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LCA FOR RENEWABLE RESOURCES

Life cycle assessment of bio-based ethanol produced from different agricultural feedstocks

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Abstract
Purpose Bio-based products are often considered sustainable due to their renewable nature. However, the environmental performance of products needs to be assessed considering a life cycle perspective to get a complete picture of potential benefits and trade-offs. We present a life cycle assessment of the global commodity ethanol, produced from different feedstock and geographical origin. The aim is to understand the main drivers for environmental impacts in the production of bio-based ethanol as well as its relative performance compared to a fossil-based alternative.
Methods Ethanol production is assessed from cradle to gate; furthermore, end-of-life emissions are also included in order to allow a full comparison of greenhouse gas (GHG) emissions, assuming degradation of ethanol once emitted to air from household and personal care products. The functional unit is 1 kg ethanol, produced from maize grain in USA, maize stover in USA, sugarcane in North-East of Brazil and Centre-South of Brazil, and sugar beet and wheat in France. As a reference, ethanol produced from fossil ethylene in Western Europe is used. Six impact categories from the ReCiPe assessment method are considered, along with seven novel impact categories on biodiversity and ecosystem services (BES).
Results and discussion GHG emissions per kilogram bio-based ethanol range from 0.7 to 1.5 kg CO_2 eq per kg ethanol and from 1.3 to 2 kg per kg if emissions at end-of-life are included. Fossil-based ethanol involves GHG emissions of 1.3 kg CO_2 eq per kg from cradle-to-gate and 3.7 kg CO_2 eq per kg if end-of-life is included. Maize stover in USA and sugar beet in France have the lowest impact from a GHG perspective, although when other impact categories are considered trade-offs are encountered. BES impact indicators show a clear preference for fossil-based ethanol. The sensitivity analyses showed how certain methodological choices (allocation rules, land use change accounting, land use biomes), as well as some scenario choices (sugarcane harvest method, maize drying) affect the environmental performance of bio-based ethanol. Also, the uncertainty assessment showed that results for the bio-based alternatives often overlap, making it difficult to tell whether they are significantly different.
Conclusions Bio-based ethanol appears as a preferable option from a GHG perspective, but when other impacts are considered, especially those related to land use, fossil-based ethanol is preferable. A key methodological aspect that remains to be harmonised is the quantification of land use change, which has an outstanding influence in the results, especially on GHG emissions.

Keywords Bioethanol · Bio-based · Biogenic feedstock · LCA · Maize · Sugarcane · Sugar beet · Wheat

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1 Introduction

There is a global trend to move away from a dependence on a fossil fuel-based economy driven by such factors as combating climate change, the anticipation of rising prices for fossil fuels and the desire of many countries to reduce external resource

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dependency (de Jong et al. 2012). Bio-based products are often considered preferable to fossil-based alternatives due to their renewable nature; however, the environmental profile of a product can be more complex than that, and hence a life cycle approach is required to assess potential benefits as well as trade-offs (Miller et al. 2007). A typical example of a bio-based commodity chemical is ethanol, which is used mainly as a biofuel, but also as a solvent, raw material for chemical production and for human consumption (Linak et al. 2009). Around 74,000 million litres of bioethanol were produced in 2009, with USA and Brazil accounting for 88 % of the total world production (Biofuels Platform 2010). Ethanol is mainly produced by fermentation of sugars but it can also be produced synthetically from petroleum, although the latter route is rapidly declining relative to fermentation (Linak et al. 2009). Bio-based ethanol is currently produced using a variety of agricultural feedstocks such as maize, sugarcane, wheat, sugar beet, molasses, cassava as well as cellulosic biomass. Ethanol represents an interesting example of a bio-based chemical that is currently produced in high volumes and with potentially quite different environmental profiles, depending on the feedstock used and the region in the world where it is produced. To date, ethanol has been extensively investigated in life cycle assessment (LCA) studies, mainly from a transport bio-fuel perspective (Fu et al. 2003; Nguyen and Gheewala 2008a, b; Ometto et al. 2009). However, very often the literature is focused on the assessment of greenhouse gas (GHG) emissions only (Kim and Dale 2009; Silalertruksa and Gheewala 2011; Laborde 2011; USEPA 2010), and most of the cited studies tend to focus on a single agricultural feedstock for ethanol production. Recent work conducted under the UNEP/SETAC Life Cycle Initiative has led to the development of novel methods to account for impacts of land use and land use change on biodiversity and ecosystem services (BES) (Koellner et al. 2013). This is of the utmost importance in the context of bio-based products.

In this article, we present the results of an LCA study of ethanol production with different agricultural feedstock in different regions: maize grain and maize stover in the USA, sugarcane in two regions of Brazil and wheat and sugar beet in France. The aim of the study was to identify the main activities in the life cycle driving the various potential environmental impacts and to explore the differences between feedstocks, as well as to benchmark bio-based ethanol with fossil-based ethanol from a cradle-to-grave perspective. The novelty of this work lies in the following aspects: it assesses ethanol as a feedstock for the chemical industry, rather than as biofuel which has been the focus of all studies to date; it consistently assesses a relatively high number of production routes; it incorporates a full set of novel impact categories focusing on BES; and thorough sensitivity and uncertainty analyses are carried out to ensure the robustness of the results.

2 Methods

The approach taken in this case study is a descriptive (attributional) LCA and the functional unit is 1 kg of ethanol with a water content of 5 %.

2.1 Investigated countries and biomass resources

The choice of feedstock and producing countries was based on market research and production volumes, agricultural practices and on life cycle inventory data availability. Due to their global importance, production from maize in the USA and from sugarcane in Brazil was considered. Maize grain and stover were assessed separately, and for sugarcane

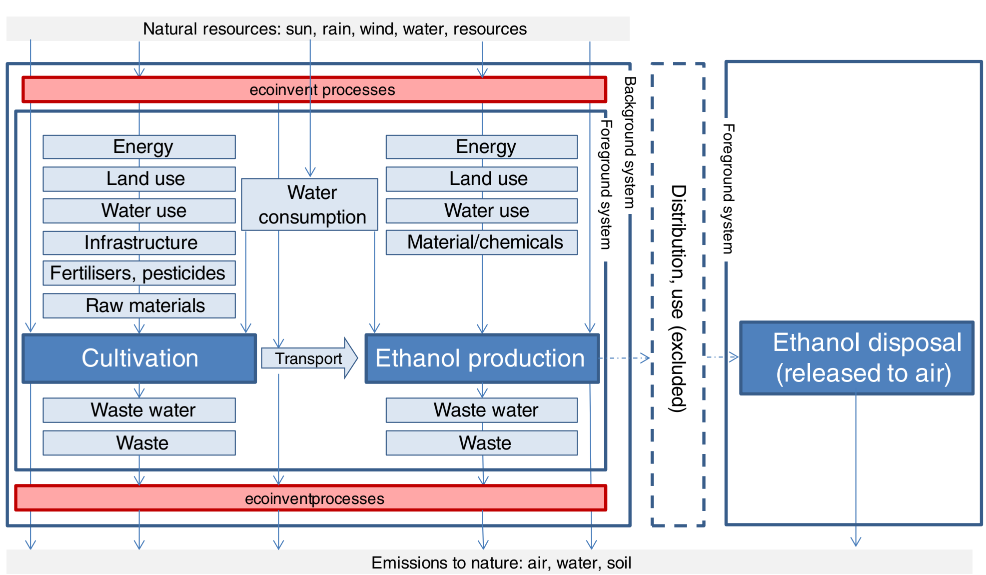

Fig. 1 Product system under study

Table 1 Key figures for the cultivation and processing of the different feedstocks under study (Flury and Jungbluth 2012)

Cultivation	Unit	Sugarcane, BR CS	Sugarcane, BR NE	Maize grain, US	Maize grain and stover, US[a]	Sugar beet, FR	Wheat, FR
Yield, main product	kg/ha	82,697	57,677	9,703	9,703	84,649	6,605
Allocation	%				81 %		100 %
Co-product					Stover		Straw
Yield, by-product	kg/ha				4,676		0
Allocation	%				19 %		0 %
Seed	kg/ha	6,616	4,614	21	21	2	135
N-fertiliser	kg/ha	76	31	157	167	103	150
P_2O_5 fertiliser	kg N/ha	25	22	67	73	68	24
K_2O fertiliser	kg P_2O_5/ha	102	75	82	120	146	23
Lime	kg/ha	305	213	366	366	0.1	12
Pesticides	kg/ha	6	4	3	2.5	3	3
Diesel	kg/ha	129	90	46	46	160	68
Land burned in pre-harvest	%	70 %	70 %				
Land transformation	ha/ha	0.032	0.032				0.016
Transport, total	tkm/ha	2,771	1,933	479	479	1,432	338
Water consumption	m3/ha	1,884	4,579	3,168	3,168	184	87
Processing	Unit	Sugarcane, BR CS	Sugarcane, BR NE	Maize grain, US	Maize stover, US[a]	Sugar beet, FR	Wheat, FR
Feedstock input	kg/kg ethanol	15.4	15.4	2.5	3.9	16.9	2.5
Chemicals	kg/kg ethanol	0.03	0.03	0.02	0.21	0.43	0.02
Electricity	kWh/kg ethanol			0.17		0.09	0.1
Heat	MJ/kg ethanol			2.6			5.1
Transport	tkm/kg ethanol	0.33	0.33	0.19	0.43	0.98	0.27
Water consumption	L/kg ethanol	−6.1	−6.1	−0.3	4.9	6.5	0.31
Allocation factors:							
Ethanol	%	50 %	50 %	84 %	87 %	89 %	73 %
Electricity	%	8 %	8 %		13 %		
Sugar	%	41 %	41 %				
Vinasse	%	0.30 %	0.30 %				
Filter cake	%	0.30 %	0.30 %				
DDGS	%			16 %			27 %
Pulp	%					11 %	
Overall ethanol yield	kg/ha	2,693	1,873	3,299	4,350	4,423	1,914

[a] Cultivation data refers to maize cultivation and harvesting of both grain and stover. Allocation is then applied to determine the burdens of stover. Processing data refers to stover only

two Brazilian regions were assessed due to climatic differences: the North-East region and the Centre-South region. Wheat and sugar beet were assessed in France as it is an important European ethanol-producing country.

2.2 System boundaries

Ethanol production from the considered feedstock was assessed from cradle-to-gate, as shown in Fig. 1. This includes cultivation of the feedstock, its transport to the mill and the processing to ethanol. Furthermore, end-of-life emissions are included in order to allow a full comparison of the GHG emissions associated with bio-based and fossil-based ethanol (see Section 2.6.5). All remaining activities between the factory gate and disposal are excluded, as these are common to all scenarios.

2.3 Allocation

When maize stover is harvested along with grain, maize cultivation becomes a multi-output process, which was modelled by means of allocation based on economic value. Also, the processing of all feedstocks to ethanol results in other co-products, as shown in Table 1. Again, allocation based on economic value was performed. Even though the ISO standards on LCA identify economic allocation as the last feasible option, the literature shows that it is one of the most common procedures for allocation (Ardente and Cellura 2011). The reasons to choose economic allocation in this study were several: first, our inventories for all bio-based ethanol production routes were built on the existing ecoinvent 2.2 datasets for bioenergy production (Jungbluth et al. 2007), which also employed economic allocation. Another practical reason was the fact that ethanol production mills usually export surplus electricity, which cannot be handled through, e.g. mass allocation. Allocation factors were nevertheless subject to a sensitivity analysis (see Section 3.4.1).

2.4 Uncertainty

Uncertainty associated with the data was assessed by estimating standard deviations with the pedigree matrix (Frischknecht et al. 2010) which were in turn propagated with the Monte Carlo analysis tool in Simapro v.7.3.2 (Pré Consultants 2012). In total, 1,000 simulations were performed for each ethanol production route.

2.5 Impact assessment methods

The following indicators from the ReCiPe method (Goedkoop et al. 2009) were considered at mid-point level: terrestrial acidification potential (TAP), freshwater eutrophication potential (FEP), marine eutrophication potential (MEP), photochemical oxidant formation potential (POFP) and agricultural land occupation (ALO). With regard to the assessment of GHG emissions, global warming potentials (GWP) from carbon dioxide (CO_2) and methane were used as proposed by Muñoz et al. (2013) for a 100-year period, accounting for methane oxidation in the atmosphere and considering biogenic CO_2 emissions as neutral, with the exception of those resulting from land use change (LUC). In particular, biogenic CO_2 and methane are applied GWP values of 0 and 25 kg CO_2 eq/kg, respectively, whereas their fossil counterparts are applied values of 1 and 27.75 kg CO_2 eq/kg, respectively. For the remaining GHG emissions, such as nitrous oxide, among others, the GWP from the climate change impact indicator in ReCiPe were used.

The following mid-point impact methods for BES were included: biodiversity damage potential (BDP) (de Baan et al. 2012), climate regulation potential (CRP) (Müller-Wenk and Brandão 2010), biotic production potential (BPP) (Brandão and Milà i Canals 2012), freshwater regulation potential (FWRP), erosion regulation potential (ERP), water purification potential through physicochemical filtration (WPP-PCF) and water purification potential through mechanical filtration (WPP-MF) (Saad et al. 2013).

Water use was inventoried and assessed but this topic has been the subject of an independent document (Flury et al. 2012); therefore, it is not reported here.

2.6 Data collection and inventory analysis

2.6.1 Cultivation

Table 1 shows the most important material inputs for the cultivation of the different feedstock. For a detailed list of data sources used to build this inventory, see Flury and Jungbluth (2012). Emissions to air of ammonia, nitrous oxide and nitrogen oxides due to fertiliser input were estimated following Nemecek et al. (2007), whereas nitrogen losses to groundwater were estimated with a factor of 30 % nitrogen input loss according to de Klein et al. (2006), with the exception of sugarcane, where a 2.5 % loss according to Stewart et al. (2003) was assumed. Emissions of phosphorus to water as a consequence of fertiliser input were also estimated following Nemecek et al. (2007). In the case of maize, emission factors from Jungbluth et al. (2007) were used. Emissions to air associated with pre-harvest burning of sugarcane fields were obtained from several sources (ADEME 2010; FAO 2010; Macedo et al. 2008; Tsao et al. 2011).

2.6.2 Land use

Land use (also called land occupation) was directly obtained from the yield and the cultivation period, which is 7 months for maize, 8 months for wheat and 5 months for sugar beet. Sugarcane is cultivated for 5.5 years, after which a fallow period of 6 months is required before fields can be replanted. The latter was also taken into account as occupation. The expected biome within each country and region was determined by expert judgement, as shown in Table 2. As it can be seen in the table, two biomes were considered for maize, assuming a 50/50 split.

Table 2 Biomes considered for land used in feedstock cultivation

Crop/ feedstock	Source country/region	Biome
Maize grain/stover	United States of America	50 % temperate broadleaf and mixed forests, 50 % temperate grasslands, savannas and shrublands
Sugar beet	France	Temperate broadleaf and mixed forests
Sugarcane	Centre-South, Brazil	Tropical and subtropical grasslands, savannas and shrublands
Sugarcane	North-East, Brazil	Deserts and xeric shrublands
Wheat	France	Temperate broadleaf and mixed forests

Land use associated to the processing of the ethanol and to background processes was included, but specific biomes were not defined. As a consequence, generic characterisation factors for BES were used for this type of land. Generic factors were also used for occupation flows associated to fossil-based ethanol.

2.6.3 Land use change

LUC (also called land transformation) is a key environmental aspect of bio-based products; however, to date, there is a lack of a standardised approach to quantify it and translate it into impacts such as GHG emissions, among others. In this study, LUC attributable to each crop was determined at a country level following the stepwise procedure by Milà i Canals et al. (2012). FAO statistics from 1990 to 2009 showed that only sugarcane in Brazil and wheat in France could be attributed LUC according to this method. In that period of time, the harvested area of those two crops in those countries had increased (FAO 2011a), while at the same time the total cultivated area of permanent crops in Brazil, as well as the total arable land cultivated in France, had increased too (FAO 2011b). It could consequently be assumed that the increase in area cultivated for those crops had been at the expense of another land use type. The amount and type of other land use types impacted by the increase in area harvested for these crops were determined using FAO statistics. In the supplementary material we provide, the LUC estimations for all crops were assessed. The amount of LUC estimated for sugarcane in Brazil was of 0.032 ha LUC per ha cultivated in the 1990 to 2009 period, which was assumed to be forest, as this is the only land use type with a decreasing area over the same period. Research by Adami et al. (2012) has shown that from 2005 to 2010 only 0.6 % of sugar cane crop expansion in Brazil took place directly on forest. As a consequence, it can be concluded that the LUC quantified in our study corresponds to indirect expansion rather than direct expansion. As for wheat in France, the estimated LUC was of 0.016 ha LUC per ha cultivated, for the 1990 to 2009 period, which is mostly attributed to expansion over meadows and pastures.

The GHG emissions due to LUC were calculated according to Flynn et al. (2012), taking into account the climate zone, soil type, land use, crop type and country, as shown in Table 1 of the Electronic Supplementary Material (ESM). The resulting LUC emissions, expressed as CO_2 eq were 23 tonnes/ha LUC/year (for a 20-year period) in both the Centre-South and North-East regions of Brazil, and 6.6 tonnes CO_2 eq/ha LUC/year in France.

2.6.4 Ethanol production

Table 1 shows the main material inputs to the production of ethanol from the different feedstock. Data were obtained from several sources, as reported in Flury and Jungbluth (2012).

Production of ethanol based on fossil resources was modelled with the generic ecoinvent dataset 'ethanol from ethylene, at plant, RER', which represents production of ethanol in Western Europe by direct hydration of ethylene (Sutter 2007).

2.6.5 GHG emissions from end-of-life

Ethanol is not used as a fuel by Unilever but as a solvent in household and personal care products. As a consequence, rather than combusted, ethanol will be degraded after being discharged to the environment, either to air, or to water via urban sewage. In this study, we considered release to air by evaporation after use, which is a plausible assumption given the relatively low boiling point of ethanol (78.4 °C). After release, part of the ethanol is degraded abiotically to CO_2 in air but part is also expected to end up in water bodies, where it could be degraded to methane if anaerobic conditions are present (Muñoz et al. 2013).

Emissions of CO_2 and methane following release of ethanol to air were estimated by Muñoz et al. (2013) as 1.85 and 0.023 kg per kg ethanol, respectively. This equals to 0.58 and 2.49 kg CO_2 eq per kg of bio-ethanol and fossil ethanol, respectively, after applying the GWP values mentioned in Section 2.5.

3 Results and discussion

3.1 GHG emissions

Figure 2 shows the GHG emissions of ethanol production from cradle-to-gate plus the emissions from its degradation in the environment at the end-of-life. CO_2 eq emissions from cradle-to-gate range from 0.7 to 1.5 kg per kg ethanol and from 1.3 to 2 kg per kg ethanol if emissions from degradation at end-of-life

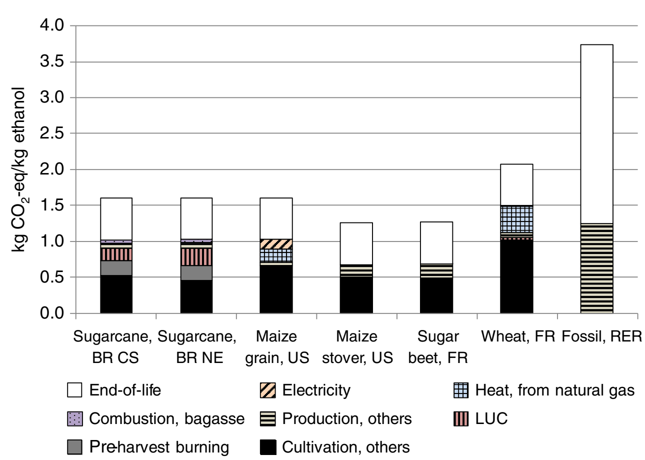

Fig. 2 GHG emissions of ethanol from different agricultural feedstocks and from fossil resources, including emissions from degradation of ethanol at the end-of-life phase (ethanol emitted to air)

Table 3 Results for the ReCiPe indicators considered, per kilogram ethanol

Impact indicator	Unit	Sugarcane, BR CS	Sugarcane, BR NE	Maize grain, US	Maize stover, US	Sugar beet, FR	Wheat, FR	Fossil, RER
Global warming potential (GWP)	Kg CO_2-eq	1.60	1.61	1.60	1.25	1.27	2.07	3.74
Photochemical oxidant formation potential (POFP)	kg NMVOC	0.029	0.029	0.003	0.003	0.005	0.004	0.006
Terrestrial acidification potential (TAP)	kg SO_2 eq	0.012	0.011	0.009	0.009	0.007	0.015	0.003
Freshwater eutrophication potential (FEP)	kg P eq	1.3E-04	1.4E-04	4.5E-04	3.9E-04	1.8E-04	2.1E-04	4.7E-04
Marine eutrophication potential (MEP)	kg N eq	0.0130	0.0120	0.0060	0.0040	0.0020	0.0170	0.0001
Agricultural land occupation (ALO)	m^2 year	2.07	2.95	1.23	0.93	0.87	2.78	0.01

are included. Even though carbon in bio-based ethanol is biogenic, there is a significant contribution from expected methane emissions due to degradation when part of the ethanol is partitioned to the water compartment (Muñoz et al. 2013). For fossil-based ethanol, emissions are of 1.3 kg CO_2-eq from cradle-to-gate and of 3.7 kg CO_2-eq when end-of-life degradation emissions are included. The latter are higher than for bio-based ethanol due to the fossil origin of the carbon. Thus from a cradle-to-grave perspective, bio-based ethanol can involve a significant (ca. 50 %) GHG reduction when compared with its fossil-based counterpart.

When the bio-based ethanol alternatives are compared, sugar beet and maize stover appear as the least GHG-intensive feedstock, whereas wheat is the most intensive, with twice as high GHG emissions per kilogram product. All the other feedstocks have similar emissions from cradle-to-gate, of around 1 kg CO_2 eq per kg. The low impact of sugar beet is closely linked to its high yield. Sugarcane in the studied regions of Brazil also has high yields, but these alternatives are penalised by the emissions related to pre-harvest burning and LUC, showing higher overall emissions than sugar beet. It can be seen that there is little difference between the Brazilian regions: the Centre-South region has higher yields, but this is achieved by means of higher inputs of agrochemicals and energy (see Table 1). The low emissions for maize stover are related to the fact that the impact of maize cultivation is shared between the two co-products (grain and stover), and the lower economic value of stover benefits the latter in terms of its share of impact. However, the difference between these two feedstocks in GHG emissions is not as big as one could expect, due to the fact that harvesting stover as opposed to leaving it in the field necessitates an additional fertiliser input to correct for the loss in nutrients.

GHG emissions from cradle-to-gate are available in the literature for some of the feedstocks, although several methodological aspects make the comparison difficult, such as how multifunctionality is dealt with (allocation/system expansion) and the inclusion of LUC emissions. Ethanol from sugarcane in Brazil has been assessed with GHG emissions per kilogram of 0.74 (Renouf et al. 2008) and 2.15 kg CO_2 eq (California EPA 2009). The first value is lower than the one obtained in this study because it excludes LUC emissions, whereas the second one is higher because it includes a global assessment of indirect LUC emissions. In our study, only indirect LUC emissions taking place in Brazil were taken into account. Previous studies on maize ethanol in the USA (California EPA 2009) highlight the same aspect, namely the importance of whether or not and how LUC emissions are dealt with. In the sensitivity analysis, further discussion on LUC and GHG emissions is provided.

3.2 Other ReCiPe impact indicators

Table 3 shows the results obtained by each alternative in the other ReCiPe indicators considered in the study. In terms of ALO, MEP and TAP, all bio-based alternatives appear to have higher impact compared to fossil-based ethanol. On the other hand in POFP all bio-based alternatives except sugarcane are preferred to fossil-based ethanol. The relatively higher impact of sugarcane is related to the emissions from pre-harvest burning. With regards to FEP, all bio-based alternatives but maize grain and stover are preferable to fossil ethanol. This is due to high emissions of P from the fossil ethanol production plant. However, the data source for these emissions is very old (Meyers 1986) and its ability to represent current industry performance is put into question. The reason for the poorer performance of maize grain and stover is the relatively higher input of phosphorus fertilisers per kilogram ethanol.

When it comes to ranking the different bio-based alternatives, again a mixed picture is obtained. For example, sugarcane performs best in FEP but worst in POFP. Wheat performs well in POFP, but it is the worst option in terms of MEP, due to nitrogen losses through wastewater generated in the mill, as well as diffuse losses from fertilisers. Maize grain and stover perform well in POFP, but as previously mentioned, they rank poorly in FEP.

3.3 Impact indicators for BES

Results for the seven BES impact indicators are shown in Fig. 3. The first observation to be highlighted is that fossil-based ethanol has a relatively negligible impact in all indicators. This is clearly

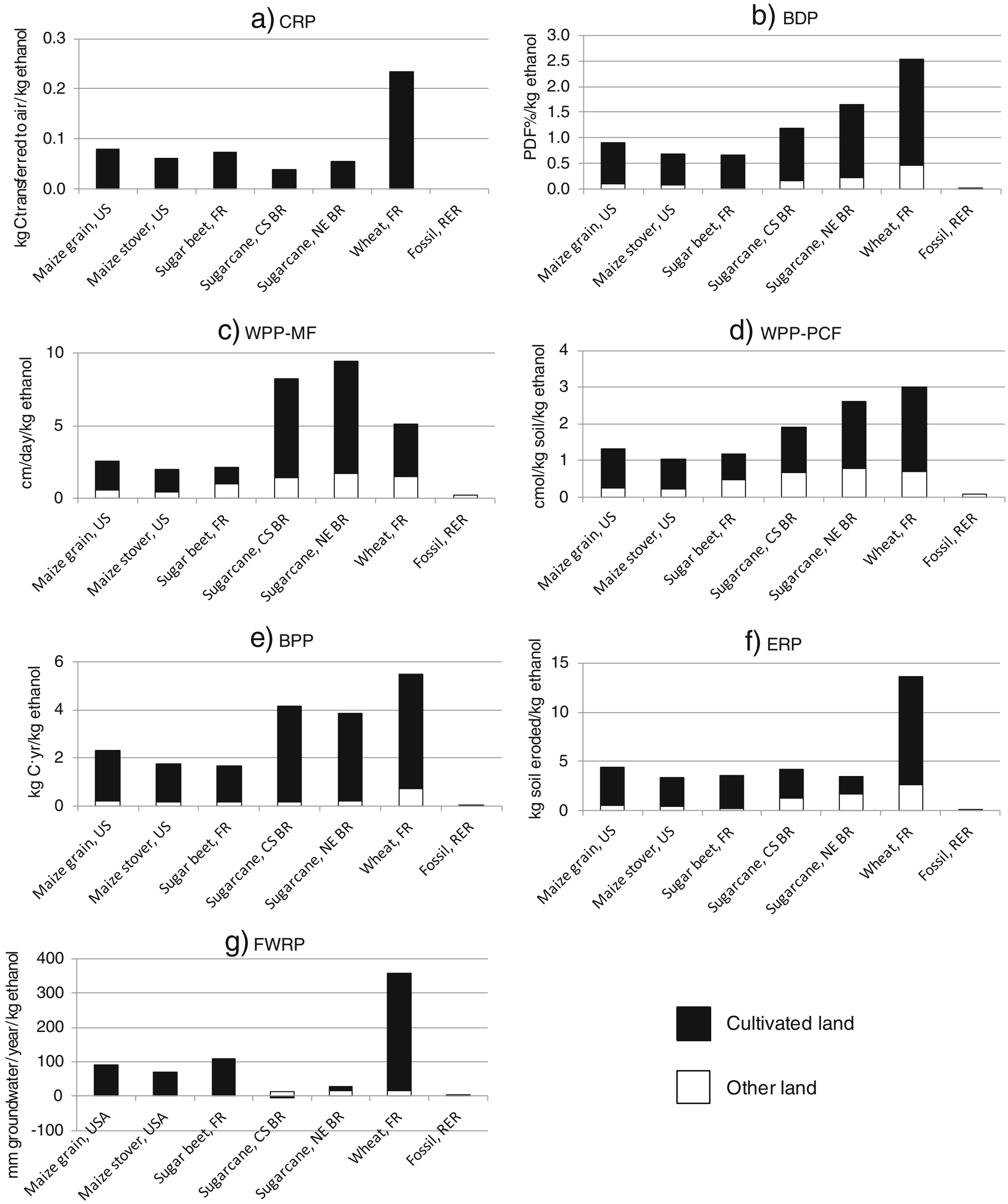

Fig. 3 Results on biodiversity and ecosystem services' impact indicators: **a** *CRP*, climate regulation potential; **b** *BDP*, biodiversity damage potential; **c** *WPP-MF* water purification potential through mechanical filtration; **d** *WPP-PCF* water purification potential through physico-chemical filtration; **e** *BPP* biotic production potential; **f** *ERP* erosion regulation potential; **g** *FWRP* freshwater regulation potential

related to the fact that land occupation per kilogram ethanol is very low when a petrochemical feedstock is used. This has also been observed in the ALO indicator in Table 3. The dominance of BES impacts by land occupation has also been reported by Milà i Canals et al. (2012).

In general, impact scores are lower (except FWRP) for production of ethanol from maize grain, maize stover and sugar beet compared with sugarcane and wheat. This is mainly driven by the overall yield of ethanol per hectare of feedstock crop. The overall yield per feedstock is determined by a number of factors including the crop's yield, allocation of the co-products grain and stover in the case of maize, the ethanol yield during ethanol production and allocation of co-products during ethanol production. The highest overall yield per hectare is obtained by the production chain from sugar beet followed by maize grain and stover (Table 1). On the other hand, the production chain with sugarcane from the North-East region of Brazil followed by wheat yield the lowest output per hectare.

In the case of CRP (Fig. 3a), Müller-Wenk and Brandão (2010) argue that the results of this impact category should be added to other GHG emissions (Fig. 1). We have not done so in Fig. 1 because CRP relates to the 'foregone sequestration', i.e. the amount of C not fixed in biomass due to a past LUC, and not to actual GHG emissions. If CRP was converted to CO_2 eq and added to the cradle-to-grave GHG emissions, the results would still be favourable for all options of bio-based ethanol when compared to fossil-based ethanol, although the performance of wheat-based ethanol would be significantly eroded, with GHG emissions only 20 % lower than those of the fossil-based ethanol.

3.4 Sensitivity analyses

Several aspects were subject to a sensitivity analysis to check their influence on the results of the ReCiPe impact indicators. Below, we describe these analyses and the results obtained.

3.4.1 Allocation in ethanol production

Due to increased demand of ethanol as a fuel and basic chemical, its price could rise, therefore affecting the economic allocation in the production process. We modelled the system assuming as that, as a worst case, 100 % of the environmental burdens are allocated to ethanol and 0 % to any co-product (DDGS, electricity, etc.). The results (Fig. 1 in ESM) showed a sharp increase in all impacts for ethanol from sugarcane, but much lower for the other feedstocks. Due to the equally valuable by-product sugar, the allocation factor for ethanol from sugarcane is doubled, resulting in a doubled score in all impacts, with the exception of GHG emissions, which increase by 63 %. However, given that there is an increasing global demand for sugar, it is unlikely that 100 % of the revenue from sugarcane products will be attributed to ethanol in the future. In terms of overall ranking of feedstocks, this sensitivity analysis does not change the main results obtained, i.e. bio-based feedstocks still present lower GHG emissions than fossil-based ethanol, but the former are still more impactful in many of the other ReCiPe impact categories.

3.4.2 Pre-harvest burning of sugar cane

The Brazilian State of Sao Paulo aims for zero pre-harvest burning in 2021. This will lead to a reduction in impacts from open burning of biomass but also to increased mechanisation. When sugarcane is modelled without pre-harvest burning, there is a reduction in impacts (Fig. 2 in ESM), most notably in POFP (66 % reduction), but also in other indicators such as GHG emissions (9 % reduction) and TAP (20 % reduction). The big reduction in the POFP category is related to the avoidance of emissions of volatile organic hydrocarbons, nitrogen oxides and sulphur dioxide from open burning, which are key contributors to this impact category. Similarly, in the climate change impact category, the reduction is associated with avoiding the dinitrogen monoxide and methane emissions

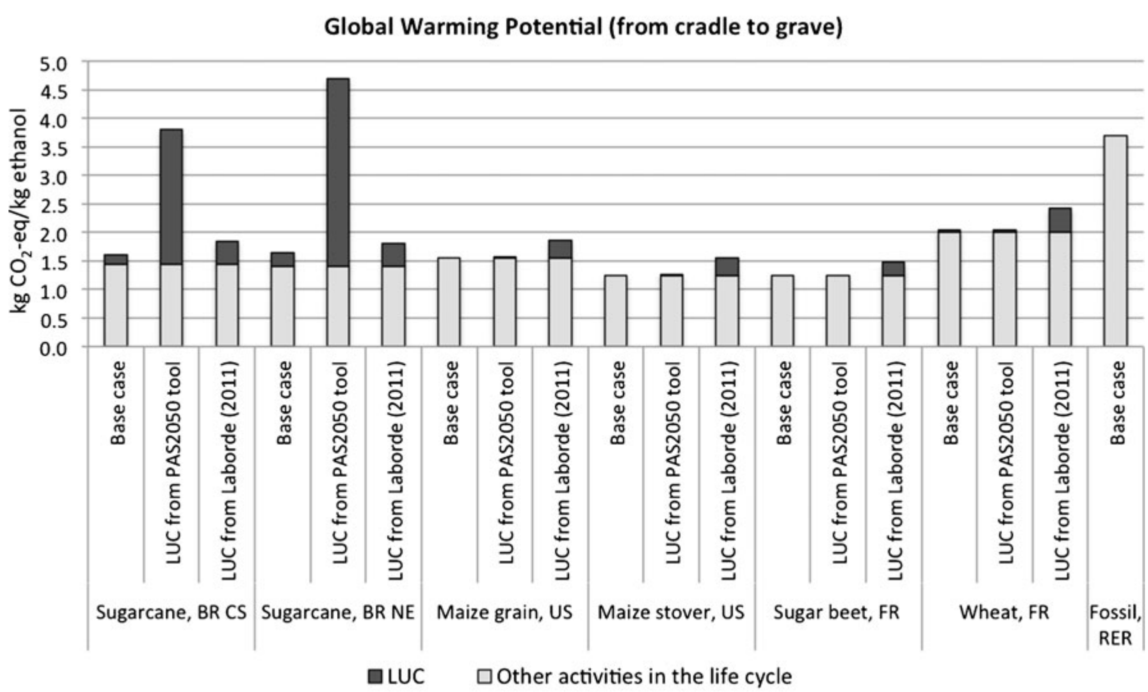

Fig. 4 Results of the sensitivity analysis on GHG emissions from LUC

from open burning, whereas in TAP the reduction is associated to lower nitrogen oxides and sulphur dioxides. Even though increased mechanisation increases fuel use, there is a net reduction in impacts with the exception of FAP, which increases by 11 % mainly due to phosphorus emissions in the background processes of the operation of a tractor. Overall, though, the ranking of bio-based feedstocks does not change based on the results of this sensitivity analysis.

3.4.3 Maize grain drying

Maize grain loses moisture naturally, but in wet weather conditions, additional drying can be necessary. As a base case, we assumed no drying, and in the sensitivity analysis, we added the drying step to check its environmental relevance. This was done by means of the ecoinvent dataset for maize drying (Nemecek et al. 2007). The results (Fig. 3 in ESM) showed that GHG emissions increase by 17 % when this step is added and makes maize grain the most GHG-intensive bio-based feedstock after wheat. In the remaining impact categories, the increase is of less magnitude and do not change the ranking of bio-based feedstocks.

3.4.4 GHG emissions from LUC

We carried out two alternative calculations of GHG emissions from LUC: (1) using the tool developed to support the application of the British standard PAS 2050–1:2012 (BSI 2012) and (2) using data from a study commissioned by the European Commission (Laborde 2011).

Both the PAS2050 tool and the method by Milà i Canals et al. (2012) are based on the use of retrospective FAOSTAT data to ascertain whether or not and how much LUC occurring in a country can be attributed to a crop. According to the PAS 2050 tool using the 'Calculation of averages' option, all crops but sugar beet in France were attributed LUC emissions, namely 11.5 tonnes of CO_2 eq/ha/year for sugarcane in Brazil, 0.01 tonnes of CO_2 eq/ha/year for maize in the USA and 0.07 tonnes of CO_2 eq/ha/year for wheat in France. In comparison, the method from Milà i Canals et al. (2012) led to GHG emissions of 0.75 tonnes CO_2 eq/ha/year for sugarcane in Brazil and of 0.1 tonnes of CO_2 eq/ha/year for wheat in France, whereas for maize no LUC was identified. We have not been able to identify all the possible reasons for these differences between the two methods, although a key factor in our opinion is that the amount of LUC is crop-specific in the PAS2050 tool, whereas in Milà i Canals et al. (2012) the amount of LUC is not crop-specific, but a country average. It can be seen in Fig. 4 that the choice of LUC estimation method has a dramatic effect in sugarcane-based ethanol from Brazil, to the extent that if the PAS 2050 method is used, its life cycle GHG emissions exceed those from fossil-based ethanol.

In a second sensitivity analysis, we replaced the GHG emissions estimated with our base-case method with those by Laborde (2011), who estimated global GHG emission factors from LUC for ethanol from maize, sugarcane, sugar beet and wheat based on general equilibrium modelling. Due to lack of detailed data in that study, it was not possible to distinguish between maize grain and stover; therefore, the same factor was used for both. The resulting GHG emissions (Fig. 4) are higher for all feedstocks, from 9 to 24 % when compared to the original results. In the latter, LUC emissions for Brazilian sugarcane were of 0.16 and 0.24 kg CO_2 eq/kg ethanol for the Centre-South and North-East regions, respectively, whereas the value from Laborde (2011) is higher, of 0.4 kg CO_2 eq/kg ethanol.[1] For wheat, the difference is even higher, since the original emissions from the method by Milà i Canals et al. (2012) were of 0.04 kg CO_2 eq/kg ethanol whereas the value from Laborde (2011) is of 0.42 kg CO_2 eq/kg ethanol.

3.4.5 Choice of biomes

The choice of biome for agricultural land used in the foreground system was not always straightforward. A sensitivity analysis was carried out for sugarcane in North-East Brazil, assuming 'Tropical and Subtropical Moist Broadleaf Forests' instead of 'Deserts and Xeric shrublands'. This resulted in impacts substantially rising for some indicators such as 76, 35 and 78 % in CRP, BPP and ERP, respectively. In CRP, BPP and ERP, it involved the highest impact of all alternatives, and in FWRP, impacts increased to similar levels to those from maize grain, maize stover and sugar beet. On the other hand, other impacts reduced with WPP-MF, WPP-PCF and BDP lowering to similar levels as wheat.

3.5 Uncertainty analysis

Figure 5 in the ESM shows the results of the uncertainty analysis for the ReCiPe impact indicators. The dots in the figure represent the probabilistic mean, whereas the vertical lines cover the 2.5th and 97.5th percentiles.

When GHG emissions are compared in Fig. 5 in the ESM, it can be seen that all bio-based ethanol alternatives are likely to involve lower emissions than the fossil alternative, but it is difficult to distinguish a clear difference between the different bio-based alternatives, due to the overlap in distributions.

As for fossil ethanol, it shows comparatively lower uncertainty than the bio-based feedstocks. The generally higher uncertainty for bio-based materials is consistent with the notion in LCA that agricultural systems usually involve a high variability and uncertainty (see e.g. Roches et al. 2010; Röös et al. 2010; Nemecek et al. 2012).

[1] The GHG factors in the original source are given per MJ ethanol. They are transformed on a per kilogram basis using a calorific low value of 29.96 MJ/kg ethanol (Kosaric et al. 2001).

Overall, like in the deterministic modelling, the fossil ethanol shows clear advantages in MEP, TAP and ALO, whereas the best-performing bio-based alternatives in these indicators are sugarcane and sugar beet (MEP) and sugar beet (TAP, ALO).

4 Conclusions

LCA was applied to assess the global bio-based commodity ethanol, from several biomass sources and regions in the world. Bio-based ethanol was also compared to its fossil-based counterpart, produced from ethylene.

The results have shown that GHG emissions from cradle-to-gate for the bio-based ethanol production routes assessed vary by a factor of up to 5, with the sugar beet route in France showing the lowest emissions and wheat showing the highest, unless GHG emissions from LUC are assessed with the PAS 2050 tool for horticultural products, in which case sugarcane-based ethanol from Brazil shows the highest GHG emissions. Looking at cradle-to-grave emissions, all bio-based ethanol production routes involve lower emissions than fossil-based ethanol, with the exception of sugarcane-based ethanol, provided that GHG emissions from LUC are assessed with the PAS 2050 tool for horticultural products.

When other impact indicators are considered, trade-offs appear between bio-based and fossil-based ethanol. The latter performs better in all BES impact indicators as well as in agricultural land occupation, marine eutrophication and terrestrial acidification. A parallel water footprint study also clearly showed a much lower water demand and impacts on water scarcity from fossil-based ethanol (Flury et al. 2012). In terms of the novel BES impact categories considered in this study, they allowed to clearly identifying fossil-based ethanol as the preferred alternative, and this is linked to its lower land occupation. On the other hand, when comparing bio-based ethanol production routes, the choice of biomes was not straightforward. This is an area that requires further guidance for practitioners.

This latter conclusion also points to the general issue of trading climate change for land use impacts when transitioning to a bio-based economy. This transition may be unavoidable in the face of climate change and fossil resource depletion, but adequate consideration of impacts on BES will be required in order to minimise detrimental trade-offs.

The biomass feedstock showing the best environmental performance in most indicators assessed was sugar beet, and this is clearly linked to the high yields of this crop. With regard to ethanol from sugarcane grown in the two assessed regions of Brazil, differences were found to be small in all impact indicators. However, water consumption was found to be three times higher in the North-East region when compared to the Centre-South region by Flury et al. (2012).

In general, we can conclude that environmental impacts from bio-based ethanol are highly dependent on the characteristics of the production chain considered, as well as on modelling and scenario choices, such as the choice of allocation factors, or the need to dry the product after harvest. However, key factors found in this study to describe the environmental impacts of bio-based products are the net product yield per hectare and year (combination of agricultural yield plus yield in the processing stage) and especially modelling of emissions caused by LUC. Given the increasing importance of bio-based products in the global economy, we call the LCA community to initiate a debate to harmonise the modelling of LUC emissions.

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