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A Review of Environmental Life Cycle Assessments of Liquid Transportation Biofuels in the Pan American Region

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Abstract Life-cycle assessment (LCA) has been applied to many biofuel and bioenergy systems to determine potential environmental impacts, but the conclusions have varied. Different methodologies and processes for conducting LCA of biofuels make the results difficult to compare, in-turn making it difficult to make the best possible and informed decision. Of particular importance are the wide variability in country-specific conditions, modeling assumptions, data quality, chosen impact categories and indicators, scale of production, system boundaries, and co-product allocation. This study has a double purpose: conducting a critical evaluation comparing environmental LCA of biofuels from several conversion pathways and in several countries in the Pan American region using both qualitative and quantitative analyses, and making recommendations for harmonization with respect to biofuel LCA study features, such as study assumptions, inventory data, impact indicators, and reporting practices. The environmental management implications are discussed within the context of different national and international regulatory environments using a case study. The results from this study highlight LCA methodology choices that cause high variability in results and limit comparability among different studies, even among the same biofuel pathway, and recommendations are provided for improvement.

Keywords Life-cycle assessment · Biofuels · Bioenergy · Sustainability · Pan American region

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Abbreviations

BD	Biodiesel
HRJ	Hydro-renewable jet fuel
DAYCENT	Daily Century
dLUC	Direct land-use change
EBAMM	ERG Biofuel Analysis Meta-Model
EPA	Environmental Protection Agency
EtOH	Ethanol
EU-RED	European Union-Renewable Energy Directive
GHG	Greenhouse gases
GREET	Greenhouse gases, regulated emissions, and energy use in transportation
GWP	Global warming potential
IPCC	Intergovernmental Panel on Climate Change
ISCC	International Sustainability & Carbon Certification
LCA	Life-cycle assessment
LCI	Life-cycle inventory
LCIA	Life-cycle impact assessment
RSB	Roundtable on sustainable biomaterials
US-RFS	United States-Renewable Fuel Standard

Introduction to Issues of LCA of Biofuels

Life-cycle assessment (LCA) is a well-established methodology to comprehensively determine potential environmental and human health impacts of a product throughout its life cycle, starting with extraction of raw materials, then including manufacturing, transport and use, and ending with disposal of residues at end of life (Allen and Shonnard 2002). LCA is useful to gain an understanding of a product system, to identify the most relevant environmental impacts, to guide product improvement, for stakeholder communication, and decision-making. It has emerged as an important part of environmental management since the first studies were conducted in the 1960s focusing on the cumulative energy demand for chemical intermediates and products (SAIC 2006). Due to the energy crisis in the early 1970s, energy applications of LCA increased, and when global environmental challenges emerged in the late 1980s, interest in LCA again increased. The first formal LCA methodology guidance documents (SETAC 1993) were followed by the publication of the internationally agreed-upon LCA standards, the ISO 14040 series which laid a general framework and requirements (ISO 14040 1997; ISO 14041 1998; ISO 14042 1998; ISO 14043 1998; ISO 14040 2006; ISO 14044 2006; SETAC 1991, 1993). These documents have provided critical guidelines in research and helped establish LCA as a professional practice.

Interest in achieving environmental sustainability for biofuels and bioenergy has provided additional momentum to study biofuel pathways using LCA. Partly in response to policy and regulation, emissions of anthropogenic (man-made) greenhouse gases (GHG) have been a common feature of biofuel LCA. There is little doubt that biofuel policy and regulation have in turn been influenced by a scientific consensus that the Earth's atmosphere and oceans have warmed, extent of snow and ice cover has diminished, sea level has risen, and concentrations of CO_2 and other GHG have increased since about 1850 (IPCC 2013). As presented in Solomon et al. in this special feature, energy policy in many Pan American countries mandates the use of LCA to demonstrate savings of GHG emissions for biofuels. These eligible biofuels will count toward production targets that transportation fuel producers are obligated to achieve (Moser et al. 2014). For example, in the United States, the Renewable Fuels Standard 2 (RFS2) defines a methodology to assess GHG emissions of biofuel pathways, including indirect land-use change emissions of CO_2 (emissions resulting from conversion of natural lands to food production as a result of biofuel expansion). Furthermore, RFS2 mandates 20 % GHG emission savings for conventional biofuels (corn ethanol, soybean biodiesel), 50 % for advanced biofuels (sugar cane ethanol, hydrotreated esters of fatty acids, HEFA), and 60 % for cellulosic biofuels (cellulosic ethanol, pyrolysis-based hydrocarbon biofuels, gasification-based hydrocarbon biofuels) (Moser et al. 2014)—and see Shonnard et al. (2012) for a summary of biofuel processing options. These LCA requirements will likely affect production systems throughout the Pan American region for countries exporting biofuels to the U.S. through the RFS2 guidelines, or to the European Union, through their Renewable Energy Directive (EU-RED). This has already been demonstrated in Argentina, where exports of soybean biodiesel to the EU were restricted before new calculations were certified and due to restrictions on GHG emissions as calculated under EU-RED guidelines (Hilbert and Galligani 2014).

The two main established governmental standards that influence the practice of LCA for biofuels globally (Moser et al. 2014) are RFS2 and the EU Renewable Energy Directive 2009/28/EC and Fuel Quality Directive 98/70/EC through 2009/30/EC (RED 2009, 2012). The RFS2 mandates consequential biofuel LCA modeling, in which effects beyond the biofuel pathway, such as indirect land-use change and displacement of existing market items with co-products, are added to the GHG inventory attributed to the biofuel (EPA 2010). The U.S. Environmental Protection Agency (EPA) is responsible for determining whether biofuel pathways achieve GHG reduction targets mandated in the RFS2. In carrying this out, EPA uses several LCA models and sub-models. The Greenhouse gases, Regulated Emissions, and Energy use in Transportation (GREET) model from Argonne National Laboratory (ANL 2014) is used for assessing the direct biofuel pathway, while the DAYCENT model provides soil biogeochemical process emissions such as N_2O from N fertilizer application and soil carbon dynamics (CFR 2010). Indirect land-use change effects and their emissions are determined using domestic and global commodity market models, such as the Forestry and Agricultural Sector Optimization Model (FASOM) and the integrated Food and Agricultural Policy and Research Institute (FAPRI) models (CFR 2010). EPA RFS2 LCA approach employs "system expansion" to account for co-products generated during biofuel production and credits avoided GHG emissions from co-product displacement effects to the Renewable Identification Number (RIN) generating biofuels (see Solomon et al. in this special feature for a discussion of RIN in RFS2).

The EU-RED differs from the RFS2 biofuel LCA approach in several ways. The EU-RED employs energy allocation to distribute GHG emissions among products and co-products in a biofuel pathway. Direct land-use change (dLUC, emissions when land converts to biofuels) GHG emissions are included using the IPCC "tier 1" estimation method and carbon stock data for different land types (IPCC 2006a, b), but iLUC effects are not currently included. Finally, whereas in RFS2, the US-EPA determines each pathway's GHG emissions and qualifies biofuel pathways, the EU-RED allows compliance with mandated sustainability criteria using voluntary certification standards (Moser et al. 2014). There currently are six voluntary certificates that may qualify under the EU-RED meta-standard. A review of 13 Latin American and Caribbean countries showed that of a total 177 certified biofuel entities (biomass growers, biofuel facilities, supply chain companies, etc.), a large majority (139) qualified under EU-RED (Solomon and Bailis 2014). A competent review of sustainability standards and certification of biofuels is provided in Moser et al. (2014) and Diaz-Chavez (2014).

Biofuel LCA can be a very complicated analysis and, depending on study scope, may include over 100 unit processes, thousands of inventory elements, and multiple midpoint or endpoint impact categories. Aspects of LCA methodology such as choice of system boundary, source of inventory data for unit process inputs, and decisions on co-product allocation can all have a profound effect on study results (Allen and Shonnard 2002; Cherubini et al. 2009; Larson 2006). Larson (2006) reviewed a number of liquid biofuel LCAs from the North America and the European Union (EU). That study revealed a wide range of GHG emissions and energy demand results due to variability of several study features, such as climate-active species included N_2O emission assumptions, co-product allocation method, and soil carbon dynamics. Beyond these biofuel LCA topics, Cherubini et al. (2009) evaluated key issues influencing LCA outcomes for liquid biofuel and bioenergy systems (biopower and heat) and the need to model them accurately. These issues included biomass type and supply chains, soil carbon pools, CH_4 emissions, effects of residue removal on soil N and C balances, fossil reference system features, functional unit selection (a preference for land area), crop yields, and fertilizer inputs. They also noted the potential for trade-offs between GHG emissions and fossil energy reductions and potential increases in acidification, eutrophication, and local air pollutants when bioenergy replaces fossil energy systems. Cherubini and Strømman (2011) reviewed 94 LCAs of biomass energy, mostly from the EU and with contributions from North America, Asia, but with very few from South America, Africa, and Oceania. The study provided qualitative rather than quantitative evaluations of the LCA results from this literature, and it discussed the key LCA issues and features as well as the approaches taken to address them. Information was presented on the study locations, biofuel and bioenergy pathways, feedstock types, choice of functional unit, impact categories, allocation method, fossil reference system, and land-use change. Relations of methodology choices with policy maker's requirements were described, highlighting shortcomings and future research directions. Within the study reported in this article, the focus is not only on the qualitative differences within the context of biofuel LCAs but also on the quantitative differences between different feedstocks biofuel pathways, and with a dedicated focus on the Pan American region, which has not occurred before.

As noted previously, choice of system boundary will have a large effect on study results depending on whether only impacts directly linked to the biofuel pathway are considered (attributional LCA modeling) or whether indirect effects beyond the pathway are considered (consequential LCA modeling) (Allen et al. 2009). Indirect effects are most often associated with indirect land-use change (iLUC) emissions of CO_2 due to the market-driven demand for more land to compensate for food production lost to biofuels (Fargione et al. 2008; Searchinger et al. 2008). In addition to that, inventory data within life-cycle inventory databases, [ecoinvent™ (SCLCI 2014), US Life Cycle Inventory (NREL 2014), GREET (ANL 2014), GaBi (PE International 2014), among others], are not necessarily compatible with each other due to differences in data formatting and quality requirements, geographical and technological coverage, allocation procedures, and time relevance. Several studies concluded that the choice of method to allocate inventory data among biofuel pathway products and co-products has an overwhelming effect on LCA results (Bailis and Baka 2010; Larson 2006; Wang et al. 2011b). Finally, LCA software packages (SimaPro, GREET, GaBi, GHGenius, BioGrace) may yield variable results for the same biofuel pathway because of differences in life-cycle inventory databases, in their treatment of biogenic carbon, in how recycle of material is handled, impact assessment methods used, and because there is no common agreement in relation to emission factors for such items as electricity and N_2O emissions from soil (Fan et al. 2012).

Research Objectives

Despite the fact that there are some good reviews discussing the variation of LCA results due to methodological differences as discussed above, an in-depth review for the Pan American region is missing in the literature. The Pan American region is of particular interest as a study focus because of its dominance in global biofuel production (OECD 2014). Yet despite the large number of Pan American biofuel LCAs, no comprehensive review of the literature has occurred, in contrast to what occurs for the US and EU biofuel and bioenergy LCA literature (Larson, 2006; Cherubini et al. 2009). This review builds on prior work and expands the scope of study with a more detailed review and analysis including aspects of policy-driven LCA approaches (through the case study presented), more impact categories, and statistical analyses of LCA results, especially for GHG emissions. Furthermore, in this work, we focus on two research questions to address in the reviewed articles, in the context of Pan American countries: (1) What LCA methodology choices are used to determine the potential environmental impacts of the biofuel production systems in the Pan America region? (2) How frequently is policy-driven LCA employed in Pan American biofuel and what is the magnitude of change in LCA results when it is employed? One Pan American case study directly addresses the latter question. The article ends with recommendations for improving biofuel LCA through research and other actions.

Research Methods

To answer the two research questions, we conducted a literature review by means of search engines of scientific publishers including Elsevier/ScienceDirect, SpringerLink, Redalyc, and the American Chemical Society, and then performed a case study. Studies not reported in journals, such as governmental analyses, were searched by means of the Google Scholar search engine. Studies performed in countries out of the Pan American region were discarded. We considered studies in English, Spanish, and Portuguese languages, since these are the main languages in the Pan American region. The time frame considered articles published from 2000 to the present in order to consider the most recent studies.

A total of 74 articles were found and analyzed according to a number of LCA features (see "Introduction" section), including the geographic location, feedstock used, the types of biofuel produced, the functional unit, the chosen life-cycle impact assessment methodology and impact categories, the allocation criteria, the system boundaries, and the regulatory frameworks guiding the studies. These articles represent LCA studies of biofuels production in Argentina, Brazil, Canada, Colombia, Cuba, Chile, Costa Rica, Ecuador, Mexico, Peru, and the US. *Qualitative analyses* of the articles determined how often the articles aligned with certain LCA features. An overview of the evaluated studies is provided in Table 1 of the electronic supplementary material. To undertake a *quantitative analysis* of the environmental profile of biofuels, the results on GHG emissions were conveyed in "box and whisker plots" showing the medians, interquartile ranges, minimums, maximums, and non-typical data (Cleary 2009; Muench and Guenther 2013). The medians separates the higher and lowers halves of a set of results, and the interquartile ranges represent the points lying between the lower and upper quartiles, Q_1 and Q_3, respectively. The whiskers represent the maximums and minimums of a sample. Non-typical data are shown with a × symbol and represent points that lie outside of $Q_1 - 1.5 \cdot (Q_3 - Q_1)$ and $Q_3 + 1.5 \cdot (Q_3 - Q_1)$.

Qualitative Results

Geographic Locations

The distribution of articles among different geographic (country) locations is shown in Fig. 1a, with some articles evaluating more than one geographic area. The majority of studies were on biofuel production in the United States (32/74 articles-US) and Brazil (21/74-BR), with fewer studies on biofuels produced in Colombia (8/74-CO), Argentina (5/74-AR), Chile (3/74-CL), Mexico (3/74-MX), Canada (2/74-CA), Costa Rica (2/74-CR), Cuba (1/74-CU), Ecuador (1/74-EC), and Peru (1/74-PE). Brazil's large number of studies is a result of their long history of ethanol production and the need to understand its environmental implications. The higher number of studies in the United States is likely a result of the active research programs investigating many types of advanced biofuels and the interest by funding agencies to understand the environmental implications of future biofuel production systems with respect to meeting regulatory standards for savings in GHG emissions and other sustainability criteria.

Regulatory Framework for LCA

The ISO 14040 standards establish that the scope, assumptions, description of data quality, methodologies, and output of LCA studies should be transparent (ISO 14044 2006). The transparency of an LCA is what allows for reproduction of the work by others and for accurate comparisons and conclusions to be made; therefore, good documentation calls for a more transparent study. Nearly all the papers reviewed use ISO 14040 standards to conduct their LCAs.

Nearly half of the reviewed studies use a regulatory framework (which have predetermined functional units and allocation methods) as a guideline to perform the LCA, as shown in Fig. 1b. Of the studies mentioning regulatory framework, the most common is the RFS (20/74) due to the abundance of LCAs conducted in the US. The Low-Carbon Fuel Standard (LCFS) of the state of California in the United States provided guidance for LCAs in 5/74 articles in this review. The few LCAs used the EU-RED framework (8/74) and only 1/74 used the Roundtable on Sustainable Biofuels (now Biomaterials) (RSB) metrics. Sixty-three percent of the studies did not mention regulatory-

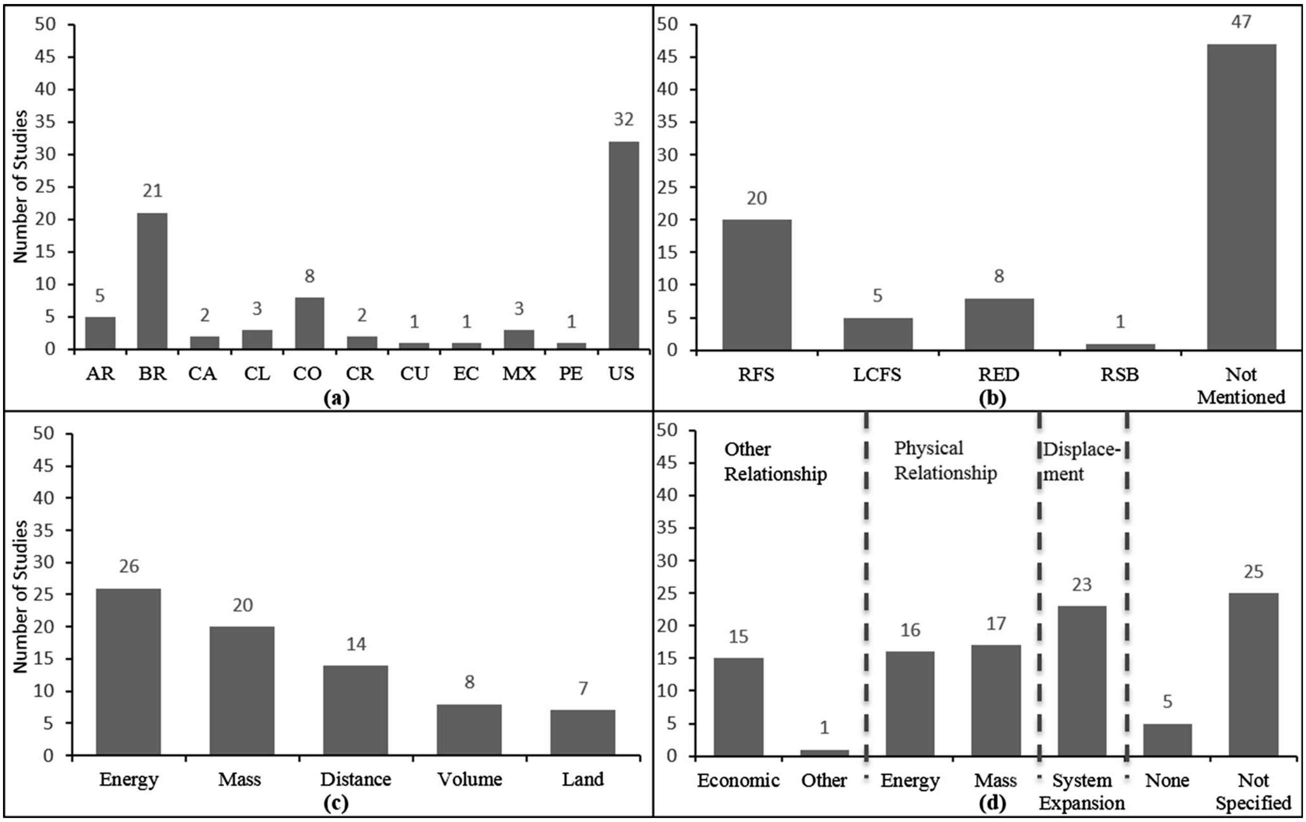


Fig. 1 Number of studies in the reviewed articles: **a** geographic locations (74 studies, 79 scenarios), **b** selected framework and methodology (74 studies, 82 scenarios), **c** functional units used (74 studies, 75 scenarios), **d** allocation methods (74 studies, 102 scenarios)

driven LCA guidance. Because a large amount of articles (27/74) mentioned some regulatory-driven guidance, this can be interpreted as policy having a significant influence on the methodology aspects of current LCAs of biofuel production systems. There are no frameworks specifically for Latin American and the use of U.S., and European frameworks for assessing the environmental sustainability of biofuels may reflect the interest of exportation of biofuels rather than local use. This concept that regulatory frameworks affect production and certification of Latin American biofuels is elaborated in the case study located at the end of this article.

Functional Units

An LCA should clearly specify the functional unit, which provides a reference to which the input data and output results are normalized and allows for comparisons among different fuel production systems (ISO 14044 2006). The review showed that the preferred functional unit is energy content of the biofuel (26/74) such as the lower heating value followed by mass of fuel (20/74), distance traveled by a vehicle (14/74) operated on pure biofuel, volume of fuel (8/74), and land area (7/74) (Fig. 1c). Most of the studies that used the energy functional unit compared the GWP of the biofuel with that of the fossil reference or against GHG emission savings targets stated by either the EU-RED or the US-RFS. Studies that used a distance-based functional unit meant to compare biofuel or their blends with the fossil reference. Studies that used a land-based functional unit compared different cropping scenarios or estimated the carbon payback time of the biofuel production system. Finally, the few studies that did not use any functional unit showed percentages of GHG reductions achieved by substituting fossil-based fuels by biofuels. The variation in functional units used makes comparison of LCA results difficult between biofuel pathways and even between the same biofuel pathways in studies conducted by different research groups. Over 60 % of the reviewed studies quantify the performance of biofuels in terms of energy giving confidence as a suitable functional unit; additionally targets for meeting global warming potential thresholds are expressed in g CO_2 eq/MJ. Thus, it would be useful to have LCA results based on MJ of produced energy. This is perhaps an area where policy-driven LCA frameworks can help (see Table 1) by standardizing the functional unit definition.

Table 1 Methodological metrics to estimate the GHG balance

Initiative	Functional unit	Allocation	Default factors	Selected time period	GHG emission reduction required
EU-RED	Energy content of fuel	Energy	Typical and default values	Annualized emissions over 20 years	35 %[a]
US-RFS2	Energy content of fuel	System expansion	EPA results to producer	100 years with 2 % discount rate or 30 year with 0 % discount rate	Conventional biofuel: 20 %[b] Advanced biofuels: 50 % Biomass-based diesel: 50 % Cellulosic biofuel: 60 %
RSB	Energy content of fuel	Economic[c]	Ecoinvent emission factors[c]	IPCC metrics	50 % for a blend of biofuels[c]

Adapted from van Dam et al. (2010)

[a] This value will rise to 50 % on January 2017 and will be 60 % on 2018 for those facilities which production starts on or after January 2017

[b] Below gasoline

[c] RSB (2011)

Allocation Methods

The partitioning of the inventory from input or output flows of a process or a product system between one or more products is called allocation (ISO 14044:2006). When the LCA study follows the recommendations of a regulatory framework such as RSB, US-RFS, or EU-RED, the allocation procedure is fixed (see Table 1). Figure 1d shows the frequency of allocation procedures reported in the reviewed studies. These results indicate that system expansion (23/74) followed by mass allocation (17/74), energy allocation (16/74), and economic allocation (15/74) are the most common methods. However, the largest number of studies (25/74) did not report the allocation method used, which was surprising because most biofuel production systems include one or more co-products which may be used as animal feed, power or heat production, or chemical intermediates. In some studies, the regulatory framework was discussed, but there was no clear indication of allocation or adhering to that framework. Different allocation criteria lead to considerably different results on the impacts even when considering the same agricultural and/or industrial assumptions (Amores et al. 2013; Bailis and Kavlak 2013; Bailis and Baka 2010; Consorcio 2012; Hilbert and Galbusera 2011; Iriarte et al. 2012; Krohn and Fripp 2012; Luo et al. 2009). When possible, different allocation criteria (mainly mass and energy) should be used in biofuel LCAs, to allow for proper comparisons among LCA results across different regulatory frameworks, and for evaluation of the final results when considering emission thresholds.

Biofuel Pathway Inputs and Sources of Inventory Data

The quality of LCA pathway inputs and inventory data will determine the quality of the study results, and it is always preferred to have site-specific inputs from biofuel producers along the supply chain. However, because advanced biofuels are not yet a commercial reality, availability of high quality inputs is often lacking, and estimation methods are largely relied on. Figure 2a shows the large variety of input and inventory sources chosen for LCA studies in the Pan American region. The most commonly cited sources of process inputs and inventory data are from the literature sources (65/74). Ecoinvent is the most commonly used life-cycle inventory database for this study group. SimaPro was considered a "data source" in Fig. 2a, when studies failed to report what databases were used within SimaPro. A discussion of LCA software used, such as SimaPro, is in "Biofuel Pathway Inputs and Sources of Inventory Data" section of the electronic supplementary material. Life-cycle inventory sources that are important in LCAs include land-use change models (such as GTAP), biogeochemical models for predicting soil organic carbon and nitrogen emissions (such as DAYCENT), and IPCC emission factors for dLUC emissions, among others. Wide variance with respect to data sources and primary data gathering methods demonstrates the need for LCAs to have the most current temporal and spatial data possible in order to generate the most accurate conclusions.

N_2O Emissions

Application of N fertilizers to biomass cultivation systems for biofuels can be an important source of GHG emissions, an important cause of groundwater contamination, and a primary reason for eutrophication of receiving waters (Cherubini and Strømman 2011). Subject to variation in nitrogen fertilizer requirements, biofuel GHG results can often be dominated by N_2O emissions. N_2O emissions are dependent on a number of soil and biomass production system parameters; soil properties, climate, irrigation and

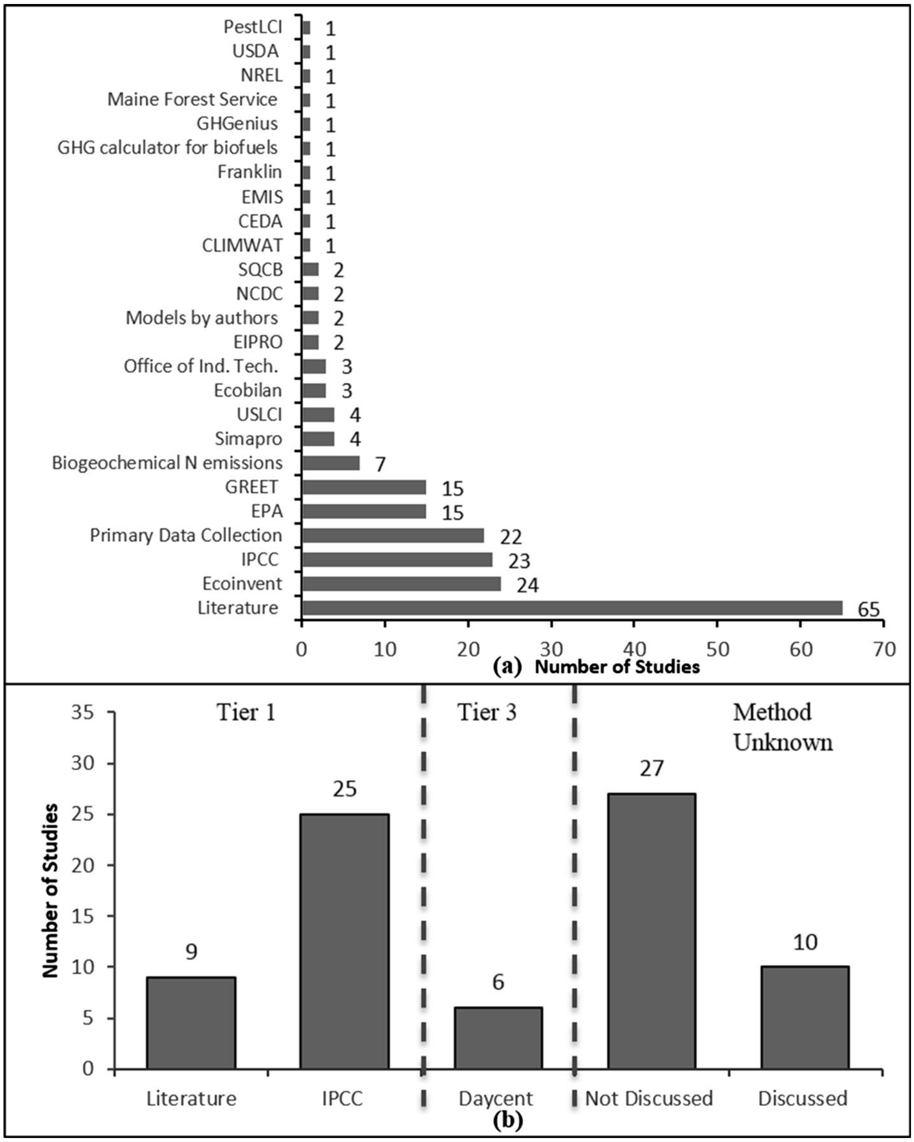


Fig. 2 **a** Number of articles using different sources of inputs and inventory data, **b** number of N_2O methodologies used according to IPCC tier categories. See electronic supplementary material glossary for more information on inputs and inventory data sources

tillage practice, and annual versus perennial crops (Cherubini and Strømman 2011). Type of N fertilizer can also impact biofuel GHG results because of the large differences in upstream emissions among different fertilizer types (Adom et al. 2012). In the impact assessment methodologies studied in this evaluation, N_2O is reported to have a global warming potential ranging from 276 to 310 times higher than that of CO_2, providing another source of variability in GHG results. Figure 2b shows the distribution of N_2O emission estimation methods categorized according to the IPCC as Tier 1, Tier 2, and Tier 3. Tier 1 is the simplest and most common method, employing a constant emission factor for both direct and indirect (NO_3^- leaching, NH_4^+ volatilization) mechanisms, 1.325 % of applied N is emitted as N in N_2O. Also, climate types are cataloged in a very wide classification that may lead to conclusions that are unrepresentative of actual conditions. For example, in a study comparing predicted emissions in two different climates, a 300 % difference was predicted between temperate-dry and temperate-humid climates (Galbusera and Hilbert 2011), none of which are representative of the actual locations, according to the authors.

Tier 2 and 3 methods are more detailed and depend heavily on site-specific data such as soil type, precipitation, climate information, etc. DAYCENT, CENTURY, and EPIC are examples of Tier 3 biogeochemical models which predict not only nitrogen cycle reactions, but also soil carbon, crop yield, and other system outcomes. Some studies use factors embedded within LCA software such as GREET and EBAMM, which use the IPCC tier 1 method. Twenty-seven of the articles do not discuss N_2O, and thus, it is unclear if these were included in the overall GHG emissions. Of the articles discussed, 10/74 mentioned how the application of fertilizer is very GHG intensive due to N_2O emissions but omits mentioning the method used to calculate those emissions. Allocation of N_2O emissions is highly dependent on yields of the crops, with most studies relying on single yield numbers, and year-to-year variations are rarely considered. Tier 1 methods were used 34/74, or 46 % of the time, whereas Tier 3 methods were only used in 8 % or 6/74 of the reviewed studies. Comparing the amount of studies that used Tier 1 over Tier 2 or 3 methods suggests that relative ease at calculating these values may be a factor. Using Tier 2 or 3 methods requires a vast amount of data and software. The lack of studies discussing N_2O emissions should be a reminder of the need for meticulous documentation of all GHG emissions.

Impact Assessment Categories and Methods

Impact categories are classified as midpoint or endpoint. The first approach focuses on potential environmental problems in the middle of the environmental cause and effect chain, while the second approach models additional mechanisms to estimate actual damage to human health, ecosystem quality, and resource depletion. Midpoint analyses are easier to model but require more knowledge of human health and ecosystem damage mechanisms by decision makers, and endpoint analyses are easier to interpret and to communicate.

Depending on the goal and scope of the study, one or more impact categories may be included in the life-cycle impact assessment (LCIA). While global warming potential (GWP) and energy consumption are often included in biofuel LCAs, a full environmental evaluation should consider other categories related to impacts to soil, water, air, human health, and ecosystems (Muench and Guenther 2013; Cherubini and Strømman 2011). The occurrence of the various impact categories in the reviewed articles is shown in Fig. 3. The articles included a wide range in impact categories, and an analysis was done to determine whether biofuels outperformed or underperformed fossil fuels most of the time. The most common categories found in these articles were GWP (GHG emissions) and energy demand (fossil and total), and both of these show biofuels generally out performing fossil fuel systems. As in other reviews of biofuel and bioenergy LCA literature (Larson 2006; Cherubini and Strømman 2011), biofuels were found in our study to underperform overall compared to fossil fuels in many categories, including acidification, eutrophication, dLUC, iLUC, and land occupation. Table 2 in the electronic supplementary material shows the impacts assessed for each reviewed article.

Figure 4 shows the frequencies of the various LCIA methodologies used in the reviewed studies and divides these into midpoint and endpoint impact indicator methods. Each of these methodologies considers a specific set of environmental impacts. For example, IPCC GWP 100a only includes GWP whereas EPA's TRACI also considers ozone depletion, acidification, cancer health impact, non-cancer effects, eutrophication, smog formation, eco toxicity, fossil fuel, land, and water uses. The GREET model and the Centrum voor Milieukunde Leiden (CML) and the Ecological Scarcity life-cycle impact assessment methods consider midpoint impact categories, while Ecoindicator 99 considers endpoint impact categories. The GREET model includes GWP, energy, and emissions of regulated pollutants contributing to acidification, smog formation, and health effects. Compared to the GREET model, CML includes also ecotoxicity related environmental impacts. The Ecological Scarcity method generates a single environmental index which requires that the impact categories be normalized according to a critical annual flow in the reference area and a set of factors that include data adapted for Switzerland. Ecoindicator 99 expresses the resource depletion as the surplus energy required for the extraction of mineral and fossil fuels in the future, the damage to ecosystem quality as the loss of species in a certain area and period of time, and the damage to human health as the number of years life lost and lived disabled (combined as Disability Adjusted Life Years, DALY) (PRé-consultants 2011). Both Ecological Scarcity and Ecoindicator 99 use varying ranges of impact categories and weighting factors for determining a single environmental score. Some methodologies such as CML 2001 and TRACI have similar categories (i.e., acidification and eutrophication) but the units used for analysis differ making comparisons between them difficult.

The single impact category present in nearly all reviewed LCAs was GWP. This was to be expected since one of the primary goals of biofuels is to reduce GHG emissions compared to conventional fossil fuels. The few studies which do not include the GWP instead focus on the energy consumption (Bruinsma 2009; da Costa et al. 2006; Pradhan et al. 2011; Velásquez et al. 2010) or water consumption (Mishra and Yeh 2011). The vast majority of biofuels outperform conventional fossil fuels within these two impacts. In other impact categories, especially those that are less studied (acidification and eutrophication) conventional fossil fuels outperform the majority of biofuels. This is mainly due to the large requirement of fertilizers for most biofuel feedstocks. In other impact categories, the results vary due to factors such as feedstock production, system boundaries, input data, transportation distances, energetic content, and blending.

Only 8 of the articles looked at the overall endpoint impacts (Cavalett et al. 2013; Consorcio 2012; Emmenegger et al. 2011; Koch 2003; Neupane et al. 2011; Yang et al.

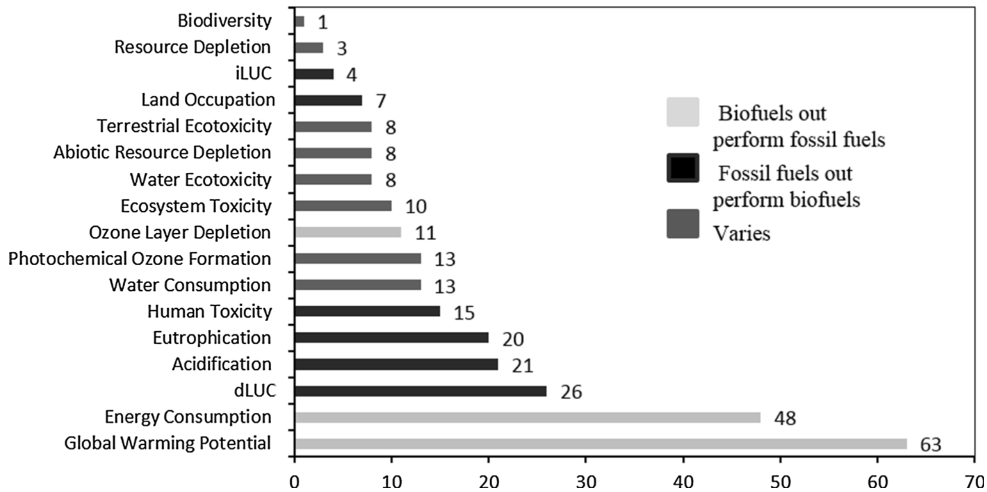


Fig. 3 Number of midpoint impacts studied within this assessment. See electronic supplementary material glossary for more information on impacts used

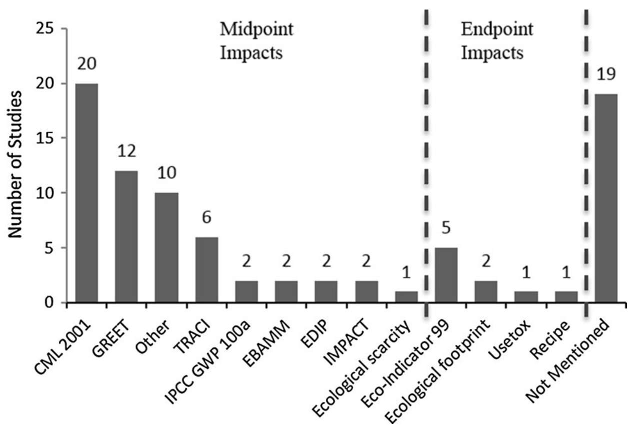


Fig. 4 Number of the articles studied using different impact assessment methods. See electronic supplementary material glossary for more information on impact assessment methods

2012), with the Cavalett et al. (2013) study performing multiple analyses comparing endpoint results from different LCA methodologies. Ecoindicator 99 is the most common LCIA method for analyzing endpoint impacts. Under this approach, biofuels seem to present a worse endpoint environmental impact than fossil fuels in part due to normalization and weighting factors strongly affecting the final results of endpoint impacts. Moreover, the factors are site specific and most are based on European conditions. No particular normalization and weighting factors for the Pan American region exist, which makes the use of the endpoint approach difficult and uncertain for this region. On the other hand, normalization values (Bare et al. 2006) and weighting factors (Thomas et al. 2007) are available for the US.

Water Consumption

Fresh water is considered a renewable, though finite, resource and as such its sustainable management must be considered. In LCAs of biofuels, some attention has been paid to land-use change and to some aspects of water degradation such as eutrophication, acidification, and aquatic ecotoxicity, but water consumption is seldom included. In the reviewed studies, less than 18 % or 13/74 papers considered water consumption, with one study comparing US and Brazilian scenarios (Chavez-Rodriguez and Nebra 2010). Most studies considering water consumption are from countries with an extensive and well-developed biofuel sector, such as the US and Brazil. Eight analyses were conducted in the United States (Chavez-Rodriguez and Nebra 2010; Chiu et al. 2009, 2012; Clarens et al. 2010; Mishra and Yeh 2011; Yang et al. 2011, 2012; Zaimes and Khanna 2013), 3 in Brazil (Cavalett et al. 2013; Chavez-Rodriguez and Nebra 2010; Ometto et al. 2009), 2 in Chile (Iriarte et al. 2010, 2012), and 1 in Argentina (Emmenegger et al. 2011). Water consumption is of particular importance where water scarcity is a prevalent issue (e.g., southwestern US, the northern region of Mexico, and the Norte Grande in Chile all deal with arid climates) and can be a limiting factor. In order to give an accurate view of the sustainability of biofuels, water consumption and its potential environmental impacts related to pressure on water availability (from ISO 14046) must be assessed, especially in water scarce and/or arid regions.

Quantitative Results

In this section, the quantitative impact that assumptions on allocation criteria and inventory data have on LCA results will be described. The focus of this section will be on GWP since most of the studies reviewed analyzed this impact category. This in no way implies that other environmental impacts are less important and is done to illustrate the source of variability in LCA studies conducted in the Pan

American region. The terms agricultural stage and industrial stage used here refer to all the activities involved in biomass production and biomass transformation into biofuels, respectively. The GHG emissions were analyzed as g CO_{2eq}/MJ. The article's original values expressed per kg of biofuel were then transformed considering a lower heating value of 26.8 MJ/kg for ethanol (Garcia et al. 2011) and of 37.1 MJ/kg for biodiesel (Iriarte et al. 2012). LCA results expressed in other functional units (such as land area) were not considered.

Effects of Allocation Method, Biomass Yields, and Pathway Inputs

This analysis considered 100 scenarios present in 32 articles. Figure 5 shows the life-cycle GHG emissions associated with biofuels production from different types of feedstock without including the dLUC emissions. Calculated GHG emissions using economic, energy, mass, or no specified allocation (Fig. 5a) tend to be greater than when using system expansion (Fig. 5b), which can even result in negative emissions (relative to the substituted system). Under system expansion, it is assumed that the co-products generated by the biofuels production system displace current products available in the market. Thus, the (relatively high) GHG emissions generated by the conventional products in the market are subtracted from the (relatively lower) total GHG emissions derived from the biofuel production system, which leads to lower GHG emissions than for attributional allocation. Variations in the GHG emission results may also be attributed to the assumptions on the agricultural stage (biomass yield and fertilizers required) and/or the technical level of the industrial stage (efficiency of the equipment). It is difficult and uncertain to identify whether allocation or differences in the inputs to the biofuel pathway causes a larger effect on the final results; however, some trends were uncovered as described next.

Allocation criteria and different assumptions on biomass cultivation and yields are responsible for the bulk of the variations on the GHG emissions from jatropha-based hydro-renewable jet (HRJ) production. Mass, economic, energy, and no specified allocation criteria present GHG emissions of 23–33 (first and second quartiles), 27–29 (second quartile), 28–40 (third quartile), and 45–78 g CO_{2eq}/MJ (fourth quartile), respectively (Fig. 5a). System expansion may generate GHG emissions benefits depending on the use of the co-products. Using these as substitutes of soybean meal or as boiler fuel resulted in GHG emission benefits of 300–391(first and second quartiles) and 134 g CO_{2eq}/MJ (third quartile), respectively, while using them as fertilizers lead to GHG emissions of 40 g CO_{2eq}/MJ (fourth quartile) (Bailis and Baka 2010; Bailis and Kavlak 2013).

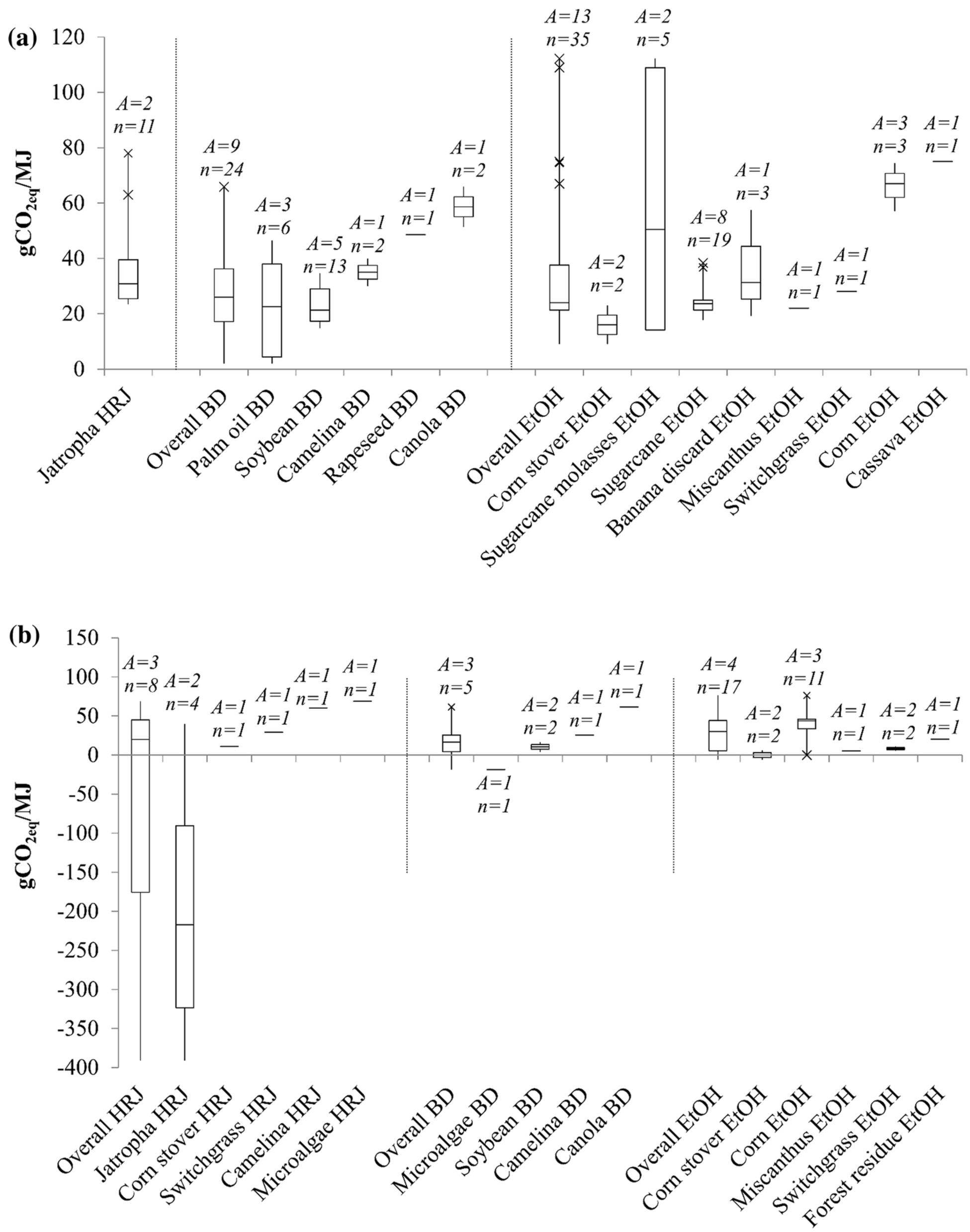
Allocation criteria and different assumptions on biomass cultivation and yields are also responsible for the variations on the GHG emissions of the biodiesel production from camelina and canola (Krohn and Fripp 2012). A similar situation occurs with the soybean-based biodiesel production. This biofuel presents GHG emissions of 15–20 (first quartile), 21–31 (second and third quartiles), and 23–35 g CO_{2eq}/MJ (third and fourth quartiles) under mass, not specified, economic, and energy allocation criteria, respectively (Hilbert and Galbusera 2011). Under system expansion, the soybean-based biodiesel achieves relatively lower GHG emissions of 4–17 g CO_{2eq}/MJ (Huo et al. 2008; Krohn and Fripp 2012).

Different assumptions on both the agricultural and industrial stages are responsible for the variations of the GHG emissions derived from the palm oil-based biodiesel production reaching GHG emissions of 2–46 and 10 g CO_{2eq}/MJ under Colombian (Castanheira and Freire 2011; Consorcio 2012), and Brazilian (de Souza et al. 2010) conditions, respectively.

The presence of non-typical data (denoted by the × symbol) for the overall ethanol production in Fig. 5a suggests that the estimated average of 24 g CO_2-eq/MJ is not representative of all feedstock sources, with the largest differences for corn, cassava, and sugarcane molasses. Allocation criteria and different assumptions on the industrial stage explain the variations on the corn-based ethanol production. Estimations of net GHG emission are 57 (first quartile) and 67–75 g CO_2-eq/MJ (second to fourth quartiles) under energy and not specified allocation criteria, respectively (Chavez-Rodriguez and Nebra 2010; Wang et al. 2012; Wu et al. 2006). Under system expansion and using natural gas for energy purposes make corn-based ethanol reach GHG emissions of 30–47 (first to fourth quartile). Different geographic locations within the US are responsible for this variation. The use of coal instead of natural gas makes the GHG emissions rise to 76 g CO_{2eq}/MJ (Liska et al. 2009).

Under Colombian conditions, the sugarcane molasses-based ethanol production reaches net GHG emissions of 14 g CO_{2eq}/MJ (first and second quartiles) with no significant differences between energy and economic allocation criteria (Consorcio 2012).The net GHG emissions of the sugarcane molasses-based ethanol production under Mexican conditions, and considering the energy allocation criteria, are 50–112 g CO_{2eq}/MJ (third and fourth quartiles) (Garcia et al. 2011). Different assumptions on the industrial stage, such as the boiler efficiencies, the electricity requirements, and the ethanol yield per ton of cane, are responsible for this variation.

Fig. 5 *Box* and *Whisker* plot showing the minimum, maximum, and non-typical data of the GHG emissions not including the LUC effect (**a**) from studies considering mass, energy, economic, or no allocation (**b**) from studies considering system expansion. *A* refers to the number of articles and *n* to the number of analyses



There seems to be a consensus on the GHG emissions derived from the ethanol production from the sugarcane juice in Brazil. Such emissions range between 18 and 28 g CO_{2eq}/MJ (first to third quartiles) depending on the cultivation and industrial conditions assumed. Higher GHG emissions of 29 and 37–38 g CO_{2eq}/MJ (fourth quartile) are estimated for Argentinean and Mexican conditions, respectively.

The banana discard-based ethanol production under Costa Rica conditions reaches GHG emissions of 19 g CO_{2eq}/MJ if no fertilizers are required, while under Ecuador conditions, the resulting GHG emissions for an organic farm and a conventional farm are 31 and 57 g CO_{2eq}/MJ, respectively (Graefe et al. 2011).

The low GHG emissions attributed to lignocellulosic ethanol (produced from corn stover, miscanthus, switchgrass, or forest residue) are mainly due to the assumption of using the residual lignin for process heat and power cogeneration (Wang et al. 2007, 2011a) and export of excess electricity to displace coal-derived or grid mix electricity.

Overall, the variability in GHG emissions suggests that process inputs, rather than LCA methodology differences, are more important for these studies. Since many LCA inputs such as crop yields undergo significant changes throughout the years from random variation in annual weather conditions, it is important to also focus on long-term studies, rather than single "snapshot" LCAs of a given biofuel pathway.

Regarding the breakdown of the GHG emissions by stage, there is significant variability in the data resulting mainly from the relatively low number of articles that analyzed this issue (see Fig. 2 in the Electronic supplementary material), which hinders reaching conclusions about the individual contributions. While in general, the agricultural stage appears to be the largest contributor to the net GHG emissions, in the case of lignocellulosic-based and soybean-based biofuels, it is lower than the industrial stage because these feedstock sources are considered as either crop residues or N-fixing crops needing low N fertilizer inputs (Agusdinata et al. 2011; Bailis and Baka 2010; Graefe et al. 2011; Luo et al. 2009).

Direct Land-Use Change (dLUC) Effects

This analysis considered 61 scenarios present in 15 articles. Initially, we will refer to the GHG emissions that do not consider the dLUC effect as the base GHG emissions (Fig. 5). The GHG emissions that do include dLUC emissions of CO_2 will be referred as the net GHG emissions. Figure 6 shows the net GHG emissions by the biofuels production from different types of feedstock. Similar to results in Fig. 5, the net GHG emissions using economic, energy, mass, or not specified allocation criteria (Fig. 6a) tend to be larger than when using system expansion (Fig. 6b). Most of the biodiesel LCAs consider low-carbon-content soils such as savannah, pastureland, or grassland as reference land types (Castanheira and Freire 2011; Galbusera and Hilbert 2011; Iriarte et al. 2012; Iriarte and Villalobos 2013), while studies on ethanol production also consider forest deforestation (Amores et al. 2013; Consorcio 2012; Garcia et al. 2011). This helps explain the apparently lower net GHG emissions for biodiesel than for ethanol (Fig. 6a). Studies that employ the GREET model (Kim and Dale 2009; Krohn and Fripp 2012; Wang et al. 2011a, 2012) include both domestic and international direct and indirect LUC, and it is not possible to extract only the dLUC portion. Overall, the effect of the dLUC is either an increase or a reduction of the base GHG emissions of the biofuel production depending on the dLUC scenario assumed.

For the jatropha-based HRJ production, the effect of the dLUC on the base GHG emissions is a reduction of about 11–27 g CO_{2eq}/MJ when the cultivation takes place on pasturelands, reaching net GHG emissions of 13–17 g CO_{2eq}/MJ (first and second quartiles). However, the cultivation on grasslands and shrub lands lead to net GHG emissions of 56 (third quartile) and 140 g CO_{2eq}/MJ (fourth quartile), respectively, which means an increase of about 16–112 g CO_{2eq}/MJ compared to base GHG emissions (Bailis and Baka 2010).

For the palm oil-based biodiesel production in Colombia, the cultivation on savannah results in net GHG benefits of 13–43 g CO_{2eq}/MJ (first to third quartiles) depending on the degradation level of the soil; in other words, its effect is a reduction of about 52–82 g CO_{2eq}/MJ in the base GHG emissions. On the other hand, the effect of the cultivation on displaced forests is an increase of about 4–85 g CO_{2eq}/MJ in the base GHG emissions, reaching net GHG emissions of 49–124 g CO_{2eq}/MJ (fourth quartile) (Castanheira and Freire 2011).The challenge in multipurpose crops that are not produced specifically for biofuel production is trying to calculate the impact on dLUC of the derivation of a co-product of the crop as in the case of soybean oil (18 % of oil in the seed) (Galbusera and Hilbert 2011). A similar situation occurs with the ethanol production from sugarcane (Amores et al. 2013) and sugarcane molasses (Amores et al. 2013; Consorcio 2012; Garcia et al. 2011). The net GHG emissions of the soybean-based biodiesel, considering the economic allocation criteria, are 7–21 (first and second quartiles) and 52–105 g CO_{2eq}/MJ (third and fourth quartiles) when the cultivation takes place on agricultural lands (changing the crop) and on pastureland, respectively. The effect of the dLUC on the base GHG emissions is then a decrease of about 1–15 g CO_2eq/MJ and an increase of about 30–83 g CO_{2eq}/MJ when cultivation takes place on agricultural land and on pasturelands, respectively (Galbusera and Hilbert 2011). However, soybean cultivation on agricultural lands may incur iLUC emissions as other lands, such as forestlands, can be converted to croplands in an attempt to tradeoff the area used for the soybean biofuel cultivation, which is a topic that requires further studies. The dLUC effect on the base GHG emissions of the rapeseed-based biodiesel production in Chile is an increase of about 7 g CO_{2eq}/MJ considering that the cultivation takes place on non-degraded grasslands, reaching net GHG emissions of 56 g CO_{2eq}/MJ (Iriarte et al. 2012).The sugarcane-based ethanol production in Brazil reaches net GHG emissions of 36–45 g CO_2eq/MJ (first quartile) when the cultivation takes place on typical savannah and/or pasturelands. The effect of the dLUC is an increase of about 17 g CO_2eq/MJ in the base GHG emissions (Souza et al. 2012). Under Mexican conditions, the GHG emissions are 65–67 (second quartile), 72–74 (third quartile), and 135–137 g CO_2eq/MJ (fourth quartile) when performing the cultivation on tropical dry forests, grasslands, and rainforests, respectively. In other words, the effect of the dLUC is an increase of 32–100 g CO_2eq/MJ on the base GHG emissions (Garcia et al. 2011). Considering direct deforestation of rainforest, the sugarcane-based ethanol production in Argentina reaches net GHG emissions up to 560 g CO_2eq/MJ (Amores et al. 2013).

The net GHG emissions of the sugarcane molasses-based ethanol production depend on the assumptions on both the industrial stage conditions and the dLUC scenarios

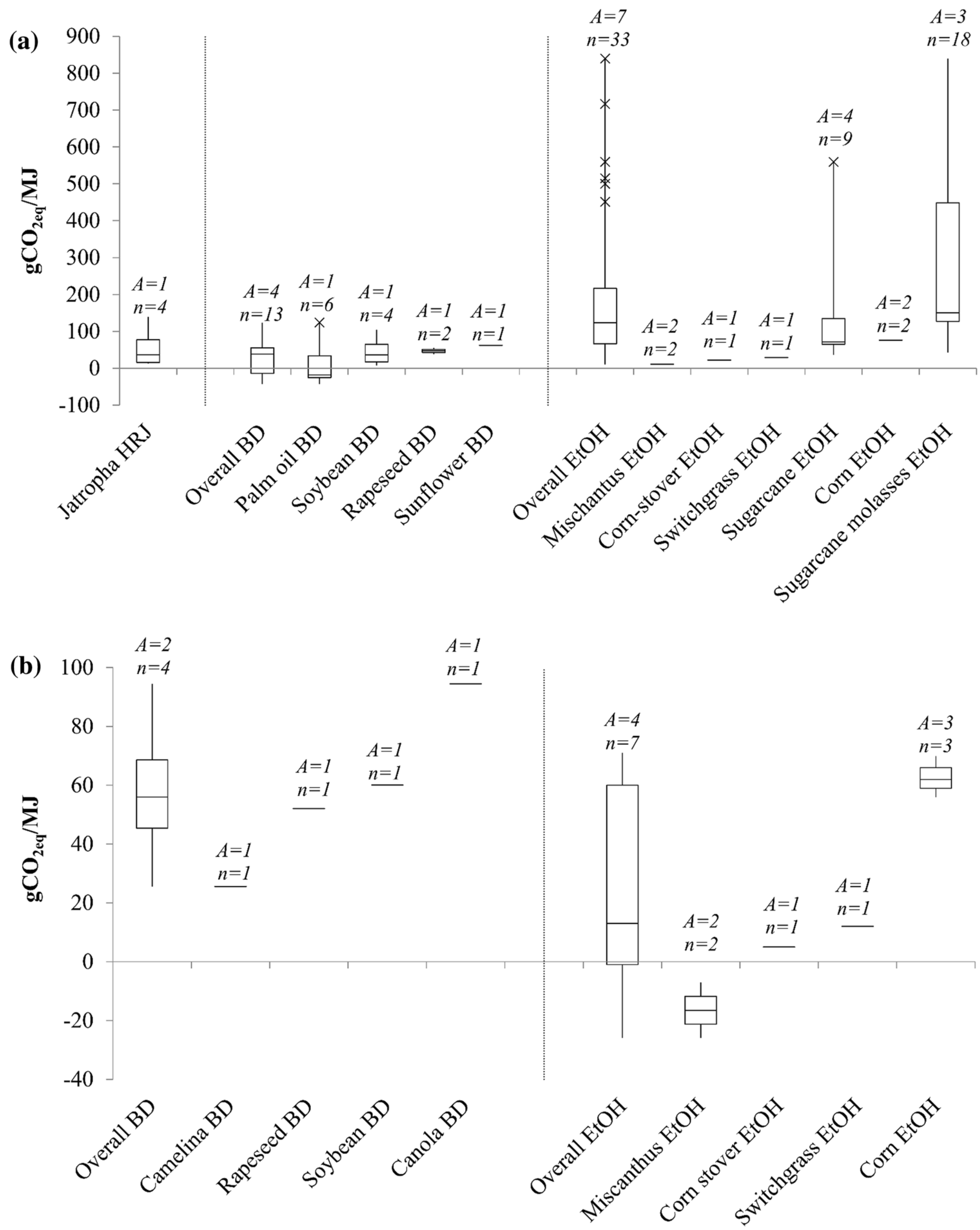


Fig. 6 *Box* and *Whisker* plot showing the minimum, maximum, and non-typical data of the GHG emissions including the dLUC effect (**a**) from studies considering mass, energy, economic, or no allocation (**b**) from studies considering system expansion. *A* refers to the number of articles and *n* to the number of analysis. Corn, corn stover, miscanthus, and switchgrass include also iLUC GHG emissions

considered. The use of all of the sugarcane molasses leads to net GHG emissions ranging between 43 and 123 g CO_{2eq}/MJ (first quartile) depending on the dLUC scenario assumed (Consorcio 2012; Garcia et al. 2011). Similarly, the use of a portion of the molasses leads to net emissions of 140–224 g CO_2eq/MJ (second and third quartiles) (Garcia et al. 2011). These trends were estimated for Mexico and Colombia. When the dLUC involves direct deforestation under Argentinean conditions, the net GHG emissions range from 440 to 839 g CO_2eq/MJ (third and fourth quartiles) depending on the allocation criteria used (Amores et al. 2013) (Fig. 6a). Overall, the effect of the dLUC on the base GHG emissions of the sugarcane molasses-based ethanol production in Colombia is an increase of about 27 g CO_{2eq}/MJ when cultivation takes place on shrublands (Consorcio 2012). In the case of Mexico, the effect of the dLUC is an increase of about 97–116, 28–32, and 24–41 g CO_2eq/MJ when cultivation takes place on rainforests, tropical dry forests, and grasslands, respectively (Garcia et al. 2011).

Studies on the ethanol production from corn, corn stover, miscanthus, and switchgrass did not specify the

LUC scenario considered. Furthermore, two studies gathered both the dLUC and the iLUC emissions of GHG as simply LUC GHG emissions (Wang et al. 2011, 2012). Thus, Kim and Dale (2009) estimated that the corn-based ethanol production reaches average net GHG emissions of 56 g CO_2eq/MJ, while Wang et al. (2011, 2012) estimated total GHG emissions of 62–70 g CO_2eq/MJ when including both dLUC and iLUC GHG emissions. The effect of the dLUC and the iLUC on the base GHG emissions derived from the corn-based ethanol production is an estimated 9 g CO_2eq/MJ increase (Wang et al. 2012). Other studies, however, have estimated that the dLUC and the iLUC emissions of GHG derived from the corn-based ethanol production may be higher, ranging from 20 up to 104 CO_2eq/MJ depending on the above- and below-ground carbon content of the soil and the treatment of the emissions at different times (Wang et al. 2011). The net GHG emissions of the ethanol production from corn stover and switchgrass are 5 and 12 g CO_2eq/MJ (Wang et al. 2012), respectively, including both the dLUC and the iLUC effects. On the other hand, the miscanthus-based ethanol production results in GHG emissions benefits (negative emissions) of 7 and 26 g CO_2eq/MJ considering and not considering the iLUC effect, respectively (Scown et al. 2012; Wang et al. 2012). Overall, the LUC effect on the base GHG emissions (including both the dLUC and the iLUC effect) of ethanol production from switchgrass and corn stover is almost null, while for miscanthus, the effect is a reduction of 12 g CO_2eq/MJ (Wang et al. 2012).

Regarding the breakdown of the GHG emissions by stage, adding the GHG emissions derived from the dLUC to the agricultural emissions makes this stage the major contributor to net GHG emissions for biofuels production for all types of feedstock, with the exception of corn and corn stover (see Fig. 3 in the Electronic supplementary material). In the case of corn, some analyses assume old industrial conditions that require coal for energy production (Kim and Dale 2005), which explains the high contribution of the industrial stage. However, the use of more recent data that reflect the current corn-based ethanol production leads to a different trend where the agricultural stage is the major contributor to the net GHG emissions (Wang et al. 2011, 2012). On the other hand, considering corn stover as a residue results in GHG emissions of the agricultural stage coming mainly from the supplement of fertilizers to compensate the nutrient loss from stover removal (Wang et al. 2011a, 2012).

Regulatory Frameworks and Certification Schemes for Biofuel Sustainability

Currently, several regulatory frameworks and certification schemes are available that aim to assess the sustainability of biofuels production. The Testing Framework for Sustainable Biomass (TFSB) or "Cramer Criteria," and the EU-RED are examples of regulatory frameworks, while the Roundtable on Sustainable Biomaterials (RSB), the International Sustainability & Carbon Certification (ISCC), and the Global Bioenergy Partnership (GBEP) are examples of certification schemes (BEFSCI 2011). These initiatives analyze a range of factors associated with the biofuel's supply chain including air quality, biodiversity, energy security, GHG emissions, land-use change, soil quality, and water use, and in all cases rely on LCA results. The most critical factor in these certification schemes and regulatory frameworks is the GHG emissions. The main regulatory framework and certification schemes explicitly state the guidelines to be used in the LCA.

Table 1 shows a brief comparison of these metrics along with those developed by the RSB. The guidelines developed by the US-EPA and the EU-RED are the most commonly employed in LCAs conducted in the Pan American countries, as shown in Fig. 1b. Several of the certification schemes listed above require production to meet or exceed the regulatory frameworks of the EU-RED. "Regulatory Frameworks and Certification Schemes for Biofuel Sustainability" section in the electronic supplementary material discusses how dLUC can affect the ability of biofuels to meet certification schemes, such as EU-RED.

The use of certification schemes and/or regulatory frameworks allows for comparison and assessment of environmental management between biofuel production systems. Several of the certification schemes (RSB, ISCC, Bonsucro) have criteria that must be met concerning land use, soil, water, air, waste, and several other social and environmental indicators (Solomon and Bailis 2014). These certification schemes with a focus on environmental quality could help ensure that the best possible environmental management of these biofuel systems across the Pan American region is being used.

Case Study: The GWP of Jatropha HRJ Production in the Yucatan State of Mexico—Effects of Regulation-Driven Allocation Requirements

Introduction

"Qualitative Results" and "Quantitative Results" sections presented a wide range of biofuel LCA results when different study assumptions are used. In addition, it was mentioned in "Regulatory Frameworks and Certification Schemes for Biofuel Sustainability" section that certification schemes and regulatory frameworks have the potential to standardize biofuel LCA around a set of accepted practices. This section presents a case study LCA of hydro-renewable jet (HRJ) produced from *Jatropha* oil in Mexico using LCA methods required by US-EPA RFS2 and EU-RED and compares GHG results to each other and to fossil jet. Rather than making LCA results agree, the LCA results diverge because the LCA methods are different between the regulatory frameworks in the US and the EU.

Jatropha curcas (referred to as jatropha) is a shrub plant that produces seeds with high oil (40 wt%) content that can be grown on marginal soils and therefore can help restore eroded areas. The tallest variety grows to 6 m height, has adapted to a variety of climate conditions (from subtropical to arid), and can grow in low fertility soil (FACT 2010). The most suitable climate conditions for jatropha cultivation are within a belt extending from 30N to 35S straddling the equator. Seed productivities have historically been between 0.3 and 6 dry ton/ha/year depending on rainfall and soil quality. The entire plant has been used for erosion control, as a hedge plant, medicinal use, and for firewood. The fruit of the plant has been used as a combustion source and fertilizer. The seed oil has been used in lamps, for cooking, as an engine fuel, and for soap making; the seed cake has been used as a fertilizer, an input for biogas and charcoal production, and for combustion (FACT 2010).

This case study presents results from a jatropha cultivation project in the Yucatan region of Mexico with conversion of extracted oil to hydro-renewable jet (HRJ) fuel in Cancun. The effects of LCA allocation method are explored—system expansion versus energy allocation conducted, according to both US and EU frameworks.

LCA Methods

Goal and Scope

The goal of this limited LCA is to evaluate the greenhouse gas emissions associated with the production of HRJ derived from jatropha oil grown on marginal agriculture lands in the Yucatan peninsula of Mexico. The study scope is cradle-to-grave starting with jatropha cultivation and concluding with combustion of HRJ in jet engines. Both attributional and consequential modelings were done depending on allocation (energy and system expansion, respectively). Results of the LCA for the proposed HRJ are compared with impacts of producing, and using fossil jet fuel and savings of GHGs are computed.

Production Site and Carbon Stocks

Figure 7 shows the locations of jatropha cultivation, oil extraction at Uman, and HRJ production at Cancun. The land area bordered by green in this figure is the location of the proposed plantations of jatropha. A report by the Universidad Autonoma de Chapingo from June 2010 cataloged the canopy cover and carbon content of above- and below-ground biomass of native vegetation in the jatropha plantation area. Table 3 in the Electronic supplementary material details the carbon content for several land categories from acreage in the green bordered area from Fig. 7. The carbon stock values are used to estimate direct land-use change (dLUC) emissions in this study. This analysis assumes cultivation on 55,000 ha with an average annual yield of 10 metric ton/ha of wet seeds to produce a total wet weight of 550,000 ton of seeds. No indirect LUC effects are included because Jatropha will not be grown on agricultural lands. The study assumes oil extraction will occur at Uman, and the resulting jatropha oil will be transported by truck to Cancun for processing to HRJ.

Biofuel Pathway, Functional Unit, and Allocation Methods

The major life-cycle stages for this study are shown in Fig. 8; jatropha cultivation and harvesting, jatropha seed and shell transport, jatropha oil extraction at Uman, jatropha oil transport, jatropha HRJ production in Cancun, and HRJ combustion. The study assumes that the seed will be dried at the site of harvesting using natural gas and transported by truck along with the shell and husk to a processing plant in Uman, 95 km distant on average from harvesting sites. The plantation will utilize wastewater from adjacent pig farms for irrigation and soil nitrogen amendment. Additional chemical fertilizer will also be required—data for this requirement were provided by the company “KUO Bioenergía.” The jatropha oil is extracted from the seeds at the plant through mechanically pressing the seeds, application of heat, and hexane extraction.

The base line analysis assumes that the residual shell, husk, and seed cake are combusted to generate electricity for internal use in oil extraction and for export to the Yucatan grid. The functional unit for this LCA is 1 MJ of energy released upon combustion of each fuel product, HRJ and petroleum jet fuel. Energy allocation was used to attribute environmental burdens to pathway co-products (see Fig. 8); system expansion was also employed as an alternative allocation scenario to conform to the US-EPA RFS2. A pathway diagram for system expansion, along with further details of the allocation methods and factors are presented in the Electronic supplementary material (Fig. 5).

Process Inputs and Inventory Data

Inputs at each stage of the HRJ life cycle were developed on an annual basis for all 55,000 ha of jatropha plantation and conversion of the entire amount of jatropha oil to HRJ. All input data to the HRJ life cycle were obtained from

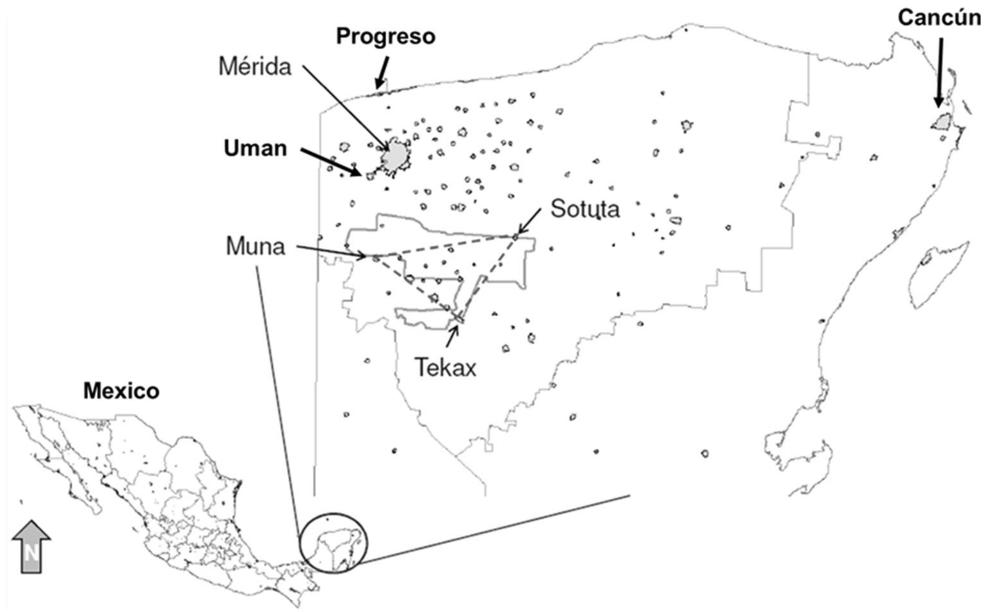


Fig. 7 Locations of Jatropha cultivation, oil extraction at Uman, and HRJ production at Cancun

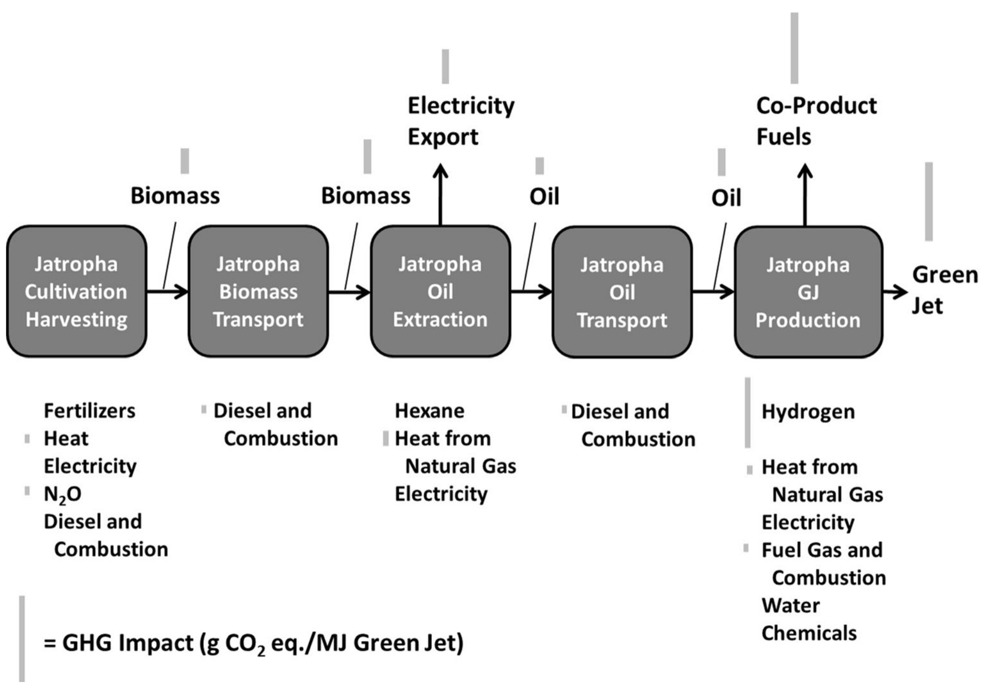


Fig. 8 Life-cycle stages for the analysis of HRJ from jatropha using energy allocation according to the USA framework. The *gray bars* indicate greenhouse gas (GHG) emission impacts, and height of bar is proportional to the degree of impact. GHG impacts accumulate as material moves through the life cycle due to the input and use of material and energy (shown below each stage). Impacts exit the HRJ product life cycle when co-products are created and exported, and are allocated to HRJ and co-products using energy allocation by considering the material flows of products and their lower heating value

original documents of the company KUO Bioenergía, and from the company UOP for the HRJ production processes, and standards based upon IPCC guidelines for N_2O emissions from fertilizer application. Unless otherwise noted, the ecoinvent^{TM} database from SimaPro 7.2 was utilized to develop inventory data for all inputs to the life cycle. Input tables for each stage of the pathway are included in the Electronic supplementary material (Tables 3–8). The allocation according to energy of the inventory data to HRJ from the various life-cycle stages is shown in the Electronic supplementary material in Table 9.

Impact Assessment

Environmental impacts are limited to global warming which was calculated using the IPCC 2007 GWP 100a method in SimaPro 7.2 version. In this method, CO_2 has a global warming potential (GWP) of 1, $CH_4 = 25$, and $N_2O = 298$. A full accounting of the GWP of solvents and refrigerants is included in the analysis using inventory elements from the ecoinvent^{TM} database in SimaPro 7.2. The annual GHG emissions are divided by the energy content (in MJ) of the annual HRJ production to arrive at the desired result: g CO_2

eq/MJ HRJ. Emissions of CO_2 from combustion processes involving biomass fuel or HRJ are *not* counted toward the GHG totals, because they are considered carbon neutral, unlike fossil CO_2 emissions which *are* counted.

GHG Results

The GHG emissions for Yucatan jatropha HRJ for the different allocation methods are shown in Table 2 along with savings compared to fossil jet fuel. The GHG results are organized based on major stages of production along the biofuel pathway. Allocation methods are correlated with regulatory frameworks in the US and EU. The US Department of Energy (US-DoE) uses energy allocation in which *all co-products* are included in the allocation factor calculations. US-EPA uses system expansion which is the method employed to determine whether biofuels qualify toward targets established in the RFS2. EU-RED is energy allocation in which co-products from oil extraction are considered as products with negligible value (wastes) and *are not included* in the calculation of allocation factors. As explained in the Electronic supplementary material (ESM) ("Energy allocation according to the EU-RED" section), the allocation case of EU-RED includes some minor changes to inputs to the HRJ pathway relative to the US-DoE and US-EPA cases, but otherwise, the same inputs were used (see Tables 11, 12 in ESM).

The highest emission stages are HRJ production and jatropha cultivation, with all other stages, including dLUC, being of minor importance. In the cultivation stage, diesel fuel use, electricity, fertilizers, and N_2O emissions are the most important inputs and source of inventory data. In the HRJ production stage, H_2 generation and process heat, both from natural gas, are the two dominant inputs. In the US-EPA case using system expansion, very large emission credits for co-products from oil extraction and HRJ production dominate the results. Savings of GHG emissions compared to fossil jet fuel are greater than $\sim$80 % for all cases and significantly larger for the system expansion case.

Interpretation of GHG Results

The GWP results from this study achieve large savings compared to fossil jet fuel and would qualify as an advanced biofuel under the RFS2 standard (>50 % savings) and also under EU-RED (>35 % savings). The different requirements for each of the allocation approaches add effort and complexity to the LCA. The US-DoE energy allocation is an often-used approach in which all co-products of economic value are allowed in the calculation of the allocation factors. The EU-RED case does not allow energy allocation for certain co-products such as extraction residues which are considered wastes according to the regulation, though they may be economically viable as a renewable energy source in certain cases. In system expansion (the US-EPA case), the calculated changes to the environment are attributed to the system as a whole, with no possibility of dividing the total impact among co-products. Attributional modeling, on the other hand, does not attempt predicting actual changes to the environment due to limited modeling of market product displacements; however, all products and co-products are assigned environmental burdens, proportionally to their mass, volume,

Table 2 GHG emissions for Jatropha HRJ for three allocation methods compared to petroleum jet fuel

GHG emissions	Fossil jet[a]	US-DoE	US-EPA	EU-RED
Jatropha cultivation/harvest (HRA)	6.8	1.5	7.8	1.8
Jatropha seed, shell transport (RMT)	1.3	0.5	2.5	0.4
Jatropha oil extraction		1	5.2	0.2
Jatropha oil transport		0.7	1.3	0.7
HRJ production from jatropha oil (LFP)	6	16.4	30.7	14.6
Co-product credit extraction stage			−61.4	
Co-product credit GJ production stage			−70	
Final production transport	1			
Fossil jet fuel combustion	77.7			
dLUC		−0.8	−4.1	−3.2
Total	92.9	19.3	−88.0	14.5
GHG emissions savings (%)		79.2	194.7	84.4

[a] From Skone and Gerdes (2008)

RMA raw material acquisition, *RMT* raw material transport, *LFP* liquid fuel production, *US-DoE* energy allocation, *US-EPA* system expansion (displacement) allocation, *EU-RED* energy allocation; electricity export from oil extraction not included energy content, or market price. Each allocation has advantages and limitations, and therefore, for the near term, it is likely that biofuel environmental LCAs for any potential biofuel feedstock pathway will have to conform to one or more regulatory frameworks with regard to allocation and other LCA methodology aspects. As a final note, cultivation of jatropha at this study site was halted in 2013 due to lower than acceptable yields from these marginal lands. This testifies to the importance of updating system inputs through time before considering LCA results representative of real conditions. As of this writing, cultivation of jatropha for the purpose of evaluation for feasibility as a biofuel feedstock continues at other study sites in the state of Yucatan.

Recommendations for Research and Other Efforts to Enhance LCA and Improve Environmental Management of Biofuels and Bioenergy

This critical evaluation of biofuel LCAs in the Pan American region reveals a wide range of important study features such as regulatory-driven frameworks, modeling assumptions (study scope, functional units, and allocation), inventory databases, environmental impacts assessed, and biofuel pathways in different countries. When considering the most commonly encountered potential environmental impact, global warming potential, these LCA characteristics caused a wide range in GHG emission results, even when the results are converted to the same basis (g CO_2 eq/MJ biofuel). Normally, these complications make it very difficult to compare environmental performance between different biofuel pathways in different locations, and more importantly, make it difficult to understand the main drivers for environmental damage so that improvements to biofuel production systems can be achieved. However, in our quantitative analysis of GHG emission results from many of the articles studies, we were able to determine the magnitude of change in emissions for two main study features, allocation, and LUC, admittedly with some significant effort. Other aspects of biofuel LCA such as crop yield, inputs to the agricultural and industrial stages, could have been quantified on a statistical basis but were beyond the scope of this study. It is also important to note that while the bioenergy sector has adopted, many of the best practices of conducting LCAs, broader adoption of more progressive mandates such as peer review of industrial process LCAs prior to certification, will require higher confidence in the results of the methodology.

In order to take full advantage of LCA as an environmental management tool, we offer a number of recommendations to guide future research with the goal of improving the quality and utility of study results.

- *Biofuel LCA Guidance Frameworks* A number of regulatory-driven biofuel LCA frameworks and certification schemes are currently in effect throughout the Pan American region. Based on our evaluation of the literature and the case study, these frameworks and schemes are one of the most important reasons for divergence in LCA study results. Future research should quantify the impact of these frameworks and schemes for the same pathways in order to isolate this single study variable. The use of Product Category Rule (PCR) for biofuel production can help with comparisons between different pathways and feedstocks due to the data collection and other LCA methods being standardized. PCR's follow ISO standards and can therefore be used within other frameworks and certification schemes. Understanding the variations within other frameworks and certification schemes will become increasingly important as the research community develops new methods for incorporating local sociological values and the wide variety of sustainability metrics into LCAs.
- *Life-Cycle Inventory Data Quality* There is a need to improve inventory data quality so that the output from LCA of biofuels and bioenergy systems are more accurate and useful. Inventory data reside in the industrial sphere and also in the context of cultivation systems and ecosystems. In the past, industry-funded confidential benchmarking studies have shown that there is a wide range of energy efficiency and extent of pollution control for industrial production of key inputs needed for biofuel and bioenergy production (fertilizers, industrial chemicals, electricity, etc.). There is also a great need of field validated data for carbon stocks and N_2O emissions from fertilizer use, both of which have a large impact on GHG emissions and water quality. Research is needed to understand this variability and to incorporate it on a statistical basis for use in uncertainty analysis in LCA. This will be particularly challenging with respect to the need to incorporate sociological indicators and metrics associated with labor rights, land-use change, and other impacts on rural communities.
- *Cultivation Systems* Important changes will occur to soils when land transitions into bioenergy cropping on a large scale. These changes will affect inputs to biofuel and bioenergy LCA in the form of inventory data used in the analysis of GHG emissions and other important categories of environmental sustainability. Research needs to continue and to be expanded in regional scope in both experimental and modeling aspects on soil properties such as organic matter, carbon, cycling of nutrients (N, P, K), management of water, erosion, emissions, and yields. This research should be conducted in a coordinated way

on multiple cropping systems, various climates, and in multiple locations throughout the Pan American region in order to capture the effects of local conditions that impact a variety of processes (e.g., transpiration rates, water yield relationships drought tolerances.)

- *Life-Cycle Impact Categories* The focus in biofuel and bioenergy LCA has been on GHG emissions due to the mandates in regulatory frameworks. However, LCA should anticipate future environmental issues, and therefore, the scope of environmental impacts must expand. Future research should continue with GHG emissions but also expand with increasing momentum into water availability and degradation issues (nutrient runoff and management), biodiversity, criteria air pollutants (PM, H_2S, Hg, etc.), and human/ecosystem toxicity.
- *Systems Analysis for Sustainability* Although beyond the scope of topics in this review, it is worth noting that in the opinion of the authors, LCA is an ideal platform to integrate information and data across the entire biofuel pathways, and depending on system boundary, may also include data from technical and natural systems outside of the direct pathway. LCA not only is capable of including this indirect data into the analysis but also is able to contribute to meta analyses by contributing environmental assessments for a full spectrum sustainability analysis. The meta analyses would include techno-economic analyses, regional and global economic analyses, environmental impact analyses, and societal impact analyses. Future research should address the data, framework, and methodology issues in systems analysis for sustainability. This will become increasingly important as biofuels and bioenergy in general become deeply imbedded in the global energy markets which will make it more difficult to understand coupling between the integrated systems (e.g., water-energy nexus.)
- *Carbon Neutrality* Most of the articles reviewed assumed carbon neutrality in the carbon cycle of the biofuel production. The carbon neutrality assumption eases the analysis toward the GWP impact. However, this assumption does not consider the timing of the CO_2 emissions, since when the biofuel is burned, the carbon stored is released instantly as CO_2 to the atmosphere while the carbon sequestration process in the next cycle of biomass production can take a longer time period.
- *Outreach* Another topic beyond the scope of issues in this LCA review is outreach, but the authors consider this a high priority. Programs should be considered for translating LCA research out to the professional communities who are impacted by study results, to policy makers, and also to the general public. As done effectively in agricultural and forestry industries, outreach into the biofuels production, community would be an effective mechanism to disseminate the best sustainable practices.

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