Energy Balance & Greenhouse Gas Emissions of Biofuels from a Life Cycle Perspective

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Introduction

Policies and targets for biofuels have been set in several countries around the globe. The main drivers for the setting of such policies are potential contributions to energy security, climate change mitigation and rural development. However, there is ongoing and intense debate over whether biofuels are really capable of meeting these expectations. In particular, the sustainability profile of biofuels has been recently questioned by several researchers (e.g. Doornbosch and Steenblik 2007; Fargione et al. 2008; Searchinger et al. 2008). The most frequently cited issues of concern include direct and indirect land use impacts, carbon stock decreases, water depletion and pollution, biodiversity loss, and air quality degradation. In addition to these environmental problems, critics point to potential economic and social conflicts deriving from energy/food source competition. As a consequence, policies supporting biofuels are currently being challenged.

In response to a call for considering all stages of biofuel production, the Life Cycle Assessment (LCA) methodology has been increasingly used to assess the potential benefits and/or undesired side effects of biofuels. The LCA methodology studies and evaluates the environmental flows related to a product or a service during all life cycle stages, from the extraction of raw materials to the end of life. It is regulated by the ISO 14040:2006 and 14044:2006 standards which provide the principles, framework, requirements and guidelines for conducting an LCA study. LCA is more frequently used to support policy making in many countries and thematic areas such as eco-design, integrated product policy, waste prevention and recycling, and sustainable use of natural resources.

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Explicit reference to LCA is made in the European Commission Renewable Energy Sources Directive proposal, the US Energy Independency and Security Act, the German Sustainable Biofuel Obligation draft, the Swiss directive on mineral oil tax redemption for biofuels, and the UK Renewable Transport Fuel Obligation.

The present chapter evaluates and critically assesses the most relevant existing LCA studies in the area of biofuels, particularly those relating to the energy balance and greenhouse gas (GHG) emissions of biofuels produced from a range of crops and other biomass feedstocks using various conversion technologies. Through this analysis, we aim to:

- identify the most rigorous methodologies, by evaluating the main hypothesis and assumptions, the quality of data, the transparency of the calculation processes, and the replicability of results
- determine a range of results, for each indicator, pathway, and crop
- pinpoint the main parameters and life cycle stages critically influencing the results
- highlight the main research gaps, and suggest possible measures for improvement
- derive a list of key messages and recommendations for policy makers, the industry and the research community

Method of review and selected studies

In total, 30 studies on different types of biofuels from various crops and technology conversion processes were short-listed after an extensive Internet search and targeted requests to experts in the field of bio-energy and life cycle analysis. The reports selected are amongst the most recent and internationally acknowledged reference studies in the sector, and guarantee a certain balance in terms of geographical scope and feedstock chain coverage. The screening process has taken into account over 60 different types of documents including technical reports, scientific articles and publicly available executive summaries of proprietary reports. The full spectrum of studies reviewed is reported in the list of references. The documents used as a basis for the determination of the range of results are briefly presented in Table 5.8 at the end of this chapter.

In addition to the studies listed in the table, some other articles and reports which were excluded during the screening phase[1] have still been useful as a cross-reference in order to compare and validate some critical parameters (e.g. assumptions on the fertilizer rate, yields, or efficiencies of conversion processes).

Determination of the range of results for energy consumption and GHG emissions

Wherever possible, a comparable sample of studies was selected for a certain number of biofuel and crop pathways after a filtering process based on the following main parameters: biofuel type, feedstock type, geographical scope and conversion technology process. Results are provided for both first- and second-generation biofuels, as well as for both ethanol and biodiesel fuels.

Due to the limited sample of studies focusing on stationary applications, the box 5.1

General observations

A preliminary analysis of the all studies reviewed indicates a number of concerns regarding the quantity and quality of LCA and other environmental impact studies of biofuels. The following is a list of our general observations:

- The number of full LCA studies continues to increase but is still relatively small.
- The majority of studies are limited to European or US conditions, and are based on western agricultural processes and average conversion technologies. The only notable exception so far is Brazil.
- The majority of studies refer to the so called first-generation technologies. Nevertheless, a small number of reports are appearing which investigate second generation technologies.
- Most studies focus on the more "traditional" feedstocks such as corn, sugarcane, rapeseed, wheat. Very few studies have assessed life cycle environmental impacts for new crops more recently evaluated for biofuel production, such as jatropha and sweet sorghum. Due to confidentiality reasons, it was not possible to show the results contained in these studies in the present analysis.
- Most studies only include energy consumption (sometimes only non-renewable energy, sometimes total energy) and CO_2 emissions. A few studies also include other relevant impact indicators as acidification potential, eutrophication potential, ozone depletion potential and various toxicity potentials. However, quite remarkably, very few studies include water use impacts.
- Methodologies to develop biodiversity quality indicators are still under discussion, and this is reflected in the current review where no study presents results in terms of biodiversity.[2]
- Very few studies take into account land use impacts driven by biofuel crop production. More specifically, only one third of the studies define an alternative land use reference system and calculate the carbon stock. Potential impacts in terms of indirect land use change driven by increased bioenergy demand are not considered in the sample analyzed.
- The transparency level of reports is quite heterogeneous with respect to hypothesis and assumptions, yields, heating values, emission factors, and other background methodological choices. Very few studies include a data quality review according to the requirements of the ISO standards for LCA.
- Heterogeneity observed in terms of treatment of co-products and allocation methods followed.
- Social issues are very often overlooked in the studies. This is not surprising, given the purely environmental focus of LCA technique.
- Many databases and LCA softwares are used to model data. In particular, some of the life cycle inventory databases used in the studies appear relatively old. This affects the quality of results, regardless of the quality of the primary data collected.

focus of the comparison below will be restricted to transportation use only. Given the small number of studies presenting results, impact category indicators other than fossil energy consumption and GHG emissions are reviewed separately.

In order to summarize and compare results, energy consumption and GHG emissions are expressed here in terms of percent improvement with respect to conventional fuels (gasoline and diesel). This percentage is sometimes expressed as such in the original studies. However, other studies report GHG emissions in grams carbon dioxide or carbon dioxide equivalent (g CO_2, g CO_2eq).

The well-to-tank (WTT) studies relate emissions with respect to a megajoule (MJ) of fuel (g CO_2eq MJ^{-1}), but in order to compare fossil and alternative fuels, they also include the direct emissions of gasoline or diesel during the use phase in the motor combustion (i.e. well-to-wheel). The well-to-wheel (WTW) studies express results per kilometer (km), calculated as the sum of the tank-to-wheel (TTW) direct emissions (in gCO_2eq km^{-1}) plus the indirect emissions calculated as the product of the WTT emissions (g CO_2eq MJ^{-1}) and the TTW energy consumed by the vehicle per unit of distance covered (MJ km^{-1}).[3] In both cases the percentage improvement has been calculated by dividing the biofuel emissions by the corresponding fossil fuel emissions.

In a few other cases where published results were expressed in other units (e.g. per ton of fuel), the percentage improvement had to be recalculated using standard fuel properties and standard tank-to-wheel fuel energy use. This obviously introduced a certain level of (additional) uncertainty. Therefore the numbers presented in the following tables should be read only as indicative values.

First-generation biofuels

In the following paragraphs, we provide ranges of energy and GHG savings for first-generation ethanol (Table 5.1) and biodiesel (Table 5.2) pathways. Results are expressed as percentage of improvement with respect to conventional fuels. The values in brackets correspond respectively to the lower and upper limits of error bars, as reported in the study. The ranges not in brackets are instead related to different scenarios or assumptions in the respective study.

The ethanol feedstocks investigated include corn, wheat, sugarcane and sugar beet. Not enough studies were collected to allow any meaningful comparison for other crops such as barley, cassava, rye, whey and sweet sorghum. As for potato, the results shown in the three studies collected were very inconsistent[4]; therefore it was decided not to report them here.

For biodiesel, we report for rapeseed, soybean, sunflower and palm oil pathways. Lack of published data precluded any comparison with other biodiesel feedstocks such as coconut, jatropha, tallow and used vegetable oils.

Corn. The results for ethanol from corn show a large heterogeneity amongst the studies. In the case of fossil energy consumption, they range from improvements of 60% or more improvement in the most favorable cases, to only 26% as the lowest performance compared to conventional gasoline. Analyses of GHG emissions show, in some cases, negative results.

In their review of some U.S. studies, Farrell et al. (2006) explain negative results for ethanol by the lack of impact allocation to co-products and the use of old data. Farrell et al. also show in their sensitivity and modeling analysis that if consistent assumptions are used, comparable results can be extrapolated from the different studies. However, they also highlight the importance of the energy mix. Several studies clearly indicate that the use of coal as process fuel leads to a worsened perormance of corn ethanol with respect to gasoline. This is reflected in the article of Wang et al. (2007) in which the ethanol performance in terms of GHG emissions ranges from slightly negative values to

Table 5.1 Range of results for 1st generation bio-ethanol from maize, wheat, sugar cane, and sugar beet. Reported values do not include GHG emissions associated with land use change

Author	Year	Scope	Fossil Energy Improvement	GHG Improvement	Type
Farrell et al.[5]	2006	USA	34% ; 16 %[6]	13% ; -2%[7]	maize
Grood & Heywood	2007	USA	68%[8]	20% (-47%, +58%)[9]	maize
Unnasch & Pont	2007	USA	33-64%	-5%, +30%[10]	maize
Wang et al.	2007	USA	36% (30-70%)[11]	19% (-3%, +52%)[12]	maize
De Oliveira et al.	2005	USA	26%	-4%	maize
Shapouri et al.	2002	USA	39%	35%	maize
Zah et al.	2007	US ,China (util)	37%[13]	18%	maize
Quirin et al.	2004	Various	16-85%	18-90%	wheat
Elsayed et al.	2003	Various	61%	64%	wheat
Edwards et al.	2007	Europe +	42% (22-115%)[14]	32%[15]	wheat
S&T Consultants	2006	Canada	61%	48%	wheat
Lechon et al	2005	Spain	42%	78%	wheat
Ecobilan	2002	France	57%	60%	wheat
Various (Ecofys & SenterNovum)	2005	Europe	40%	32%	wheat
De Castro	2007	Brazil, Africa	90%	>100%	cane
Smeets et al.	2006	Brazil	>90%	85 - 90%	cane
Edwards et al.	2007	Europe +	>90-100%+	~87%	cane
Unnasch & Pont	2007	USA	86%	84%	cane
De Oliveira et al.	2005	Brazil, USA	78%	>70%	cane
Macedo et al.	2004	Brazil	91%	86%	cane
Zah et al.	2007	Brazil, China	89%[16]	85%	cane
Smeets et al.	2006	NA	NA	~35 - 55%	beet
Edwards et al.	2007	Europe +	48% (24-73%)[17]	48% (32-65%)[18]	beet
Ecobilan	2002	France	58%	61%	beet
Elsayed et al.	2003	Various	~58%	51%	beet
Zah et al.	2007	China	73%[19]	65%	beet
Gnansounou & Dauriat	2004	Switzerland	85%	40%	beet

significant improvements depending on the type of technology used in the milling plant (wet vs. dry), the state-of-the art of the milling plants (current average vs. 2010 average technology), the process fuel used (natural gas, coal or renewables), the use of co-generation or not, and the fate of distiller grains and solubles (DGS). The type and intensity of cultivation is an even more important factor which can heavily affect the GHG emission balance.

Wheat. All studies of wheat ethanol converge on indicating that the use of wheat ethanol leads to net benefits compared to conventional gasoline, but there are significant variations in the values provided.

sugarcane. Ethanol from sugarcane is the pathway with the most convergent results. Most of studies analyzed are energy and GHG balance studies, with only the EMPA report (Zah et al. 2007) showing a wider set of environmental impact category indicators. As shown in the table, all studies agree that ethanol from sugarcane can allow GHG emission reduction of over 70% compared to conventional gasoline. Higher values (also beyond 100%) are due to credits for co-products in the sugarcane industry. This reflects the recent trend in Brazilian industry towards more integrated concepts combining the production of ethanol with other non-energy products and selling surplus electricity to the grid.

Sugar beet. The last investigated pathway is ethanol from sugar beet. As opposed to other crop pathways like corn and wheat, results seem to converge around an average improvement in fossil energy consumption and GHG emissions of at least 50% compared to conventional gasoline.

Rapeseed. The studies reviewed estimate a net GHG saving for the biodiesel produced under this pathway. Results range from a minimum benefit of approximately 20% to a maximum of about 80% compared to conventional diesel, with most studies converging around the 40-60% interval. The two studies showing the most diverging results are, on one hand, EMPA which offers the more pessimistic results and, on the other, Ecobilan which presents the most favorable ratio for rapeseed-oil methyl ester. Several studies hint to the fact that changes in agricultural practices and technology improvement in the fertilizer industry can lead to a significant reduction of overall GHG emissions balance of 1st generation biofuels.

Soybean. The main findings for soybean biodiesel show wide variation, ranging from significant improvements to considerable net worsening. The main reasons which explain such huge differences are the agricultural yields and the assumptions made on allocation of impacts and the fate of glycerine. The latter originates as a co-product during the production of soybean oil methyl ester and can be used to substitute animal feed or as a chemical. A third factor which influences the results is the type of energy system used in the transformation process, and particularly the existence of modern natural gas-fired co-generation plants (as reported for example in Lechón et al. 2006) versus older and less-efficient facilities (petroleum is reported as an input for the esterification process in Unnasch and Pont 2007). The end of life of solvents used to extract soybean oil (e.g. whether the hexane is recovered or not) has some influence on results, too. The particularly negative performance depicted by Zah et al. (2007) in the Brazilian case is

Table 5.2 Range of results for rapeseed, soybean, sunflower, palm oil based on a selected number of studies (w/o land use change)

Author	Year	Scope	Fossil Energy Improvement	GHG Improvement	Type
de Castro	2007	Brazil/ Africa	NA	~20 - 40%	rapeseed
Quirin et al.	2004	various	~60% (to > 100%)	~20 - 85%	rapeseed
Elsayed et al.	2003	various	65%	53%	rapeseed
Puppan	2001	Belgium/ Germany	55%	45%	rapeseed
Edwards et al.	2007	Europe/ Brazil	56 - 61%	41-47%	rapeseed
Lechon et al.	2006	/ Spain	79%	56%	rapeseed
Ecobilan	2002	France	80%	~80%	rapeseed
Choudhury et al.	2002	Europe	43%	~55% (30-85%)	rapeseed
Zah et al.	2007	various	46-54% [20]	64%	rapeseed
Various (Ecofys, SenterNovem)	2005	various	57%	40%	rapeseed
De Castro	2007	Brazil/ Africa	NA	53-78%	soybean
Larson	2005	Europe/ N. Amer.	-70%	45-75%	soybean
Quirin et al.	2004	various	>100%	68-110%	soybean
Edwards et al.	2007	Europe/ Brazil	67%	67%	soybean
Unnasch and Pont	2007		10%	10%	soybean
Lechon	2006		79%	56%	soybean
Zah et al.	2007		27% (BR) - ~ 40% (USA)	-17% (BR) - ~40% (USA)	soybean
Quirin et al.	2004	various	72-139%	35-110%	sun-flower
Edwards et al.	2007	Europe/ Brazil	67%	67%	sun-flower
Lechon et al.	2006	Spain	76%	66%	sun-flower
Ecobilan	2002	France	83%	83%	sun-flower
Reinhardt et al.	2007	Average production	7%	31%	Palm oil
Unnasch and Pont	2007		10%	8-12%	Palm oil
Lechin et al.	2006	Thailand/ Spain	64%	40%	Palm oil
Zah et al.	2007	Malaysia/ China	64%	70%	Palm oil
Beer et al.	2007	Indonesia & Malaysia	NA	~80% (-868% w/ rainforest conversion; 2070% w/ peat forest conversion	Palm oil

Table 5.3 Range of results for second-generation biofuels (ethanol and biodiesel) based on a selected number of studies (w/o land use change)

Author	Year	Scope	Fossil Energy Improvement	GHG Improvement	Fuel & Feedstock
Farrell et al.	2006	USA	93%	88%	Cellulosic ethanol from switchgrass
Quirin et al.	2004	various	58%-138% (EtOH) 72 - 100% (BtL)	15-115% (EtOH) 200% (BtL)	Ethanol and BtL from lignocellulose
Elsayed et al.	2003	various	~100%	84%	Cellulosic ethanol from wheat straw
Edwards et al.	2007	Europe/ Brazil	76-91%	76--88%	Cellulosic ethanol from wheat straw, wood
Grood and Haywood	2007	USA (AL, IA)	76%	93-98%	Cellulosic ethanol from switchgrass
Unnasch and Pont	2007	USA (CA) +	70-80%	10-102%	Cellulosic ethanol from poplar, switchgrass, forest residue
Wang et al.	2007	USA	93%	86%	Cellulosic ethanol
Veeraraghavan & Riera-Palou	2006	UK	78-102%	88-98%	Cellulosic ethanol from wheat straw
Choudhury et al.	2002	Europe	NA	~70% (poplar) - 80% (residual)	Cellulosic ethanol from poplar and Fischer Tropsch diesel from residual wood
Jungbluth et al.	2008	Europe	37-61%	28-69%	BtL using straw, forest wood, or short-rotation wood
Zah et al.	2007	Swiss +	73-79%	65%	Cellulosic ethanol from grass and wood
Reinhardt et al.	2006	Germany/ Central Europe	50-80%	60-115%	BtL diesel from different types of biomass
Baitz et al.	2004	Germany	NA	61-91%	Sundiesel produced via gasification & Fischer Tropsch synthesis of wood waste

due to the computation of the emissions caused by the conversion of natural vegetation (jungle) in agricultural land. A more in-depth comparative analysis is impeded by the lack of disaggregated information on the assumptions taken on each life cycle stage. In this regard, only the studies of Lechón et al. (2006) and, to a lesser extent, Zah et al. (2007) provide details on assumptions retained and the sources consulted.

Sunflower. According to most of the studies reported, the use of sunflower biodiesel to replace conventional diesel leads to significant environmental improvements in terms of both energy savings and GHG emissions. However, a wide range of values is observed, which depends on the different assumptions retained in the studies.

Palm oil. Palm oil based diesel compares favorably to conventional diesel, in terms of GHG emissions. However, if previously non-cultivated areas are converted for palm oil production, the net resulting balance can be dramatically negative. This issue is well described in Beer et al. (2007), which compares a base case scenario from cropland with palm oil from cleared rainforest and cleared peat forest. Results change from 80% improvement to over -800 and -2000%, respectively.

Second-Generation Biofuels

The main results for second-generation technologies are reported in Table 5.3, both for the ethanol and biodiesel pathways. In general, studies find considerable net improvements for second generation technologies. However, there are some remarkable exceptions. In particular, Jungbluth et al. (2008) highlight that, when the full life cycle is taken into account, some biomass-to-liquid fuels (BtL) from agricultural biomass (particularly short-rotation wood) do not produce significant energy and GHG emission savings compared to the fossil fuel reference. On the other hand, BtL processes using wood from forestry or biomass residues show a much better profile, thus representing a good possibility to reduce GHG emissions. Based on this outcome the authors conclude that the use of agricultural biomass needs further improvement in order for BtL fuels to fully compete with fossil alternatives.

Other life cycle impact indicators

Besides GHG emissions, other environmental impacts can arise from feedstock production, processing, and distribution of biofuels. In particular, the impacts on land used for feedstock production, and the corresponding effects on water, soil quality, and biodiversity can be substantial. These impacts depend on various factors including feedstock, cultivation practice, land management, location, and downstream processing routes.

The number of studies that assess a wider set of impact category indicators besides global warming potential is still limited; only seven of the reviewed studies compare results for a minimum of five impact category indicators (Table 5.4). Less than one third of the studies reviewed present results for acidification and eutrophication potential[21] only six assess the toxicity potential (either human toxicity, eco-toxicity, or both), seven include summer smog (i.e. Photochemical ozone potential), four include ozone depletion, and just three address abiotic resource depletion potential.[22] Any possibility of grouping is further impeded by the fact that the studies reviewed diverge in the goal and scope, system boundaries, product system, functional units, and presentation of results. Also, background assumptions are not always clearly reported.

The lack of a consistent set of studies focusing on a wider set of environmental impact indicators is interesting from a methodological point of view. In principle, the goal of LCA is to depict a comprehensive picture of the product system investigated in order to assess trade-offs between different environmental impact indicators. In the case of biofuels, however, this does not yet seem to be the normal practice.

Given the small number of comparable studies presenting results for non-GHG environmental impacts, it is very difficult to provide a reasonable range of results for different crops and technology conversion pathways. Nevertheless, some general indications and conclusions can be drawn from the analysis of the documents.

Acidification Potential. Most studies indicate that biofuels underperform conventional fuels in terms of acidification potential. This is mainly due to the manufacturing and use of synthetic fertilizers. In the case of rapeseed biodiesel, the impact ranges from -30 to -60% compared to conventional diesel (Puppán 2001; SenterNovem 2005; Zah 2007). This range is attributable to the different geographical contexts, time horizons and sulphur content of the substituted conventional diesel of the studies considered. Puppán (2001) compares the results of two studies carried out at the end of the nineties in Belgium and Germany respectively. While the Belgian study presents very unfavorable results in terms of acidification potential for rapeseed, the German one depicts a more

Table 5.4 Number of studies presenting results for the indicators other than GHG

Indicator	No. of Studies
Acidification Potential	8
Eutrophication Potential	8
Toxicity Potential	6
Photochemical ozone potential	7
Ozone depletion potential	4
Abiotic resource depletion	3

optimistic scenario. A comparison of the fertilizer rate assumed in the two studies indicates that the Belgium study systematically adopted higher fertilizer and pesticide application rates, varying from +30% to ten times more depending on the chemical component. As reported by Puppán (2001), the difference is due to the more favorable agricultural conditions for rapeseed growing in Germany compared to Belgium. The particularly bad agricultural conditions of Belgian rapeseed cultivation seem to be the primary cause of higher inputs, which explains why the results of the Belgian study are very different from those found in similar studies conducted in Germany and the United States. Thus, any generalization of results must be done very carefully, taking into account the precise scope of the study.

All studies concur that if proper agricultural practices are followed biofuels can improve their environmental profile. For example, Kägi et al. (2007) reported that organic agriculture can substantially decrease the acidification potential for rapeseed in the cultivation phase, leading to only 9% of the impacts compared to intensive agricultural practice. Lechón et al. (2007) suggested that the use of other than synthetic fertilizers like ashes from residual biomass combustion can be another option to reduce the impacts of biofuels.

Slightly better but still negative values are observed for ethanol from wheat, where the biofuel is responsible for a 25% increase compared to conventional gasoline, according to SenterNovem (2005). Again, the emissions of ammonia, nitrogen (NOx) and sulphur (SOx) oxides from fertilized fields are responsible for this effect. The German study reported in Puppán (2001), also shows more favorable results for ethanol from wheat, leading to roughly the same values as the conventional fuel. According to Kägi et al. (2007) bio-agriculture can lead to reducing impacts by 70% per kg of dry matter of harvested product compared to intensive culture.

Unfavorable results are found in Lechón et al. (2007) who show significantly pejorative results both in the case of bioethanol and biodiesel. However, these are not easily comparable to the ones observed in the other studies, since the authors assume a blend of different crops (wheat and barley in the case of ethanol; soybean, rapeseed, palm oil and sunflower in the case of biodiesel). Gnansounou and Dauriat (2004) also show dramatically worse results in terms of acidification for cereals compared to gasoline. They state that the combustion phase also plays a role since current engines are optimized to run on conventional gasoline and not on ethanol, which has a higher oxygen content and can lead to higher tailpipe emissions. Negative results are found also for corn and soybean in Kim and Dale (2005). On the contrary, used vegetable oils lead to a 9% improvement according to Nieder and Narodoslasky (2004).

Other Environmental Impacts. The studies show agreement on many of the environmental impacts investigated. Pessimistic findings are found for eutrophication and observed for almost all pathways and crops in the sample. A convergence is also observable in the case of ozone depletion, where all studies attribute pejorative results to biofuels compared to traditional fuels. As far as summer smog is concerned, on average slightly favorable results are observed in almost all studies for biofuels compared to fossil fuels. Notable exceptions are Zah et al. (2007) and Lechón et al. (2007), which estimate negative results for various ethanol and biodiesel chains.

Other impact categories show divergent result, including results for toxicity. For example, while Zah et al. (2007) and Puppán (2001) report substantial benefits in the case of rapeseed compared to conventional diesel, very unfavorable results are provided by SenterNovem (2005). Lechón et al. (2007) present negative results for biodiesel and favorable results for bioethanol compared to gasoline. However, it is worth highlighting that at present there is no general consensus on the characterization factors to be used while assessing toxicity effects. Therefore the results of different studies should be interpreted and compared carefully in order to avoid misleading conclusions.

Similarly, the LCA community has not agreed yet upon indicators on soil quality preservation and biodiversity. Measurement of land use impacts on biodiversity is in fact a complex task, and additionally there is no unanimous definition of biodiversity (Koellner and Scholz 2008). At present there are some methodological attempts to set up and include an impact indicator on biodiversity, but scientific consensus has not been reached yet as regards species vs. ecosystem level indicators. [23]

Finally, with respect to second generation biofuels, few studies present a comprehensive assessment including a wider set of environmental impact indicators. Zah et al. (2007) assess ethanol from grass and wood. Their results are slightly favorable to bioethanol in terms of acidification potential, summer smog and very favorable for eco-toxicity. However, in the case of eutrophication, bioethanol from grass and wood underperform conventional gasoline. In their LCA of SunDiesel, Baitz et al. (2004) show very encouraging results ranging from 5 to 42% improvement for acidification, 3 to 29% for eutrophication and 89% to 94% in the case of summer smog, depending on the scenario under consideration. Reinhardt et al. (2006) assess different routes for biomass-to-liquid (BtL) fuels. All investigated pathways present favorable results in terms of summer smog, mixed results for acidification and eco-toxicity, and unfavorable results in terms of eutrophication.

Land Use Change. Land use has also been introduced as an impact category in LCA (Koellner and Scholz 2008). Recently, several studies have emphasized the importance of land use change on the overall GHG balances (e.g. Fargione et al. 2008; Searchinger et al. 2008). Land use change for biofuel production can occur in two ways: directly, when non-crop land is converted to energy crop lands (e.g. grassland is used to plant rapeseed for biodiesel), or indirectly, when existing food and feed crop acreage is converted for use as energy crops, thus inducing new production of the food/feed crop elsewhere, at the expense of native habitats, to meet total demand. Second order effects may also occur (e.g. expanded soybean production in pastureland leads to the conversion of rainforests into pastureland).

Although it is very difficult to model the complex interactions in the agricultural markets between demand for different crops and land-use change, there is mounting concern that current biofuel policies do not adequately take into account the risk of GHG emissions occurring indirectly (IEA 2008). Some recent studies draw the conclusion that the effects of land use may completely offset the potential GHG emission reduction of biofuels, and even substantially increase emissions compared to conventional transport fuels (e.g. Fargione et al. 2008).

Our review showed that only a few studies take into account direct land use impacts driven by biofuel crop production. More specifically, less than one third of the studies define an alternative land use reference system and calculate the carbon stock. Potential impacts in terms of indirect land use change are not considered in the studies. Although an assessment of these impacts is important, their omission is not surprising as by its very nature LCA was not designed to address this specific issue. LCA was developed as a method to compare the environmental profiles of products and services on a “per-unit” basis (“functional unit”), and is, in most applications, a static approach. With competing uses of biomass in different sectors, the scope of LCA needs extension to address cross-sectoral issues. A "full-scope" LCA would avoid the problem of indirect effects through accounting of all relevant flows and resources as well as the competing uses, so that it would develop from product-oriented "per-unit" focus into a more material/energy-flow system approach (OEKO 2008).

The approach to combine an LCA study with a macro-economic agro-modeling is a step forward. With this approach, changes in land use of different regions can be connected to information on carbon stocks and carbon release data to provide land-use related GHG emissions due to changes in biofuel support policies. This would allow the evaluation of potential absolute impacts of mass-scale deployment of biofuels, as opposed to the marginal impacts assessed by traditional LCA studies.

Traditional LCA is best able to assess the contribution of the studied product or system to environmental effects on a global scale, such as global warming or ozone depletion. In addition, it is suitable to calculate primary energy consumption and total fossil energy depletion, therefore offering a measure for energy security. LCA can also provide indicators for environmental impacts which are relevant on a regional and/or local scale, such as for example acidification, eutrophication, photochemical ozone creation, human and eco-toxicity. However, it is worth noting that these category indicators represent an aggregated global measure of the potential impact. Since actual impacts depend on specific concentrations and receptor response pathways, LCA results cannot be used for the assessment of local pollution or site-specific effects, which may however have significant policy relevance.

Key determinants of results, methodological aspects and relevant site-dependent settings

Despite some discrepancies in results and regardless of the crops analyzed, most sources converge on the fact that the agricultural and transformation phases, and an isolated number of variables within these two phases, account for the vast majority of total impacts over the life cycle of bio-energy products. The distribution of impact share within these two phases varies from study to study and depends largely on both the type of feedstock and the impact indicator analyzed.

The agricultural phase is responsible for a relevant share of GHG emissions and is by far the dominant contributor to acidification and eutrophication, largely due to the emissions of nitrous oxide (N_2O), other nitrogen gases (i.e. NOx) and SOx associated with the use of fertilizers. Another relevant issue in the agricultural phase is the fate of co-products such as straw for cereal crops. The treatment of co-products and the way impacts are allocated to them can significantly change the overall results of the analysis.

The impacts of energy use are significant in the technology conversion phase, in particular in the case of ethanol production. The quantity and type of process energy used (e.g. heat and power from coal, natural gas or bagasse) can significantly affect the overall results, as reported by Wang et al. (2007). In their comparative analysis of well-to-wheels GHG emissions by fuel ethanol relative to gasoline the authors show that if coal is used as the process fuel in corn ethanol plants, the GHG emissions of corn ethanol are 3% higher than gasoline, where-as when biomass (such as wood chips) is used as a process energy fuel, an emission reduction of more than 50% can be achieved. The allocation of impacts to co-products can also be very significant in this phase of the life cycle.

Cross-cutting aspects affecting LCA results are the life cycle inventories databases used for modeling upstream processes and the life cycle impact assessment method indicators applied. The review showed that updated and scientifically acknowledged databases co-exist with somewhat older ones. This affects the reliability of results, regardless of the quality of the primary data collected. Finally, other parameters influencing the contributions to the overall LCA results are the assumptions made on vehicle performance and on biofuel transport distances. However, usually these parameters do not change results significantly over the whole life cycle.

Each of these key determinants are discussed in greater detail in the following sections.

The N_2O balance

Anthropogenic N_2O emissions are caused by two processes: nitrogen fertilizer production and field application. The field emissions constitute a serious uncertainty source in the LCA results of many biofuel pathways. Direct N_2O emissions occur from nitrification and denitrification at the site, while indirect N_2O emissions are associated with the volatilization and leaching of nitrogen which is converted into N_2O following atmospheric deposition or in waterways. The significance of these emissions is indicated in the very high characterization factor of N_2O in terms of greenhouse gas equivalent emissions: according to the IPCC 1 kg of N_2O has the same effect of 298 kg of CO_2 emissions over a time horizon of 100 years (Solomon et al. 2007). As a consequence, even small changes in the N balance and rate of N_2O emissions can significantly affect the overall GHG balance results for biofuels.

The impacts of N_2O emissions are particularly relevant for grain or seed-based biofuels, given the high rates of fertilizers they require. This issue has been discussed in most of the studies analyzed, and has been identified as a major cause of uncertainty in previous literature reviews (Larson 2005; von Blottnitz and Curran 2006). However, the use of fertilizers and related N balance and N_2O emissions strongly depend on site-specific aspects, thus it is difficult to identify representative average emission factors. Currently, the most commonly applied scientific method is the one developed by the IPCC, which provides a global average emission factor of 1% of applied fertilizer. This has the advantage to be acknowledged at the international level as a common reference thus facilitating the comparability of results, but is also affected by some limitations. In particular it cannot distinguish between crops or soils.

In the JRC-Eucar-Concawe study, Edwards et al. (2007) has applied a methodology developed by the Institute for Environment and Sustainability at the EC's Joint Research Centre (JRC) of Ispra that combines GIS information on soil, daily climate, and crop distribution with EU national data on fertilizer use and farm calendar. This results in more favorable estimates than the IPCC method. The authors of the study find that while the IPCC assumes that the emissions are proportional to the nitrogen fertilizer rate, the JRC model reveals a higher contribution of other factors like soil type, climate and ground cover. This finding is further supported by other researchers (Ecobilan 2002; Lechón et al. 2005, 2006). These studies adopt crop-specific nitrogen emission balance methods in their base case analysis (based on field measurements conducted in France, Germany, Spain and UK[24] and IPCC factors in the sensitivity analysis. More favorable results are obtained in the base case compared to sensitivity analysis.

A more in-depth analysis was carried out by comparing the main background assumptions retained in the previously mentioned studies in order to assess which of those have the largest influence. The results of this comparison are summarized in Table 5.6 using the example of biodiesel from rapeseed. As the table shows, two main aspects underlie the discrepancies observed: 1) the methodology followed for assessing N_2O emissions from fertilizers, and 2) the assumptions for the treatment of by-products in the technology conversion phase. In particular, the emission factor applied by Ecobilan (2002) is 0.50%, while Zah et al. (2007) adopt a range of 1.6-3.5%. SenterNovem (2005) and Edwards et al. (2007) adopt similar values, while Lechón et al. (2005, 2006) are in line with Ecobilan (2002). Various studies suggest that the IPCC percentages are too low. For example, Parkin and Kaspar (2006) found soil N_2O emission rates three times higher than the IPCC default emission factor for corn in central Iowa. Crutzen et al. (2007) suggested that the N_2O conversion factor should be in the range of 3-5% on average.

Due to the high global warming potential of N_2O, even a small change can lead to relevant variations, therefore more research is needed on the sources of N_2O and the nitrogen cycle. Nevertheless, all studies seem to concur that, if new N_2O abatement technologies are applied in the fertilizer industry and nutrient management practice is improved in the fields, biofuels can improve their environmental profile.

Co-Product Allocations

Biofuel production, as with many other industrial processes, can be classified as a multi-input/multi-output product system. Therefore, to correctly evaluate the impacts of biofuels, co-products need to be taken into account as well. This can be done through two main methodological procedures: system expansion or allocation.

Allocation is the method by which input energy and material flows and output emissions are distributed among the product and co-product(s) (ISO 14044: 2006; Kodera 2007; Veeraraghavan 2007). The ISO 14044 standard provides guidance on allocation methods and states the following options in order of preference:

- Wherever possible, allocation should be avoided by either sub-division (i.e. subdivide the fuel life cycle process into sub-processes not requiring allocation), or substitution (i.e. expand the system boundaries to include co-product function)

- Where allocation cannot be avoided, the inputs and outputs of the system should be partitioned between its different products or functions in a way that reflects the underlying physical relationships between them (i.e. they should reflect the way in which the inputs and outputs are changed by quantitative changes in the products or functions delivered by the system)

- If physical criteria are not feasible then other criteria should be used (i.e. allocate inputs and outputs to the product and co-product(s) in a way (e.g. economic allocation) which reflects other relationships between them)

Several allocation methods have been applied in the reviewed studies. As shown in Table 5.5, only one study applied sub-division, twelve studies applied system expansion, one study applied energy allocation, and three studies applied the economic allocation. In addition, six studies adopted a "mixed approach" (e.g. using system expansion as a base case and carrying out sensitivity analysis with allocation methods, or combining different methods in different life cycle phases, including no allocation at all). As already mentioned, the choice of the allocation method has considerable impacts on the final results, and is also an area where great discrepancies have been observed amongst the reviewed studies. In the proposed example of rapeseed biodiesel (see Table 5.6), the methodology used significantly affects the comparability of results.

Table 5.5 Allocation methods applied in the reviewed studies

Subdivision	1
System Expansion	12
Mass- based	0
Energy content-based	1
Economic allocation	3
Mixed methods	6
No allocation	1
Not applicable/ not available	6

Different hypothesis and scenarios are built in the studies analyzed. In particular, while in Ecobilan (2002) 46% of the impacts in the pressing phase are allocated in mass to

Table 5.6 Identification of the main areas of convergence and divergence in background assumptions for a selected number of studies in the case of rapeseed

Green: quite consistent background assumptions, no area of concern

Orange: some discrepancies, affects results to some extent

Red: high inconsistency area, affects results significantly

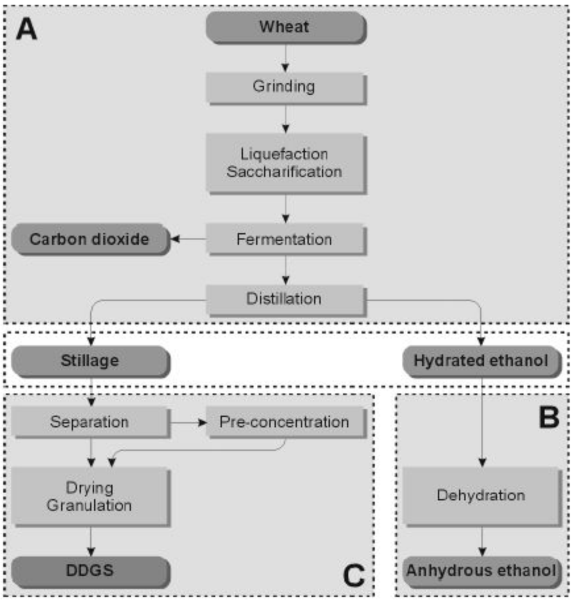
Agricultural Phase					Conversion phase	
Land Use reference scenario	Crop yield	Fertilizer	N balance	Allocation	Energy fuel used	Co-product allocation
Green	Green	Orange	Red	Orange	Green	Red

rapeseed, Zah et al. (2007) adopt an economic allocation where 87% of impacts are assigned to rapeseed-oil methyl ester. These different hypothesis and methodological choices lead to very different results. In the Ecobilan study, rapeseed biodiesel leads to an overall improvement of GHG emissions of 80% compared to conventional diesel, while Zah et al. (2007) obtain significantly lower values (ranging from 23 to 41%). The influence of the allocation method on final results is an issue that has been and is still extensively debated in the LCA community (e.g. Weidema 2001).

Several reviewed studies have tried to assess the variability range in function of different allocation methods. An example is provided by Gnansounou and Dauriat (2005), in which the authors assess the influence of the allocation method on the energy ratio of the fuel (ratio between MJ content of the fuel divided by MJ of primary energy spent for its production). The study takes into account bioethanol production from wheat in a Swiss plant with a production capacity of 50 ML year^{-1}. The graph and table reported in Figure 5.1 are extracted from the authors' article. The two values in brackets show the results for allocation between anhydrous ethanol and Distilled Dry Grains and Solubles (DDGS).

All methods have advantages and drawbacks. Furthermore, a distinction has to be made between analytical and regulatory purposes. Amongst the various options, subdivision is often not applicable in LCA studies; it can be applied only when the sub-processes are both physically separate and economically independent of each other (Kodera 2007).

Figure 5.1 Effects of different allocation methods on results (source: Gnansounou and Dauriat 2005) Values given in brackets reflect results with allocation between anhydrous ethanol and DDGS



Allocation method		Energy MJ L^{-1}	Allocation % ethanol	Allocation % stillage	E/R MJ MJ^{-1}
w/o allocation	A	19.35	100	0	0.70
	B	1.93	100	0	
	C	8.95	100	0	
economic value	A	19.35	95	5	1.08
	B	1.93	100	0	(0.83)
	C	8.95	0	100	
energy content	A	19.35	61	39	1.54
	B	1.93	100	0	
	C	8.95	0	100	
system extension	A	19.53	81	19	1.21
	B	1.93	100	0	
	C	8.95	0	100	
mass	A	19.35	12	88	5.01
	B	1.93	100	0	(1.56)
	C	8.95	0	100	

The substitution method and system expansion is preferred by ISO; however, it appears more appropriate for analysis than for regulatory purposes (Hodson 2008), as it requires arguable hypotheses about the substituted product, implicitly assumes that co-products are sold on the market, and may trigger perverse incentives to maximize the production of co-products. Economic allocation reflects more properly the actual market conditions. However, it also significantly increases the volatility of results and therefore their uncertainty. Ideally, this approach would require analysts to re-conduct the LCA study several times and adjust the results accordingly. This looks very difficult for regulatory implementation purposes. Mass allocation turns out to be much more generous to biofuels than other methods (Ecobilan 2002 and 2006).

For regulatory purposes, a more pragmatic approach might be to use energy allocation. Depending on use of co-products, this gives comparable results to those of the substitution method (Hodson 2008). Both the European Commission Proposal for the Directive on Renewable Energy (EC 2008) and the draft for the German Sustainable Biofuels Ordinance (Fehrenbach et al. 2008) apply the energy allocation method. The UK Renewable Transport Fuel Obligation uses a mixed allocation method instead (Chalmers 2008).

Finally, exergy allocation is rather overlooked in the analyzed studies. In principle, allocation based on the exergy content could reduce some of the limitations of energy allocation, which treats different forms of energy (e.g. electricity, heat and fuel) as equal.

Given its large influence on results, choice of the applied allocation method must be clearly presented in publication. Moreover, LCA studies should include sensitivity analysis within the interpretation stage comparing results in function of the different methods used, as recommended by ISO 14044.

Process energy

The type and quantity of process energy used can significantly affect the overall results of biofuel LCA. For example, the use of coal or lignite can totally offset ethanol GHG emission reduction potential with respect to gasoline. On the contrary, the use of biomass or other renewable energy improves the environmental profile of the produced biofuel.

Our review revealed a wide range of discrepancies in energy consumption rates. This can be explained by the fact that, while some studies focus on state-of-the-art installations properly designed for ethanol production, others study older, inefficient plants, sometimes converted to biofuel manufacturing.

We report a comparative exercise carried out by Unnasch and Pont (2007) in Table 5.7. The example refers to the quantification of energy consumption in the transesterification process for the production of biodiesel. To estimate energy consumption in the form of methanol, steam (assumed to come from natural gas fired boilers) and electricity, the authors consulted the JRC and NREL reports and compared them against the default values contained in the GREET model (version 1.7). The last row indicates the inputs ultimately retained by Unnasch and Pont (2007) in their study. The values for methanol energy content agree reasonably well. For steam, the two referenced studies are in fairly close

Table 5.7 Energy inputs for transesterification process reported in a set of studies (source: Unnasch and Pont 2007)

	Energy inputs (MJ kg^{-1} biodiesel, LHV)			
	Methanol	Steam	Electricity	Total
GREET 1.7	1.86	4.33	0.79	6.98
EUCAR/ JRC	2.19	1.53	0.11	3.83
NREL	NA	1.37	0.10	NA
TIAX	2.03	1.45	0.11	3.59

agreement, but the GREET value is a factor of three higher. For electricity, the values reported in the two studies are again in close agreement, while the GREET value is approximately a factor of 7 higher. This issue needs to be properly considered when comparing results. Moreover, it would be worth taking into account not only the current average status of technology and process energy mixes, but also the technology improvement potential and best practices under a "consequential LCA" approach. This of course applies not only to biofuels but also to fossil fuels.

Cross-cutting Aspects

Other methodological aspects which have been identified during the review process as potentially affecting full comparability of results relate to the use of different life cycle inventory databases and life cycle impact assessment methods.

Life cycle inventory datasets are commonly used to support LCA modeling of upstream processes or to compensate for the lack of primary data. Many LCA studies combine primary data with secondary data available in a series of LCI databases. Although the studies reviewed build upon some of the most widely acknowledged and commonly used databases for life cycle modeling,[25] the use of different data sources for the same unit process always introduces certain variability in results. This is particularly true for some older databases which might not contain updated data, new materials and processes, and best available technologies. In addition to temporal representativeness, geographic and technological representativeness of life cycle inventory data is also an issue. The topic of harmonization of life cycle inventories has been largely debated in the LCA community. There has been much effort at the international, country, and industry level to provide practitioners with a matched set of data for LCA modeling, but such a dataset does not yet exist and more research is needed.

As for life cycle impact assessment methods, the review revealed differences in energy consumption and GHG emission accounting. While some studies report only fossil energy consumption, others take into account total primary energy consumption (renewable and non renewable) thus not allowing an immediate comparison. With respect to global warming potential (GWP), most studies only take into account the contribution of CO_2, N_2O and CH_4. These gases are often called direct GHGs because they impact climate directly (Larson 2005). The IPCC method for calculating GWP actually includes a list of over 60 gases, some of which having a global warming potential over 10,000 times higher than CO_2. Delucchi (2006) calculated that the impact of taking into account other "indirect" GHGs (e.g. NOx) would change GHG emissions by 3 to 5% relative to an accounting that considers only direct GHG emissions. Higher impacts are calculated if aerosol emissions from diesel engines (soot or black carbon) are considered. The GWP of black carbon has been estimated to be 680 on a 100-year basis (Larson 2005).

Additional aspects of process improvement to be considered include the reference scenarios modeled, heating values, and the productive life of plantations. Studies diverged significantly as to the technology reference scenario modeled. For example, Ecobilan (2002) model best practices while the majority of studies were based on average practices. In some cases, pejorative assumptions were considered, while others focused on potential technological improvements. Heating values also varied. Some studies use the low (net) calorific value (e.g. Choudhury et al. 2002; Edwards et al. 2007), some use the high (gross) calorific value, and others use both (e.g. Elsayed 2003). Even though this is not a big issue by itself, for completeness of information it would be worth using both in the same study. An example of how GHG emission results would change based on different assumptions regarding the productive life of plantations is provided in Beer et al. (2007).

Conclusions

Assessing the environmental performances of biofuels is a complex task, as it implies covering many different feedstock systems, conversion technologies, land-use and land-use change related issues, as well as aspects related to the substituted products, including fossil transport fuels, as well as animal feed and electricity. The present review identified ranges of fossil energy consumption and GHG emission results for selected biofuel pathways relative to conventional fossil fuels (both gasoline and diesel), based largely on the assumption of unchanged land use. For certain pathways and crops (e.g. ethanol from sugarcane, sugar beet and, to a lesser extent, wheat) a good convergence of results was found, while in other cases (e.g. ethanol from corn, diesel from soybean and palm oil) the variability was far higher. Because of confidentiality reasons or lack of data, we did not include studies on biofuels from feedstocks grown on degraded or abandoned land (e.g. cassava, jatropha, and sweet sorghum). More analysis is recommended in this area.

Most LCA studies are based on current technologies. However, energy and climate change policies imply long-term decisions. Therefore, future studies should aim to address future technological developments and improvements. For instance, the trend towards system integration into multi-fuel/multi-product bio-refineries has not so far been taken into account. Parametric LCA[26] might be a useful tool to predict possible improvements, while maintaining a reasonable level of uncertainty of results.

Most assumptions and data used in LCA studies thus far are related to Europe or the US and rely on western technology patterns.[27] This affects the representativeness of the studies. More data needs to be gathered on other world regions, including developing countries. In this respect, we share the view of Larson (2005) who already recommended carrying out country or region-specific studies. UNEP is engaged in a targeted research project on biofuels together with UNIDO and FAO, which has the aim to help overcome some of the mentioned limitations by allowing for coherent data gathering on a series of pathways and crops with focus on developing countries.

LCA also provides an indicative comparative assessment of biofuels with respect to non-GHG environmental impact indicators, such as acidification, eutrophication, summer smog and toxicity. However, LCA aggregates results over time and space, and gives therefore only a potential and indicative measure of impact. It should be combined with other environmental assessment tools looking at local and regional impacts. Possible trade-off judgments (e.g. between GHG vs. non-GHG impacts) ultimately remain a decision-making issue, which depends on national and sometimes local circumstances.

LCA results can be combined with data on land carbon storage in order to take direct land-use change effects into account.

Furthermore, recent studies on potential indirect land-use change identify and focus on a real concern, i.e. the risk that biofuel deployment could accelerate and worsen the current unsustainable trends of deforestation and depletion of natural resources in a framework of accelerated growing population, and food and feed demand. LCA per se does not assess absolute impacts of large-scale deployment of a certain technology or product, but it can be combined with other assessment tools to do so (e.g. agro-economic market models). The very recent estimates of indirect-land use changes due to biofuels diffusion are an important step in this direction. However, further research on agro-economic modeling looking at worldwide impacts of large scale biofuel diffusion is needed to properly address this challenging issue. At the same time, LCA models need to be improved and further developed to treat future technologies with a minimized level of uncertainty.

We recommend the land use change GHG contribution to be always presented in a transparent and disaggregated way from the rest of the life cycle; and all the assumptions about new and former land use to be clearly reported.

LCAs are already currently used in regulatory proposals to set environmental criteria and standards for biofuels. As shown in this report, LCA studies feature a wide range of sometimes diverging results. This partly reflects the complexity and technological and geographical scope dependency of the modeled reality. However, it is also the result of the many different methodological and numerical assumptions required to perform a LCA analysis.

There is a clear need to reach consensus on how to carry out LCAs on biofuels, driven by national and international legislation which include GHG emission reduction goals. This implies reaching agreement on

Table 5.8 Documents short-listed and used as a basis for determination of the range of results. (rvo= recycled vegetable oil, ww= wood waste)

No	Author	Org	Year	Pp	Type	Feedstock(s)	System Bound
1	de Castro	DGIS/ DMW/IB	2007	42	review	sugar cane/ molasses/ maize/ cassava/ jatropha/ rapeseed/ soybean	WTT
2	Farrell et al	Univ. California	2006	3	review + energy & GHG	maize/switch grass	various
3	Smeets et al	SenterNovem	2006	136	review	sugar cane	WTW
4	Larson		2005	43	review	various	WTW
5	Quirin et al	IFEU for FVV/ UFOP	2004	66	review	sugar cane/ maize/ wheat/ beet/ ligno-cell/ potato/ molasses/ rapeseed/ sunflower/ soybean/ coconut oil/ rvo/ animal grease/ cooking grease/ org. Residue/ cult. biomass	WTT & TTW
6	Elsayed et al	DTI	2003	341	review + energy &GHG	rapeseed/ rvo/ wood chip (residue, woodland mngt, & short-rotation coppice)/ miscanthus/ straw/ ligno-cell/ beet/ wheat	WTT
7	Puppán	Buapest Univ Tech	2001	22	review	rapeseed/ beet/ winter wheat/ potato	WTW
8	Beer et al	CSIRO	2007	126	GHG	rapeseed/ tallow/ rvo/ palm oil	WTW
9	Edwards et al.	CONCAWE/ EUROCAR/ JRC	2007 update	140	energy & GHG	Ethanol: wheat/ beet/ straw/ ww/ sugar cane. Methanol: ww/ farmed wood. Diesel: rapeseed/ sunflower	WTT & TTW
10	Grood and Heywood	MIT	2007	25	energy & GHG	maize/ switchgrass	WTT
11	Reinhardt et al	IFEU for WWF (DEU)	2007	50	energy & GHG	palm oil	WTW
12	Unnasch and Pont	TIAX for Ca. Energy Comm	2007	243	energy & GHG	maize/ sugar cane/ soybean/ palm oil/ waste material	WTW
13	Wang et al	Argonne Nat. Lab	2007	13	energy & GHG	maize/ switchgrass	WTW
14	S&T Consultants	NRC, Canada	2006	44+ 118	energy & GHG	maize/ wheat/ cellulose	WTW
15	Lechón et al.	Environment Ministry of Spain	2006	141	energy & GHG	imp soy oil (40%)/ dom. sunflower oil (10%)/ imp. Palm (25%)/dom & imp rapeseed (25%)	WTW

No	Author	Org	Year	Pp	Type	Feedstock(s)	System Bound
16	Veeraraghavan and Riera-Palou	Shell Global Solutions Intl	2006	64	energy & GHG	wheat straw	WTT and TTW
17	de Oliveira et al	WS Univ	2005	11	energy & GHG	sugarcane/ maize	WTW
18	Lechón et al.	Environ. Min Spain	2005	114	energy & GHG	wheat/ barley	WTW
19	Macedo et al	Univ Campinas	2004	32	energy & GHG	sugar cane	WTW
20	Choudhury et al	LB Systemtechnik/ GM	2002	138	energy & GHG	woody biomass/ beet/ lignocell/ rapeseed/ biogas	WTT and TTW
21	Ecobilan PwC	ADEME/ DIREM	2002	132 + annex	energy & GHG	Ethanol & ETBE: wheat/ beet. Diesel & MTBE: sunflower/ rapeseed	WTT
22	Pimental and Patzek	Cornell Univ/ Univ. California	2005	12	energy	Ethanol: maize/ switchgrass/ wood. Biodiesel: soybean/ sunflower	WTT
23	Shapouri et al	USDA	2002	14	energy	maize	WTT
24	Jungbluth et al	ESU Services	2008	24	LCA	wood & forest residues (also via black liquor)/ agric. Residues and by-products/ barley/ wheat/ sorghum/ Jerusalem artichoke	WTW
25	Zah et al	EMPA	2007	206	LCA	Biodiesel: rapeseed/ palm/ soybean. Ethanol: grass/ potato/ beets/ whey/ wood/ sorghum/ rye/ maize/ sugar cane. Methane: grass/ manure/ biowaste/ sewage sludge/ wood	WTW
26	Reinhardt et al	IFEU	2006	100	LCA	wood/ ww/ straw/ cereals/ agr. waste/ short-rotation/ Triticale	WTW
27	Ecofyf/ SenterNovem	Min. Housing, Spatial Plan & Envir, NL	2005	145	LCA	Ethanol: wheat. Biodiesel: rapeseed ETBE: ethanol from wheat	WTW
28	Baitz et al	Daimler Chrysler/ Volkswagen	2004	7	LCA	ww/ standing timber	WTW
29	Gnansounou and Dauriat	EPFL	2004	75	LCA	maize/ potato/beet/ molasses, etc. Inhouse & imp.	WTW
30	Fu et al	NRC, Canada	2003	5	LCA	wood and agric wastes compared against cult feedstock	WTW

approaches and assumptions on a wide set of key parameters (e.g. allocation rules of impacts on co-products, N_2O emission rates, land use carbon stock, technology progress, etc). Ideally this should happen in a multi-stakeholder process at international level. The involvement of relevant stake-holders in defining assumptions for LCA is included in the stage of Product Category Rules (PCR) definition in the ISO 14025 standard on ISO-type III environmental product declarations. The standard also gives guidelines on third-party review and validation of underlying LCA data and results. The latter are obviously an important element of any future possible biofuel certification scheme. The biofuel community should look at and profit from the experiences gained in the area of eco-labeling and environmental product declarations. Harmonization is a goal of both the UNEP Life Cycle Initiative and the European Platform for LCA, as well as the international network GEDnet (Global Type III Environmental Product Declarations Network).

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Notes

[1] Excluded articles and reports include Life Cycle Inventories, cradle-to-gate studies, or others in which determination of range of results was impeded by lack of sufficient quantitative information.

[2] The report by Zah et al (2007) contains results on biodiversity loss due to land use change, but the applied methods are still inappropriate as acknowledged by the lead author

[3] In principle, the TTW energy consumed can vary slightly as a function of the fuel due to differences in combustion efficiency; however, in practice, several WTW studies actually use a constant energy consumption rate dependant on general power train improvements over time. In such a case, WTT and WTW studies lead to the same results expressed in terms of percentage improvement.

4. For example, the estimated benefit in terms of GHG emission savings range from 7-90%

5. Values reported are for "ethanol today" and "CO2 intensive" scenarios, respectively.

6. The savings is 95% if calculated as a ratio of petroleum input (MJ) per MJ of ethanol only.

7. As reported in the Excel workbook dated 25 December 2005. More favorable results are found in the updated

version of the supporting online material issued on 13 July, 2006, reflecting EBAMM 1.1 calculations.

8. Reflects best current practice for Iowa corn

9. Average value for Iowa corn ethanol with 20% credits for co-products. Range from -47% for Georgia corn without allocation to co-products to +58% for Iowa corn with credit allocated to distilled dry grains and solubles production

10. Mid-west corn with co-product allocation. -5% if coal is used. California corn = -30% to +50%

11. Results differ with energy source used (minimum value for coal, maximum for biomass). Previous works from Wang (as reported in other sources) indicate improvements ranging from 15-40%

12. Current average reported in table. Results range from -3% if coal is used to +52% if biomass (I.e. Woodchips) is used as process fuel.

13. On a Well-to-Wheel (WTW) basis.

14. 42% is average case based on use of natural gas in the processing phase. The best case scenario is straw CH15P with DDGS used as fuel

15. Values reported are for conventional gas boilers; a wider range is found with other energy sources (i.e. Coal or straw CHP) Range in brackets includes lignite vs. Straw CHP, both with DDGS.

16. Non-renewable energy from Wheel-to-Tank (WTT)

17. 25% if pulp to fodder, 73% if pulp to heat

18. 32% if pulp to fodder, 65% if pulp to heat

19. On a Wheel-to-Tank (WTT) basis

20. Energy improvement is for non-renewable energy Wheel-to-Tank (WTT) basis. For both energy improvement and GHG improvement, the first number in the range reflects average production of rapeseed in Europe, the latter reflects the case for Switzerland.

21. Other studies were identified during the screening process, but were excluded in the short-listing phase because they were not fully compliant with the guiding criteria expressed in paragraph 4. Nevertheless, some finding will be used to complement and comment on the results from the 30 studies included in the present review.

22. A description of the most commonly used Life Cycle Impact category indicators and characterization models is available from the UNEP/SETAC Life Cycle Initiative web site (http://lcinitiative.unep.fr).

23. A good overview of ongoing efforts to determine a suitable indicator is reported in Trydeman et al. (2007). See also Koeller and Scholz 2008.

24. For a complete discussion of the methodologies adopted and a description of the references retained in the studies, please see Lechon et al. 2005, p. 26; and Ecobilan 2002, p. 22.

25. Amongst the most frequently reported studies, we found ECOINVENT, DEAM, NREL, UMBERTO, GREET, LEM.

26. In a parametric LCA model a certain number of parameters are selected, which are deemed to be relevant in terms of goal and scope of the study. The variation via parameters is possible on multiple levels including material, manufacturing process, and component or sub-assembly levels. The strength of parameters is the ability to make different versions of the model, allowing the modeller to create multiple scenarios almost automatically.

27. With the exception of Brazil

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