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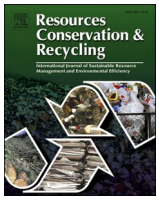

Full length article

Cradle-to-grave life cycle assessment of single-use cups made from PLA, PP and PET

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ABSTRACT

Polylactide (PLA) is both bio-based and biodegradable and has therefore attracted increased attention for single-use plastics applications. Under the context of the recent EU Plastics and Bioeconomy strategies, this study uses life cycle assessment (LCA) to assess the environmental footprint of single-use drinking cups made from PLA, including 13 environmental impact categories. Land use changes (LUCs) were assessed based on a deterministic model. The manufacturing phase was modeled based on primary production data stemming directly from the industry. The end-of-life (EoL) impacts were assessed using the EASETECH. PLA cups were then compared with their petrochemical counterparts polyethylene terephthalate (PET) and polypropylene (PP) cups. Based on the available data quality of the petrochemical polymers, six impact categories were compared. For PLA cups, the process energy use in the conversion from biomass to PLA polymer was identified as the main environmental hotspot, followed by the electricity consumption of thermoforming of the cups. It was found that the biomass acquisition phase has a limited overall impact. LUCs contribute to a negligible impact in all impact categories except for climate change and photochemical ozone formation. Compared to PET cups, the current PLA cups offer environmental impact savings for climate change even including the impacts of LUC. Compared to both PET and PP cups, PLA cups offer savings for fossil fuels resource use but lead to higher impacts for photochemical ozone formation, acidification and terrestrial eutrophication.

1. Introduction

In 2015, the global plastic production amounted to approximately 380 Mt (million metric tonnes) (Geyer et al., 2017), requiring approx. 6% of the global crude oil production (World Economic Forum, 2016) and generated an annual waste equivalent to the annual production due to the streams from previous years (Jambeck et al., 2015). In addition, the global demand for plastics is expected to double in the next 20 years (European Commission, 2018a). To achieve such increased demand while combating climate change and plastics littering, novel polymers that are both bio-based and biodegradable, such as PLA (polylactic acid), have attracted much attention for single-use plastics applications.

However, it is still to be questioned whether PLA is really a solution for the environmental issues caused by single-use plastics applications from a policy perspective (European Commission, 2019). For this type of policy-context decision-makings, environmental Life Cycle Assessment (LCA) (ISO14040:2006; ISO 14,044:2006) is a typical and widely applied tool (Finkbeiner, 2014; European Commission, 2010).

To completely abandon the use of single-use cups is difficult in the short term, especially in the applications with stringent hygiene requirements. For this reason, there is still need to search for the alternatives for a more sustainable solution for single-use cups. LCAs on PLA cups have been extensively published in the last decade. The major peer-reviewed studies are summarized in Table 1. For single-use cups, most studies consider PET, PP and PS as the main petrochemical counterparts of PLA. All studies considered cradle-to-grave comparisons. The common key conclusions of these studies can be highlighted as follows:

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Table 1
Main characteristics and scopes of the LCAs of PLA cups in the literature*. "x" stands for indicators or processes included in the scope of that study. PS=polystyrene, PC=Polycarbonate, PE=Polyethylene, PET=Polyethylene terephthalate.

Study	Petrochemical counterparts	Impact categories assessed													End of Life options			
		Climate change	Acidification	Eutrophication	Energy depletion	Respiratory effects	Ionizing radiation	Mineral depletion	Ozone layer depletion	Ecotoxicity	Human toxicity	Photochemical oxidant formation	Land use	Water use	Incineration	Landfilling	Composting or anaerobic digestion	Recycling
(Uihlein et al., 2008), original study	PS	x	x	x	x	x	x	x	x	x	x		x		x			
(Binder and Woods, 2009), original study	PP and PET	x	x	x	x							x		x		x		
(Vercalsteren et al., 2010), original study	PC, PP, PE-coated cardboard	x	x	x	x	x		x	x	x	x				x		x	
(Van der Harst and Potting, 2013); Review	PP, PET and PS	x													x	x		x
(van der Harst et al., 2014), Original study	PS		x	x	x				x	x	x	x			x		x	x
(Potting and van der Harst, 2015) Original study	PS	x	x	x	x			x	x	x	x	x			x		x	x

* List of articles retrieved from Scopus database on 4th June 2019. The inserted query was: TITLE-ABS-KEY ((lca OR (life AND cycle AND assessment)) AND (cups OR cup) AND (pla OR (lactic AND acid))) AND (LIMIT-TO (LANGUAGE, "English")). The list has been extended by including the technical report authored by (Binder and Woods, 2009) and the review report by (Van der Harst and Potting, 2013). All LCA studies listed here are cradle-to-grave studies.

- compared to PET cups, PLA cups offer lower environmental impacts in terms of fossil fuels depletion and climate change (Binder and Woods, 2009; Potting and van der Harst, 2015; Uihlein et al., 2008); PLA cups generally perform worse than their petrochemical counterparts for many impacts other than fossil fuel depletion and climate change (Potting and van der Harst, 2015; Uihlein et al., 2008; Vercalsteren et al., 2010);
- the conversion steps from biomass to PLA polymer dominate the overall environmental impact of PLA cups from cradle to grave (Binder and Woods, 2009; Potting and van der Harst, 2015); and
- from an environmental point of view, composting is worse than recycling and incineration for PLA cups, because 1) it does not lead to the displacement of products like energy and virgin plastic materials whose production is highly impacting on the environment (Potting and van der Harst, 2015; van der Harst et al., 2014) and 2) PLA does not contain nutrients, and therefore the production of the fertilizers avoided by the produced compost cannot be credited (van der Harst et al., 2014).

However, some common limitations are observed from the LCA studies published:

- None of the studies reviewed has investigated the environmental impact caused by land use changes (LUCs). The use of arable land to supply the biomass for PLA production generates a flow of emissions resulting from additional demand for arable land (IPCC, 2019). Globally, land use changes account for approx. 13% of annual net CO_2 emissions (data for 2007–2016); these emissions are essentially due to deforestation (IPCC, 2019).
- Most published LCA studies do not include key resources such as land use and water use for bio-based plastics (see Table 1), whereas these are important resource indicators for the bio-based sustainable innovation (Broeren et al., 2016). Two studies reported either land use or water use (see Table 1) but only the aggregated inventory data were presented rather than the associated environmental impacts, e.g. land occupation reported by (Uihlein et al., 2008) and water consumption reported by (Binder and Woods, 2009). This is partly due to the lack of consensus on the impact assessment models for land and water uses in the past.
- The published LCAs often simplified the end of life (EoL) models, especially the modeling of heterogeneous materials, their actual compound compositions (e.g. additives), and potential contaminations. Yet, these are highly variable from one polymer to another and therefore, such differences should be accounted for. Nevertheless, assessing properly heterogeneous material flows taking into account their physical and chemical compositions is not straightforward by using LCA software like Simapro or GaBi (Clavreul et al., 2014). For example, the contamination and the heterogeneity of the material flows were often overlooked (for example in the studies investigating composting listed in Table 1). These two aspects can be improved by using a dedicated EoL model such as EASETECH, a model that allows to account for the compositions of the waste flows and for their carbon degradation rates (Clavreul et al., 2014).
- The published LCAs of PLA cups listed in Table 1 used PLA production data of a decade years old. The use of updated data is particularly important for novel technologies because the production processes have larger potentials of improvement than mature and established productions. For PLA polymer, the climate change impacts published in 2011 was 60% lower compared to that published in 2006 (Van der Harst and Potting, 2013). This result was largely due to improved lactic acid production (e.g. better energy optimization).

To bridge these above-mentioned knowledge gaps, this study aims to provide a more complete environmental impact assessment by assessing 13 environmental impact categories, including the effects of land use changes, using the most up-to-date production data directly gathered

Table 3
Environmental impact categories assessed and the impact assessment models*. Categories marked with "X" are selected for comparison with PET and PP.

Impact Category	Unit	Impact assessment models	Comparison with PP and PET
Climate change	kg CO_2eq	IPCC 2013, GWP 100a with carbon climate feedback (Hartmann et al., 2013)	X
Ozone depletion	kg CFC-11 eq	(Ramanathan and Feng, 2009)	
Particulate matter	kg PM2.5 eq	(Rabl et al., 2014)	X
Ionizing radiation Human Health (HH)	kBq U235 eq	(Frischknecht et al., 2000)	
Photochemical ozone formation	kg NMVOC eq	(van Zelm et al., 2008)	X
Acidification	molc H+ eq	(Posch et al., 2008)	X
Terrestrial eutrophication	molc N eq	(Posch et al., 2008)	X
Freshwater eutrophication	kg P eq	(Goedkoop et al., 2009)	
Marine eutrophication	kg N eq	(Goedkoop et al., 2009)	
Land transformation	kg C deficit	Soil Organic Matter model (Milà i Canals et al., 2007)	
Water use	m3	AWARE factors (Boulay et al., 2015)	
Resource use, minerals and metals	kg Sb eq	(van Oers et al., 2002)	
Resource use, fossil fuels	MJ	(van Oers et al., 2002)	X

*The impact assessment models that differ compared to PEFCR guidance version 6.3 are the ones related to land transformation and particulate matter, which have been based on PEF guidance (European Commission, 2012) because the sufficient inventory details were not available for these two impact categories based on version 6.3 of the PEFCR guidance.

Table 2
Mass per functional unit based on material substitution factors in terms of stiffness.

Material	Young's modulus (MPa) tested under ISO 527–2:2012 standard conditions (ISO, 2012a)	Density(g/cm^3)	Stiffness Material Index	Mass per FU adopted in this study. PET and PP cups calculated based on MSF benchmarked with the weight of PLA (kg/1000 cups)	Observed weights of single-use cups available in the market (kg/1000 cups)
PLA	3500 (Corbion, 2019)	1.25 (Corbion, 2019)	12.1	4.6	4.1–4.7
PET	2200 (Throne, 2008)	1.30 (Throne, 2008)	10.0	5.5	5.5–6.4
PP	1800 (Molgroup, 2017)	0.86 (Molgroup, 2017)	14.1	3.9	2.9–3.5

from the two major polymer producers of PLA, and assessing the EoL impacts using a model handling heterogeneous material flows. By accounting for all these aspects, it is possible to provide a comprehensive answer to whether PLA cups can really offer environmental benefits in the near future. Such a full picture is in the interest of the EU policy audience as support for future bioeconomy and plastic strategies (European Commission, 2019). In particular, most of the methodological choices adopted in this LCA e.g. the selection of the spectrum of environmental impact categories are based on the latest European Commission Product Environmental Footprint Category Rules (PEFCR) guidance method, version 6.3 (European Commission, 2018b) whose aim is to increase the environmental comparability of products using LCA.

2. Material and methods

ISO 14,040 and ISO 14,044 (ISO, 2006a, 2006b) were followed for this LCA. The recommendations of the latest version available of PEFCR guidance by the time the study was prepared (European Commission, 2018b) were also followed as closely as possible.

2.1. Goal and scope definition

The targeted audience of this LCA is EU policy makers, who are interested in an environmental sustainability assessment of innovative single-use bio-based products to be used as the basis for possible future policy intervention (European Commission, 2019). In particular, such a policy decision needs scientific evidence about the main sources of environmental impact caused by bio-based materials and the potential reduction of environmental impact allowed by replacing the conventional materials currently used for the same applications (Moretti et al., 2020; Vera et al., 2020).

Accordingly, this study aims to support policy makers by 1) highlighting the environmental hotspots of PLA single-use cups and 2) comparing the environmental performances of single-use plastic cups made from PLA and its fossil fuel counterparts. To reach these two objectives, it is important to consider the entire life cycle of the product and, therefore, a cradle-to-grave LCA is performed.

For PLA cups, the LCA is modeled considering a short-term decision context (current/near-future situation) and therefore, the next 5–10 years are selected as temporal scope. It is assumed that the cups are sold, used and disposed of in Europe. The supply chain can, however, be global. The technological scope is the market-weighted average mix of the near future commercialized technology.

The goal of the study requires to identify the main materials currently employed for single-use cups, which PLA could potentially replace in the near future. Based on the interview with the industrial partners, PLA cups are mainly designed to compete with PP and PET. The comparison is therefore carried out considering cups made from these two materials as petrochemical counterparts. Based on the targeted audience and the geographical scope defined, the European production of PET and PP is considered.

The functional unit (FU) is defined as *1000 single-use cups with 200 ml volume used to contain cold drinks*. Based on the market survey conducted in the EU BIOSPRI project (European Commission, 2019), the volume of 200 ml is chosen as one of the most common volumes offered on the market for single-use cups.

The determination of the weights of the cups and, therefore, the reference flow of each product system is one of the main critical assumptions in a comparative LCA for single-use cups (Van der Harst and Potting, 2013). Based on the market survey (see appendix for more details), a 200-ml cup is the lightest when made from PP (2.9–3.5 g), followed by PLA (4.1–4.7 g) and the heaviest for PET (5.5–6.4 g). However, these weights are not directly operational in defining the functional unit. Large ranges were observed for all three cups, possibly not only due to specific market demand but also the constraints of processing technologies, which are not directly associated with technical performance of the material.

In this LCA, a theoretical approach is adopted to estimate the weight required for a single-use drinking cup made from different materials. The theoretical estimation is preferred to measure the weights of the real products for two reasons: 1) it avoids the selection of a value in the range observed in the market, which may not be caused the material properties but rather by the limitation of the processing equipment and techniques; and 2) it allows comparison based on the same functionality performance for a primary property, e.g. stiffness or strength of the material.

We assumed a stiffness-constraint design for the cup because stiffness is the primary desired property for a cup (Engelmann, 2012). Material stiffness is a measure of how a material resists deformation when a force is applied: stiffer means less flexible. The axial stiffness of an element in tension is the ratio of the cross-sectional area times Young's modulus and divided by the length of the element. Accordingly, Ashby (2005) proposed a so-called Material Index (MI) to indicate the stiffness performance of a material as a function of both Young modulus E and density ρ of that material (see Eq. (1)). The higher the MI, the better (stiffer) the material, which in turn leads to a lower mass requirement.

$$MI_{stiffness} = \frac{E^{1/3}}{\rho} \quad (1)$$

MIs allow making comparative analyses for novel materials by defining the *Material Substitution Factor* (MSF). MSF represents the ratio between the minimum masses needed by two materials to satisfy the same design constraint (e.g., stiffness-constraint design). MSF is defined as the ratio between the MI of the reference material and the MI of the new material (see Eq. (2)). When MSF is greater than 1, it means that the new material (Material A in Eq. (2)) needs to be heavier than the reference material in order to fulfill the same performance (e.g. the same stiffness performance).

$$MSF_{stiffness} = \frac{MI_{ref}}{MI_A} \quad (2)$$

First, the weight of the 200 ml PLA cups was assumed to be 4.6 g, which corresponds to the weight of a PLA cup supplied by a company who supported this study, and the value is within the range of the weights observed in products available in the market (4.1–4.7 g). Given the FU of 1000 cups, it leads to 4.6 kg of PLA cups. Then, applying the concept of MSF, the theoretical weights of PP and PET cups (the reference materials) were determined by assuming 4.6 g for the weight of the PLA cup. Table 2 provides the data used to calculate the MSFs, MIs and the weights of PP and PET cups for the same functional unit.

Although the calculated values are slightly higher for PP cups, the other values are in line with the ranges of weight observed for the products in the market. A sensitivity analysis is conducted to assess the variation of the results when the weights observed on the market are used instead of the theoretical estimation (see Discussion section).

Based on the interest of EU policy makers, the selection of the impact categories is based on the recommendations of PEFCR guidance version 6.3 (European Commission, 2018b). However, the impact categories related to toxicity are excluded because their models are so far uncertain and, therefore, could lead to distorted results (Zampori et al., 2016). Hence, thirteen environmental impact categories (see Table 3) were selected for PLA cups. Table 3 highlights also the six impact categories that are compared to the petrochemical cups. Not all impact categories were compared due to lack of data for the petrochemical polymers for all impact categories (see the Section 2.2.1.2 for details).

2.2. Life cycle inventory

In this section, the life cycle inventory is detailed. Section 2.2.1 describes the sub-processes of the three product systems, the data and the assumptions made. Section 2.2.2 describes the modeling of land use changes, and Section 2.2.3 describes the inventory model of the end-of-

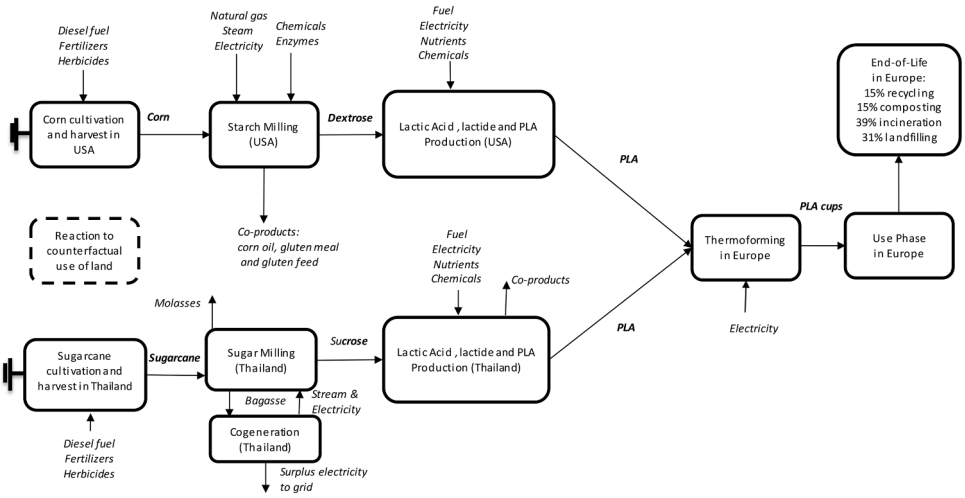

Fig. 1. Process flow diagram illustrating life-cycle stages and unit processes of PLA cups. The dashed box represents a counterfactual unit process for the impact of land use change.

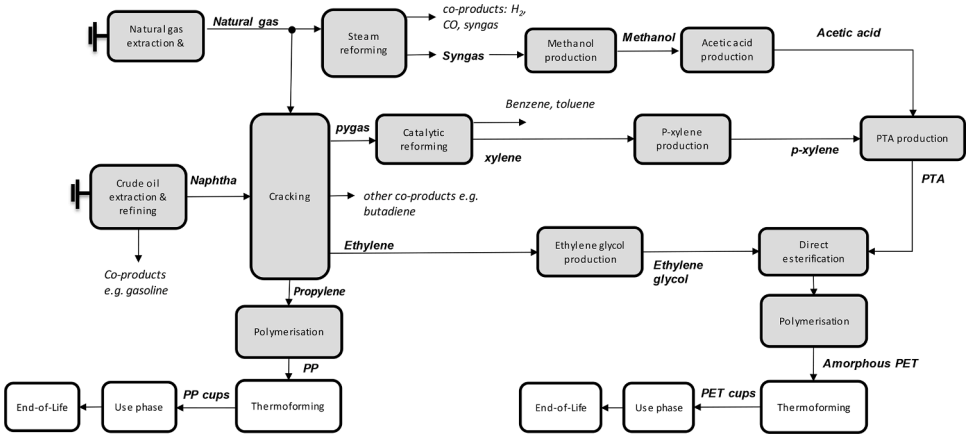

Fig. 2. Process flow diagram for petrochemical cups made of PP and PET produced in Europe. Gray boxes represent processes included in the PlasticsEurope's datasets for PP and PET. End of life: 30% recycling, 39% incineration and 31% landfilling.

life waste management. In Sections 2.2.4 and 2.2.5, the model of biogenic carbon and the allocation principles are explained.

The generic system boundaries are described as follows (with the assumption that the use phase has negligible impacts):

1) The material production phase. For PLA cups, it covers the phases from the cradle (land use and biomass cultivation), all the conversion steps (including all the biomass processing, fermentation, purification, and all logistics services) to the delivery of the polymer at the factory gate. For PET and PP cups, it starts with crude oil extraction and includes oil refining and the distribution of the final polymer.
2) The manufacturing of the final product (cups). This is mainly the thermoforming of polymer granulates into the final product cups. Transportation of polymers and cups distribution are also included in this step.
3) The end-of-life waste management phase (recycling, incineration and landfilling, plus composting only for PLA cups).

Concerning the selection of the data, the following hierarchy is used:

I Primary data are preferred because they are both site-specific and representative in terms of geographic, technological, and temporal scopes.

II If primary data are not available, either secondary data from the literature or modified LCI data based on site-specific information are used. These data are therefore only partially specific to the facility under assessment and are mixed with proxy data, for example representing average data from similar industries.

III The least preferred option is data from generic LCI databases. Such data are not specific of the facility under assessment and are selected based on the best proxy.

2.2.1. Flow diagrams and cradle-to-gate modeling

2.2.1.1. From biomass to PLA cups. Based on the goal and scope of this LCA, PLA production should reflect the near future technological level in Europe. Fig. 1 shows the process flow diagram of cradle-to-grave PLA cups.

From biomass to PLA polymer

To date, two major producers offer the polymer and the monomer (lactide) at a commercial scale. Based on the announced nameplate capacities, NatureWorks LLC has an the annual capacity of 150 kilotonnes (kt) PLA from corn in the US accounting for 67% of the global PLA capacity; Total Corbion PLA produces 75 kt of lactide per year made from Thai sugarcane accounting for 33% of the global capacity.

For corn-based PLA, the life cycle starts with corn cultivation (and

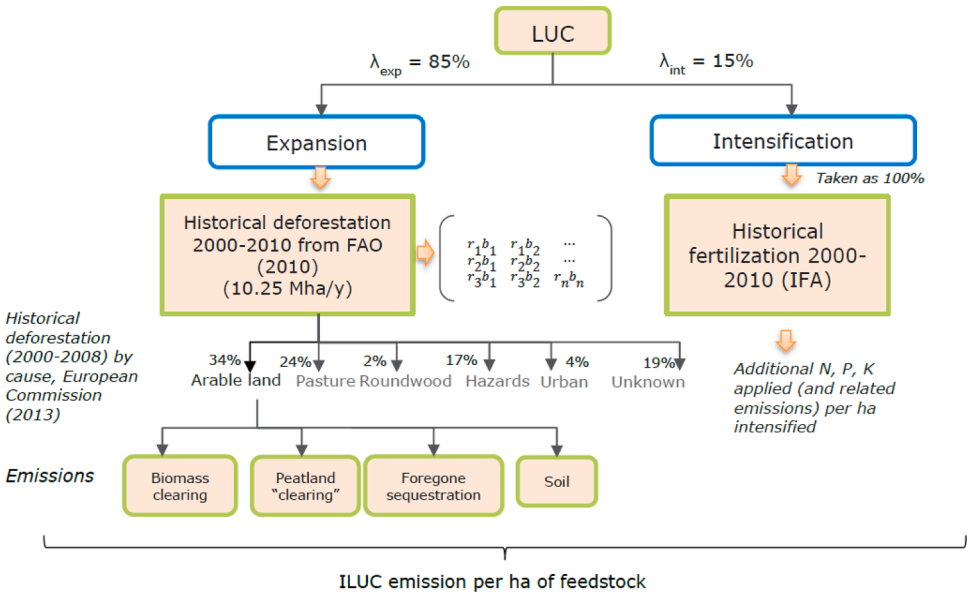

Fig. 3. The land use change model applied in this study. λ is the share of the response to an increase in demand of arable land. r is region and b refers to biome.

harvesting), which requires corn seeds, fertilizers, pesticides, irrigation, limestone, electricity and fuels (e.g. diesel) (Vink and Davies, 2015). During cultivation, dinitrogen oxide, nitrogen oxides, nitrates and phosphates are released (Vink and Davies, 2015). The harvested corn is transported to the wet milling facility. For each kg of PLA, 2.7 kg of corn is hydrolyzed to obtain dextrose using a wet milling process (Vink and Davies, 2015). The milling process requires both steam, which is generated by natural gas, and electricity, which is locally sourced from the grid. The dextrose is then transported by pipeline to the biorefinery where the lactic acid fermentation process occurs (Vink and Davies, 2015). From lactic acid, lactide is manufactured and then polymerized to PLA. For all conversion processes to obtain PLA from US corn, the dataset "*Ingeo Polylactide (PLA) biopolymer production*" from GaBi ts 8 was modified with updated 2018 background data and site-specific data for Iowa/Nebraska corn production provided by the company.

Sugarcane cultivation requires similar inputs to the ones listed for corn. A typical practice for sugarcane cultivation is the use of the filter cake (residue from sugar milling) as a soil conditioner (Morão and de Bie, 2019). Differently from corn, only 10% of Thai sugarcane needs irrigation and harvesting is 90% manual (Morão and de Bie, 2019). The harvested sugarcane is processed in a sugar mill to liberate the sugar. The steam and electricity needed for the sugarcane milling are obtained combusting bagasse, which is a fibrous residue of sugarcane juice extraction. The surplus electricity is sold to the local Thai grid operator. After the milling process, PLA is obtained by fermenting the sucrose content of sugarcane. For PLA from Thai sugarcane, an aggregated inventory dataset[1] was provided by Total-Corbion based on a recently published LCA (Morão and de Bie, 2019). For Thai sugarcane cultivation, the process "*Sugar cane, at farm/TH Economic*" from Agri-footprint version 2.0 was modified this with more site-specific and up-to-date data for the yield, irrigation water consumption, and harvesting practices (Morão and de Bie, 2019).

From PLA polymer to PLA cups

The manufacturing of the cups is assumed to occur in Europe and therefore, the PLA polymer is assumed to be transported to Europe by transoceanic containers from Thailand and the US. The distances which are assumed for the transportation of the PLA polymer were based on the actual location of production. For PLA from corn, this distance corresponds to 2000 km by freight train, and 6000 km by transoceanic ship. For PLA from sugarcane, the transportation process includes 200 km by freight train, and 17,000 km by transoceanic ship.

Once in Europe, the polymer is thermoformed into plastic cups. The electricity consumption for PLA thermoforming is assumed to be 1.23 kWh per kg of PLA processed. This figure was obtained by multiplying two values. The first value is the electricity consumption for thermoforming PET reported in (Ecoemballages and Elipso, 2015). The second value is the ratio between the electricity needed for thermoforming 1 kg of PLA and the one needed for thermoforming 1 kg of PET, which was retrieved from (Suwanmanee et al., 2013).

Distribution of cups

The cups are then distributed in Europe. This distribution is assumed to occur in two steps: first, from the producers to the distribution centers and then, from distribution centers to the final users. The distance of the first transportation step is assumed to be 850 km as done in a similar LCA of PLA cups commercialized in Europe (Vercalsteren et al., 2006). The distance covered in the second step is assumed to be 250 km and is based on the PEFCR's default distribution when no specific data are available (European Commission, 2018b).

2.2.1.2. From crude oil to PET and PP cups. Fig. 2 shows the production chain of PET and PP. In the EU, these two polymers are mainly derived by cracking naphtha, which is a byproduct of from crude oil refinery. According to the goal of assessing cups for the European market, the petrochemical polymers are assumed to be both distributed and thermoformed into cups in Europe.

LCI data of PP and PET are limited in the public domain. In the reviewed LCAs of cups in the scientific literature, the petrochemical PP and PET data are usually retrieved from either PlasticsEurope's Eco-Profiles or Gabi. In our study, the data for European average petrochemical polymers production are retrieved from PlasticsEurope (2014, 2012). We compared the LCA results of PET and PP polymers published by PlasticsEurope's and GaBi 2017 database to evaluate the consequences of choosing the other database i.e. Gabi. The geographic scopes of the two datasets are not the same (average Europe for PlasticsEurope vs. Germany for Gabi). The comparison has highlighted that the differences in the impact assessment results between the two LCI datasets are significantly large. This fact cannot be linked only to the different

[1] The dataset provided by the company included the impact of direct land use change calculated using specific sourcing data as detailed in (Morão and de Bie, 2019). To avoid double counting, such an impact was removed from the dataset.

geographical scope (also resulting in different order of magnitude for several impact categories; see Appendix). On this basis, it was decided to exclude the impact categories affected by large and unexplainable variability. The comparisons with PET and PP cups are therefore only carried out for the following six impact categories: climate change, particulate matter, photochemical ozone formation, acidification, terrestrial eutrophication and resource use of fossil fuels (for impact assessment models adopted, see Table 3). The large variability observed shows that there is a lack of harmonization in the adopted inventory modeling and data categorization by Gabi and PlasticsEurope. Hence, for the categories presenting these largest variations, the comparison performed in the reviewed LCAs might change if the other main database was chosen to retrieve the impact of the petrochemical polymer.

Since the precise data for the average European distribution of PP and PET polymers used for cups were not available, the transportation of PP and PET polymers is modeled according to PEFCR default scenario. Hence, it was assumed as follows: 230 km by freight 32-t lorry plus 280 km by freight train and 360 km by freight inland waterways barge.

The polymers are then thermoformed into cups. The process *Extrusion of plastic sheets and thermoforming, inline {FR}| processing* of Ecoinvent 3.3 was used. To account for the different geographic scope and the specific polymer processed, the electricity input was modified as follows. First, the electricity consumption was adjusted to 1.41 and 1.03 kWh per kg of polymer respectively for PP and PET, based on the literature (Wilkinson, 2007). Second, the electricity dataset representative of France was changed to the Ecoinvent 3.3 one named *Electricity, medium voltage {Europe without Switzerland}*. The distribution of the cups was assumed to be the same as for PLA cups.

2.2.2. Modeling land use changes

The impact of displacing land as additional arable land demanded for corn cultivation in the US and sugarcane cultivation in Thailand, i.e., the so-called land use changes (LUC), were taken into account based on the approach described in (European Commission, 2019) updating the deterministic approach presented in (Tonini et al., 2016). The proposed approach to count for the impacts of LUC is not based on PEFCR method, which does not cover indirect land use change. In a nutshell, the approach used in this study (see Fig. 3) is based on an analysis of the global deforestation that occurred between 2000 and 2010, and considers two key reactions to an increased demand for arable land, namely arable land expansion (85% of the response) and agricultural intensification (15% of the response; here translated as an additional fertilizer demand only). These two shares are based on (Marelli et al., 2011). The former is further split into two reactions, namely: land clearing and foregone sequestration. It addition, the methodology considers that only 34% of the 2000–2010 deforestation was due to the demand for cultivated land (European Commission, 2013).

All carbon (carbon dioxide; CO_2, methane; CH_4 and carbon monoxide; CO), nitrogen (ammonia; NH_3, dinitrogen monoxide; N_2O, nitrogen oxides; NO_x and nitrates; NO_3^-) and phosphorus flows occurring as a result of the expansion and intensification responses were taken into account. When translated into CO_2 eq, the approach resulted in an emission factor of 4.0 t CO_2 eq ha^{-1} demanded y^{-1}, which is in the same order of magnitude as the factor derived by Tonini et al. (2016) (4.1 t CO_2 eq ha^{-1} y^{-1}), but slightly above the factors derived by (Schmidt, J. H., Munoz, I., 2014) (1.7 t CO_2 eq ha^{-1} y^{-1}; value for "world average arable land"). The LUC implication of PLA were derived considering some of the key data used in this study, namely a US corn yield of 10.3 t dry matter ha^{-1} y^{-1} (15% moisture) (Vink et al., 2015), a Thai sugarcane yield of 22.5 t dry matter ha^{-1} y^{-1} (70% moisture) (Durlinger et al., 2017).

2.2.3. Modeling end-of-life

After the cups are used, it was assumed that they are managed the same way as all mixed plastics waste in Europe. In particular, the 2018 *European EoL mix* for plastic wastes was made of 30% mechanical recycling, 39% incineration and 31% landfilling (European Commission, 2018a). The share of (mechanical) recycling is representative of PP and PET cups, but not for PLA cups, which are still a niche product. Mechanical recycling of PLA cups is feasible at a small scale but there is a lack of proper infrastructure for its recycling at large scale (Hottle et al., 2017). Moreover, PLA is developed under the intention to be disposed of by industrial composting. It was, therefore, assumed that a near-future package of the waste management of PLA could consist of 15% industrial composting, 15% mechanical recycling, 39% of incineration and 31% landfilling.

As mentioned in the introduction, using the EASETECH model (Clavreul et al., 2014), it is possible to account for the difference of material composition between PP, PET and PLA cups. The EoL of the cups is assessed considering the associated organic contamination. In particular, based on (Petersen et al., 2012), 100 g of organic contamination was assumed per 1 kg of PLA. The chemical composition of this added organic contamination was based on (Petersen et al., 2012). For the petrochemical plastics, the composition of the waste flow (PP or PET waste with contamination) was based on measured values from a real municipal solid waste in Denmark (Götze et al., 2016). The main EASETECH input data for the different cups are summarised in Table 4.

For recycling, the *sorting and technology efficiency* (defined as the share of the collected plastic both sorted and technically possible to recycle) for mechanical recycling was assumed to be 70% (Plastics Recyclers Europe, 2017). The rejects of the recycling process (30%) are treated with incineration (average EU representing both with and

Table 4
Main EASETECH data inputs for PLA, PP and PET cups including contamination. C=Carbon, P= Phosphorus, N=Nitrogen, K=Potassium. Organic contamination was added to PLA cups to obtain the physic-chemical characterization of the waste flow (PLA cups plus contamination). For PP and PET cups, the organic contamination was already included in the waste characterization of the waste flow in EASETECH.

Material	Water (% of total)	Total solids (% of total)	Volatile solids (% of total solids)	C (% of total solids)	P (mg per kg of total solids)	N (% of total solids)	K (mg per kg of total solids)	Energy content LHV (MJ/kg total solids)	Sources
PLA (including contamination)	6.6	93.4	99.4	50.0	383.9	0.3	909.1	18.82	Compiled based on pure PLA and assumed organic contamination
PLA excluding contamination	0.1	99.9	100.0	50.2	0.0	0.0	0.0	18.72	(Boonmee et al., 2016; Hermann et al., 2011; Kolstad et al., 2012)
Organic contamination	72.0	28.0	93.9	41.2	4222.5	3.2	10,000.0	19.87	(Petersen et al., 2012)
PP (including contamination)	3.3	96.8	94.0	77.6	516.3	0.4	440.5	37.84	(Götze et al., 2016)
PET (including contamination)	3.3	96.8	99.0	64.0	75.0	0.2	204.2	22.91	(Götze et al., 2016)

without energy recovery). Water, steam, electricity, and chemicals consumed in the recycling process were based on (Rigamonti et al., 2014). The market substitution rate assumed was 81% (Rigamonti et al., 2014). The substituted polymers were "granulate PP at plant" for PP and "bottle grade PET granulate at plant" for both PET and PLA. Hence, for PLA, it was assumed that 1 kg of recycled PLA substitutes 0.81 kg of PET on the market as a result of a marginal replacement (see Section 2.2.5 for Multifunctionality and Allocation). The impacts of the substituted plastic materials were determined based on PlasticsEurope's Eco-Profiles.

Based on the current practices in Europe, the municipal solid waste incineration (MSWI), both with and without energy recovery, was assumed to have an average heat efficiency of 22% and an electrical efficiency of 9% based on the average of 314 waste incineration plants in 17 European countries during 2007–2010 (CEWEP, 2013; EUROSTAT, 2017). About 60% of incineration plants are equipped with energy recovery units, recovering 15% electricity and 35% of heat for every GJ of waste incinerated; the remaining 40% incineration plants are without energy recovery (CEWEP, 2013; EUROSTAT, 2017).

The landfill modeling includes construction and operation, direct emissions and collection of landfill gas (flaring and upgrading to energy production) as well as the collection of leachate. It was assumed that 29% of the collected landfill gas is upgraded and used to produce electricity with a conversion efficiency of 37% (Christensen, 2010). There is no landfill gas generated from PP and PET cups. The amount of methane gas produced from PLA is calculated based on the 1st order decay rate for methane generation in landfilling (0.011 1/s) reported by (Kolstad et al., 2012). Based on the EASETECH model, the gas collection varies over the lifetime of the landfill with rates between 35 and 75% during the first 55 years (three periods) and no gas collection for the last 45 years (OpenLCA Nexus, 2015). For the collected landfill gas, it is assumed that 49% is leaked, 29% is used for energy recovery and 22% of the gas is flared (OpenLCA Nexus, 2015). The landfill gas, which is not collected, is assumed to be partly oxidized in the landfill top covers and depends on the age of the landfill.

For industrial composting of PLA, it was assumed that 80% of the volatile solids would degrade and the 57.1% of the biogenic carbon is oxidised into carbon dioxide (Pradhan et al., 2010). A reject rate of 30% of the biodegradable polymer was assumed based on expert consultations. The rejected materials were assumed to be incinerated. Default process emissions in EASETECH were used for compost plants including the indirect emissions originated from the use of electricity and diesel used for wheeled loaders. About 30% of the original carbon of PLA is stored in the compost.

EASETECH contains a detailed module for the modeling of carbon sequestration in soil and field emissions of applying compost in soil for fertilizer substitution. In this module, the substitution of fertilizer was accounted for as a credit by applying system expansion by substitution. Compost and fertilizers are not the same product. So, this case is different from the substitution of the same product. For example, in the case of incineration, electricity substitutes the electricity from the grid and the end user does not notice any difference. When the two products are different, it is necessary to expand the boundaries to guarantee that the overall resource consumption and environmental emissions can be fairly compared (ISO, 2012b). Following ISO recommendations, it is therefore necessary to adjust the performance differences by including inside the boundaries the additional unit process that is directly affected by the switch of product (ISO, 2012b). For composting, it is necessary to include the differences of the use phase inside the system boundaries.

For this reason, in the EASETECH module, the use of compost is seen as part of the waste management system of composting, and therefore, is included in the system boundary. Consistently with the inclusion of the application of compost in the LCA, the credit for substituting fertilizers includes the avoidance of field emissions from applying synthetic fertilizers. In this way, the difference in the field emissions between the application of compost and fertilizers is also accounted for. This is important because the substituted product would have generated very dissimilar emissions compared to the ones resulting from the application of compost. In addition, the difference in the operation of diesel-powered spreaders from land application of organic fractions instead of fertilizers was also accounted for. The substitution of the production and application of synthetic fertilizers originated from the nutrient contents of the organic contamination, is a major difference compared to the studies investigating composing listed in Table 1.

2.2.4. Biogenic carbon

Biogenic carbon removals were taken into account in this LCA. In the inventory modeling, the biogenic carbon removal is modeled by assigning a negative CO_2 emission to the PLA cups based on their embedded carbon content (1.83 kg CO_2eq./kg PLA). The biogenic carbon that is fully oxidized during the manufacturing processes (e.g. combustion of bagasse) does not contribute to the net GHG emissions of PLA cups. The embedded biogenic carbon in the cup was assumed to become carbon dioxide when full oxidized (e.g. incineration), carbon

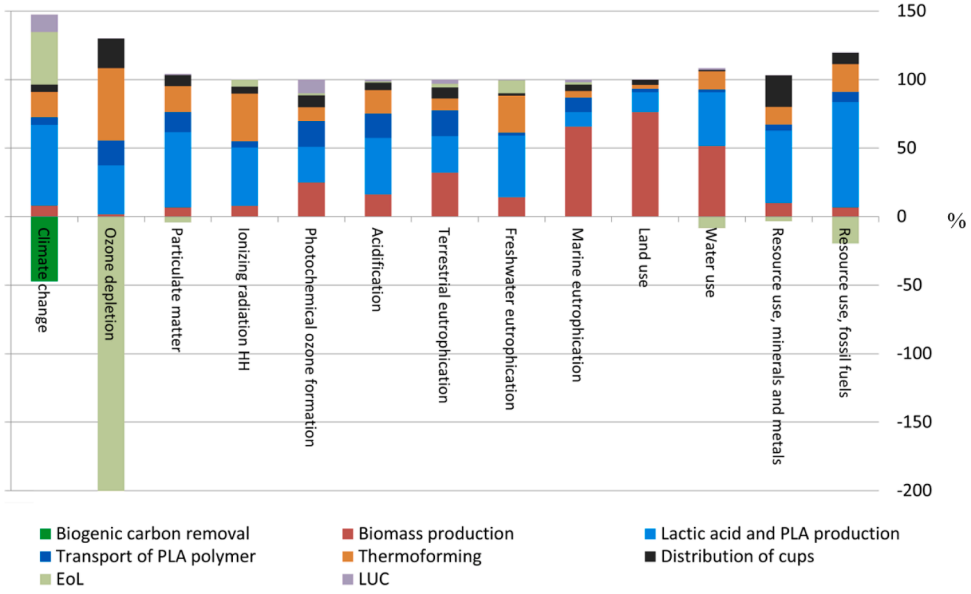

Fig. 4. Breakdown of the cradle-to-grave characterized midpoint environmental impact of one functional unit of PLA cups including LUC.

Table 5
Detailed breakdown and interpretation for each impact categories for one functional unit of PLA cups.

Impact category	Total mid-point environmental impacts(per functional unit)	Unit process/life cycle stages (See also Fig. 4 to interpret the percentages)	Major Activity-level contribution	Major Elementary-flow level contribution
Climate change	17.48 kg CO_2 eq	59% lactic acid and PLA production	60% lactic acid production (fuels, electricity and chemicals such as lime and sulfuric acid);	CO_2 of fossil origins (~90%)
			25% lactide production (natural gas and electricity	CO_2 of fossil origins (~90%)
		19% thermoforming	91% electricity	CO_2 of fossil origins (~90%)
		12.5% land use change	>99% land expansion See section 3.1.1.1	CO_2, land use change (77%) CH_4, land use change (12%)
		39% EoL	See section 3.1.1.2	−1.28 kg fossil CO_2eq plus 8.1 kg biogenic CO_2eq per FU
Ozone depletion	−7.99E-07 kg CFC-11 eq	53% thermoforming	79% electricity	Ethane, 1,2-dichloro-1,1,2,2-tetra-fluoro-,CFC-114 (70%)
		40% transportation	Production of diesel and heavy fuel oil used for transporting the polymer (46%) and distribute the cups (64%).	Methane, bromotrifluoro-, Halon 1301 (96%)
Particulate matter	6.28E-03 kg PM2.5 eq	55% lactic acid and PLA production	n.d.	particulates <2.5 um (70%), SO_2 (26%) and particulates <10 um (5%)
		19% thermoforming	98% electricity	55% SO_2, 42% particulates <2.5 um and NO_x 3%
		23% transportation	Combustion of diesel and heavy fuel oil for transporting the polymer (64%) and distribute the cups (36%).	particulates (<2.5 um) (75%) and SO_2 released (16%)
Ionizing radiation HH	1.29E+00 kBq U235 eq	42.5% lactic acid and PLA production	n.d.	58% ^{222}Rn to air and 42% ^{14}C to air
		35% thermoforming	100% electricity	92% ^{14}C to air
Photochemical ozone formation	6.87E-02 kg NMVOC eq	26% Lactic acid and PLA production	n.d.	NO_x (59%), NMVOC (29%) and SO_2 (9%)
		25% Biomass production	n.d.	81% NO_x
		19% transportation of polymer	Combustion of diesel and heavy fuel oil for transporting the polymer (69%) and distribute the cups (31%).	85% NO_x
		10% thermoforming	88% electricity	82% NO_x
Acidification	9.98E-02 molc H+ eq	42% Lactic acid and PLA production	Chemicals used during the conversion process (percentages per type n.d.)	SO_2 (78%) and NO_x (10%)
		17% Thermoforming	96% electricity	77% SO_2 and 21% NO_x
		18% transportation of PLA polymer	Transporting by container ships (~100% diesel)	~100% SO_2
		16.5% Biomass production	fertilizer use (59%), manure (10%) and energy such as electricity and diesel (25%)	~80% ammonia to air, ~10% SO_2 to air and ~10 NO_x
Terrestrial eutrophication	2.61E-01 molc N eq	33% biomass production	~85% fertilizers, ~10% manure and ~10% diesel	~85% ammonia to air
		lactic acid and PLA production (27%)	Lactic acid production (85%), lactide production (8%), polymerisation for (4%) and milling (3%).	Mainly NO_x
		19% transportation of polymer	Combustion of diesel and heavy fuel oil for transporting the polymer (77%) and distribute the cups (23%).	~95% NO_x
Freshwater eutrophication	1.07E-03 kg P eq	45% lactic acid and PLA production	n.d	~80% phosphate to water and ~10% P component to soil caused by fertilizer applied
		27% thermoforming	99% electricity	98.5% phosphate to water
		14% biomass production	n.d.	Phosphate to water (47%) and phosphorous to water (50%)
Marine eutrophication	4.18E-02 kg N eq	65.5% biomass production	n.d.	Nitrate to water (90%)
		15%Transportation	Combustion of diesel and heavy fuel oil for transporting the polymer (70%) and distribute the cups (30%).	~99.5% NOx to air
Land use	91.98 kg C deficit	76% biomass production and 14.5% lactic acid and PLA production	n.d.	n.d.
Water use	5.66 m^3	51.5% biomass production and 39.5% lactic acid and PLA production	n.d.	n.d.
Resource use, minerals and metals	1.50E-05 kg Sb eq	53% lactic acid and PLA production	n.d.	Mainly Gold (25%), Copper (15%), Cadmium (14%) and Lead (12%)
		27.5% transportation	Mostly caused by the distribution of the final product (84%)	Mainly Cadmium (35%), lead (24%) and Gold (15%)
Resource use, fossil fuels	177.48 MJ	77% lactic acid and PLA	61% lactic acid fermentation	n.d.
			27% lactide production	n.d.
		20% thermoforming	98% electricity	38% hard coal, 30% brown coal, 25% natural gas

Note: n.d. not possible to be disclosed due to confidentiality. Abbreviation: SO_2 =Sulfur dioxide, NO_x =nitrogen oxides, NMVOC = Non-methane volatile organic compound.

Table 6
Environmental impact of waste management per FU of PLA cups (4.6 kg).

Impact category	Unit	Recycling	Incineration	Landfilling	Industrial composting
Climate change	kg CO_2eq	−2.6E+00	7.0E+00	1.4E+01	5.2E-01
Ozone depletion	kg CFC-11 eq	−2.9E-05	−1.3E-07	8.3E-06	−2.8E-08
Particulate matter	kg PM2.5 eq	−1.0E-03	−2.0E-04	−1.1E-04	−4.8E-06
Ionizing radiation HH	kBq U235 eq	−7.3E-02	1.9E-01	−1.5E-02	7.2E-02
Photochemical ozone formation	kg NMVOC eq	−1.4E-02	4.0E-03	4.2E-03	1.2E-03
Acidification	molc H+ eq	−1.4E-02	6.6E-03	−1.9E-04	3.2E-03
Terrestrial eutrophication	molc N eq	−2.6E-02	2.4E-02	1.7E-04	1.0E-02
Freshwater eutrophication	kg P eq	−2.5E-04	3.3E-04	−1.4E-05	8.8E-05
Marine eutrophication	kg N eq	−2.6E-03	1.9E-03	1.5E-04	2.2E-03
Water use	m3	−1.2E+00	−6.5E-01	−2.6E-01	2.4E-01
Resource use, minerals and metals	kg Sb eq	−2.4E-07	−9.0E-07	−2.1E-07	−2.5E-07
Resource use, fossil fuels	MJ	−1.6E+02	−2.5E+01	−1.7E+00	−4.6E+00

monoxide or methane when partially oxidized (e.g. landfilling), or stored in the solid form (e.g. in the compost).

2.2.5. *Multifunctionality and allocation*

According to ISO 14,044:2006, multifunctionality shall be solved by avoiding allocation using subdivision or expanding the system boundaries (ISO, 2006b). When this is not possible allocation is the only possible option.

In the production of PLA from corn, the starch milling process has multiple outputs: corn oil (3 wt%), gluten meal (5 wt%) and gluten feed (27 wt%) along with starch (64 wt%). This multi-output process was addressed through sub-division into 11 sub-processes, as reported by Vink and Davies (2015). When it was not possible to further subdivide to avoid allocation, mass allocation was applied (Vink and Davies, 2015).

In the production of sugarcane-based PLA, the surplus electricity is sold to the Thai grid operator. System expansion with substitution was applied to the sold electricity by assuming the replacement of the local electricity production (Morão and de Bie, 2019).

As it can be noticed from the process flow diagram (Fig. 2), the petrochemical plastics industry is characterized by multiple co-productions. Co-productions in LCA should be addressed every time possible through sub-division or system expansion (ISO, 2006b). However, the unit processes of the plastic industry even if sub-divided as much as possible still remain mostly multi-functional and it is not possible to enlarge the functional unit to include all the co-functions (it does not fit our LCA goal). For this reason, PlasticsEurope applies allocation to deal with the multifunctionality issues (PlasticsEurope, 2011). The principle used for the selection of the allocation was described in the Eco-Profile documentation. The allocation (mass, energy, molar or economic value) was selected based on the closest "representation of physical causality" i.e. the one reflecting "the goal of the production process" (PlasticsEurope, 2011). .Unfortunately, the allocation applied to each unit process and their selection criteria are not fully transparent for all steps in PlasticsEurope's ecoprofiles.

For End-of-Life waste management phase, according to the formula recommended by the PEFCR guidance, the co-products of the EoL processes should be modeled by substitution. Some of the included EoL-treatment technologies produce energy (electricity and/or heat by e.g. MSWI or combusting landfill gas). The energy produced (electricity and heat) was assumed to substitute the "marginal European technologies". This substituted energy was defined by combining the marginal energy technology for each EU country to an "EU marginal mix". The marginal EU marginal electricity process was based on a methodology reported in (Itten et al., 2014) with data from 2014/2015 extrapolated to 2017. The marginal heat in Europe was assumed to be natural gas, as natural gas remains the largest contributor to heat production in Europe (Honoré, 2018). The dataset was acquired from the Ecoinvent database 3.3 (data excludes Switzerland).

Similarly, the recycling of plastics allows savings of virgin plastic and therefore substitution was applied also to the recycling process (assumed substitution of PET for PLA and PET cups and PP for PP cups). See the detailed substitution rates described in Section 2.2.3.

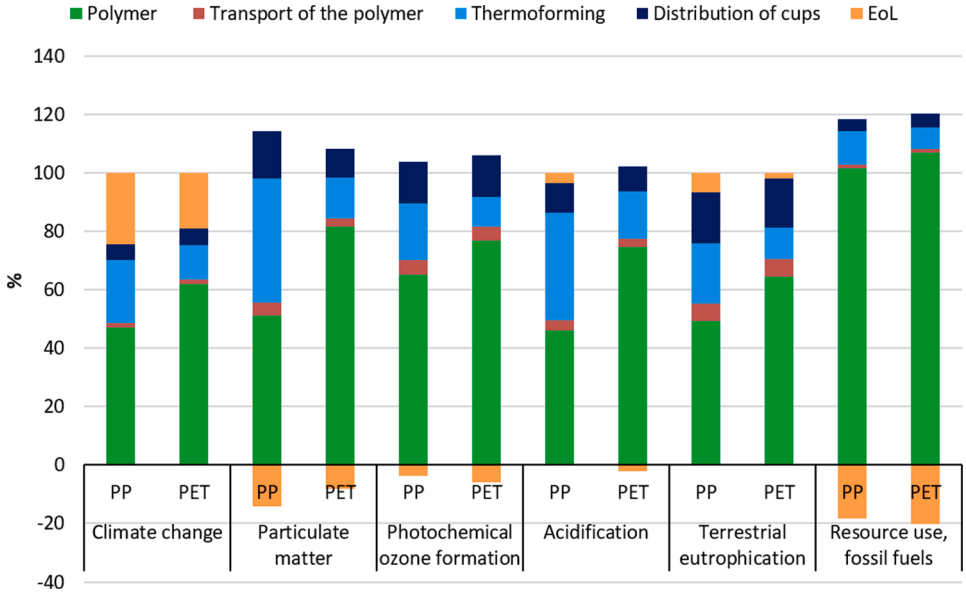

Fig. 5. Breakdown (%) of the cradle to grave environmental impact of PP cups (PP) and PET cups (PET).

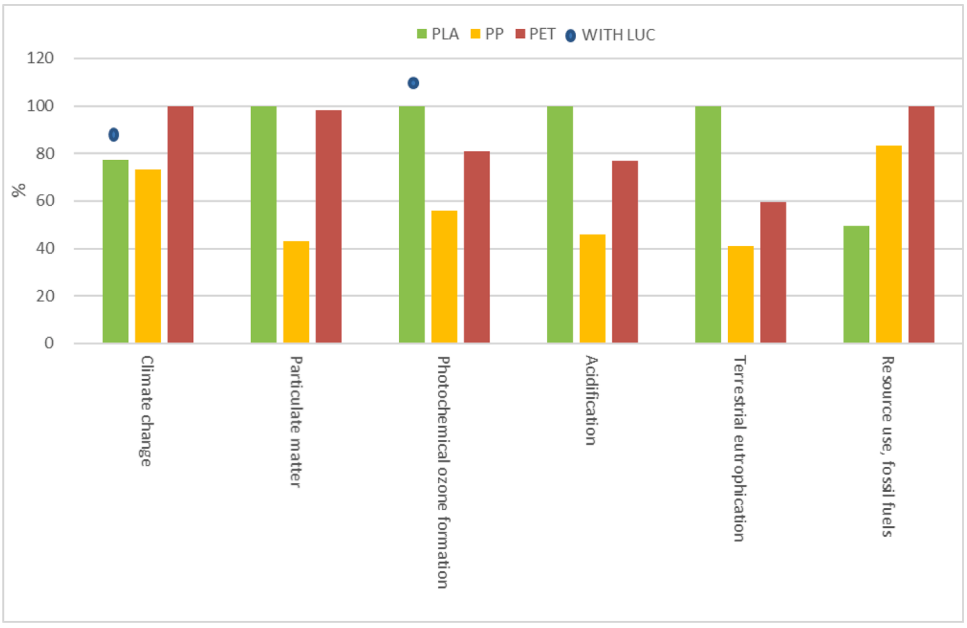

Fig. 6. Comparison of cradle-to-grave environmental impact 1000 single-use cups made from PLA, PP and PET (dots marking the impact of PLA with LUC).

3. Results

3.1. Impact assessment and interpretation

3.1.1. PLA cups

The overall breakdown for the cradle-to-grave impact of 1000 single-use cups is presented in Fig. 4 and numerically in the Appendix. Table 5 shows the detailed contributions to the environmental impact at both activity and elementary flow levels.

The production of lactic acid and PLA represents the major environmental burden in the cradle-to-grave life cycle of PLA cups. In particular, it causes 59% of climate change, 55% of particulate matter, 53% of resource use (minerals and metals) and 77% of fossil resource use. These impacts are largely caused by the production of the process heat and electricity required for fermentation, purification and polymerisation (see Table 5).

Thermoforming of cups is the second major source of impact. It is responsible for 53% of ozone layer depletion, 35% of ionizing radiation, 27% of freshwater eutrophication and 20% of fossil resource use. The electricity production dominates (80% - 100%) these impacts (see Table 5).

Biomass production has a relatively small contribution in most impact categories but not for marine eutrophication (66%), land use (76%), water use (52%), terrestrial eutrophication (33%) and photochemical ozone formation (25%). The eutrophication impacts are mainly caused by the production of the fertilizers and their application, which lead to ammonia, phosphate and nitrate emissions (see Table 5).

From an environmental point of view, the two transportation steps,

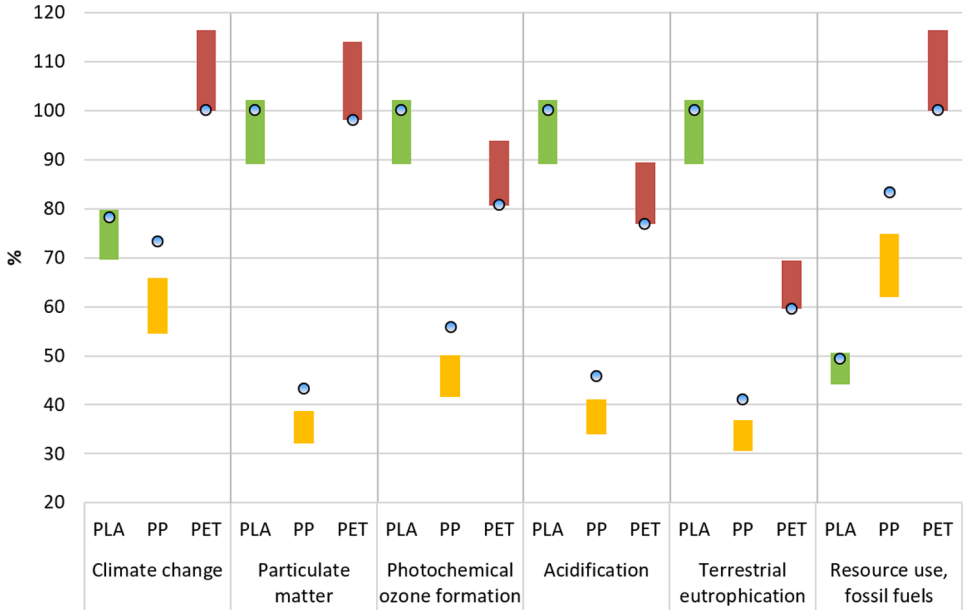

Fig. 7. Sensitivity analysis on the weight of cups per functional unit. The bars represent the ranges of variation of the environmental impact when adopting the range of weights observed on the market. The dots represent the baseline results, which used the mass per FU based on stiffness-MSFs. 100% values as in Fig. 6 (the cup with the highest baseline impact in each impact category).

even if different for distance and mode of transport, are equally important and their environmental relevance depends on the impact category considered. The transportation of PLA polymer from US or Asia to Europe is an important source of impact in the following categories: photochemical ozone formation (21%), terrestrial eutrophication (19%), acidification (18%), ozone depletion and particulate matter (14%), climate change and marine eutrophication (11%). The distribution of cups generates more than 10% impact only in ozone depletion (22%) and use of minerals and metals (23%). For all these impact categories, the main cause of impact is identified in the emissions caused by the combustion of petroleum fuels.

3.1.1.1. Land use changes of PLA. The impact of LUC is negligible (about 1% on average) for almost all impact categories except for climate change and photochemical ozone formation, where it represents respectively 13% and 10% of the cradle-to-grave impact (see Fig. 4). In these two impact categories, LUC is dominated by land expansion (>99%). For climate change, the LUC impact is essentially due to the releases of CO_2 resulting from land clearing. For photochemical ozone formation, LUC is mainly due to the CO released during land clearing (burning the cleared biomass). In particular, the LUC impact caused by US corn production is about 1.6 times higher than the ones from Thai sugarcane per kg of PLA. The main reason can be found in the higher yield per metric ton of dry matter in the case of sugarcane compared to corn.

3.1.1.2. End-of-life of PLA cups. As detailed in Section 2.2.3, the EoL mix for PLA was based on the current European EoL mix for plastic wastes and assumed to be 15% mechanical recycling, 15% industrial composting, 39% incineration and 31% landfilling. From Fig. 4, it can be seen that the overall contribution of EoL is relatively small for most of the impact categories. Nevertheless, EoL contributes to climate change impact significantly (39%) due to the GHGs emitted from incineration and landfilling due to landfill gas combustion-flaring) and direct emission of methane from the gas upgrading). Several impact categories show negative values for EoL (see Fig. 4). This is caused mainly by PLA recycling that allows savings of virgin plastic and waste incineration that recovers heat and electricity (see the assumed displaced productions in Section 2.2.3).

Table 6 shows the environmental impact results for the four investigated EoL options. Recycling is the most preferable option in all impact categories because of the substitution of virgin PET, which leads to a negative impact for recycling. Among incineration, landfill and industrial composting, the preferred option depends on the impact category.

When climate change is in focus, industrial composting seems preferable to incineration but the scope of such a comparison needs to be carefully examined before similar studies are compared. If the scope was limited to the EoL stage, and if the biogenic carbon emissions were accounted for during the composting and incineration processes, whereas no biogenic carbon removals were taken into account (because it's beyond the scope of only EoL stage), our study would have drawn the similar conclusion that incineration of PLA is much favorable than composting (van der Harst et al., 2014). Because the environmental credit received from energy recovered from incineration is substantial.

However, some other studies, e.g. (Vercalsteren et al., 2006) investigated the entire life cycle and came to a conclusion that is in line with our finding: composting is preferable compared to incineration for climate change. To identify the best end of life option, it is therefore important to account for the entire life cycle and to model the biogenic emissions of the end of life stage consistently with what applied in the previous life cycles.

The impact of composting (0.52 kg of CO_2eq per functional unit) is mainly caused by the emissions from the application of compost on the field (2.16 kg of CO_2eq) and the treatment of rejects (1.08 kg of CO_2eq). The credit for the substitution of fertilizers (production and field application), which is based on the NPK content of the contamination (see Table 4), leads to a credit of 2.77 kg of CO_2eq.

3.2. PP and PET cups

As detailed in the methodology section, only six impact categories are in focus for the petrochemical cups. The breakdown for the cradle-to-grave impact of PP and PET cups is presented in Fig. 5 and, numerically, in Appendix. The impacts of the petrochemical cups are dominated by the production of the polymer (more than 60% and 45% of impact respectively for PET cups and PP cups in all the six impact category considered). Thermoforming is the second most important environmental hotspot (due to electricity consumption). In particular, for PP cups, the impact of thermoforming is significant for particulate matter and acidification, where it represents 43% and 37% respectively. EoL is impacting relevantly only on climate change (24% and 19% respectively for PP and PET cups) mainly due to incineration.

3.3. Comparing PLA cups with PP and PET cups

Fig. 6 shows the comparison of single-use cups made from PLA, PP, and PET. Compared to their petrochemical counterparts, PLA cups offer environmental benefits in terms of fossil fuels depletion: 41% lower than

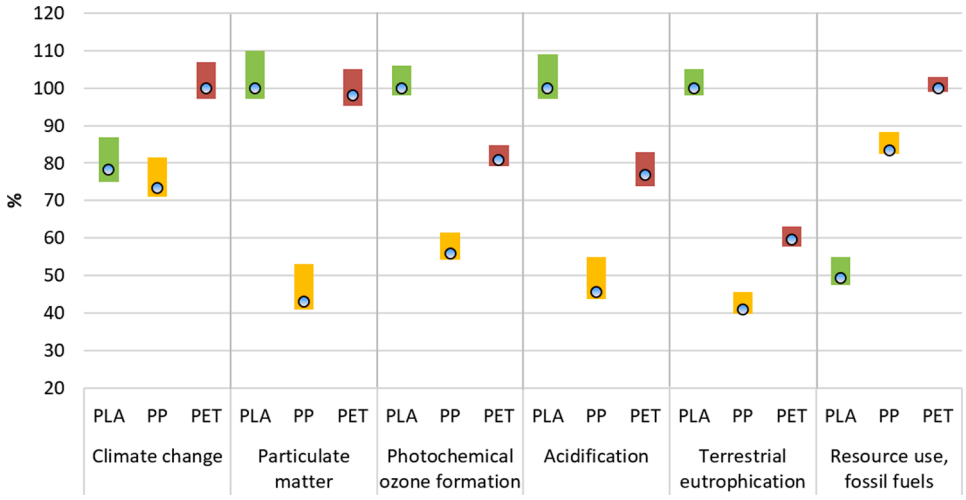

Fig. 8. Sensitivity analysis of electricity used for thermoforming per functional unit. The bars represent the ranges of variation of the environmental impact. The dots represent the baseline results. 100% values as Fig. 6 (the baseline value for the cups with the highest impact).

PP and 51% lower than PET. In all other five impact categories compared, PLA cups offer environmental impact savings compared to PET but they are less favorable compared to PP.

Compared to PET cups, PLA cups are more favorable for climate change, by offering 22% lower GHG emissions. By adding the impact of LUC to PLA cups, this figure becomes 12%. Compared to PP cups, PLA cups have slightly higher climate change impact, but the difference is marginal (5%). However, once LUC is accounted for PLA cups, this difference becomes substantially higher (15%).

In the remaining four impact categories, petrochemical PP cups have significantly lower environmental impacts compared to both PLA and PET cups. From Fig. 6, it can be seen that PP cups have about 40–60% lower impacts compared to PLA cups and 20–60% compared to PET cups in the remaining four impact categories. PP cups outperform PET in all the six impact categories and outperform PLA in four impact categories.

One of the major environmental advantages of PP cups is attributed to the low density, resulting in a lower mass requirement to fulfill the functional unit. As a consequence, a lower impact is observed in all life cycle stages for PP, i.e. polymer manufacturing, processing, transportation, and EoL waste management.

4. Discussion

4.1. Sensitivity analysis

4.1.1. Sensitivity of the key assumptions

In the goal and scope section, it was explained that the weight of the three cups in comparison had been estimated theoretically. For PLA cups, the assumed weight was within the observed market range (4.1–4.7 g per cup). For PP cups, the weights from the theoretical estimation were higher than observed in the market (2.9–3.5 g). For PET cups, the weight estimated in the baseline was in the lower bound of the range observed in the market (5.5- 6.4 g). Fig. 7 shows how the results would vary if the weight of a functional unit of cups was based on the aforementioned weight ranges.

For PLA cups, the overall impacts would vary between - 11% and + 2%. The heavier design of PET cups (6.4 g per cup) would lead to 16% higher impacts. For PP, the impact would decrease by 10–25%. Even considering such variations, PLA cups still show the lowest fossil resources use of all three options. PP is still the preferred option in five categories out of six. Overall, the conclusions of the comparison are not affected by the weights of the single-use cups.

The electricity used for the thermoforming process was one of the main sources of environmental impact (see Table 5). An inline thermoforming was assumed for the baseline calculations. However, when a non-inline process is used for thermoforming, the consumption of electricity can be 16–53% higher than in an inline process (European Commission, 2019). Moreover, based on the company data collected during this project, compared to the baseline values assumed, the electricity consumptions of an inline thermoforming could also be 14%, 23% and 19% lower respectively for PP, PET and PLA (European Commission, 2019).

Fig. 8 shows the variation of the environmental impact considering the possible ranges of the electricity consumption of thermoforming. A variation between −5% and +10% is observed in the cradle-to-grave results. This confirms the robustness of the conclusions of the final comparison. The only exceptions are PP cups whose acidification and particulate matter impacts can increase by 20–25%. Nevertheless, PP cups are still by far the best in these two impact categories.

4.1.2. Future scenarios

4.1.2.1. A better EoL waste management. In this section, two sensitive aspects for the future are analyzed: the waste management system (EoL was an important impact for climate change; Fig. 4) and electricity production (affecting most impact categories; Table 5).

In the study, the current EU EoL mix was considered. Nevertheless, in the near future, the EU aims to reduce the share of landfill and the recycling rate is expected to increase (European Commission, 2018a). Hence, it is interesting to assess how a future change in the EoL mix would affect the comparison of the cups.

PLA is designed to biodegrade under industrial composting conditions, and its recycling is also feasible. However, appropriate infrastructure for logistics and governance of both composting and recycling are still absent (see the section limitation of the study). In the future, an hypothetically best possible scenario for PLA cups “intended EoL” could be represented by 50% recycling plus 50% composting. For petrochemical cups, this hypothetically ideal EoL waste management would be 100% recycling.

When these hypothetically best possible EoLs are assumed (see Fig. 9), PLA cups become better than PP cups for climate change, allowing 12% reduction (with LUC impact) or 32% reduction (without LUC). The fossil fuel resource of PLA cups become 36% lower than that of PP cups. On the other hand, PLA still performs worse than PP in the

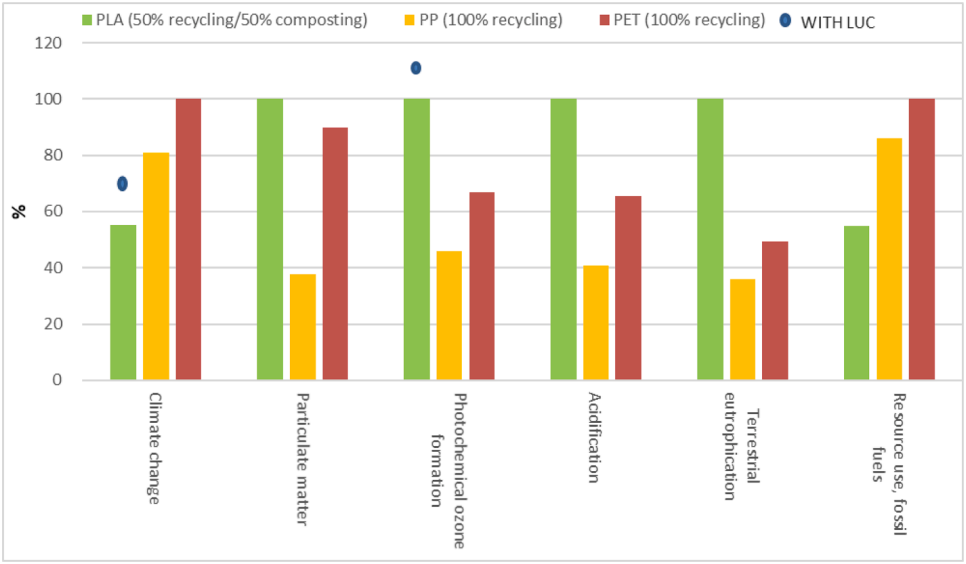

Fig. 9. Comparison of cradle-to-grave environmental impact for 1000 single-use cups by assuming the EoL of PLA: 50% recycling and 50% composting and the EoL of PP and PET: 100% recycling (highest impact as 100%). Dots marking the impact of PLA with LUC.

other four impact categories, where the EoL share was negligible (see Fig. 4).

4.1.2.2. Impact of a possible future renewable electricity use for PLA production. The material manufacturing phase, including lactic acid and PLA polymer production, is responsible for a large part of the impact of PLA cups (see Fig. 4). The consumption of electricity causes an important part of the impact of monomer and polymer production (see Table 5). Given the expected decarbonization of electricity in the future, it is important to understand how much the current environmental footprint of PLA could be affected by using more renewable electricity in the material manufacturing phase. For the PLA from the US, we assume that 100% renewable electricity from photovoltaics could be supplied for lactic acid and PLA polymer production processes. The percentage of the impact caused by electricity for PLA from US corn was retrieved from Ecoinvent 3.3 (Polylactide, granulate {GLO}| production | Alloc Def). Such a dataset refers to the older data of PLA processing compared to the one used in this study (which was not possible to modify because provided as aggregated inventory data). The 33% PLA from Thailand was instead not modified because 1) the breakdown at the activity level was not available (aggregated inventory data was provided) and 2) the electricity is already partly produced from renewables (burning bagasse) and a credit was accounted for the surplus supplied to the grid. This is taken as an extreme to show the extent to which a different electricity source could change the results and has some limitations (for example, the environmental impact of the energy storage system which is needed for an intermittent renewable is not taken into account).

By varying the electricity source, PLA cups become the most favorable option in terms of climate change impact, allowing a 25% reduction of GHG emissions and a 58% reduction in fossil resource use compared to PP cups. If LUC is taken into account, the GHG emission reduction would still be 10%. In the remaining four impact categories, PLA is still less favorable compared to petrochemical PP (see details in Appendix). Nevertheless, such comparison should be read as indicative only and incomplete, as the possible developments in the petrochemical plastics sector are not accounted for. In the last twenty years, the impact of petrochemical polymers has also shown significant improvements in environmental footprints by energy and material efficiency measures and emission controls (van der Harst et al., 2014). Unfortunately, the PlasticsEurope's eco-profiles are not transparent enough to allow the reconstruction of future impacts similarly to what done for PLA in this study.

4.2. Comparing with similar LCAs reported in the literature

The conducted LCA shows that the impact of PLA cups is dominated by the production of lactic acid and PLA. This is in line with the findings of the studies previously published (Binder and Woods, 2009; Potting and van der Harst, 2015), which as well identified the conversion of biomass to PLA (and in particular to lactic acid) as the most environmentally impacting life cycle stage.

In contrast to previous research (Potting and van der Harst, 2015; van der Harst et al., 2014), our analysis shows that composting is not a worse option than incineration and that the best option is highly variable depending on the impact category assessed. Sometimes, this difference originates from different scopes and how the biogenic carbon is accounted for in the entire life cycle (see Section 3.1.1.2). In other cases, the difference is led by the multiple impact categories that bring different insights. For example, it was found by this study that industrial composting is favored over incineration for ionizing radiations but has a higher impact on marine eutrophication, which was not discovered in earlier studies. Moreover, the heterogeneity of the materials and the contamination assumed in the PLA waste leads to a much close-to-reality assessment than a simplified assumption of pure material treated in the EoL phase.

4.3. Limitations of the study

The main limitations of the presented study are reflected as the following:

1 For a multifunctional process, if there is more than one suitable allocation method, ISO 14,044:2006 requires a sensitivity analysis. Unfortunately, it was not possible to perform such an analysis for some unit processes within the production processes of the three polymers. The main reason was that the LCI datasets for the three polymers were available only at aggregated level. Nevertheless, most products requiring allocation were by-products (e.g., gluten meal for corn-based PLA) with lower economic-physical significance than the products focused on (e.g. starch used for lactic acid fermentation) and therefore less affected by allocation choices. On the other hand, the allocation may have significantly affected the results if substitution was applied. For example, it would have been interesting to assess the variations in the PLA impact by using allocation instead of substitution for the electricity surplus of the biomass CHP burning bagasse from sugar cane.

2 Due to missing/unreliable data for PP and PET for the other seven impact categories, it was not possible to compare the total environmental footprints. To allow a broader and reliable comparison between bio-based and petrochemical materials, more transparent datasets in the public domain are needed for the production of PP and PET. This would allow a more comprehensive comparison on a fair and symmetric basis (Carus et al., 2019). Moreover, it was assumed that PP and PET cups are produced in Europe by using polymers produced in the EU. For a more precise assessment, it would be important to investigate 1) the percentage of petrochemical cups that are produced outside the EU and 2) the availability of LCI data for PP and PET produced in these other countries.

3 At present, the waste management system in the EU is not ready for composting and recycling of PLA on a large scale. PLA is certified as a compostable material and can biodegrade in commercial-scale composting facilities. However, in many commercial composting facilities, the conditions required for the biodegradation of PLA are not met. One of the main reasons is that PLA does not fully biodegrade because large industrial composting facilities run on a short-term batch process for 3–5 weeks while for PLA, it requires up to 10 weeks to reach a satisfactory disintegration and biodegradation (63–97%)(European Commission, 2019). PLA is also technically recyclable but the recycling of PLA does not take place because the current infrastructure is facing too many hurdles (Hottle et al., 2017). To recycle PLA, the PLA stream needs to be sorted out from either mixed plastics waste, or from the waste polyesters streams which are currently often recycled on a large scale (e.g. PET) (Hottle et al., 2017). At the current state, given the limitations of the sorting techniques, mechanical recycling of PLA is unlikely to be realized in the near future. Moreover, given the deficient waste collection labeling presently in the EU, PLA creates cross-contamination problems for the established mechanical recycling system of conventional plastics (European Commission, 2018a). Other critical assumptions of the EoL modeling include the marginal energy technologies, the efficiencies of the EoL treatment plants, the quantity of rejects and carbon biodegradation of PLA in industrial composting, and the carbon biodegradation of PLA during landfill.

4 There is growing concern about plastic littering. Nevertheless, this impact is not modeled in this LCA neither for PLA nor for petrochemical plastics due to missing emission data (i.e. quantity of littered plastic cups for both PLA and petrochemical polymers) and a lack of impact assessment models. The emissions (type, size and volume of the macro, micro or nanoplastics), the hazards and toxicities, the effect and the disintegration/degradation pathways are not well understood yet. For these reasons, although this environmental issue is central to the public discussion about plastic items,

this is not addressed in the current environmental impact assessment but should be included in the future assessment when the method and data are available.

5. Conclusions

In this study, the cradle-to-grave environmental footprint of PLA cups was assessed for thirteen impact categories. The lactic acid fermentation and PLA polymer production processes were identified as the main source of environmental impacts that are responsible for about 60% of climate change, 43% of ionization radiation, 40% of acidification, 45% of freshwater eutrophication, 53% of resource use of minerals and 77% of resource use of fossil fuels. A lion's share of these environmental impacts are caused by the process heat and electricity consumption. Biomass production (cultivation and harvesting) contributes substantially (more than 50%) to three impact categories: namely marine eutrophication, land use and water use. However, it does not have a determining role in the remaining ten impact categories. The impact of LUC is negligible overall but significant for climate change and photochemical ozone formation, where it represents 13% and 10% of the cradle-to-grave impact, respectively.

It is not possible to assess all 13 impact categories for petrochemical PP and PET due to large variability of the impacts reported by literature and databases that cannot be justified straightforwardly. The comparison of PLA cups with its petrochemical PP and PET cups was limited to six of impact categories. Nevertheless, the current study confirmed that, even by taking into account LUC and potential contaminations in the waste, PLA cups offer better performances in terms of climate change (22% lower impact) compared to PET cups and offer less fossil resource depletion compared to both PET cups (52% lower) and PP cups (41% lower). However, PLA cups have significantly higher impacts compared to PET and PP cups for photochemical ozone formation, acidification and terrestrial eutrophication. PP cups have better performance than PET cups in all the six impact categories compared.

We conclude that current PLA cups can be considered environmentally better than PET cups, but still not yet outperform PP cups. The main reason can be found in the low weight of the PP cups due to low density of the material. Two tipping points could be a better waste management of PLA by only composting and recycling, and the use of renewable electricity in the lactic acid production phase. Together, these measures could offer PLA to be environmentally preferable also compared to PP cups even including the impacts of LUC. Although, the full implementation of recycling and composting PLA on a large scale is still limited by many technical and governance hurdles.

The comparison was limited to a much more restricted spectrum of impact categories than the comprehensive one recommended by the PEFCR guidance. Using more transparent datasets would allow to increase the completeness of the comparison e.g. by implementing more consistent assumptions for both bio-based and petrochemical materials. To broader the analysis to more environmental impact categories, we recommend therefore that more consistent LCI data are urgently needed for the petrochemical counterparts, especially in the context of benchmark establishment for the future policies of biobased economy.

CRediT authorship contribution statement

Christian Moretti: Conceptualization, Methodology, Software, Data curation, Formal analysis, Investigation, Visualization, Writing – original draft, Writing – review & editing. **Lorie Hamelin:** Methodology, Data curation, Formal analysis, Investigation, Visualization, Writing – original draft, Writing – review & editing. **Line Geest Jakobsen:** Methodology, Data curtion, Formal analysis, Investigation, Visualization, Writing – original draft, Writing – review & editing. **Martin H Junginger:** Conceptualization, Data curation, Supervision, Funding acquisition, Resources, Writing – review & editing. **Maria Magnea Steingrimsdottir:** Methodology, Data curation, Formal analysis, Investigation. **Linda Høibye:** Conceptualization, Project administration, Funding acquisition, Resources, Supervision, Writing – original draft. **Li Shen:** Conceptualization, Methodology, Project administration, Funding acquisition, Resources, Supervision, Writing – original draft, Writing – review & editing.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Supplementary materials

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