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Research paper

Environmental sustainability assessment of bioeconomy value chains

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ABSTRACT

Bioeconomy has gained political momentum since 2012 when the European Commission adopted the strategy "Innovating for Sustainable Growth: A Bioeconomy for Europe". Assessing the environmental performance of different bioeconomy value chains (divided in three pillars: food and feed, bio-based products and bioenergy) is key to facilitate solid and evidence-based policy making. The objectives of this work were: (1) to map and analyse accessible LCA data related to bioeconomy value chains in order to identify knowledge gaps; (2) provide a more robust and complete picture of the environmental performance of three bioeconomy value chains (i.e. one per each bioeconomy pillar). This analysis reveals that apart from few products (such as liquid biofuels, some biopolymers and food crops) the environmental assessment of bioeconomy value chains is still incipient and limited to few indicators (e.g. Global Warming Potential and energy efficiency). In this study, a harmonised procedure – the Product Environmental Footprint (PEF), which includes fourteen impact categories – is used to estimate the environmental performance of three exemplary case studies which are inter-related due to the use of sugar as feedstock: sugar (food and feed), bio-based ethanol (bioenergy) and polyhydroxyalkanoates (bio-based product). Results highlight the strong need for methodological harmonisation and coherence for LCA of bioeconomy value chains.

1. Introduction

The bioeconomy concept refers to the sustainable exploitation of renewable biological resources for the production of food and feed, bio-based products and bioenergy [1] – the "three pillars" of the bioeconomy. It includes several industries and sectors: agriculture, forestry, fisheries, food, pulp and paper production and part of the chemical, biotechnological and energy industries. The European bioeconomy has gained political momentum and strategic importance. In 2012, the EU bioeconomy had a turnover of nearly 2 € trillion, employed more than 22 million people (i.e. 9% of total employment in the EU) and presented a strong innovation potential [2]. In the same year, the European Commission reaffirmed its commitment to the bioeconomy through the communication: "Innovating for Sustainable Growth: A Bioeconomy for Europe" [3], highlighting the unique opportunity to accomplish economic growth while guarantying resource security and efficiency through smart and sustainable use of renewable biological resources. So far, 16 countries in the EU have adopted action plans and measures in support of the bioeconomy.

This communication includes both a strategy and an action plan. Three of the main challenges addressed in the strategy – the management of natural resources sustainably, the reduction of the dependence on non-renewable resources and the mitigation and adaptation to climate change – are directly related to a progressive switch from the current fossil fuel-based economy to a more bio-based one. Towards this change, it is essential that an increasing share of biomass is made available to meet European demand for production of food and feed, bio-based products and bioenergy.

Such an increasing mobilisation and use of biomass has economic, social and environmental implications. This paper focused on the environmental implications. Assessing the environmental performance of different bioeconomy value chains is important to facilitate evidence-based policy making. The broadly accepted and extensively used Life Cycle Assessment (LCA) methodology was selected to quantify impacts along bioeconomy value chains. It includes all processes from the extraction of resources to the end-of-life – "from cradle to grave" [4] within the boundaries of the

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study. A remarkable number of LCA-based studies have been conducted to evaluate the environmental profile of different bioeconomy value chains. However, these are often based on different methodological assumptions and use data of different nature and quality. This makes the results from these evaluations virtually incomparable. A clear quantitative understanding of the environmental aspects of bioeconomy value chains is thus currently missing.

The objectives of this work were: (1) to map and analyse accessible LCA data relative to bioeconomy value chains in order to identify knowledge gaps; (2) to provide a more robust and complete picture of the environmental performance of three exemplary bioeconomy value chains (i.e. one per each bioeconomy pillar).

This paper is organised as follows: Section 2 describes the methodology and the criteria used for mapping existing LCA-based studies and perform new ones. Section 3 provides the results of the mapping and the LCA modelling for three exemplary case studies which are inter-related due to the use of sugar as feedstock: sugar production (for the food and feed pillar); bioalcohols production via fermentation (for the bioenergy pillar); and polyhydroxyalkanoates (PHAs) production (for the bio-based product pillar). The conclusions of the work are drawn in Section 4.

2. Methodological approach

2.1. LCA mapping exercise

This paper builds upon work conducted within the FP7 project "Set-up of a Bioeconomy Observatory – Bioeconomy Information System and Observatory (BISO)", [5]. In particular, a list of key bioeconomy value chains (see Table 1) for each pillar was selected and analysed to identify existing and prospective technologies for biomass conversion and measure its environmental performance. The criteria for selecting the value chains are: importance in the global market, representativeness and/or relevance for possible competition with similar fossil-based products.

LCA is a widely accepted decision support method to assess environmental impacts along all stages of the life-cycle of a given product system. Different impact assessment methods can be used when conducting LCA studies. Each of these methods has a specific set of impact categories and characterisation factors. The most used impact assessment methods include: ReCiPe, CML2001, Eco-indicator 99, IMPACT 2002+ and TRACI [6,7]. In addition to LCA, several other life cycle-based environmental accounting methods and standards exist. The European Commission recommends the use of the LCA-based Product Environmental Footprint (PEF) [8] method to evaluate the environmental performance of product-system supply chains [9]. A comparison of the robustness of the PEF against other most used methods and standards for environmental accounting can be found in Ref. [10]. The PEF method was used as the reference for our LCA data mapping. It includes fourteen impact categories in order to provide comprehensive evaluation of the environmental performance of value chains (see Table 2).

The literature review conducted in this study revealed that it is a common practice to limit the number of impact categories considered to facilitate the overall assessment and interpretation of the results, as well as to limit data collection efforts. However, such an approach can lead to inaccurate and misleading conclusions [11]. Thus, all fourteen PEF-recommended impact categories were considered in this assessment of three exemplary bio-based value chains. The identification of knowledge gaps in the reviewed literature was done through a mapping of accessible LCA studies that provided an evaluation of the environmental performance of the selected bioeconomy value chains. The selection of these LCA studies was performed using the following criteria:

- studies conducted under the EU framework programmes for research [12];
- peer-reviewed literature;
- priority was given to studies accounting for the highest number of impact categories and studies reporting environmental impacts calculated in line with the PEF methodology;
- studies with obsolete, incomparable (i.e. percentages or weighted figures) or dubious quality data were excluded.

The LCA data mapping was performed by identifying the minimum and maximum reported values for each impact category (see Section 3). The purpose of this study was not to discuss the correctness of the methodological assumptions and choices done in the reported studies. However, a discussion on the effect of some key LCA assumptions on the final result is provided in Section 3 and some recommendations are given in Section 4.

2.2. LCA of exemplary bioeconomy value chains

The objective of performing a LCA can be either (1) measure the consequences of altering a system, or (2) analyze the environmental impacts along a product's life cycle. These two goals are frequently tackled by consequential LCA and attributional LCA, respectively [13]. The comparison of data and results obtained under such different methodological assumptions is challenging and sometimes even impossible. For that reason, the second objective of this paper was to develop comprehensive LCAs of selected bioeconomy value chains (one for each pillar) using the

Table 1
Selected bioeconomy value chains within the BISO FP7 project.

Food & feed	Bioenergy		Bio-based products
Product	Product	Via	Product
Eggs	Biodiesel	Transesterification	Lactic acid
Milk	Bio-based alcohols	Fermentation	Acetic acid
Sugar	Small-scale heat	Direct combustion	Adipic acid
Tomato	Large-scale heat	Direct combustion	Succinic acid
Wheat	Electricity	Direct combustion	1,3-Propanediol
Wine	CHP	Direct combustion	Glycerol
	Biofuels	Gasification	Polylactic acid (PLA)
	Hydrogen	Gasification	Polyhydroxyalkanoates (PHAs)
	CHP	Gasification	Amino acids
	Biodiesel	Hydrogenation	Paper
	CHP/Fuel	Torrefaction	
	CHP	Anaerobic digestion	
	CHP/H_2	Pyrolysis	

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Table 2
Impact Categories in the PEF methodology, acronyms and normalization factors.

Impact category	Impact assessment model	Impact category units	Acronym	Normalisation factor
Climate change	Bern model	kg CO_2 eq.	CC	4.6×10^{12}
Ozone depletion	EDIP model	kg CFC-11 eq.	OD	1.08×10^{7}
Ecotoxicity for aquatic fresh water	USEtox model	CTUe[a]	Fecotox	4.36×10^{12}
Human toxicity – cancer effects	USEtox model	CTUh[b]	HH,ce	1.84×10^{4}
Human toxicity – non-cancer effects	USEtox model	CTUh[b]	HH,nce	2.66×10^{5}
Particulate matter/respiratory inorganics	RiskPoll model	kg $PM_{2.5}$ eq.	PM	1.9×10^{9}
Ionising radiation – human health effects	Human health effect model	kg U^{235} eq. (to air)	IR,hh	5.64×10^{11}
Photochemical ozone formation	LOTOS-EUROS model	kg NMVOC eq.	POF	1.58×10^{10}
Acidification	Accumulated exceedance model	mol H+ eq.	A	2.36×10^{10}
Eutrophication – terrestrial	Accumulated exceedance model	mol N eq.	TE	8.76×10^{10}
Eutrophication – aquatic – freshwater	EUTREND model	kg P eq.	FE	7.41×10^{8}
Eutrophication – aquatic – marine	EUTREND model	kg N eq.	ME	8.44×10^{9}
Resource depletion – water	Swiss ecoscarcity model	m^3 water used	WRD	4.06×10^{10}
Resource depletion – mineral, fossil	CML2002 model	kg antimony (Sb) eq.	MFRRD	5.03×10^{7}
Land transformation	Soil organic matter model	kg C (deficit)	LT	3.74×10^{13}

[a] Comparative Toxic Unit for ecosystems.
[b] Comparative Toxic Unit for humans.

same methodological assumptions along the construction of the LCA models that characterised each product (as shown in Section 3).

The LCAs conducted in this study are based on the guidelines provided by the "methodology for environmental sustainability assessment" developed under the framework of the BISO project [14]. This methodology is largely based upon the LCA guidelines provided by the EC PEF method and the International Reference Life Cycle Data System (ILCD) Handbook [15] in order to ensure consistent, robust and quality-assured life cycle results of bio-based products and their supply chains.

The assessment of each bio-based product was conducted using the software package SimaPro v.8 [16]. Ecoinvent v.3 database [17] was mostly used for developing the life cycle inventories and data gaps were filled with other bibliographic data.

To make results cross-comparable, a normalisation was also performed. The normalisation factors that were used express the total impact occurring in the EU for each impact category in 2010 and relates to the domestic inventory reported by the JRC [18] (see Table 2). Domestic emissions correspond to all emissions originated from activities taking place within the European territory.

The mapping and LCA modelling results are included in Section 3 for each of the selected example bioeconomy value chains: sugar production from the food and feed pillar; bioalcohols production via fermentation from the bioenergy pillar; and polyhydroxyalkanoates (PHAs) production from the bio-based product pillar.

3. Case studies and results

3.1. Food & feed pillar: sugar

Sugar for use in food corresponds mainly to crystalised white sugar. It is extracted from the stem of sugar cane or the root of sugar beet through a refining process. The European Union is the world's biggest producer of beet sugar and the largest importer of raw cane sugar for refining (although a small amount is cultivated in overseas European territories). In 2013, the EU-28 production of sugar (from sugar beet) reached approximately 17.5 million tonnes [19].

Fig. 1 shows the processes involved in the cultivation of sugar crops and their processing and refining into crystalised sugar. Since 2006, the EU sugar market is regulated by production quotas, a minimum beet price and trade mechanisms. However, out-of-quota industrial white sugar does not have a fixed buy price. The total EU production quota is 13.5 million tonnes of sugar (2013) [19]. Sugar imports (3.3 million tonnes) are mainly in the form of raw cane sugar for refining (64%) and white sugar (26%), from the African, Caribbean and Pacific (ACP) states and Least Developed Countries (LDC) which benefit from quota-free, duty-free access to the EU market. Imports from other countries are subject to high import duties (339 € per tonne on raw cane sugar for refining and 419 € per tonne on white sugar). Apart from food applications, sugar and sugar molasses are also used in the production of bio-based products (see section 3.3) such as biopolymers, organic acids and amino acids, and biofuels (see section 3.2), such as ethanol, through fermentation processes.

3.1.1. Review of LCA data on sugar production

Multiple studies investigating the environmental impact of sugar production can be found in literature. Roy et al. [20] reviewed LCA studies on some food products including sugar, analysing the causes for the high variability in results. In the present LCA mapping exercise, six separate studies were considered [21–26], investigating twenty nine sugar production systems.

Newly developed varieties of both sugar cane (herbicide and draught tolerance) and sugar beet (herbicide resistance and adaptation to dryer tropical areas) could result in significantly lower LCA environmental impact of sugar production. These were however not considered in this study due to lack of published data.

A strong variability in the number of impact categories that were taken into account is observed in the summary results of the environmental mapping (see Fig. 2). Large variations can also be observed in the type of units considered and in the impact values (see Table 3). These variations reflect different system boundaries and allocation methods, used in the studies. The variations are particularly significant for CC, A and MFRRD, where values range from positive (i.e. net environmental loads) to negative (i.e. net environmental benefits). Negative values for CC and MFRRD can be explained by the utilisation of by-products (bagasse from cane sugar) to produce energy while the negative values for A are associated with local phosphate fertilisers production practices specific to a case study in Queensland, Australia [25].

None of the studies investigated the totality of the fourteen impact categories. Seven impact categories (OD, HH,nce, PM, IR,hh, TE, ME and LT) were consistently ignored and four more categories (Fecotox, HH,ce, POF and WRD) appeared in only one publication. Fig. 2 indicates that CC is the most frequently reported impact category, followed by A, FE and MFRRD. While studies that investigated CC and FE, consistently used accounting units in line with PEF, other categories such as A and MFRRD were constantly reported in other units.

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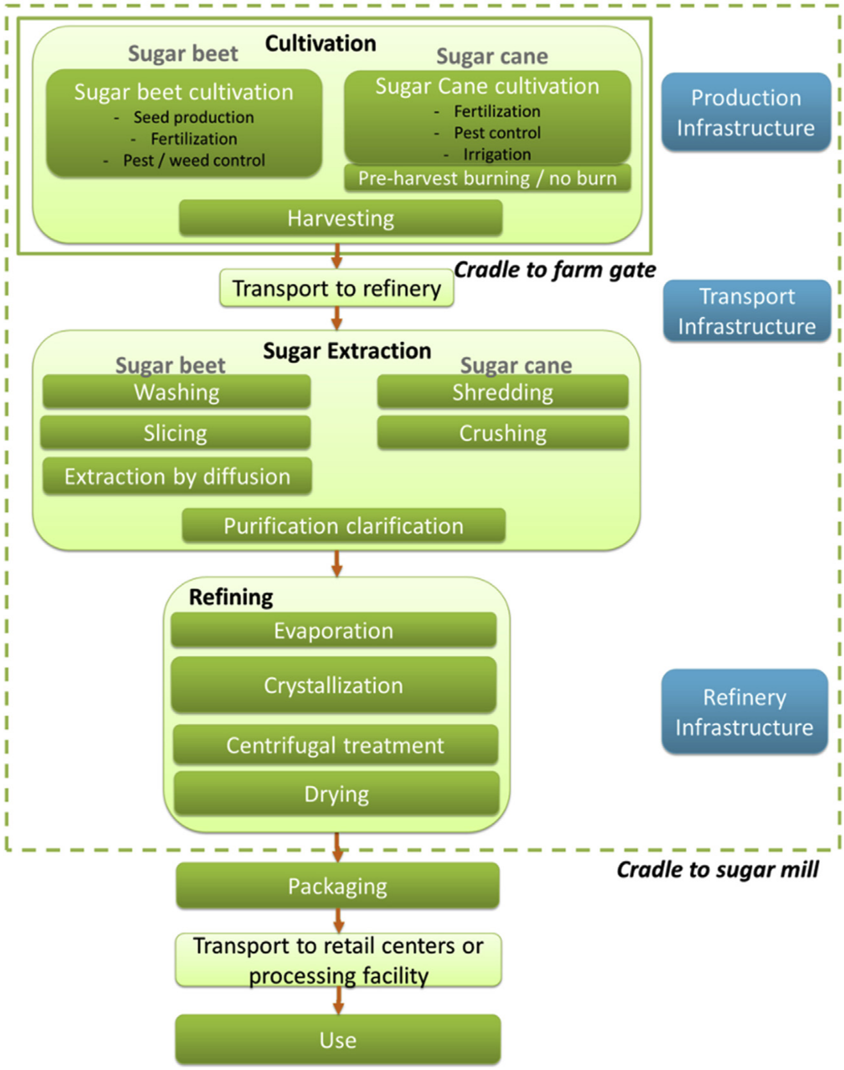

Fig. 1. Processes and system boundaries in sugar production.

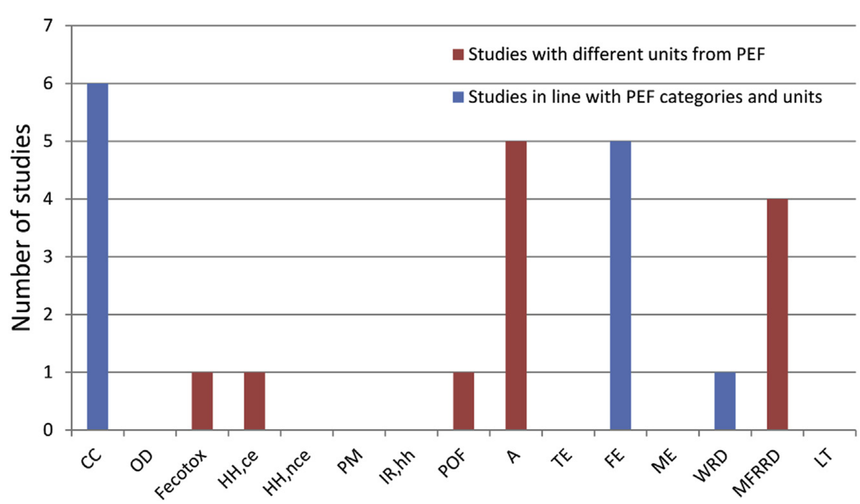

Fig. 2. Sugar mapping results.

Table 3
Mapping of results from LCA studies in sugar production (1 kg of extractable sugar).

IMPACT CATEGORY (Units in line with PEF)	Impact values (Per FU)	Studies with different units from PEF	Impact values (Per FU)
CC (kg CO_2 eq.)	-5×10^{-2}–1.3	–	–
OD (kg CFC-11 eq.)	–	–	–
Fecotox (CTUe)	–	kg 1.4 DB eq	1.0×10^{-2}
HH, ce (CTUh)	–	kg 1.4 DB eq	2.2×10^{-3}
HH, nce (CTUh)	–	–	–
PM (kg $PM_{2.5}$ eq.)	–	–	–
IR, hh (kg U^{235} eq.)	–	–	–
POF (kg NMVOC eq.)	–	kg C_2H_4	2.4×10^{-5}
A (mol H+ eq.)	–	kg SO_2 eq	-4.7×10^{-3}–1.3×10^{-2}
TE (mol N eq.)	–	–	–
FE (kg P eq.)	1.4×10^{-4}–4.2×10^{-3}	–	–
ME (kg N eq.)	–	–	-
WRD (m^3 used)	7.4×10^{-3}–9.0×10^{-1}	–	–
MFRRD (kg Sb eq.)		MJ	−10.1–6.3
LT (kg deficit)	–	–	–

3.1.2. Sugar production modelling exercise

The environmental impacts associated with the production of one kg of sugar, from cradle to sugar mill gate (Fig. 1), using two different sugar crops were estimated: (1) sugar beet grown in Germany (using the agri-footprint database – beet sugar production DE), and (2) sugar cane from Brazil (using the Ecoinvent database cane sugar production with ethanol as by-product). For both scenarios an economic allocation of impacts was used (i.e. environmental impacts were divided between the co-products according to their market price). For the sugar beet scenario, 83% to sugar, 1.8% to molasses, 0% to pulps, and 10% for lime fertilizer production; for the sugar cane scenario, 80–85% to sugar and 10–11% to ethanol. Results are shown in Table 4.

The environmental impacts estimated for the two scenarios showed that the highest values were mostly associated with the sugar production from sugar cane. With respect to the categories FE, OD and IR,hh, sugar from sugar beet provided significantly lower values (between 26 times lower for FE to 7 times for OD and IR,hh). The environmental impacts associated with the transport of sugar from non-EU countries to the EU were, however, not included in the comparison. The estimated impacts for sugar cane were therefore underestimated. On the other hand, the sugar production from sugar beet resulted in higher environmental impacts for ME, TE and A (about 4 times more), CC (2 times) and HH,nce (about 1.5 times).

The normalised results in Fig. 3 show that amongst all the impact categories studied for sugar production, HH (non-cancer effects) and WRD present the highest relative contribution to the overall European environmental impact.

Table 4
Environmental impact results for sugar production from sugar beet and from sugar cane (Functional Unit (FU) = 1 kg of extractable sugar produced).

Impact category	Unit	Total	
		Sugar beet	Sugar cane
CC	kg CO_2 eq	0.76	0.36
OD	kg CFC-11 eq	2.37×10^{-9}	1.78×10^{-8}
Fecotox	CTUe	2.81	4.6
HH,ce	CTUh	1.87×10^{-8}	3.77×10^{-8}
HH,nce	CTUh	2.46×10^{-6}	1.68×10^{-6}
PM	kg $PM_{2.5}$ eq	7.29×10^{-4}	2.25×10^{-3}
IR,hh	kBq U^{235} eq	2.90×10^{-3}	2.09×10^{-2}
POF	kg NMVOC eq	1.32×10^{-3}	2.73×10^{-3}
A	molc H+ eq	3.18×10^{-2}	8.14×10^{-3}
TE	molc N eq	0.14	3.30×10^{-2}
FE	kg P eq	4.89×10^{-6}	1.29×10^{-4}
ME	kg N eq	1.59×10^{-2}	3.69×10^{-3}
WRD	m^3 water eq	2.66×10^{-2}	0.16
MFRRD	kg Sb eq	5.51×10^{-6}	3.09×10^{-5}
LT	kg C deficit	9.04	9.94

These modelled impacts were then related to those identified in the mapping exercise, by making use of reported values only in line with the PEF categories and units. This comparison shows that for CC, the modelled values fall within the ranges identified in the literature review. This fact indicates a fairly good level of consistency across studies for this most commonly investigated impact. However, values for FE for both sugar cane and sugar beet where below the minimum reported in the literature (with sugar beet values being 28 time inferior to the minimum) and WRD values for sugar beet were also below those in the literature (about 3 times lower), which observation could indicate potential inconsistencies in less reported indicators.

Indeed, large discrepancies were identified in the environmental impacts for sugar produced from sugar beet across the Ecoinvent database (data specific to Switzerland) and the agri-footprint (data specific to Germany) life cycle inventory databases. The categories for WRD and HH,nce were of particular concern. Data from Ecoinvent reported very high values of WRD (26 times higher than in agri-footprint). Those were not associated with the sugar beet production phase, but rather with the production of electricity in Switzerland. Unlike agri-footprint, Ecoinvent also reported negative HH,nce impacts, assuming that some heavy metals were exported from the soil into the sugar beets [27]. These negative values would likely become positive if the study boundary of the systems were extended to the consumer. All these discrepancies led to the selection of the agri-footprint database as the source of data for sugar production from sugar beet.

3.2. Bioenergy pillar: bioalcohols via fermentation

Bioalcohols, as petrol substitutes, can be produced by fermentation (see Fig. 4) from a wide range of biomass feedstock: energy crops (e.g. sugar cane, corn, sugar beet, wheat), lignocellulosic material, wastes/residues, microalgae and even by-products of other processes such as glycerol [28] (by-product from the transesterification process to produce biodiesel). Bio-based ethanol is the most common biofuel in the world: its global production has been estimated at 67 Mt in 2011.

3.2.1. Review of LCA data on bio-based ethanol

Many review studies have been performed in recent years concerning the life cycle impacts of bio-based ethanol. The review of von Blottnitz et al. [29] gives a qualitative overview of the many

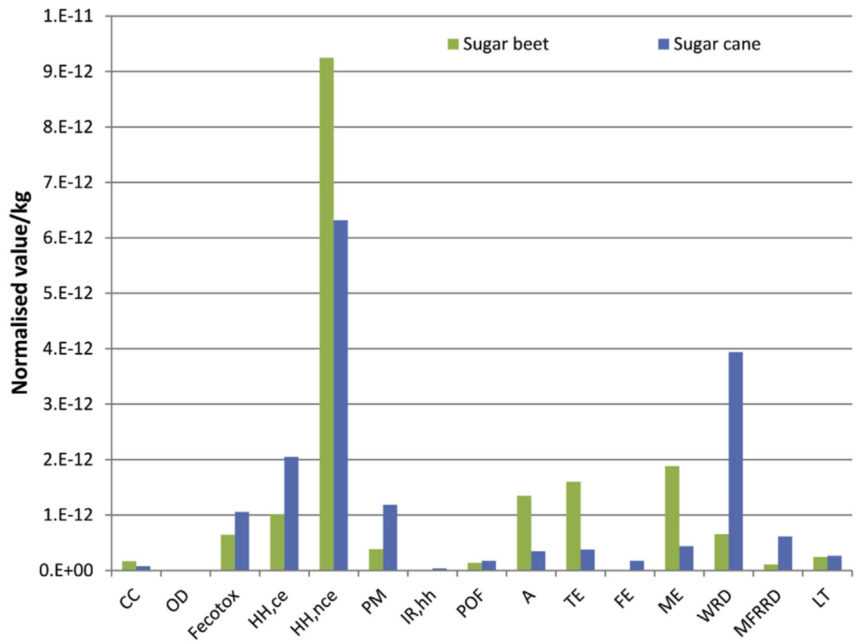

Fig. 3. Normalized environmental impacts of sugar production.

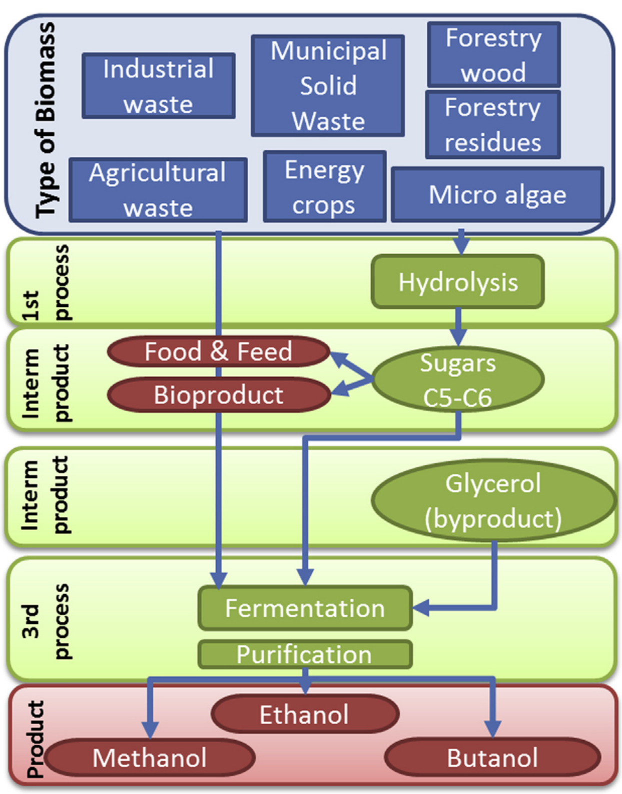

Fig. 4. Flowchart of the bio-based alcohols production process.

studies that report data on the biomass-to-fuel ethanol pathway categorised by feedstock and location. Cherubini and Stromman [11], in a more recent review, reported a large portion of the existing scientific literature on the different biomass energy uses. This review classified qualitatively the studies by the type of feedstock, the methodology (specifying the functional unit, the reference system, the allocation method and whether Land Use Change is considered), the location and the criteria included in the scope of the study (i.e. impact categories). A specific review of second generation bio-based ethanol done by Wilose et al., [30] classifies the publications by feedstock, components of the value chain, methodology and the impact categories studied. All these reviews stress the challenge and usefulness of comparing results from different LCA studies, the need for more coherent methodological standards for sustainability assessments and for evaluating additional issues beyond energy efficiency and greenhouse gas (GHG) emissions.

More than forty studies were considered in the mapping performed in the current study. Twenty three of them (reporting on more than hundred cases) are included in the results shown in Table 5 [28,31–52].

The mapping (Fig. 5) confirms that CC is the most frequently reported impact category (all of the twenty three studies). The reason for this coverage is that the European regulations on emissions from bio-fuels apply to GHGs only [53]. Among other conditions [54], in order to define a biofuel as “sustainable” and thus receive government support (and count towards national renewable energy targets), it should demonstrate at least 35% lower GHG emissions throughout the life cycle compared to fossil fuels.

The analysis also shows that additional impact categories are gradually being included besides CC but still extra efforts are needed to reach sufficient degree of inclusiveness. Only one study reported up to ten impact categories [46].

The variability found in the mapping results (Table 5) is particularly significant for some impact categories, such as CC and TE, for which the estimated impact scores can vary from positive values (i.e. net environmental loads) to negative values (i.e. net environmental benefits). This is mainly due to: 1) the way CO_2 sequestration during the biomass growth is accounted for calculating the CC impact score [51] and 2) the inclusion of winter cover crop practices when estimating TE [50]. The consistency and robustness of the data are not homogeneous across the whole set of impact categories, since some of them include high number of case studies (e.g. CC with more than hundred values), while others

Table 5
Mapping of results from LCA studies on bio-based ethanol value chain (ranges shown based on the functional unit (FU) = driving a flexible fuel vehicle for 1 km using bio-based ethanol).

Impact category (Units in line with PEF)	Impact values (Per FU)	Studies with different units from PEF	Impact values (Per FU)
CC (kg CO_2 eq.)	-1.23–1.59	–	–
OD (kg CFC-11 eq.)	1.2×10^{-8}–2.7×10^{-1}	–	–
Fecotox (CTUe)	–	kg 1,4 DB eq.	1.2×10^{-3}–18.4
HH, ce (CTUh)	–	kg benzo[a]pyrene	6.1×10^{-9}
HH, nce (CTUh)	–	kg 1,4 DB eq.	1.6×10^{-4}–0.41
PM (kg $PM_{2.5}$ eq.)	–	kg dust	6.9×10^{-4}
IR, hh (kg U^{235} eq.)	–	–	–
POF (kg NMVOC eq.)	2.1×10^{-4}–2.9×10^{-3}	kg C_2H_4 eq.	6×10^{-6}–1.1×10^{-3}
A (mol H+ eq.)		kg SO_2 eq.	6.3×10^{-5}–1.3×10^{-2}
TE (mol N eq.)	-2.4×10^{-4}–3.6×10^{-3}	–	–
FE (kg P eq.)	8.9×10^{-6}–1.3×10^{-3}	–	–
ME (kg N eq.)	1.4×10^{-4}–1.3×10^{-3}	–	–
WRD (m^3 used)	9.3×10^{-4}–4×10^{-3}	–	–
MFRRD (kg Sb eq.)	1.6×10^{-4}–0.29	coal eq. or oil eq.	–
LT (kg deficit)	–	m^2 year	6.3×10^{-4}–2.19

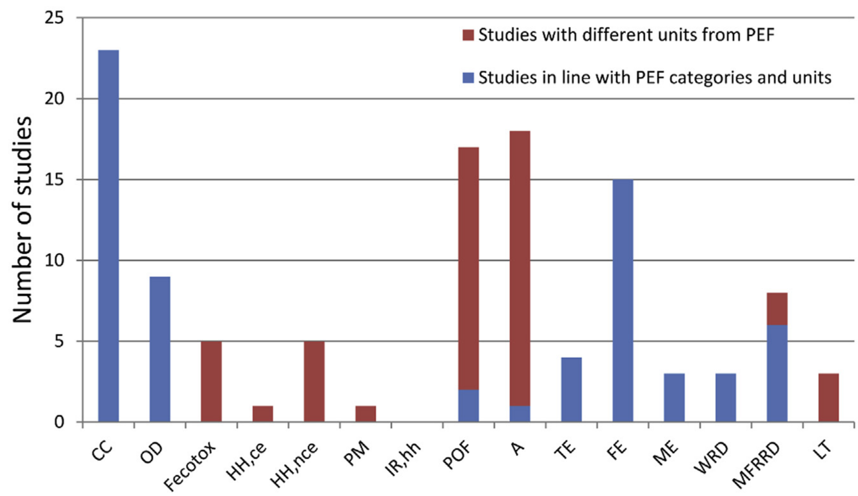

Fig. 5. Bio-based ethanol mapping results.

include just few of them (e.g. HH,ce and PM with one value).

3.2.2. Bio-based ethanol production modelling exercise

Concerning the system boundaries, for biofuels, Well to Tank – WTT (i.e. cradle-to-gate) or Well to Wheel – WTW (i.e. cradle-to-grave) boundaries can be used (Fig. 6). Depending on the feedstock, the WTW includes the cultivation-harvesting or collection of the feedstock, preprocessing, transport, hydrolysis/saccharification, fermentation, purification, transport to the end user and the use phase in a vehicle.

In this study WTW was performed, and the functional unit was defined as 1 km driven by a passenger car (flex-fuel vehicle – FFV). Data from the agricultural stage and the ethanol production process were obtained from Ecoinvent [55], for two different feedstocks (scenarios A and B). Scenario A is based on the “global market” process for ethanol production that considers feedstock from different locations (mainly sugarcane from Brazil and corn from USA) and its transport to Europe where the use phase is assumed to take place. Scenario B models bio-based ethanol using woodchips produced in Europe as feedstock. For both scenarios the economic allocation approach was used. For the ethanol use phase in the vehicle, the E85 blend (i.e. fuel blend consisting of 85% volume fraction ethanol from biomass and 15% volume fraction low-sulphur gasoline) was selected. Consumption and emission data

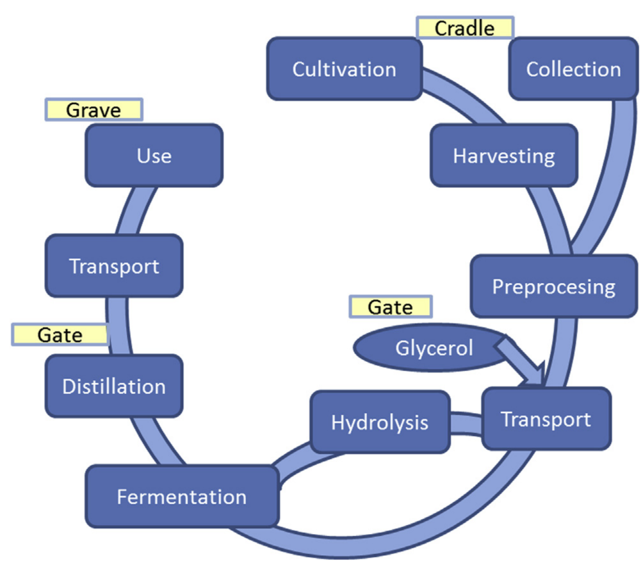

Fig. 6. System boundaries of the bio-based ethanol value chain.

were acquired from the emission tests reported by Dardiotis et al. [56] and Suarez-Bertoa et al. [57]. Fuel consumption of E85 is considered 0.109 L km^{-1} [56]. The aggregated results from both scenarios are shown in Table 6. These include the contribution of the E85 production (including petrol and ethanol production), and the use phase (that includes fuel combustion, wear emissions from the car/road and car maintenance). Infrastructure (that includes road and car construction) is not considered in this study.

Analysing the contribution of the different stages of the bio-based ethanol life cycle, the main contributor to all impact categories is the ethanol production (including sugar production, fermentation and ethanol separation).

The normalised values (see Fig. 7), representing the relative contribution of the system assessed to the total environmental impacts caused by European domestic emissions, show that the impact categories with the highest values are resource depletion (both WRD and MFRRD), Fecotox and HH,nce, mainly due to the ethanol production, but also with high contribution from the use phase in the case of Fecotox and MFRRD. HH,ce and PM impact categories present also high values and the ethanol production is again the main contributor.

Biofuels are generally reported in bibliography as good options to reduce CC compared to conventional fossil fuels. It is however important to carefully evaluate whether trade-offs may occur, along with the extent of their impacts. The conventional petrol fuel was therefore included as a benchmark in this study (total normalized emissions presented in Fig. 7). The LCA of bio-based ethanol report lower values (in both scenarios) for IR,hh, OD and CC, but higher values for the remaining impact categories compared to petrol. In the case of CC, the use phase CO_2 eq. emissions per km from petrol are around six times higher than the ones from E85—210 g km^{-1}, and 37 g km^{-1}, respectively. When considering the CO_2 eq. emissions along the whole LCA chain, the difference between bio-based ethanol and petrol is markedly reduced. The total CO_2 eq. emissions for the petrol are 270 g km^{-1} and those for E85 are 140 g km^{-1} for Scenario A and 120 g km^{-1} for Scenario B.

3.3. Bio-based products pillar: polyhydroxyalkanoates (PHAs)

Polyhydroxyalkanoates (PHAs) are biodegradable polymers that can replace petrochemical polymers currently used in several applications such as coatings and packaging. PHAs are produced by bacteria via fermentation of sugars, fatty acids and waste streams. The polymers are accumulated as bacterial intracellular granules during nutrient depletion phases or during an abrupt increase of carbon supply. They are normally produced in two fed-batch steps (a growth step and a polymer accumulation step). Fig. 8 shows several possible production pathways for PHAs that result from the conversion of different biomass feedstock. After fermentation, the produced PHA polymer must be extracted from the microbial intercellular organelles and purified. This extraction is typically performed by solvent extraction, but other technologies have been proposed to increase extraction efficiency such as enzymatic, mechanical and chemical cell disruption and supercritical extraction.

The actual full scale production capacity of PHA stands at 32,000 tonnes per year [58] and the efficiency of the production process is expected to increase in the years to come.

3.3.1. Review of LCA data on PHAs

The review analysis revealed the existence of twelve LCA [59–70] studies for PHAs that use a cradle-to-factory gate approach (see Fig. 9), three studies that consider end-of-life options (from cradle-to-grave) [71–73] and two review papers that summarise and compare LCA data for bio-based polymers such as polylactic acid (PLA) and PHAs [74,75].

The majority of the studies present results based on data from PHA production in laboratory or pilot facilities and most of them consider sucrose (from sugar cane) and glucose (from corn starch) as feedstock. Other feedstock materials are also used: glucose from corn stover (a less mature technology, but with potential for improvements), vegetable oils (soybean or rapeseed oils), wastewater, whey, biogas and genetically modified corn (in this case the PHA is produced within the corn cells).

Following the selection procedure described in Section 2, only eight studies present relevant data for the LCA mapping analyses. Few impact categories are reported in literature. Results in Fig. 10 show that the most common impact analysed is CC (reported in seven studies) followed by A (three studies). No data was found for the impact categories: HH,ce, PM, IR,hh and WRD.

Table 7 shows only results of the selected cradle-to-gate LCA studies where a high variability of results was found for CC. Lower CC impact values (i.e. below 0.49 kg CO_2eq kg $polymer^{-1}$) were reported by studies that account for carbon uptake during biomass growth, thereby considering carbon storage in the bio-based polymer. Low impacts (e.g. from −2.3–2.3 kg CO_2eq kg $polymer^{-1}$) were also reported when waste streams, such as bagasse or lignin-rich wastes, are burned for energy recovery. The highest CC impact values were reported for PHA produced from rapeseed oils

Table 6
Environmental impact results for bio-based ethanol production from different feedstock — Scenario A: global market process. Scenario B: woodchips produced in Europe (Functional Unit (FU) = driving a flexible fuel vehicle for 1 km using bio-based ethanol).

Impact category	Unit	Total		E85 production			Use phase
		Scenario A	Scenario B	Petrol	Ethanol Scenario A	Ethanol Scenario B	
CC	kg CO_2 eq	0.14	0.12	1.05×10^{-2}	9.28×10^{-2}	7.43×10^{-2}	3.71×10^{-2}
OD	kg CFC-11 eq	8.26×10^{-9}	8.12×10^{-9}	3.20×10^{-9}	4.46×10^{-9}	4.32×10^{-9}	6.02×10^{-10}
Fecotox	CTUe	1.19	0.97	1.95×10^{-2}	0.71	0.49	0.46
HH,ce	CTUh	5.81×10^{-9}	5.18×10^{-9}	2.11×10^{-10}	4.95×10^{-9}	4.33×10^{-9}	6.41×10^{-10}
HH,nce	CTUh	1.18×10^{-7}	9.00×10^{-8}	1.08×10^{-9}	1.08×10^{-7}	7.98×10^{-8}	9.10×10^{-9}
PM	kg $PM_{2.5}$ eq	2.44×10^{-4}	3.50×10^{-4}	7.46×10^{-6}	1.86×10^{-4}	2.92×10^{-4}	5.04×10^{-5}
IR hh	kBq U^{235} eq	1.20×10^{-2}	1.40×10^{-2}	2.98×10^{-3}	7.44×10^{-3}	9.44×10^{-3}	1.57×10^{-3}
POF	kg NMVOC eq	7.28×10^{-4}	9.90×10^{-4}	6.43×10^{-5}	4.94×10^{-4}	7.57×10^{-4}	1.69×10^{-4}
A	molc H+ eq	1.33×10^{-3}	1.02×10^{-3}	7.49×10^{-5}	1.15×10^{-3}	8.37×10^{-4}	1.07×10^{-4}
TE	molc N eq	4.37×10^{-3}	2.94×10^{-3}	1.14×10^{-4}	3.97×10^{-3}	2.54×10^{-3}	2.89×10^{-4}
FE	kg P eq	3.02×10^{-5}	2.18×10^{-5}	9.33×10^{-7}	2.62×10^{-3}	1.79×10^{-5}	3.05×10^{-6}
ME	kg N eq	5.32×10^{-4}	2.46×10^{-4}	1.05×10^{-5}	5.09×10^{-4}	2.24×10^{-4}	1.20×10^{-5}
WRD	m^3 water eq	6.76×10^{-2}	7.19×10^{-2}	3.87×10^{-3}	5.10×10^{-2}	5.53×10^{-2}	1.27×10^{-2}
MFRRD	kg Sb eq	1.01×10^{-5}	1.06×10^{-5}	2.62×10^{-7}	6.24×10^{-6}	6.72×10^{-6}	3.61×10^{-6}
LT	kg C deficit	1.20	4.47	2.96×10^{-2}	1.17	4.44	6.72×10^{-3}

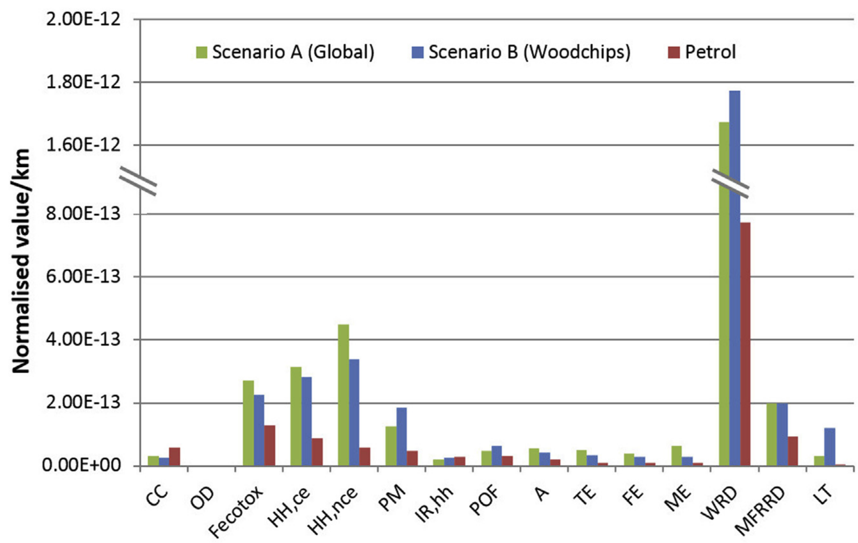

Fig. 7. Normalized environmental impacts of bio-based ethanol production.

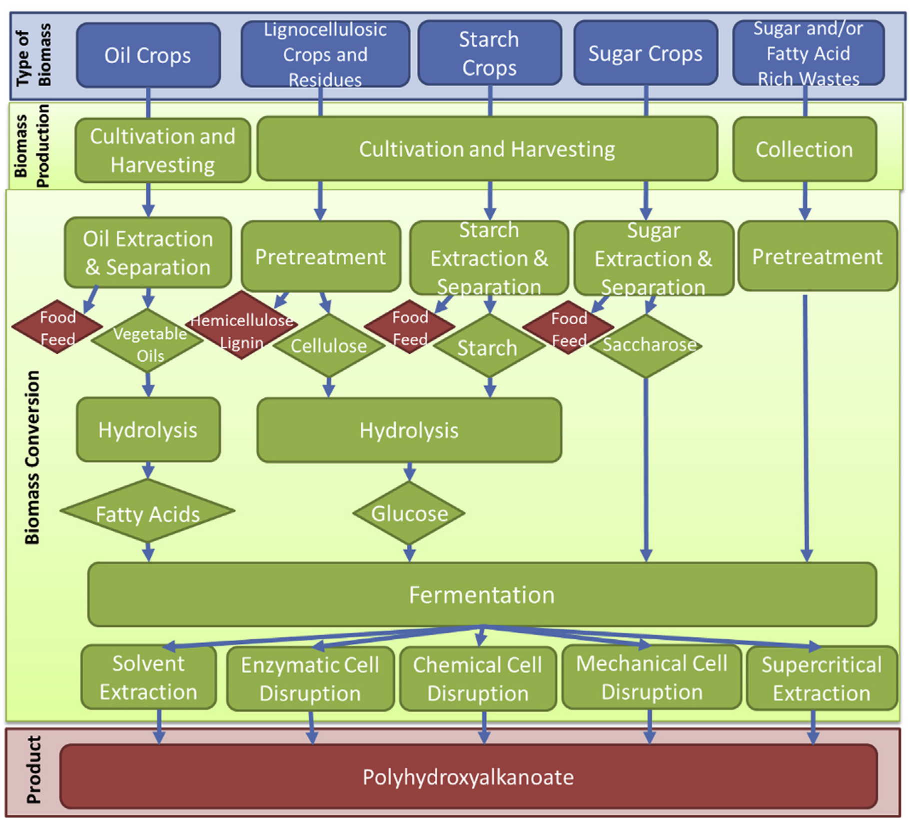

Fig. 8. Flowchart of the polyhydroxyalkanoates production process.

(from 5 to 6.9 kg CO_2eq kg polymer^{-1}) due to lower productivities compared to systems that use sugar feedstock.

The methodology used to allocate impacts amongst the different system outputs had also important impact upon the final LCA results. During the PHAs life cycle, other by-products are produced — corn meals (in the corn wet mill) or energy surplus. In the analysed PHAs studies it was typically considered that when an energy surplus was generated, credits were assigned to the system, to account for the avoided impacts of producing the same amount of energy in another external reference system. On the other hand, when a feed by-product was generated (for example, a corn mill) a mass-based or economic allocation was typically applied.

3.3.2. PHAs production modelling exercise

Two main scenarios were considered differing on the feedstock used: (1) sugar from corn starch (from Germany) and (2) sugar produced from sugar cane (from Brazil, already analysed in section 3.1). A cradle-to-gate (until the PHA factory gate) approach was

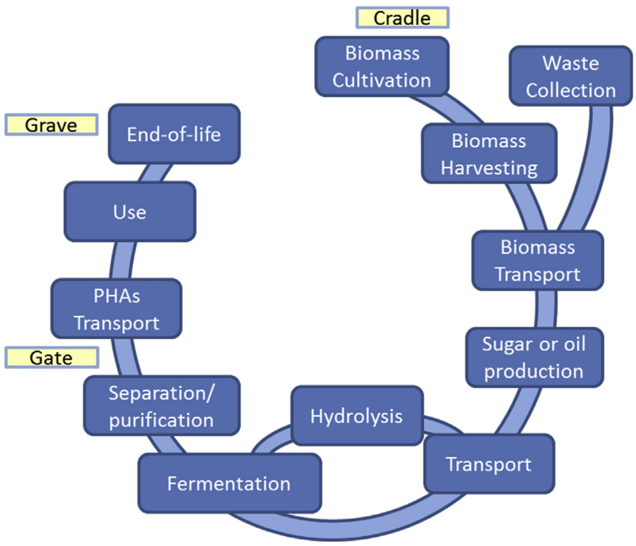

Fig. 9. System boundaries of the polyhydroxyalkanoates value chain.

and energy data for PHA production were collected from the BREW project [62] and Harding et al. [63]. All data on energy and resources extraction/production were collected from the Ecoinvent database. PHA production data refers to the use of a pure bacterial culture for converting refined sugars into PHA (data from a scale-up scenario to produce 1000 kg of PHB using *Cupriavidus necator* bacteria [63]). PHA production was considered to take place in Europe within 200 km from all the materials suppliers, except the production of sugar from sugar cane, which was considered to take place in Brazil. For the corn scenario a conversion of 0.95 g of sugar per g of starch was assumed (taking also into account 14% mass fraction of water in corn starch) [62]. For sugarcane and corn starch production Ecoinvent uses economic allocation to divide the impacts between the main product and by-products. Infrastructure (that includes reactors and other production facilities) was not considered in this study.

The LCA results presented in Table 8 show that for most categories the system based on corn starch results in higher impacts (e.g. LT and ME are 2.2 and 3.1 times higher than for sugar cane respectively), which is in accordance with the results reported in the literature. This observation can be explained by the lower

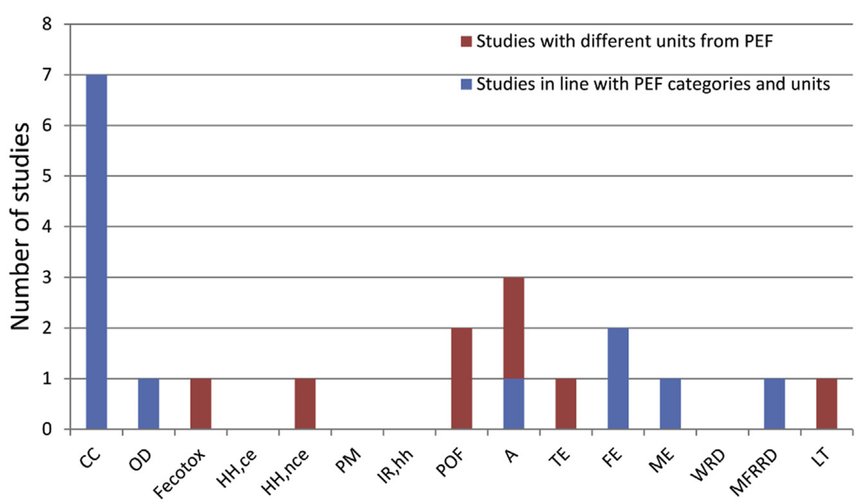

Fig. 10. Polyhydroxyalkanoates mapping results.

Table 7
Mapping of results from LCA studies in polyhydroxyalkanoates value chain (Functional Unit (FU) = 1 kg of polymer produced).

Impact category (Units in line with PEF)	Impact values (Per FU)	Studies with different units from PEF	Impact values (Per FU)
CC (kg CO_2 eq.)	−2.3–6.9	–	–
OD (kg CFC-11 eq.)	1.7×10^{-7}	–	–
Fecotox (CTUe)	–	kg 1.4 DB eq.	0.11
HH,ce (CTUh)	–	–	–
HH,nce (CTUh)	–	kg 1.4 DB eq.	0.86
PM (kg $PM_{2.5}$ eq.)	–	-	–
IR,hh (kg U^{235} eq.)	–	–	–
POF (kg NMVOC eq.)	–	kg C_2H_4	7.8×10^{-4}–4.9×10^{-3}
A (mol H+ eq.)	0.81–2.14	kg SO_2 eq.	1.6×10^{-2}–2.8×10^{-2}
TE (mol N eq.)	–	kg 1.4 DB eq.	9.0×10^{-3}
FE (kg P eq.)	1.8×10^{-4}–1.7×10^{-3}	–	–
ME (kg N eq.)	1.9×10^{-3}	–	
WRD (m^3 used)	–	–	–
MFRRD (kg Sb eq.)	2.2×10^{-2}	–	–
LT (kg deficit)	–	m^2 year	1.6–18.8

used to analyse the impacts of producing 1 kg of PHA. All material

agricultural efficiency of the corn system that yields 4.8 tonne of

Table 8
Environmental impact results for polyhydroxyalkanoates production from different feedstock: sugarcane, corn starch and wastewater. (Functional Unit = 1 kg of polymer produced).

Impact category	Unit	Feedstock	
		Sugar cane	Corn starch
CC	kg CO_2 eq	2.72	4.26
OD	kg CFC-11 eq	2.55×10^{-7}	3.21×10^{-7}
Fecotox	CTUe	11.79	17.05
HH,ce	CTUh	1.20×10^{-7}	1.84×10^{-7}
HH,nce	CTUh	3.29×10^{-6}	3.60×10^{-6}
PM	kg $PM_{2.5}$ eq	4.72×10^{-3}	1.90×10^{-3}
IR hh	kBq U^{235} eq	0.42	0.66
POF	kg NMVOC eq	1.00×10^{-2}	8.73×10^{-3}
A	molc H+ eq	2.57×10^{-2}	3.25×10^{-2}
TE	molc N eq	7.86×10^{-2}	0.10
FE	kg P eq	7.81×10^{-4}	1.47×10^{-3}
ME	kg N eq	9.95×10^{-3}	3.07×10^{-2}
WRD	m^3 water eq	2.35	3.91
MFRRD	kg Sb eq	3.18×10^{-5}	2.49×10^{-5}
LT	kg C deficit	17.35	37.79

sugar per hectare (corresponding to 5.9 tonne of wet starch per hectare) compare to 11.1 tonne per hectare for sugar cane. When assessing the contribution of each production step to the final impacts, the results show that the production of the energy (the electricity and steam consumed in the process) and the agriculture production of sugar cane or corn starch are the main contributors. For example, for OD, IR and WRD more than half of the impacts are related to the production energy that is consumed in the process. For CC, 66% of the impact in the sugar cane case study is associated with the energy consumed, while for the corn system more than half of CC comes from starch production. For the impact categories ME, TE, HH and PM, more than two-thirds (up to 91% for ME) of the impacts are associated with the sugar production stage (both from sugar cane and starch). This fact reveals the importance of the agriculture phase (described in the food section) when compared with the PHA production phase (that includes fermentation of sugars and separation of PHA).

Regarding the subsequent life cycle phases (i.e. use stage and end-of-life) the PHAs polymers defer greatly from the bio-based ethanol case study (see section 3.2) because PHAs can be shaped into different products for different applications (packaging, medical devices, etc.), while the end-of-life depends on their biodegradability potential. For example, Pietrini, et al. [71] compared biocomposite materials of PHAs and fossil-based polymers used in personal computers housing and concluded that the use of PHAs biocomposite materials presented environmental benefits compare to the fossil materials. Literature comparisons of different end-of-life options [72,73] suggest that anaerobic digestion (with energy recovery) and composting (the so-obtained compost is then used as fertilizer) appear as the best options for PHAs and other biodegradable materials.

The normalised results in Fig. 11 show that from amongst all impact categories studied for PHAs production, WRD and HH (either cancer effects, or non-cancer effects) present the highest relative contribution to the overall European environmental impacts for both sugar cane and corn starch systems.

4. General discussion

Below are general points of discussion which could help soften or avoid some of the inconsistencies detected during this analysis.

- It is important to be coherent with the methodological choices made along the analysis. It was noticed that often in literature LCA results are compared although these have been calculated using different methodological assumptions (e.g. allocation and substitution). This should be avoided as much as possible since, as stated by Ref. [15], it is incorrect to compare one system using allocation among the co-products with one system using substitution for the same co-products.
- It is a common practice to narrow down the number of impact categories analysed (compared to that indicated by the chosen LCA impact assessment method) when evaluating the environmental performance of a system. Excluding impact categories from the analyses should be supported by appropriate documents, since it negatively affects the comprehensiveness of the environmental assessment, the comparability among studies and the final interpretation [8].
- A knowledge gap was identified for some impact categories. While categories such as HH (both cancer and non-cancer effects), Fecotox, and WRD are often excluded, they could be important for specific value chains. The exclusion could be due to difficulty and/or poor robustness of their calculation models. As an example, the Ecoscarcity model, used to calculate the WRD impact category, does not consider the water used by hydroelectric facilities as water depleted. However, the LCA software used in this study gives a characterisation factor for this specific flow, resulting in an overestimation of this impact (i.e. in our examples is around one order of magnitude). Research efforts are ongoing to improve the overall robustness of the impact assessment models in order to include the missing categories in future LCA studies.
- Increased harmonisation and transparency in the reporting of LCA data and results is needed. The recommendations given by LCA guidelines and standards such as ISO 14040 and/or ILCD handbook on reporting of results should preferably be followed, and the nomenclature used by the chosen LCA impact assessment method should be used consistently when reporting environmental impacts. In order to increase transparency of LCA studies and their results, it is also important to always include or reference the Life Cycle Inventory (LCI) used to calculate the LCA results.

5. Conclusions

In this study a LCA data mapping and modelling exercise was performed for three different bioeconomy value chains. The mapping exercise identified maximum and minimum of environmental impact values reported for each analysed product.

Most of the data used to map the environmental performance of the example case studies was obtained from peer-reviewed articles published in scientific journals. Regarding the publicly available LCA data from EU research projects, they were found to be generally limited and/or aggregated (e.g. reported as percentages), which fact reduced the feasibility of performing comparisons with data from other studies.

The analyses performed for each bioeconomy value chain, show high variability in published LCA results. The reviewed individual LCA studies differ from each other with respect to several methodological assumptions: definition of the system boundaries, functional unit, energy recovery, carbon emissions and storage and allocation methods. All these differences influence considerably the environmental performance and the overall interpretation and comparison of results become rather challenging. In addition, low uniformity in the way the results are presented has been observed (e.g. different terminologies are used for the same impact). This makes it even harder to conduct comparative analyses and could even lead to misinterpretation of results. All these issues highlight

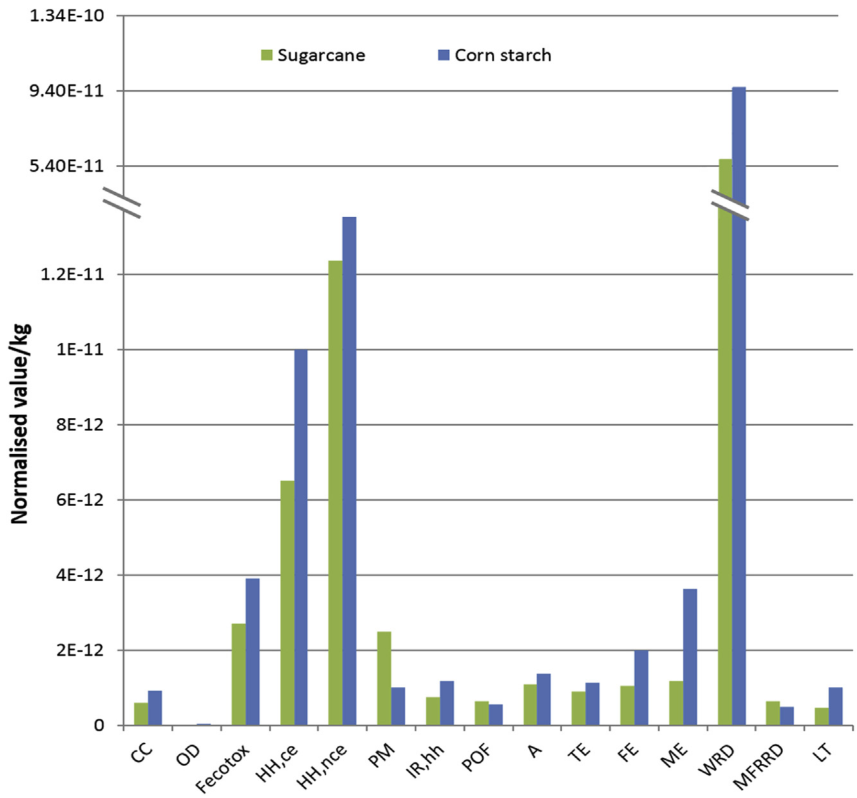

Fig. 11. Normalized environmental impacts of polyhydroxyalkanoates production.

the strong need for methodological harmonisation and coherence for LCA of bioeconomy value chains.

Disclaimer

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