupled with the increasing demand for alternative transportation fuels from developed societies, can be the reason why recent scientific literature on LCA is focusing more on biomass use for transportation service rather than for stationary applications.

As shown in the right hand part of Fig. 2, the studies cover a wide spectrum of biomass raw materials. A broad distinction can be done between feedstocks which come from dedicated crops (with occupancy of land and a possible competition with food and feed crops for available biomass and fertile land) and forest and the residues from agricultural, forestry and industrial activities, which can be available without upstream concerns. Lignocellulosic biomass is the most investigated type of feedstock, probably because this is the most abundant biomass resource in the world and is locally available in most of the countries. Many studies also analyze agricultural crops like sugar, starch and oil crops, while other feedstocks like sugar cane and palm oil are restricted to geographical areas with suitable climate conditions. A limited number of LCA studies based non-conventional crops like jatropha and algae currently exists (Kadam, 2002b; Lam et al., 2009; Lardon et al., 2009; Ndong et al., 2009).

3.1.3. Impact categories

The LCA studies reviewed in this work cover different types of impact categories. As already mentioned, some of these studies take the form of complete LCA, while others are limited to GHG and/or energy balance. However, if a study goes beyond Global Warming Potential (GWP) and energy balance, it usually estimates more than one additional impact category. About 90% of the studies include GHG emissions in their evaluation, while primary energy demand is estimated by 71% of studies. In addition to GHGs, a certain number of studies (20%) estimates other airborne emissions like NO_x, PM_{10}, SO_x, and others. Other impact categories, like acidification, eutrophication, etc., are estimated by 20–40% of the studies. In general, grey literature mainly focuses on GHG and energy balances, providing less attention to other environmental impacts.

Several studies report results after weighting, i.e. according to classification methods which use arbitrary units, like the method of ecological scarcity or the Ecoindicator 99. These methods sum together different environmental impacts to provide a limited number of aggregated figures, expressed in non-conventional units. In some cases, such an aggregation may lead to partial or even misleading conclusions and reduces comparison possibilities across studies, because of the loss of specific information. For these reasons, final figures should always be reported with defined units which can be mathematically manipulated.

Relatively few studies (9%) included in their impact assessment the land use category. This is an indicator particularly important for bioenergy systems based on dedicated crops or forest resources, since land use may lead to substantial impacts, particularly on biodiversity and on soil quality. The capital environmental importance of land use impacts contrasts with the lack of studies addressing this issue. This is particularly due to the fact that there is no widely accepted methodology for including land use impacts in LCA, despite some recent efforts (Dubreuil et al., 2007; Koellner and Scholz, 2008; Scholz, 2007). For the same reason, none of the reviewed studies included in the assessment the potential impact of bioenergy on biodiversity, despite an existing accurate methodology (Michelsen, 2008).

Another indicator which received little attention in the literature is the share of urban/local emissions in the life cycle of transportation biofuels, e.g., the urban tailpipe emissions from biofuel combustion in road vehicles. In most circumstances, studies neglect vehicle emissions from biofuel combustion, while others estimate only GHG emissions (N_2O and CH_4). Very few studies consider other gas species like SO_2, PM_{10} and NO_x, and include them in the overall final assessment. With the exception of few studies, information about urban or local emissions is not usually provided. For bioethanol, results reveal that ethanol combustion in cars usually reduce airborne emissions when compared to gasoline (Brinkman et al., 2005; Wu et al., 2008). When biodiesel is used to replace conventional diesel in road vehicle, marginal reductions in CO, uncombusted hydrocarbons and particulate emissions are observed, while SO_2 emissions are effectively eliminated; by contrast, NO_x emissions are slightly higher (Mortimer et al., 2003).

3.2. Qualitative interpretation of results

LCA results of the reviewed studies can be grouped in three broad categories: energy balance, GHG balance and other life-cycle impact categories. All the papers have different goal, scope and objectives. This implies that when comparing LCA results reported by different authors and sources, a wide range of final outcomes can be observed, even for apparently similar bioenergy chains. This variance can be attributable to differing data sources and ages, key input parameter values, agricultural managements, and other intrinsic factors. In addition to them, methodological issues like definition of system boundaries, allocation procedure, reference systems, and other indirect effects such as land-use change and N soil emissions can contribute to widen the range of final results and their uncertainty. In order to completely understand this wide variation, investigation into numerical input assumptions is always required as well as into the calculation methodologies that were used to generate the results. As already mentioned before, this paper focuses on a qualitative interpretation of the results, supported by selected examples and references. Unit based ranges for bioenergy chains are already available in many papers, as will be referred to in the following paragraphs.

3.2.1. Energy balance

Many bioenergy LCA studies include primary energy analysis in their assessment, in order to quantify the possible non-renewable energy savings of the bioenergy system. In particular, there are eight reviewed studies that only focus on energy analysis.

Different indicators can be used for this purpose, and the energy analysis approach usually evaluates all the energy inputs along the full chain, from agricultural cultivation, transportation, processing and final distribution. The resulting cumulative primary energy demand is sometimes used to calculate the EROI (Energy Return on Investment) index (Hammerschlag, 2006); this index is the ratio between energy out (i.e., the energy content of the biofuel) and the non-renewable energy in required along the full life cycle. The cumulative energy demand can be even divided into fossil and renewable. The energy balances and savings of the most common biofuel systems can be found in (Quirin et al., 2004; Shapouri et al., 2002), while ranges on biomass for heat and power production are available in (Cherubini et al., 2009).

In general, because of lower conversion efficiencies, bioenergy systems are affected by a larger cumulative primary energy demand than conventional/fossil energy systems, but it is mainly constituted by the renewable energy fraction of the feedstock, while the fossil energy consumption is significantly smaller (Cherubini and Jungmeier, 2010; Cherubini and Ulgiati, 2010). In bioenergy systems, the fossil energy demand is predominantly affected by fossil fuel energy inputs during cultivation or processing. Among transportation biofuels, bioethanol from sugar cane is the most efficient option for replacing fossil energy, thanks to both the high yields and the possibility to use its residues (i.e. bagasse) to run the processing plant. Transportation biofuels produced in temperate regions replace much less fossil energy. Most of the reviewed studies concluded that bioenergy saves fossil energy, besides two exceptions. A study located in the US states that energy outputs from ethanol produced using corn, switchgrass, and wood biomass were each less than the respective fossil energy inputs; within the same paper, a similar result was achieved in the production of biodiesel from soybeans and sunflower (Pimentel and Patzek, 2005). In this case, our understanding is that the cumulative energy demand is not allocated among the co-products. Similarly, another study focusing on bioethanol production from cassava in Thailand shows that, when an allocation step is not undertaken and coal is used to feed the conversion plant, the biofuel production consumes more fossil energy than it substitutes (Papong and Malakul, 2010).

In general, the fossil energy input is higher for production of transportation biofuels from oil or starch crops than for biomass-derived electricity/heat generation (usually produced from wood combustion). The reason is twofold: oil and starch crops need higher cultivation inputs than woody crops (Kim and Dale, 2008; Zah et al., 2007), and the production of transportation biofuels usually involves more energy intensive stages (Botha and von Blottnitz, 2006; Cherubini et al., 2009).

3.2.2. GHG balance

About 90% of the reviewed studies accounted for GHG emissions along the entire bioenergy chain to estimate its GWP. There is clear scientific evidence that emissions of greenhouse gasses, such as carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O), arising from fossil fuel combustion and land-use change as a result of human activities, are perturbing the Earth's climate. Mitigation of climate change is therefore one of the main driving forces for development and deployment of bioenergy systems. LCA studies report results with different indices and indicators, often based on different functional units, and use different reference systems to estimate GHG emission savings. This means that outcomes are often not immediately comparable and of difficult interpretation. Moreover, there is a wide variation on the methodology used to estimate GHG emissions, mainly due to the selection of system boundaries, allocation procedures, inclusion of land-use change effects and others. As a consequence, this indicator has a higher degree of divergence across studies than the energy analysis; this is why regulatory agencies and organizations recently proposed methodological standards for calculating the C footprint of products (EU, 2009; ISO, 2009; PAS2050, 2008). Wide ranges summarizing GHG balances from published studies for transportation biofuels are available in (Hossain and Davies, 2010; Quirin et al., 2004), while for heat and/or power production see (Cherubini et al., 2009; Varun et al., 2009).

Bioenergy systems generally ensure GHG emission savings when compared to conventional fossil reference systems. For example, net GHG emissions from generation of a unit of electricity from biomass are usually 5–10% of those from fossil fuel-based electricity generation (Cherubini et al., 2009; Varun et al., 2009). The ratio will be more favourable (lower), if biomass is produced with low energy input (or derived from residue streams), converted efficiently (ideally in CHP applications) and if the fossil fuel reference use is inefficient and based on a carbon-intensive fuel such as coal. However, the inclusion in the GHG balance of indirect effects is of capital importance, given their potential large influence on final results (Johnson, 2009; Searchinger et al., 2008). If compared with other renewable sources, electricity from biomass generally has higher GHG emissions than hydro, wind and geothermal derived electricity, while it is comparable with photovoltaic power production systems (Cherubini et al., 2009; Varun et al., 2009).

Concerning transportation biofuels, bioethanol from sugar cane is again the most efficient pathway in mitigating climate change. However, no one of the reviewed studies on bioethanol from sugar cane took into account the depletion of carbon pools which occurs when sugar cane plantation replaces tropical forests. Such a land-use change (i.e. deforestation) is deemed to cause a decrease of up to 31 t C/ha in soil C pools and a decrease of up to 120 t C/ha in above ground standing biomass pools (IPCC, 2006). If included in the assessment, these figures may even counterbalance the final results. Bioethanol from other sugar and starch crops in temperate climate and biodiesel from oil crops usually achieve 40–65% of the GHG emissions of conventional fossil fuels (Gartner et al., 2003; Gnansounou et al., 2009; Panichelli et al., 2009). GHG emissions may considerably vary even for a defined bioenergy chain. For instance, many studies found a reduction in GHG emissions when bioethanol from corn is used to replace fossil gasoline (Kim and Dale, 2008; Varela et al., 2005; Wu et al., 2008), while there is another study which reports an opposite trend (Delucchi, 2005).

Transportation biofuels produced from residue streams and second generation raw materials (e.g., lignocellulosic biomass, algae, jatropha oil, etc.) usually have larger GHG savings than first generation biofuels. For instance, several studies investigated 2nd generation biofuels finding a strong reduction in GHG emissions (Fleming et al., 2006; González-García et al., 2009b; Spatari et al., 2005; Spatari et al., 2010; van Vliet et al., 2009; Williams et al., 2009). There are some exceptions to this. For instance, when the energy used to feed the biomass conversion process comes from C-intensive fossil sources (e.g. coal), the bioenergy system can release more GHG emissions than its fossil alternative (Fu et al., 2003; Papong and Malakul, 2010). It should be mentioned that all the reviewed studies assumed that CO_2 emissions from biomass combustion are climate neutral.

3.2.3. Other environmental impacts

The types of environmental impacts assessed in the reviewed studies were previously presented. Since these impacts are reported with even less uniformity than GHG and energy indicators, and are even more affected by site specific assumptions, it is not easy to draw simplified figures. In general, bioenergy studies which examined life cycle consequences on human and ecosystem toxicity as well as on other impact categories concluded that most, but not all, bioenergy systems lead to increased impacts when compared to fossil reference systems (Halleux et al., 2008; Kaltschmitt et al., 1997; Zah et al., 2007). This applies particularly to bioenergy crops, where intensive agricultural practices coupled with use of fertilizers (especially nitrogen based) can cause environmental concerns in soils, water bodies and atmosphere. Usually, increased emissions occur in impact categories directly affected by N-based emissions like acidification, eutrophication and photo smog formation, (Carpentieri et al., 2005; González-García et al., 2009a; Kim and Dale, 2008; Luo et al., 2009c). In addition, some papers reports an increase in toxicological impacts, carcinogenic and heavy metals emissions, mainly due to biomass combustion (Halleux et al., 2008; Luo et al., 2009c; Uihlein and Schebek, 2009; Uihlein et al., 2008). Similar drawbacks are found in papers assessing upcoming biorefinery complexes as well: results reveal better environmental performances than conventional product alternatives for GHG emissions, but generally biorefinery systems have larger eutrophication and acidification impact potentials, as reported in (Cherubini and Jungmeier, 2010; Cherubini and Ulgiati, 2010), or greater fossil energy use, carcinogenic emissions and respiratory effects, as noted in (Uihlein and Schebek, 2009). Few studies included in the assessment additional impact categories like water use/consumption (Ramjeawon, 2008; Williams et al., 2009) and contribution of bioenergy to road traffic (Thornley et al., 2009).

3.3. Key methodological issues and assumptions

After reviewing this large number of studies, it was possible to identify some key open methodological issues and assumptions with a large influence on final results. These aspects are discussed hereinafter.

3.3.1. Functional unit

In the literature, there is not uniformity concerning the functional unit chosen by the different analysts, and this makes LCA results of difficult comparison. According to the reviewed studies, 4 types of functional units can be identified in LCA of bioenergy systems:

1. Input unit related: the functional unit is the unit of input biomass, either in mass or energy unit. With this type of functional unit results are independent from conversion processes and type of end-products. This unit can be selected by studies which aim at comparing the best uses for a given biomass feedstock.
2. Output unit related: here the functional unit is the unit of output, like unit of heat or power produced or km of transportation service. This type of functional unit is usually selected by studies aiming at comparing the provision of a given service from different feedstocks.
3. Unit of agricultural land: this functional unit refers to the hectare of agricultural land needed to produce the biomass feedstock. This unit should be the first parameter to take into account when biomass is produced from dedicated energy crops.
4. Year: results of the assessment may be even reported on a year basis. This type of functional unit is used in studies characterized by multiple final products, since it allows avoiding an allocation step.

Fig. 3 shows how many times a functional unit was selected by the reviewed papers. Some papers even report final outcomes according to two or more functional units. Output unit related functional unit is chosen by the majority of the studies, while relatively few studies show results per unit of agricultural land, even if they are based on biomass derived from dedicated crops. This is an extremely important parameter since biomass can compete against food, feed or fibre production under land-availability constraints: the available area for the production of biomass raw materials may be sometimes the biggest bottleneck for the production of bioenergy. Moreover, this indicator could be used to answer the question of relative land-use efficiency (i.e. the use of scarce land resources as efficiently as possible), as discussed in a following section.

Concerning studies on transportation biofuels only, Fig. 3 shows that use of the different units is evenly distributed. This is somewhat surprising, since LCA results for transportation biofuels should be preferably expressed on a per vehicle-km basis. In fact, the adoption of this functional unit ensures that all the life cycle stages (distribution and biofuel combustion) are included, that biofuel mechanical efficiency is considered, and that results are comparable with conventional fossil systems.

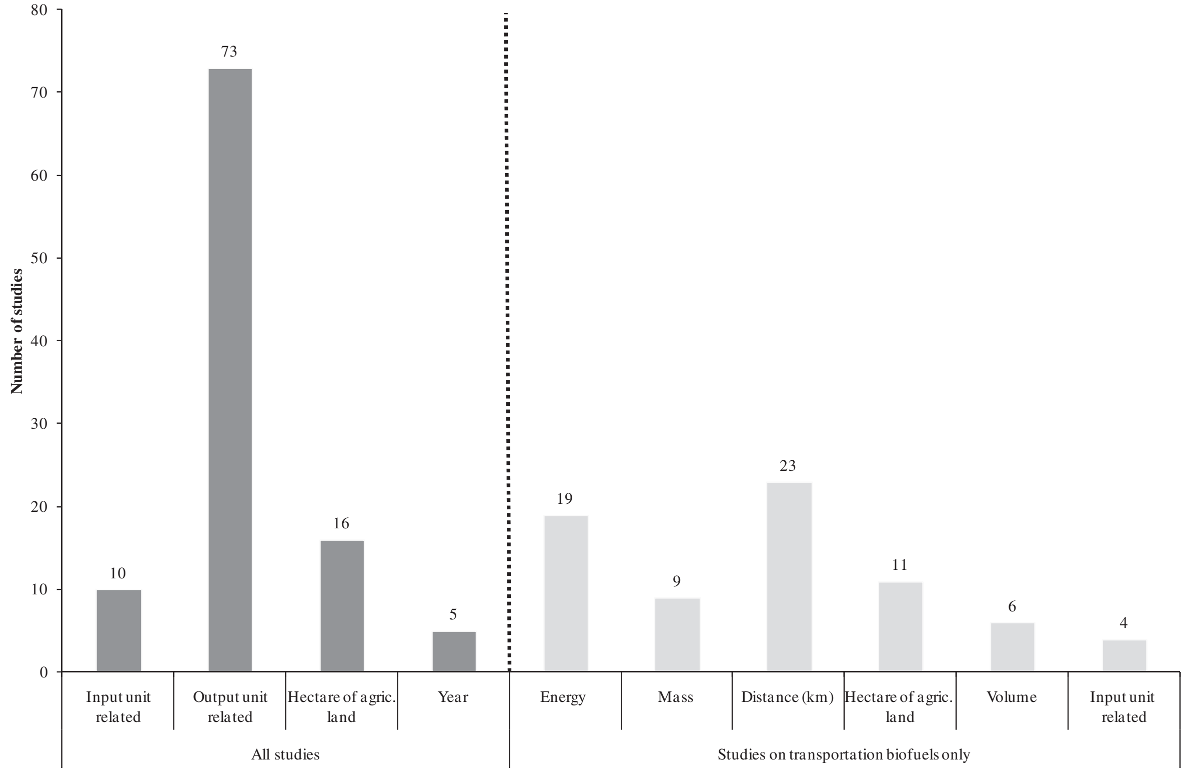

Fig. 3. Choice of functional unit in the reviewed studies.

The functional unit may play an important role when dealing with allocation issues, especially for systems with multiple co-products (e.g. biorefinery). For instance, existing LCA studies on biorefinery managed to avoid allocation by selecting a proper functional unit, i.e. reporting final results per unit of input biomass (Pettersson and Harvey, 2010; Uihlein and Schebek, 2009) or on a year basis (Cherubini and Jungmeier, 2010; Cherubini and Ulgiati, 2010).

The choice of the functional unit may also have an impact on the interpretation of final results. For instance, Lettens et al. (2003) investigated the GHG and energy balance of bioelectricity production from different kinds of short rotation coppice, i.e. hardwood species and other lignocellulosic crops (miscanthus and willow), and their LCA results give rise to a duplex interpretation: hardwood species cause the largest GHG savings per output energy unit, while the other lignocellulosic crops save more GHG emissions per unit of agricultural land. This means that results should be preferably shown using several functional units, which become indicators: in general, the limiting factor of the system should be identified and used as the reference indicator of the assessment. This enhances a better understanding of the system under study and avoids misleading conclusions.

Nevertheless, methodological standards for bioenergy systems generally neglect the importance of reporting final results according to more than one functional unit. For instance, within Annex V of the last directive of the European parliament, it is recommended to express GHG emissions of biofuels per MJ of unit output; GHG emissions per km are only allowed when differences between fuels in useful work done must be taken into account and results per dedicated agricultural land are not even mentioned (EU, 2009).

3.3.2. Reference system

According to the purpose of the study, the LCA can be carried out using different methods. In general, a distinction between attributional and consequential LCA is done. Attributional LCA describes the environmentally relevant flows to and from a life-cycle and its sub-systems, while a consequential LCA describes how environmental relevant flows will change in response to possible decisions (Finnveden et al., 2009). In general, the attributional method is the most used in LCA, but in LCA of bioenergy systems the consequential method appears as the most broadly applied: 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, since consequential LCA is more relevant for decision making. However, this approach is preferable within certain limits, i.e. the uncertainties in the consequential modelling should not outweigh the insight gained from it.

The reference system should always refer to the scope and geographical context of the study. In general, the bioenergy system is compared with a fossil reference system producing the same amount of products and services. It should be noticed that when production of feedstocks for bioenergy uses land previously dedicated to other purposes or when the same feedstock is used for another task, the reference system should include an alternative land use or an alternative biomass use, respectively. This requirement may lead to increase the uncertainty of the assessment, making the adoption of consequential LCA questionable. Similarly, when the bioenergy pathway delivers some co-products able to replace existing products, the reference substituted products should be defined in the fossil reference system and emissions for their production accounted for.

A few number of studies (12%) use another biomass source or biomass conversion technology as reference system. This data can be interpreted as an indication: bioenergy already reached a certain degree of development and some research activities are now focusing on improving environmental performances of existing technologies. For instance, a recent paper compares wood combustion in new stoves for space heating with old stoves (Solli et al., 2009), while second generation bioethanol production from lignocellulosic sources are compared with first generation bioethanol from corn (Luo et al., 2009b; Williams et al., 2009). Some studies (13%) do not include a reference system in their assessment at all.

The definition of the reference system may also play a key role in the estimation of the environmental impact savings of the bioenergy chain. According to the assumptions made, results can widely differ. In fact, fossil-derived electricity can be assumed to be produced from oil, natural gas, coal or other sources, all of which having different GHG emission factors. An example can be found in a recent paper, where GHG emission savings of bioelectricity production from black liquor are estimated using electricity coming from different fossil sources as reference (Pettersson and Harvey, 2010). Clearly, savings are much larger if coal electricity is displaced rather than natural gas electricity.

The definition of a fossil reference system is particularly used by legislations and acts, which usually set specific fraction of GHG emission savings which bioenergy systems must achieve (see for instance the EU directive and the US Energy Independence and Security Act).

3.3.3. Change in carbon pools and land-use changes

Generally, organic C is stored in five different pools: above ground vegetation, below ground vegetation, dead wood, litter and soil. When changing land utilization, these storage pools can change until a new equilibrium is reached. This is an important aspect because of the large quantities of these storage pools, especially soil organic carbon (SOC): this stock of carbon is so large that even relatively small percentage increases or decreases in their size can have relevance in the GHG balance. Land-use changes (LUC) are therefore deemed especially important, and their effects can consistently reduce GHG savings of bioenergy systems based on dedicated crops or agricultural and forest residues, depending on the nature of the changes and the period of time assumed. A distinction is generally done between direct and indirect LUC.

Direct LUC: Direct LUC occurs when new agricultural land is taken into production and feedstock for biofuel purposes displaces a prior land use (e.g. conversion of forest land to sugarcane plantations), thereby generating possible changes to the carbon stock of that land. Among the reviewed studies, land-use change effects were addressed in 22 circumstances (about 23% of the studies). These papers estimate changes in C pools, while three papers included in the assessment other LUC-induced aspects besides C pools, such as the variation in N_2O soil emissions and other effects like additional fertilizer manufacture, honey production, etc. (Cherubini and Ulgiati, 2010; Gabrielle and Gagnaire, 2008; Gartner et al., 2003).

Depending on the earlier use of the land and the crop to be established, the reviewed studies reveal that LUC can be a benefit or a disadvantage:

- When a forest is converted to agricultural land for biofuel production a loss of carbon stocks, in addition to a decrease in biodiversity, is expected; this loss of C affects the whole GHG balance and may even make the bioenergy system worse than its respective fossil reference. Some examples are (Panichelli et al., 2009; Reijnders and Huijbregts, 2008b).
- When set-aside land is taken into production, or perennial herbaceous crops replace annual row crops, the carbon stock may increase; this means that atmospheric CO_2 is sequestered from the atmosphere and stored into soil organic carbon, with a positive effect on the GHG balance of the bioenergy system (Cherubini and Jungmeier, 2010; Spatari et al., 2010; Styles and Jones, 2007; Wang et al., 1999; Wu et al., 2008).

The changes of carbon in soil and other pools are very site-specific and highly dependent on former and current agronomic practices, climate, and soil characteristics. The approach generally used in the literature to estimate LUC effects is to quantify the increase or decrease of a carbon pool (both above and below ground) for a certain period of time, and then include this C loss as CO_2 emissions in accordance to the selected functional unit. This means that LUC effects are amortized over an assumed time horizon, spreading out an emission that mainly occurs in a short period of time over a longer time frame. This approach underestimates the true climate change effects of LUC, since the effect of a GHG increases with the time it remains in the atmosphere. Efforts to overcome this inconsistency can be recognized both in the recent literature (Kendall et al., 2009; O'Hare et al., 2009) and methodological standards (ISO, 2009). Changes in carbon pools are usually estimated by means of literature references or software tools able to model soil carbon dynamics. In addition, IPCC provides default values by which it is possible to estimate the annual effect of direct LUC (IPCC, 2006). The use of IPCC default values is recommended by most of the methodological standards, which suggest the use of annualized emissions over an arbitrary time frame, usually 20 years (EU, 2009; PAS2050, 2008). In particular, PAS2050 provides tables for conversion of forest land and grassland to agricultural land, disregarding SOC changes for agricultural soils, while the ISO GHG protocol stresses the importance of defining proper time boundaries for the assessment, in order to include future emissions (ISO, 2009). The EU directive has a specific land use section, which provides guidelines to estimate GHG emissions induced by LUC, which are straight-line amortized over 20 years (EU, 2009).

Indirect LUC: Indirect LUC (or leakage) occurs when land currently used for feed or food crops is changed into bioenergy feedstock production and the demand for the previous land use (i.e. feed, food) remains, the displaced agricultural production will move to other places (for instance, expansion of agricultural land after deforestation) (Gnansonou et al., 2008). When bioenergy crops are cultivated on fallow, marginal or degraded land where previously no conventional crops were grown, and proper management strategies are implemented, no indirect LUC occurs and the GHG balance can even turn favourably, as in the case of perennial grasses discussed above. Even if none of the reviewed studies addressed this issue and no methodological standards exist, GHG emissions from indirect LUC are deemed to be even more important than emissions from direct LUC. Some authors elaborated a range of values to show the magnitude of this effect (Fargione et al., 2008; Fritsche, 2008; Searchinger et al., 2008). However, these models likely estimate GHG emissions from LUC with significant inaccuracy, and further research is needed before we can be reasonably sure of the indirect effects of biofuels (Liska and Perrin, 2009).

3.3.4. Non-CO_2 emissions from soils

The contribution to net GHG emissions of N_2O, which evolves from nitrogen fertiliser application and organic matter decomposition in soil, emerges as an important variable in LCA studies. Emissions from fields vary depending on soil type, climate, crop, tillage method, and fertiliser and manure application rates. The uncertainties in actual emissions are magnified by the high global warming potential of N_2O, 298 times greater than CO_2. The impacts of N_2O emissions are especially significant for annual biofuel crops, since fertilisation rates are larger for these than for perennial energy crops. Crops grown in high rainfall environments or under flood irrigation have the highest N_2O emissions, as denitrification, the major process leading to N_2O production, is favoured under moist soil conditions where oxygen availability is low (Wrage et al., 2005). Almost the totality of the reviewed studies based on agricultural crops included estimations of N_2O soil emissions in their assessments, and most of them show their relevant contributions to the final GHG balance (CONCAWE, 2006; Kim and Dale, 2008; Lettens et al., 2003; Panichelli et al., 2009; Reijnders and Huijbregts, 2008b). These emissions are generally quantified as a fraction of fertilizer nitrogen content and are based on literature references such as IPCC default factors (IPCC, 2006). Utilization of these factors is also recommended by PAS2050, while the other methodological standards, including the EU directive, do not explicitly mention N-based soil emissions. IPCC data estimate that about 1.0–1.5% of N in synthetic fertilizer is emitted as N in N_2O in temperate regions. A recent paper, which used a different procedure for estimating this emission, proposes a value of 3–5% (Crutzen et al., 2007). If this "extra" N_2O emission is included in GHG balances of biomass systems, Crutzen et al. (2007) state that the global warming benefits of most first generation biofuels are completely annulled. As a consequence, this study is frequently cited as evidence against the use of biofuels as an effective means for mitigating climate change; by contrast, other studies claim that Crutzen et al. (2007) apply an uncertain approach, questionable assumptions and inappropriate, selective comparisons to reach their conclusions (North-Energy, 2008; RFA, 2008). Application of fertilizers also affects other environmental impacts besides GHG emissions, like acidification and eutrophication. In fact, N-based fertilizers enhance volatilization of ammonia from soils and leaching of nitrates to groundwater. These indirect emissions are responsible for the higher impacts which bioenergy systems usually have in these categories when they are compared to fossil reference systems (Cherubini and Jungmeier, 2010; Gasol et al., 2009; Kim and Dale, 2008).

Concerning CH_4 emissions, cultivation of agricultural and energy crops can reduce the oxidation of methane in aerobic soils, and thereby increase the concentration of methane in the atmosphere. However, this effect usually has a small contribution to life cycle GHG emissions of the bioenergy chain (Delucchi, 2003).

3.3.5. Effects of agricultural residue removal

There is an ongoing debate on the actual possibilities of crop residue removal from agricultural cropping systems for bioenergy production (Lal, 2005; Wilhelm et al., 2004). In order to estimate possible effects, a reference use for agricultural residues must be firstly defined: crop residues can be mainly used as fodder for animals or ploughed back to the field to maintain soil quality. In this case, current experimental evidences on the effect of residue removal on processes like soil organic turnover, soil erosion or crop yields are not consistent because of the strong influence of local conditions (climate, soil type and crop management). In addition, the removal of crop residues for bioenergy production may influence many environmental aspects like N_2O soil emissions, leaching of nitrate and changes in soil carbon pools. There are few references on these effects in the scientific literature, and the patterns are not consistent across references. The use of agricultural residues for bioenergy purposes was investigated by 18 of the reviewed studies, but most of them ignored environmental impact consequences of residue removal, except three studies: one of them assumes that 50% of the residues are left on the field to maintain SOC levels (Spatari et al., 2010), while the others extend the investigation to other aspects, like effects on grain yields, SOC, and N cycle (Cherubini and Ulgiati, 2010; Gabrielle and Gagnaire, 2008).

3.3.6. Allocation

Allocation in LCA is carried out to attribute shares of the total environmental impact to the different products of a system. This concept is extremely important for bioenergy systems, which are usually characterized by multiple products (e.g. electricity and heat from CHP application, rape-cake and glycerine from biodiesel production), and has a large influence on final results. A detailed discussion of the possible allocation methods, with their advantages and disadvantages, is out of the scope of this paper and can be found elsewhere in the literature (Curran, 2007; Ekvall and Finnveden, 2001; Frischknecht, 2000; Heijungs and Guinée, 2007). Fig. 4 reports the abundance of the allocation criteria used in the reviewed studies. This figure shows that there is even distribution among the possible allocation alternatives and the issue of the most suitable allocation procedure is still open. Most of the papers expand system boundaries and then apply substitution method, while others share the environmental burdens of the system among the different co-products by doing partitioning methods (either based on mass, energy or economic market values). Some papers make an attempt to avoid allocation using a suitable functional unit (e.g. reporting results per unit of input biomass, per hectare of agricultural land or per year). A few number of papers explicitly decided to ignore any allocation step, and allocate all the environmental impacts to the main product. Motivations for this assumption can be different: Fu et al. (2003) argue that this is a conservative estimate and should be adopted when sufficient markets for the co-products do not exist yet, while Woods and Bauen (2003) used this approach because of the high uncertainty associated with any of the different allocation methods. Ten papers were found to deal with more than one allocation criterion, with the findings compared in a sensitivity analysis.

The publicly available methodological standards try to overcome such a divergence on allocation methods, proposing specific procedures. However, they all recommend different approaches. The EU directive suggests to allocate GHG emissions according to the energy content of co-products (EU, 2009), the PAS2050 guide recommends to avoid allocation by expanding system boundaries and, if not possible, to use the economic allocation approach (PAS2050, 2008), while the recent ISO GHG protocol recommends to avoid allocation via process subdivision, system expansion or avoided burden and, as the latter possibility, to use market or energy value of co-products as allocation criteria (ISO, 2009).

3.4. Efficient biomass use: vehicle vs. stationary applications

Since competition for biomass resources will be inevitable, it is important to make a selection of the best applications able to ensure the greatest GHG emission savings. The issue is whether biomass should be used as a biofuel in stationary energy systems for CHP or as a feedstock for transportation biofuel production. Relatively few papers among the reviewed studies made an attempt to compare alternative biomass uses (Botha and von Blottnitz, 2006; Cherubini et al., 2009; CONCAWE, 2006; Elsayed et al., 2003; Greene, 2004; Kaltschmitt et al., 1997; Searcy and Flynn, 2008; Uihlein et al., 2008). In order to make such a comparison, a proper functional unit must be chosen, like unit of input biomass. One of the papers concludes that biomass use for electricity production enhances larger GHG savings, especially when compared to first generation biofuels (CONCAWE, 2006). Similarly, Greene (2004) suggests that bioelectricity ensures larger climate change mitigation benefits per tonne of input biomass than transportation biofuels when coal electricity is displaced, but GHG savings become comparable between the two options when natural gas-derived electricity is replaced (Greene, 2004). Another paper based on possible bioenergy uses of agricultural residues reveals that electricity production via direct firing or gasification save about three times the amount of GHG emissions saved by bioethanol and FT-diesel per unit of input biomass (coal electricity is assumed to be displaced) (Searcy and Flynn, 2008). Finally, two papers reveal that heating uses of biomass usually provide greater GHG savings per hectare than conventional biofuels and bioelectricity production systems (Cherubini et al., 2009; Kaltschmitt et al., 1997).

Besides GHG emissions, another paper based on the conversion of bagasse to electricity or bioethanol included in the assessment additional environmental impact categories (Botha and von Blottnitz, 2006). The electricity option is favoured when energy, GHG,

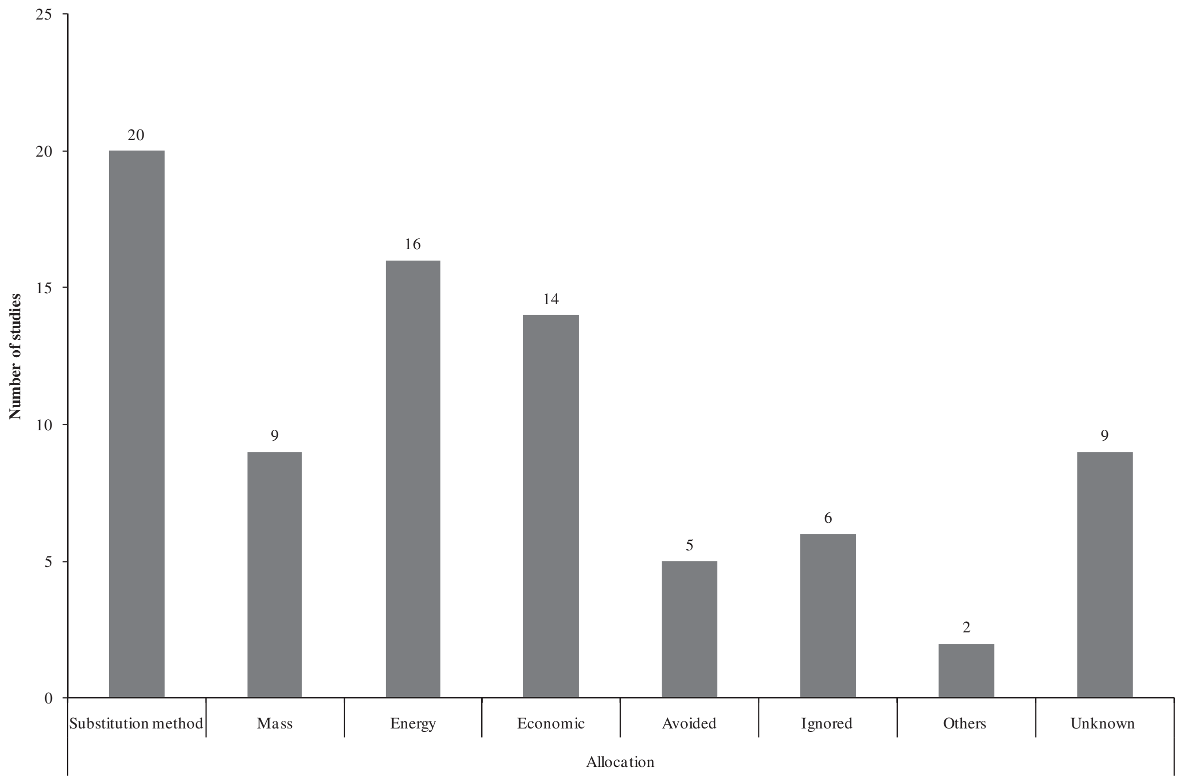

Fig. 4. Allocation criteria used in the reviewed studies.

eutrophication and acidification indicators are considered, while bioethanol production is preferred in terms of resource depletion and toxicity concerns.

3.5. Efficient land use: bioenergy vs. carbon sequestration

In the light of the future expected competition for fertile land, it is becoming increasingly important for policy makers to understand the best uses of fertile land for climate change mitigation. The key question is the following: should a piece of land be used to grow energy crops for bioenergy generation or be used to store atmospheric CO_2 in biomass carbon pools (e.g. forest)? Righelato and Spracklen (2007) argued that land used to store carbon in forest would sequester two to nine times more carbon over a 30-year period than the emissions avoided by the use of biofuel grown on the same land. The Authors emphasize that only the conversion of woody biomass may be compatible with retention of forest carbon stocks. In a recent paper, the relative benefits over 40 years of using land for bioenergy production has been compared with use of the same land for carbon sequestration (Bird et al., 2008). Results show that a combination with high yielding crop species and efficient fossil fuel substitution makes the bioenergy crop option more preferable. By contrast, low efficiency in fossil fuel replacement, independent of growth rate, means that the land is better used for carbon sequestration. Authors here conclude that bioenergy production should be preferred if biomass, from high-yielding plantations, is produced and converted efficiently, displaces GHG-intensive and low-efficiency fossil energy, and if a long term view is taken.

4. Discussion: recent trends and future challenges

In the recent years, numerous studies using a life-cycle approach to estimate environmental performances of bioenergy systems have been undertaken. An increasing number of papers dealing with lignocellulosic biomass, sugarcane or palm oil and located in developing countries was observed, especially in South-Eastern Asia. By contrast, few studies are currently available on promising feedstocks like algae and jatropha oil as well as papers based on advanced biomass processing. The review process also pointed out some methodological lacks. For instance, no one of the 9 studies on bioethanol sugarcane included in the assessment losses of carbon pools after conversion of tropical forests to sugarcane plantations; similarly, just three of the 18 studies based on agricultural residues took into account the environmental impacts related to residue removal. The number of studies located in countries with abundant biomass sources like Russia and Brazil is predicted to increase, as well as the life-cycle investigations of advanced conversion technologies (e.g. pyrolysis, gasification, torrefaction, etc.), whose data scarcity has so far hindered any comprehensive LCA (the few existing studies are mainly approximations based on mass/energy balances). Similarly, the number of papers dealing with biorefinery systems will expand in the near future; this automatically causes an increase in the importance of the role played by allocation in the determination of the final results.

Concerning the LCA outcomes, the determination of the environmental performances is complex, and different combinations of feedstocks, conversion routes, fuels, end-use applications and methodological assumptions may lead to a wide range of results. In particular, different approaches are used to deal with the indirect effects which have a large influence on final figures, and the way by which they should be estimated is still under discussion. The inclusion of these indirect effects in LCA represents the next research challenges for LCA practitioners. In fact, even though valuable improvements were achieved in determining the direct GHG emissions of bioenergy, a standard methodology for the indirect effects is still at a preliminary phase, and further research is needed. It is therefore predictable that future LCA studies will focus on reducing the uncertainties of these current key open issues, e.g.: inclusion in the assessment of indirect LUC effects and their amortization over time, estimation of bioenergy impacts on biodiversity, better determination of fertilizer induced N emissions, and others. However, standardization in GHG balance accounting (either called carbon footprint) of products is particularly perceived as urgent by policy makers, and the methodological standards provided by consultants and stakeholders try to address this need. A variety of policy objectives have motivated governments around the world to promote bioenergy and biofuels, on condition that a certain amount of GHG emission savings is achieved. This means that legislation requires a standardized GHG accounting procedure, encompassing the inclusion of indirect emissions in the life cycle of bioenergy, even if this topic is still in its scientific infancy. In order to cover this gap, several methodological standards have been proposed, as previously mentioned. In most of the cases, these guidelines tend to simplify or overlook concepts and issues of paramount importance, like indirect LUC effects and carbon storage in products. In addition, methodological standards usually limit the assessment to a very limited number of indices and indicators. On one hand, these simplifications can make the overall assessment and interpretation of final results easier, but on the other hand approximation and fixed approaches may have the drawback of misleading and inaccurate conclusions. Therefore, the formulation of regulatory standards in the presence of scientific uncertainty may lead to inefficient or counterproductive methodologies. Finding a compromise is challenging, because a certain degree of simplicity and standardization in sustainability assessment of bioenergy systems is highly desirable nowadays, especially at a governmental and political level, where the best strategies for climate change mitigation should be put into practice as soon as possible. An example of this quandary can be found in the current situation for the Clean Development Mechanism (CDM), a trading framework established by the Kyoto Protocol that allows emission-reducing projects in developing countries to earn and sell carbon credits. Despite the high growth in transportation biofuel investment and research in recent years, not a single project on transportation biofuels has been successfully registered under the CDM (Bird et al., 2008). One of the most important reasons for such an astonishing result is the lack of standard methodologies for assessing GHG balance from agricultural and forest land. In fact, while the CDM focuses on the effects of individual projects, the land use issues discussed in this paper can hardly be attributed to a single activity but tend to be the results of macroeconomic developments. Standardization in the inclusion of indirect effects in LCA may also give the possibility to establish LUC policies aiming at mitigating climate change. In fact, while deforestation and decrease of SOC are threatens for climate change, suitable land use policies may even lead to the opposite effect, given the large potential of GHG mitigation provided by CO_2 sequestration in terrestrial and vegetation carbon pools (UN-REDD, 2008; UNFCCC, 2005).

5. Conclusions

This work points out and discuss the key issues and methodological assumptions responsible for wide ranges and uncertainties in bioenergy LCA. These aspects do not make possible to provide once for ever an exact quantification of the environmental impacts of bioenergy, because too many variables are involved. Some of the key parameters (such as indirect effects) are not well known and strongly depend on local and climate conditions. Although policy makers are claiming for methodological standards, scientific research for estimating indirect effects is still at a preliminary stage. A right balance between simplicity and accuracy should therefore be purpursued.

Acknowledgements

The authors would like to thank the Norwegian research council for funding this work through the Bio-energy Innovation Centre (CenBio).

Table A1
Explanations of the acronyms and abbreviations used in Tables A2 and A3.

Acronym or Abbreviation	Explanation
Scope of the study	
GW	Global warming potential
En	Energy analysis
OG	Other gases besides GHGs (e.g., PM10, SO_x, NO_x, NH_3, etc.)
U/L	Emissions or impacts at local or urban scale
AD	Abiotic depletion potential
AP	Acidification potential
EP	Eutrophication potential
OD	Ozone layer depletion potential
PS	Photochemical smog formation potential
LU	Land use
TP	Toxicity potential (including heavy metals, carcinogenics, and others)
Type of products	
FT	Fischer–Tropsch fuels
SVO	Straight Vegetable Oil
Synthetic biofuel	This group includes methanol, DME, ETBE, H_2, SNG and others
CHP	Combined Heat and Power
Functional unit	
HA	Hectare of agricultural land
OU	Output unit
IU	Input unit
YR	Year
km	Distance (km or mi)
Reference system	
FF	Fossil fuel(s)
BD	Biodiesel
BE	Bioethanol
BG	Bioenergy (generic, CHP)
RE	Renewable energy (except biomass)
Land-use change	
LUC	Land-use change (including changes in C pools and other effects)
Allocation	
MA	Mass
EN	Energy
EC	Economic
SM	Substitution method (or system expansion)
AV	Avoided
IG	Ignored
Type of feedstocks	
Starch crops	Conventional starch crops (corn, wheat, barley, etc.)
Sugar crops, oil crops	Conventional crops (where not specified) like sugar beet, rapeseed, soya, etc.
SC	Sugar cane
PO	Palm oil
JO	Jatropha oil
AL	Algae
SRC	Short rotation coppice (willow, poplar, etc.)
WR	Wood residues
FW	Forest wood
LC	Lignocellulosic crops (switchgrass, miscanthus, etc.)
AR	Agricultural residues (corn stover, wheat and rice straw, etc.)
BL	Black liquor
MSW	Municipal solid waste
UCO	Used cooking oil
MN	Manure
BG	Bagasse

Table A2
Overview of the reviewed studies (I).

Paper	Scope of the study												Type of products									Methodology				Type of feedstocks				
													Transportation biofuels						Heat &/or power											
	Location	GW	En	OG	U/L	AD	AP	EP	OD	PS	LU	TP	Bioethanol	Bio-diesel	FT	SVO	Synthetic biofuels	Biogas	Heat	Power	CHP	Functional unit	Refer. sytem	LUC	Allocation	Starch crops	Sugar crops	Oil crops	Lignocellulosic biomass	Waste/ other
Kaltschmitt et al. (1997))	Germany	x	x				x		x				x	x					x		x	HA, OU	FF		SM	x	x	x	x	
Carpentieri et al. (2005)	Unspecified	x	x				x	x	x	x		x								x		OU	FF						SRC	
Heller et al. (2004)	USA	x	x	x																x		OU	FF	x					WR, SRC	
Pimentel and Patzek (2005)	USA		x										x	x								HA			IG	x		x	FW, LC	
Luo et al. (2009b)	Unspecified		x																			HA	BE		IG				AR	
Malça and Freire (2006)	France		x										x				x					OU	FF		5 methods	x	x			
Yáñez Angarita et al. (2009)	Brazil, Colombia		x											x								OU			EN			PO		
Khatiwada and Silveira (2009)	Nepal		x										x									HA, OU			EN, EC		SC			
Smyth et al. (2009)	Ireland		x															x				HA							LC	
Pleanjai and Gheewala (2009)	Thailand		x											x								HA	BD					PO		
Pettersson and Harvey (2010)	Scandinavia	x													x		x			x		IU	FF							BL
Stichnothe and Azapagic, (2009)	UK	x											x									OU	FF							MSW
Reijnders and Huijbregts (2008a)	South Asia	x												x								IU	FF	x				PO		
Pleanjai et al. (2009)	Thailand	x												x								km	FF		MA					UCO
Reijnders and Huijbregts (2008b)	EU, Brazil	x												x								HA, OU	FF	x	EC			x		
Higo and Dowaki, 2010)	Japan, Papua N.G.	x															x					OU							WR, SRC	
Fleming et al. (2006)	Global	x	x										x				x					km	FF						WR, SRC, LC	
Börjesson and Mattiasson (2008)	Sweden	x	x															x				HA, OU	FF		EN	x				MN
Ndong et al. (2009)	Ivory Cost	x	x											x								OU	FF	x	EN			JO		
Lam et al. (2009)	Malaysia	x	x								x			x								OU			IG			JO, PO		
Kalnes et al. (2009)	USA	x	x											x	x		x					OU			MA, SM			x		
Chester and Martin (2009)	USA	x	x																			OU	FF							MSW
van Vliet et al. (2009)	Global	x	x												x							km	FF		n.a.				SRC	
Yee et al. (2009)	Malaysia	x	x											x								OU	FF, BD	x	EN			PO		
Huo et al. (2008)	USA	x	x											x			x					OU	FF		5 methods			x		
Cherubini et al. (2009)	Global	x	x										x	x	x			x	x	x	x	HA, km	FF	x	SM	x	x, SC	x	LC, SRC	MN
Kadam, 2002a)	India	x	x	x		x	x	x				x	x									IU	FF, BG		n.a.					BG
Lardon et al. (2009)	EU	x	x			x	x	x	x	x	x	x		x								OU	FF, BD		EN			AL		
Spatari et al. (2005)	Canada	x	x	x									x									km	FF		n.a.				AR, LC	
Williams et al. (2009)	USA	x	x								x	x	x									OU	BE						FW, AR, LC	
Luterbacher et al. (2009)	Switzerland	x	x																x	x		OU			SM				WR	MN
Luo et al. (2009a)	Brazil	x	x			x	x	x	x	x		x	x									km	FF		EC		SC			
Papong and Malakul (2010)	Thailand	x	x	x		x	x	x	x	x		x	x									OU			IG, EN				LC	
Bernesson et al. (2006)	Sweden	x	x	x			x	x		x			x									OU			SM, EC, EN	x				
Talens Peiró et al. (2010)	Spain	x				x	x	x	x	x		x		x								OU			n.a.					UCO
González-García et al. (2009)	Spain	x		x		x	x	x	x	x		x	x									km	FF		MA, EC				LC	
Botha and von Blottnitz (2006)	South Africa	x				x	x	x				x	x								x	IU	Sugar							BG
González-García et al. (2009b)	Spain	x		x			x	x		x			x									km	FF		AV				LC	
Kim and Dale (2008)	USA	x	x	x			x	x		x			x									OU	FF	x	SM	x				
Spatari et al. (2010)	USA	x	x	x									x									OU	FF	x	SM				AR, LC	
Xiao et al. (2009)	China	x	x	x			x	x		x		x					x					OU							FW	
Fu et al. (2003)	Canada	x	x				x	x	x	x		x	x									km	FF		IG				LC	
Halleux et al. (2008)	Belgium	x	x				x	x				x	x	x								km	FF		SM		x	x		
Luo et al. (2009c)	USA	x				x	x	x	x	x		x	x									km	FF	x	4 methods				AR	
Searcy and Flynn (2008)	Canada	x											x		x					x		OU	FF		SM				AR	
Panichelli et al. (2009)	Argentina	x	x				x	x			x	x		x								km	FF, BD	x	EC			x		
Harding et al. (2008)	South Africa	x				x	x	x	x	x		x		x								OU	BD		MA			x		
Hu et al. (2008)	China	x	x	x										x								OU	FF		MA			x		

Table A3
Overview of the reviewed studies (II).

Paper	Scope of the study												Type of products									Methodology				Type of feedstock				
													Transportation biofuels					Heat &/or power												
	Location	GW	En	OG	U/L	AD	AP	EP	OD	PS	LU	TP	Bioethanol	Bio-diesel	FT	SVO	Synthetic biofuels	Biogas	Heat	Power	CHP	Functional unit	Refer. sytem	LUC	Allocation	Starch crops	Sugar crops	Oil crops	Lignocellulosic biomass	Waste/other
Hossain and Davies (2010)	USA	x	x	x										x		x						OU	FF, BD					x		
Cherubini and Ulgiati (2010)	Austria	x	x			x	x	x	x	x		x	x								x	YR	FF	x	AV				AR	
Cherubini and Jungmeier (2010)	Austria	x	x			x	x	x	x	x		x	x								x	YR	FF		AV				LC	
Uihlein and Schebek (2009)	Germany	x	x				x	x	x		x	x	x								x	IU	FF		AV				AR	
Corti and Lombardi (2004)	Italy	x																		x		OU	FF						SRC	
Varun et al. (2009)	Global	x																		x		OU	FF, RE						AR, SRC	MN
Petersen Raymer (2006)	Norway	x																	x			IU	FF						WR, FR	
Jungmeier and Spitzer (2001)	Austria	x																	x		x	OU	FF		AV	x		x	AR, LC, SRC	
Ishikawa et al. (2006)	Japan	x	x																		x	YR								MN
Eriksson et al. (2007)	Sweden	x	x																x		x	OU	FF, RE		SM				FW	
Goglio and Owende (2009)	Ireland	x	x																		x	HA	BG		EN				SRC	
Lettens et al. (2003)	Belgium	x	x																	x		OU, HA	FF	x					LC, SRC	
Styles and Jones (2007)	Ireland	x																		x		OU, HA	FF	x					LC, SRC	
Uihlein et al. (2008)	Germany	x	x			x	x	x	x		x	x									x	OU	FF		SM				LC	
Felder and Dones (2007)	Switzerland	x				x	x	x	x		x	x					x		x			OU, km	FF						FW	
Mann and Spath (1997)	USA	x	x	x																x		OU		x					SRC	
Thornley et al. (2009)	UK	x		x							x									x	x	OU, HA							AR, LC, SRC	
Gasol et al. (2009)	Spain	x	x			x	x	x	x	x		x									x	IU	FF, BG						SRC	
Hartmann and Kaltschmitt (1999)	Germany	x	x	x			x													x		OU	FF						FW, AR	
Kadam (2002b)	USA	x	x	x		x														x		OU	FF					AL		
Ramjeawon (2008)	Mauritius	x	x			x	x	x	x	x		x									x	OU	FF		EC					BG
Rafaschieri et al. (1999)	Italy	x	x				x	x	x	x		x								x		OU	FF						SRC	
Chevalier and Meunier (2005)	3 countries	x	x				x	x													x	YR	FF						AR	
Solli et al. (2009)	Norway	x			x		x	x	x	x		x							x			OU	BG						FW	
von Blottnitz and Curran (2007)	Global	x	x			x	x	x	x	x		x	x									HA, km	FF			x	x, SC	x	LC, SRC	
Gabrielle and Gagnaire (2008)	France	x	x				x	x	x				x									OU		x	MA				AR	
Grey literature																														
Elsayed et al. (2003)	UK	x	x										x	x		x			x	x	x	OU	FF		SM, EC	x	x	x	FW, AR, LC	MN
CONCAWE (2006)	EU	x	x										x	x	x		x	x				OU, km	FF	x	SM	x	x, SC	x	WR, AR, SRC	MN
GM et al. (2002)	EU	x	x										x		x		x	x				km	FF		n.a.		x		WR	
Quirin et al. (2004)	Global	x	x										x	x		x	x	x				HA, km	FF		SM	x	x, SC	x	WR, FW, LC	
Brinkman et al. (2005)	USA	x	x	x	x								x									km	FF		n.a.	x			LC	
den Uil et al. (2003)	Netherland	x	x				x	x		x		x			x							%	FF		n.a.				WR, FW, SRC	
Varela et al. (2005)	EU	x	x										x	x	x		x	x				km	FF		SM	x	x	x	WR, LC	UCO, MN
Wu et al. (2008)	USA	x	x	x	x								x									km	FF	x	EN	x			LC	
Greene (2004)	USA	x											x		x		x			x		IU	FF		EN				LC	
IEA (2005)	Global	x											x	x								%	FF		n.a.	x	x, SC	x	LC	
Wang et al. (1999)	USA	x	x										x									km	FF	x	EN	x			LC, SRC	
Woods and Bauen (2003)	UK	x	x										x	x	x		x					OU	FF		IG	x	x		AR, SRC	
Delucchi (2005)	4 countries	x	x	x									x		x		x					km	FF	x	SM	x			LC, SRC	
Mortimer et al. (2003)	UK	x	x											x								OU	FF		EC			x		
Gartner et al. (2003)	Germany	x	x				x	x	x	x				x								OU	FF					x		
Zah et al. (2007)	Switzerland	x	x				x	x	x	x		x	x	x			x	x				km	FF	x	EC	x	x, SC	x, PO	SRC, LC	UCO, MN
Shapouri et al. (2002)	USA		x										x									OU			4 methods	x				
Macedo et al. (2004)	Brazil	x											x									IU	FF		n.a.		SC			
Cowie (2005)	Australia	x																		x		HA, IU, OU	FF	x					FW, SRC	
Robertson (2004)	New Zealand	x																			x	YR	FF						WR	

Appendix A. Overview of the studies

This appendix reports information on each of the reviewed studies. In Table A1, the explanations of the acronyms or abbreviations used in the following tables are shown. Features of the reviewed LCA bioenergy studies are summarized in Tables A2 and A3.

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