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RESEARCH ARTICLE

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Sustainability of bio-based polyethylene

The influence of biomass sourcing and end-of-life

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Abstract

Bio-based polymers may present a sustainable, circular way to reduce the environmental impact of plastics because they are produced from biomass that absorbs CO_2 during its growth. However, sourcing (type of biomass used and cultivation location), production, and end-of-life affect the environmental impact of bio-based plastics. We assessed the effect of sourcing and end-of-life options on the environmental impact of bio-based high-density polyethylene (bio-HDPE) in 31 sourcing scenarios and five end-of-life options. Our study found that careful consideration of biomass sourcing (biomass type and production location) and end-of-life is needed to optimize the environmental impact of bio-based plastics. If these aspects are not considered, the environmental impact of bio-HDPE may exceed that of its petrochemical-based counterpart. The direct availability of fermentable sugars indicated a lower environmental impact. The production location affected the resources needed for biomass cultivation and the environmental impact of processing due to the energy mix. Recently published guidelines do not allow biogenic carbon to be accounted for during the production stage, but only upon the incineration of the plastic. Our results show that this way of attributing biogenic carbon results in an apparent disadvantage for bio-based plastics compared to petrochemical-based plastics. Furthermore, it disadvantaged mechanical recycling of bio-based plastics compared to incineration, a result out of line with circular economy principles.

KEYWORDS
bio-based plastic, biogenic carbon, circular economy, high-density polyethylene, industrial ecology, lifecycle assessment

1 | INTRODUCTION

Plastics are so ubiquitous in modern society that the time we live in may well be looked back upon as the "plastic age" (Thompson et al., 2009). Over 5 billion metric tonnes of plastics have been produced since their commercial introduction in the 1950s (Statista, 2022). However, plastics are associated with significant environmental problems, such as greenhouse gas emissions, plastic pollution (Geyer et al., 2017), and fossil fuel use (Shen et al., 2020). Bio-based plastics have been presented as a potential sustainable and circular solution to reduce the environmental impact of

plastics because they are based (at least in part) on biomass (International Standards Organisation [ISO], 2015) that absorbs carbon dioxide (CO_2) during its growth. However, bio-based plastics are not inherently sustainable or circular. Producing bio-based plastics involves the cultivation of biomass and often an extensive chemical conversion, which may lead to higher environmental impacts than petrochemical-based plastics (Walker & Rothman, 2020). Furthermore, while the molecular decomposition of bio-based plastics can be considered a circular loop (as the CO_2 they emit has been previously derived from the atmosphere (Kawashima et al., 2019), they can still contribute to plastic waste issues since they are often not biodegradable in natural environments, so the recovery of bio-based plastics at end-of-life needs to be guaranteed (Ritzen et al., 2023).

For the transition to a circular economy, both the production and end-of-life of bio-based plastics need to be addressed. Lifecycle assessment (LCA) plays a pivotal role in this process, as it can be used to compare different scenarios and find the most sustainable options. However, the methodology used in LCA also affects its outcomes, so the LCA methodology also needs to align with circular economy principles.

Published LCAs show high uncertainty and variations in outcomes for the environmental impacts of bio-based plastics. Walker and Rothman (2020) compared 50 LCAs of bio-based plastics and found variations of over 1000% for the environmental impact of the same bio-based polymer. For instance, reported global warming potential (GWP100) values for the production of 1 kg polylactic acid ranged from 0.1 to 3.1 kg CO_2-eq. They noted three important reasons for these variations: methodological inconsistencies, feedstock source, and processing. Bishop et al. (2021) analyzed the LCA methodologies of 44 LCAs for bio-based plastics. The variations in LCA outcomes were attributed to different system boundaries and different strategies for land-use change, biogenic carbon, and allocation. They also highlighted the lack of reliable data for the chemical conversion processes involved in producing bio-based monomers as a source of variations.

The effect of feedstock sourcing for bio-based plastics has also been reported in the literature by the comparison of different biomass types cultivated in different locations. Polyethylene terephthalate (PET) with wood-based terephthalic acid (TA) produced in the United States was found to have a lower environmental impact compared to PET based on wheat or corn (Akanuma et al., 2014), or corn stover (a by-product of corn grain production) (Chen et al., 2016). For the other building block of bio-based PET (monoethylene glycol), corn resulted in lower GWP100 emissions than switchgrass or wheat straw (Chen et al., 2016). Wheat-based PET yielded lower environmental impacts than sugar beet-based PET from Germany (García-Velásquez & van der Meer, 2022), or sugarcane-based PET from Brazil (Gursel et al., 2021). However, Belboom and Léonard (2016) reported a negligible 3% difference between wheat-based and sugar beet-based HDPE.

For ethanol-based polymers such as polyethylene (PE) and PET, biofuel LCAs also provide an indication of the environmental impact of different feedstock sourcing scenarios. Muñoz et al. (2014) compared six sourcing scenarios for bio-ethanol: maize grain or maize stover from the United States, sugar beet or wheat from France, and sugarcane from two regions in Brazil. Sugar beet-based ethanol from France resulted in the lowest GWP100 due to the high yield from sugar beets. Changing the feedstock for bio-ethanol from edible crops to agricultural by-products could lower GWP100 but was not beneficial for human and ecosystem health (Wietschel et al., 2021).

Today, the range of potential feedstocks for bio-based plastics is very limited. However, bio-based plastics can be produced from a wide range of biomass types in a variety of locations. This offers the opportunity to source biomass more sustainably. To the best of our knowledge, there are no publications about the effect of both biomass type and production location for bio-based polymers, as most of the aforementioned studies focus on limited scenarios and cannot be compared directly due to methodological inconsistencies.

In this article, we study the effect of biomass type, production location, and end-of-life on the environmental impact of bio-based high-density polyethylene (bio-HDPE) used in Western Europe. PE accounts for 30% of the entire plastics market (Statista, 2022). Bio-based PE is currently produced from sugarcane in Brazil and makes up 14% of the bio-based plastics market (Skoczinski et al., 2023). Bio-based PE is a so-called "drop-in" bio-based polymer, which means it is chemically identical to petrochemical-based PE (Carus et al., 2017). HDPE is a type of PE that has little branching in the polymer chain resulting in a high strength-to-density ratio. Thirty-one scenarios for bio-HDPE production, covering five types of biomass and 11 locations, are analyzed and compared to petrochemical-based HDPE (petro-HDPE). Additionally, we consider five end-of-life options for all aforementioned scenarios, as well as the effect of biogenic carbon accounting on LCA outcomes. By integrating these factors into a robust LCA framework, we provide a deeper understanding of the environmental performance of bio-based HDPE, thereby supporting more informed decision-making for sustainable plastic production and waste management and supporting the transition to a circular economy with bio-based plastics.

2 | METHODS

2.1 | Goal and scope definition

The environmental impact of bio-HDPE from different biomass resources in various locations was compared to that of petro-HDPE in an LCA. Environmental impacts were calculated using the LCA software Activity Browser (Steubing et al., 2020), with the Ecoinvent V3.9 background database (Wernet et al., 2016). The ReCiPe 2016 impact categories were used for the lifecycle impact assessment (LCIA). ReCiPe midpoint impact categories were used to study the environmental impact of the sourcing scenarios and are presented in the article. Their relative contribution to ReCiPe endpoint impact was used to select the most relevant midpoint impact categories by comparing their contribution to the total environmental impact

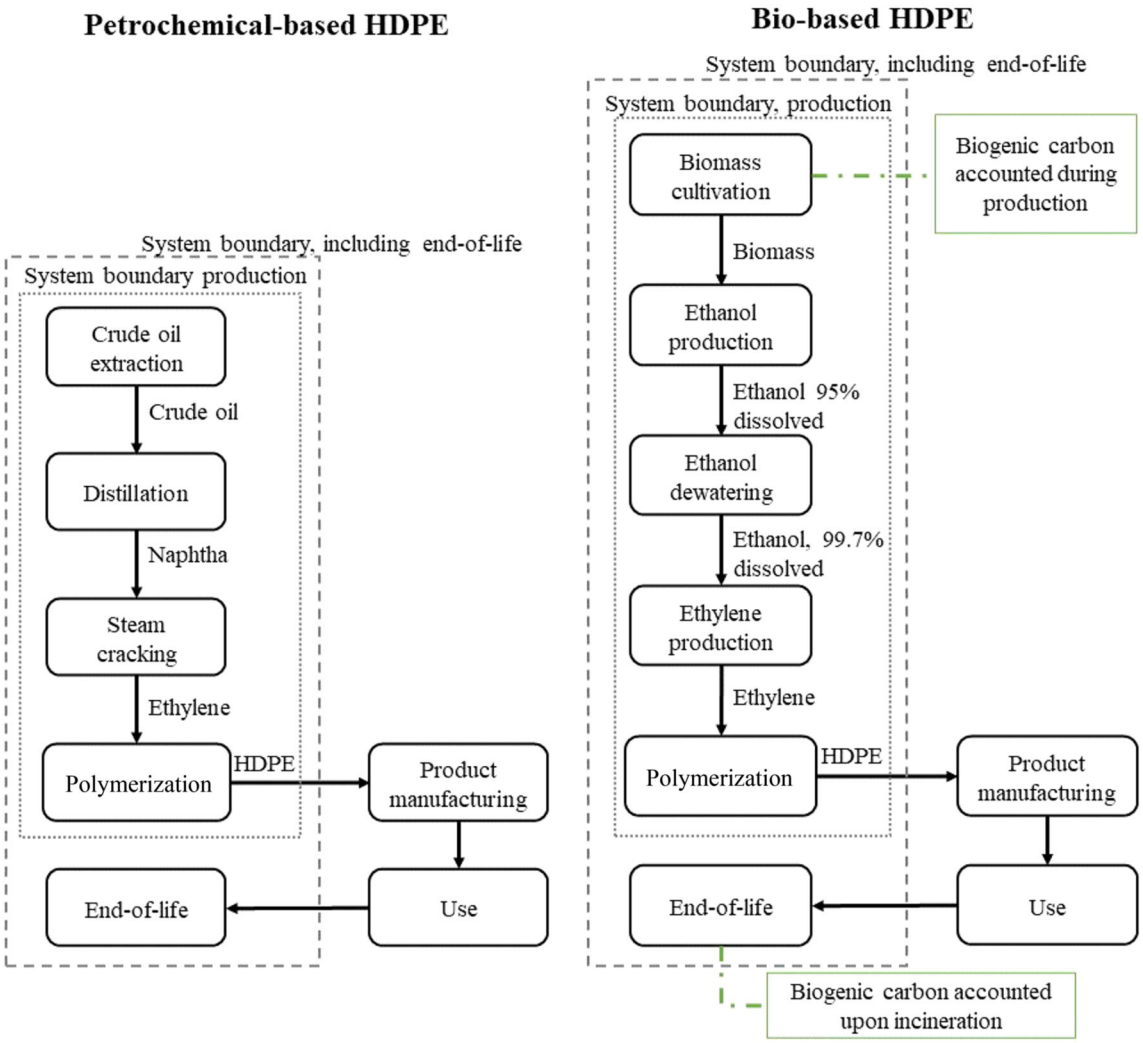

FIGURE 1 System diagram for the production of petrochemical-based high-density polyethylene (petro-HDPE) and bio-based HDPE (bio-HDPE), indicating two methods of accounting for biogenic carbon. In the first method (BC1), biogenic carbon was only accounted for when the CO_2 in the bio-based polymer is returned to the atmosphere. In second method (BC2), biogenic carbon was accounted for when it is extracted from the atmosphere, during biomass growth.

of bio-based polymer production. The JRC guidelines (JRC, 2021) dictate that the environmental impact categories discussed must account for at least 80% of the total endpoint impact. Throughout the article, we discuss the most relevant midpoint environmental impacts.

Figure 1 displays the system diagrams for petro-HDPE and bio-HDPE. The cradle-to-grave (without the use phase) environmental impact of 1 kg bio-HDPE in granulate form based on various biomass sourcing scenarios was compared to 1 kg of petro-HDPE in granulate form. Bio-HDPE was produced in the location of biomass cultivation, transported to the port of Antwerp, and used and disposed of in Europe. Since bio-HDPE is a drop-in bio-based polymer, we assumed the same manufacturing methods and use scenarios; hence, they are outside the system boundary. An overview of transport distances can be found in the supplementary information (table S1.5).

2.2 | Lifecycle inventory analysis

2.2.1 | Petrochemical-based and bio-based high-density polyethylene production

The petro-HDPE production process from the background database was used unaltered, distinguishing between a European scenario (petro-RER) and a scenario outside Europe (petro-RoW). Bio-HDPE scenarios were established based on the availability of ethanol fermentation data, resulting in five types of biomass: sugarcane (SC), maize (M), sugar beet (SB), potatoes (P), and wood (WO).

TABLE 1 Production scenario abbreviations and the corresponding biomass types and locations.

Scenario abbreviation	Resource	Location	Scenario abbreviation	Resource	Location
Petro-RoW	Crude oil	Outside of Europe	SB-CH	Sugar beet	Switzerland
Petro-RER	Crude oil	Europe	SB-DE	Sugar beet	Germany
SC-BR	Sugarcane	Brazil	SB-FR	Sugar beet	France
SC-CN[a]	Sugarcane	China	SB-SE[a]	Sugar beet	Sweden
SC-CO	Sugarcane	Colombia	P-CA	Potato	Canada
SC-IN	Sugarcane	India	P-CN	Potato	China
SC-US[a]	Sugarcane	United States	P-IN	Potato	India
M-BR	Maize	Brazil	P-US	Potato	United States
M-CA	Maize	Canada	P-CH	Potato	Switzerland
M-CN[a]	Maize	China	P-DE[a]	Potato	Germany
M-IN	Maize	India	P-FR[a]	Potato	France
M-US	Maize	United States	P-SE	Potato	Sweden
M-ZA	Maize	South Africa	WO-CA	Wood	Canada
M-CH	Maize	Switzerland	WO-CH	Wood	Switzerland
M-DE[a]	Maize	Germany	WO-DE	Wood	Germany
M-FR[a]	Maize	France	WO-SE	Wood	Sweden
SB-US	Sugar beet	United States			

[a]Scenarios from the Global Feed LCA database.

Next, all potential cultivation locations for these biomass types were found in the background database, yielding a total of 22 sourcing scenarios spanning 11 locations. The biomass cultivation data were directly used from the background database as they accurately represent the emissions of growing the biomass in that location. Additional scenarios for biomass cultivation were found in the Global Feed LCA database (The Global Feed LCA Institute, n.d.). Only locations for which scenarios from the background database were already found were included, yielding nine additional scenarios. Table 1 provides an overview of the resulting 31 production scenarios and the abbreviations used throughout the article.

For the conversion of biomass into ethanol, localized processes were not always available and the energy mix and origin of other resources for ethanol conversion were adjusted to the country of biomass cultivation. Although ethanol from (for example) sugarcane is not chemically different from ethanol based on maize, the production process of ethanol from biomass is specific to the biomass type. This means that they have different inputs and can have various by-products that need to be allocated. In some cases, such as the by-product bagasse from sugarcane ethanol production, the system boundary was expanded to include the replaced energy production from this by-product. If system boundary expansion was not practical, environmental impacts were distributed between main and byproducts using economic allocation. The amount of biomass needed to produce 1 kg of bio-HDPE therefore also varied: 18.6–23.4 kg for sugarcane, 6.7 kg for maize, 13.6 kg for sugar beet, 29.6 kg for potatoes, and 7.9 kg for wood. Brief descriptions of the various ethanol production processes can be found in the supplementary information (S1.2).

Ethylene conversion was based on industry data reported in Ita-nagy et al. (2020), adjusting the energy mix and resources for the location of production. Ethylene is produced from ethanol by catalytic dehydration, which is an acid-catalyzed, endothermic reaction (i.e., requiring energy) (Mohsenzadeh et al., 2017). For the polymerization step, we used the process for (petrochemical) HDPE production, replacing the ethylene with bio-based ethylene and adjusting for production location by changing the energy mix and the origin of other resources where possible. A complete lifecycle inventory can be found in the supplementary information (table S1.2-S1.4).

2.2.2 | Biogenic carbon accounting

Bio-based polymers act as temporary carbon storage until their biogenic carbon is reintroduced to the atmosphere, for example, through incineration, sometimes referred to as delayed emissions. In 2021, the European Joint Research Commission (JRC) published a methodology for the LCA of plastics from alternative feedstocks (including biomass) (Joint Research Commission [JRC], 2021). According to this methodology, biogenic carbon storage shall not be included in the LCI or LCIA of bio-based plastics, in accordance with the ISO 14067 standard (ISO, 2018). Instead, biogenic carbon shall be accounted for in a separate analysis to avoid double counting. However, the CO_2 that is emitted upon the incineration of the plastic is biogenic and does not contribute to the global warming potential (Joint Research Commission [JRC], 2021).

To understand the effect of this approach on biogenic carbon accounting, we also considered an alternative approach, where biogenic carbon is accounted for during production and CO_2 emitted upon incineration contributes to fossil CO_2 emissions. This resulted in two options for the biogenic carbon account. In the first method (BC1), biogenic carbon is only accounted for when the CO_2 in the bio-based polymer is returned to the atmosphere. In the second method (BC2), biogenic carbon is accounted for when it is extracted from the atmosphere, during biomass growth. Only the biogenic carbon that was converted into ethanol is taken into account. The calculations used to determine the biogenic carbon in bio-HDPE can be found in the supplementary information (S1.1). From these calculations, we determined the biogenic carbon stored in 1 kg of bio-HDPE to be 3.14 kg, which is in line with existing literature (Tsiropoulos et al., 2015).

2.2.3 | End-of-life options

There are currently three realistic end-of-life options for polyethylene: landfilling, incineration, and mechanical recycling (RameshKumar et al., 2020). In theory, there are also chemical recycling options for HDPE, but these currently do not exist at scale and there is no industry data for these processes (Davidson et al., 2021; Spierling et al., 2020). Therefore, five end-of-life scenarios were considered: mechanical recycling, incineration (with or without energy recovery), and landfilling (sanitary or unsanitary).

Mechanical recycling was modeled using the process "polyethylene production, high density, granulate, recycled." The replacement potential of recycled HDPE was implemented according to the following equation (Huysveld et al., 2022):

$$\text{Impact}_{\text{with replacement}} = \text{Impact}_{\text{reprocessing}} - A \cdot B \cdot \text{Impact}_{\text{virgin fossil production.}}$$

A is the technical substitution ratio, that is, the fraction of HDPE products that can be produced from recycled HDPE, and was set at 0.5 (Viau et al., 2020). B is the avoided virgin production, which is 0.95 kg virgin HDPE for 1 kg recycled HDPE (Wernet et al., 2016). Hence, mechanical recycling of 1 kg HDPE was assumed to avoid 0.475 kg virgin HDPE production. Recycled HDPE was assumed to replace virgin petro-HDPE since bio-based plastics only occupied 1% of the plastics market at the time of writing (Skoczinski et al., 2023).

Incineration without energy recovery was based on the process "treatment of waste polyethylene, municipal incineration." To adapt the incineration process to include energy recovery, the energy recovered from the incineration of 1 kg of HDPE was subtracted from the impact of the incineration of HDPE. The lower heating value of HDPE is 42.2 MJ kg^{-1}, or 11.67 kWh kg^{-1} (Kannan et al., 2013). For electricity recovery efficiency, a value of 22% was chosen, which is representative of European incineration facilities (Gradus et al., 2017; Merrild et al., 2008), resulting in 2.57 kWh electricity generation from 1 kg HDPE, in line with prior literature (Belboom & Léonard, 2016).

For landfilling, two scenarios were compared: a sanitary landfill and an unsanitary landfill. A sanitary landfill is lined to isolate waste from its environment (Shen et al., 2020). In an unsanitary landfill, leaking of waste into the environment (soil, water, and air) is not prevented. The degradation of polyethylene in landfill conditions is 1% in 100 years (Wernet et al., 2016), so we assumed no CO_2 emissions from landfilled HDPE.

2.2.4 | Land-use change emissions

Direct land use change (LUC) is the direct repurposing of land for the cultivation of crops, for example, the change of forest land into agricultural land or the repurposing of agricultural land for feed crops to agricultural land for crops for bio-based plastics. The ecoinvent and Global Feed LCA database both include direct LUC emission data. Indirect LUC occurs when an LUC inside the system boundary leads to an LUC outside of the system boundary. At the time of writing, there is no standard method to measure indirect LUC, and it is therefore not included in this analysis (JRC, 2021; Rosa, 2018).

2.3 | Sensitivity analysis

Data for the conversion of ethanol into ethylene are scarce and often varying (Bishop et al., 2021), and reported efficiencies and material requirements may be too optimistic or may improve in the future. Therefore, we studied the sensitivity to the two largest contributors to the environmental impact of ethylene production: ethanol conversion efficiency and electricity needed, by a +10% or −10% increment.

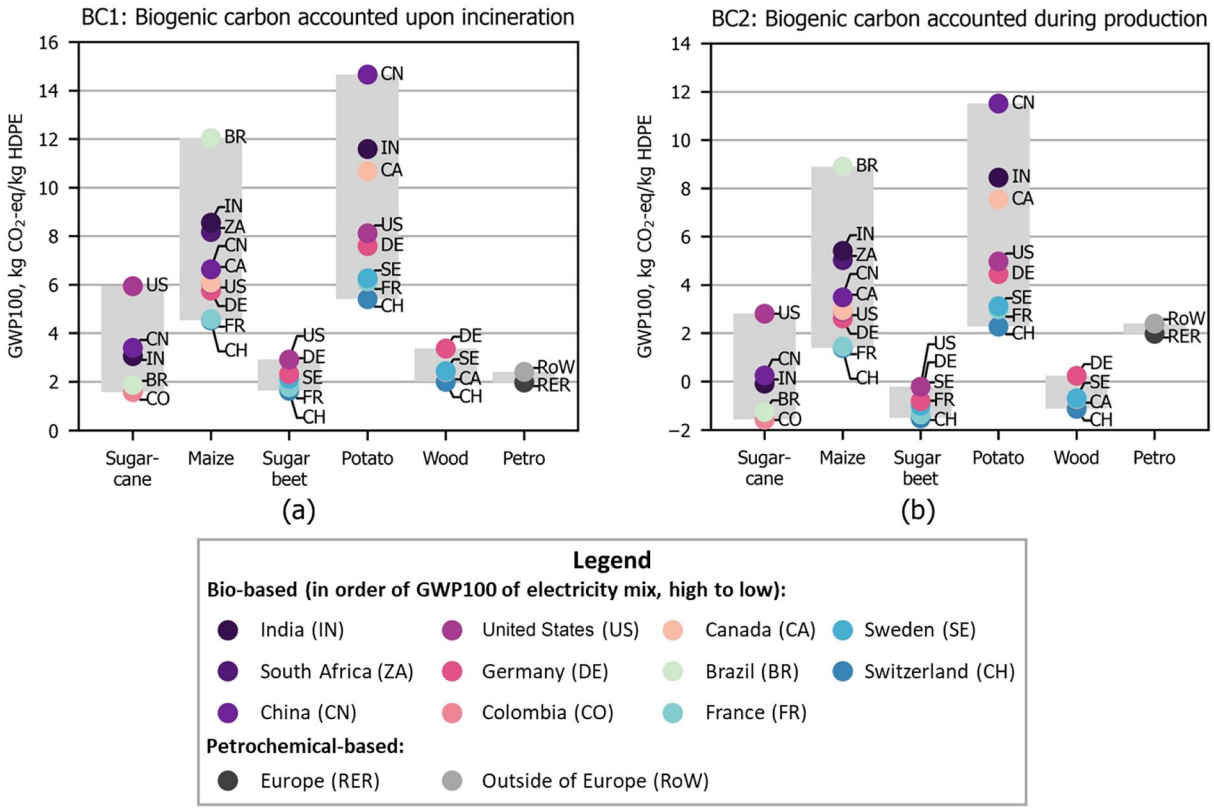

FIGURE 2 Comparison of the global warming potential (GWP100) of 1 kg petrochemical-based high-density polyethylene (petro-HDPE) and bio-based HDPE (bio-HDPE) from various resources. (a) Not accounting for biogenic carbon uptake during production, and (b) accounting for biogenic carbon uptake during production. The order and coloring of the entries in the legend correspond with the GWP100 of 1 kWh of electricity at the location. Underlying data can be found in the supplementary information (S3).

3 | RESULTS

3.1 | Production of HDPE from different resources in different locations

Eighteen ReCiPe Midpoint impact categories were calculated for each scenario. In this article, we focus on the midpoint impact categories that contribute the most to the total endpoint environmental impact: GWP100, land use, water use, particulate matter formation, and fossil fuel depletion. Combined, these impact categories account for 89%–97% of the total endpoint environmental impact of the sourcing scenarios, exceeding the 80% suggested by the JRC guidelines (JRC, 2021). The other results can be found in the supplementary information (figure S1.1-S1.2, table S1.6-S1.10, and the underlying data in S2). Of these six impact categories, we present the midpoint results.

3.1.1 | Global warming potential (GWP100)

Figure 2 displays the results for GWP100 of bio- and petro-HDPE, both without accounting for biogenic carbon uptake during production (in accordance with JRC (2021)) (Figure 2a) and with biogenic carbon uptake (Figure 2b). When excluding biogenic carbon uptake during production, four bio-HDPE scenarios resulted in lower GWP100 than petro-HDPE from Europe: sugarcane in Brazil (SC-BR) and Colombia (SC-CO) and sugar beets in Switzerland (SB-CH) and France (SB-FR). Compared to petro-HDPE from outside Europe, bio-HDPE based on sugar beets from Germany (SB-DE) and Sweden (SB-SE), and wood from Switzerland (WO-CH) also resulted in a lower GWP100. The GWP100 of bio-HDPE ranged from 1.6 kg CO_2-eq for sugar beets in Sweden to 14.7 kg CO_2-eq for potatoes in China. Figure 2b shows the GWP100 emissions of bio-HDPE and petro-HDPE when accounting for biogenic carbon during production. If biogenic carbon was accounted for in the production stage, 11 additional bio-HDPE production scenarios yielded a lower GWP100 than petro-HDPE from Europe: all scenarios produced from sugarcane, sugar beet, and wood, as well as maize from Switzerland and France and potatoes from Switzerland.

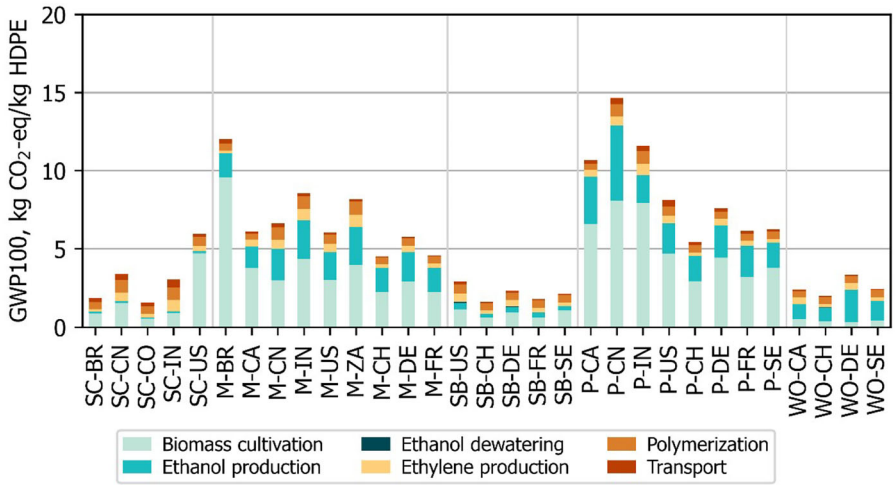

FIGURE 3 Process step contributions to the global warming potential (GWP100) of 1 kg bio-based high-density polyethylene (bio-HDPE) production, without accounting for biogenic carbon. Underlying data can be found in the supplementary information (S4). BR, Brazil; CA, Canada; CH, Switzerland; CN, China; CO, Colombia; DE, Germany; FR, France; IN, India; SE, Sweden; US, United States; ZA, South Africa.

There was a clear distinction between biomass types, where maize and potatoes resulted in a relatively high GWP100 compared to sugarcane, sugar beet, and wood. Maize- and potato-based bio-HDPE also yielded a broader variation in outcome between countries. These differences were primarily caused by the significant variation in the GWP100 of biomass cultivation (see Figure 3). Furthermore, GWP100 due to ethanol production varied: for instance, GWP100 due to ethanol production from maize and potatoes was 8.5 to 37 times higher than that of ethanol production from sugarcane, attributed to the availability of fermentable sugars. Sugar-based biomass, such as sugarcane and sugar beet, contains high amounts of sugars directly available for fermentation into ethanol (Lin & Tanaka, 2006; Muñoz et al., 2014). Maize and potatoes are starch-based materials that require enzymatic hydrolyzation into fermentable sugars, which increases the environmental impact of ethanol production (Lin & Tanaka, 2006). The environmental impact of the ethylene production and polymerization stages also varied between locations, because these processes used the local electricity mix.

The most important contributors to GWP100 were electricity, heat production, and biomass cultivation. The large contribution of electricity (accounting for up to 58% of the GWP, as seen in table S6) explains the correlation between the environmental impact of the electricity mix and the impact of the bio-HDPE. However, this correlation did not always hold. For example, the GWP100 due to bio-HDPE production from sugar beet from Switzerland was lower than that of sugar beet in France, even though the Swiss electricity mix had a higher environmental impact compared to the French electricity mix. In this case, the difference could be attributed to emissions associated with the cultivation of these crops in these locations. Similarly, the sources of heat and the efficiency of heat production also depend on the location. Transport accounted for less than 10% of the GWP100 in all but three scenarios: SC-BR, WO-CA, and WO-SE. LUC emissions accounted for less than 5% of GWP100 in all but two scenarios: P-IN (7.4%) and M-BR (16.2%).

While wood is cellulose-based biomass, which also needs additional conversion steps to yield fermentable sugars (Lin & Tanaka, 2006), it yielded relatively low GWP100, as shown in Figure 3. This could be attributed to the low maintenance of wood cultivation: Wood was not irrigated or fertilized like the other biomass types. Furthermore, relatively low quantities of wood were needed to produce 1 kg of bio-HDPE: 7.9 kg. However, the GWP100 of ethanol production from wood was 6.6–10 times that of ethanol production from sugarcane, potentially due to the additional processing steps in converting the cellulose in wood into fermentable sugars.

Some locations consistently appeared at the higher or lower end of the GWP100 results for specific biomass types. For instance, India ranked as the location with the highest GWP100 for sugarcane, maize, and potato. This was due to the environmental impact of the electricity mix of the production country, which had the highest GWP100 for 1 kWh out of all locations considered.

Table 2 compares the GWP100 of this LCIA to those reported in previous literature for each biomass type. Despite the difficulty of directly comparing bio-based plastic LCA outcomes indicated in the introduction, we note that our outcomes are largely in line with preexisting LCAs of bio-based HDPE. An exception was the Braskem LCA, which reported values twice as low as the ones in this study. However, in the Braskem LCA, −1.10 kg CO_2-eq was attributed to LUC credits, whereas in our analysis, LUC emissions were positive: 0.00097 kg CO_2-eq. Unfortunately, the lifecycle inventory for the Braskem LCA is not publicly available so the rationale for negative LUC emissions could not be derived.

TABLE 2 Comparison of the outcomes of this study with existing work.

Biomass type	This work	Other works
Sugarcane in Brazil (without biogenic carbon)	1.87 kg CO_2-eq	2.5–4.0 kg CO_2-eq (Tsiropoulos et al., 2015) 0.3 kg CO_2-eq (Liptow & Tillman, 2012) 2 kg CO_2-eq (Hermann et al., 2010) 1.3–3.6 kg CO_2-eq (Suarez et al., 2023) 1.4 kg CO_2-eq (Ita-nagy et al., 2020) 1.9 kg CO_2-eq (Kikuchi et al., 2017)
Sugarcane in Brazil (with biogenic carbon)	−1.56 kg CO_2-eq	−3.09 kg CO_2-eq (Braskem, 2022)
Sugar beet (without biogenic carbon)	1.62–2.92 kg CO_2-eq	In Belgium: 2.7 kg CO_2-eq (Belboom & Léonard, 2016)

3.1.2 | Land use

In contrast with petrochemical-based plastics, bio-based plastics require land to cultivate biomass. Figure 4a shows the LCIA results for land use. Petro-HDPE resulted in relatively low land use (0.01–0.02 m^2-year) compared to bio-HDPE. Agricultural land use for petro-HDPE was primarily attributed to wood cultivation for biofuels used in processing and the construction of pipelines and onshore wells. Land use of bio-HDPE was 11 to 1500 times as high, primarily due to biomass cultivation. The land needed depended on biomass type, location, and the amount of biomass needed to produce the ethanol required to produce 1 kg of HDPE. Sugarcane and sugar beet required relatively little agricultural land compared to maize, potato, and wood. Land use of the same biomass type also differed between locations. For instance, agricultural land needed to grow potatoes varied by a factor of 2.7 between India and China, potentially reflecting the local soil suitability and climate.

3.1.3 | Water use

The biomass used for bio-HDPE needs water to grow, either supplied passively by rain or actively through irrigation. Figure 4b shows the LCIA results for water use of bio-HDPE and petro-HDPE. Petro-HDPE production resulted in roughly 0.02 m^3 or 20 L of water use. The amount of water needed for irrigation during biomass cultivation depended on the type of crop and the climate at the location where it was grown. In the scenarios for wood, biomass was not watered, so water was only consumed during ethanol production, resulting in less than 100 L water use per kg bio-HDPE. For other biomass types, water use depended on location, over 700, 4050, 1300, and 2500 L for sugar beet, sugar cane, maize, and potatoes, respectively.

3.1.4 | Fossil fuel potential

Figure 4c shows the fossil fuel potential of bio- and petro-HDPE. When producing bio-HDPE from maize or potatoes, the location determined whether the fossil fuel potential was larger or smaller than petro-HDPE. All sugarcane-, sugar beet-, and wood-based bio-HDPE scenarios led to a lower fossil fuel potential than petro-HDPE. Other scenarios with a lower fossil fuel depletion compared to petro-HDPE were: Maize in Canada, China, the United States, Switzerland, Germany, and France, as well as potatoes in Switzerland, Germany, France, and Sweden. Similar to the GWP100 results, a relatively large spread in the outcomes for maize and potatoes was observed, with the primary contributors being biomass production and conversion into ethanol. Fossil fuel potential outcomes correlated with the environmental impact of the local electricity mix, with correlation values of > 0.9 for all biomass types except for potatoes (correlation = 0.58). This was expected since an energy mix more reliant on fossil fuels also has a higher environmental impact. Any activity that consumes energy, for example, treating biomass with agricultural machinery, harvesting it, and heating it to produce ethanol, also consumes more fossil fuels.

The most important contributor to fossil fuel potential varied per biomass type. For sugarcane-based HDPE, polymerization and transport were important contributors to fossil fuel potential. Fossil fuel depletion due to polymerization was not significantly higher compared to other biomass types, but since the total fossil fuel depletion of sugarcane-based HDPE was relatively low, it constituted a larger fraction of the total impact. Fossil fuel depletion due to transport was generally high because sugarcane is grown in regions far away from Europe. This effect was amplified by the relatively low overall fossil depletion of sugarcane-based HDPE. This was also the case for the polymerization of sugar beet-based HDPE, where polymerization constituted a significant fraction of fossil fuel depletion. However, since most of the sugar beet-based HDPE scenarios were produced in Europe, transport emissions were lower. For maize- and wood-based HDPE, ethanol production and/or biomass cultivation constituted the majority of fossil fuel depletion. Fossil fuel depletion from ethanol production could primarily be attributed to the heat used during the fer-

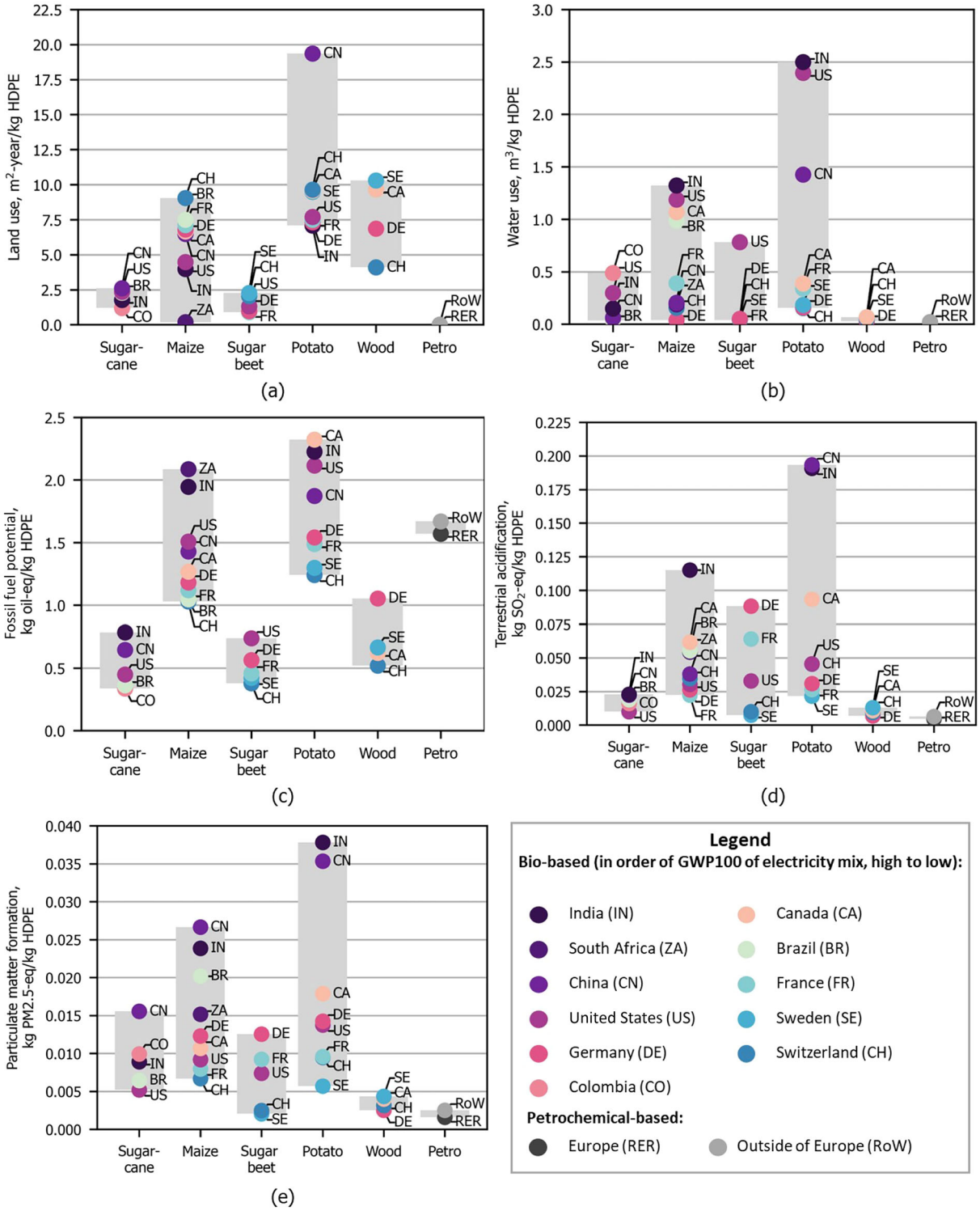

FIGURE 4 Comparison of the environmental impact of the production of 1 kg petrochemical-based high-density polyethylene (petro-HDPE) and bio-based HDPE (bio-HDPE) from various resources in the following categories: (a) land use, (b) water use, (c) fossil depletion, (d) terrestrial acidification, and (e) particulate matter formation. The order and coloring of the entries in the legend correspond with the global warming potential (GWP100) of 1 kWh of electricity at the location. Underlying data can be found in the supplementary information (S5).

mentation process (Wernet et al., 2016). For potato-based HDPE, both biomass cultivation and ethanol production were important contributors to fossil fuel depletion, due to fertilizer production, and heat, respectively.

3.1.5 | Terrestrial acidification

Figure 2d shows the terrestrial acidification due to bio- and petro-HDPE production. Terrestrial acidification is a measure of the change in soil composition due to the deposition of inorganic substances (Huijbregts et al., 2016). All bio-HDPE scenarios resulted in higher terrestrial acidification compared to petro-HDPE. For sugarcane-, maize-, sugar beet-, and potato-based HDPE, biomass cultivation was the primary contributor to terrestrial acidification. Artificial fertilizer use is a main source of acidification (Schroder et al., 2011). Notably, forests typically exhibit lower acidification (Tian & Niu, 2015), which is also observed in Figure 2d.

3.1.6 | Particulate matter formation

Figure 2e shows the particulate matter formation due to the production of bio- and petro-HDPE. Particulate matter formation refers to the formation of particles of less than 2.5 μm in the atmosphere (Huijbregts et al., 2016). Particulate matter formation of bio-HDPE was always higher compared to petro-RER HDPE. Compared to petro-RoW HDPE, only WO-DE, SB-SE, and SB-CH resulted in lower particulate matter formation. Particulate matter formation is associated with agricultural activities and road transport (Hendriks et al., 2013; Pant & Harrison, 2013). For all biomass types but wood, biomass cultivation was the main contributor to particulate matter formation (figure S2l). Within biomass cultivation, the main causes of particulate matter formation were fertilizers and agricultural vehicles. For wood-based HDPE, ethanol production caused the biggest share of particulate matter formation, due to the sulfuric acid used and gypsum produced during pre-treatment of the wood (Wooley et al., 1999).

3.2 | Effect of end-of-life options on the environmental impact

Regarding the end-of-life options of bio- and petro-HDPE, we only present the results for GWP100. GWP100 is the only impact category that considers CO_2 emissions, allowing a comparison between biogenic carbon accounting approaches. The results for other impact categories can be found in the supplementary information (Figure S1.3).

The GWP100 results are shown with accounting for biogenic carbon either at end-of-life—following JRC (2021)—(Figure 5a) or in the production stage (Figure 5b). The results of both ways of accounting for biogenic carbon led to the same final result when considering incineration (with or without energy recovery). For HDPE based on sugarcane, sugar beet, and wood, the results were more favorable than for petro-HDPE. For maize and potatoes, the result depended on the location of growth and production. When bio-HDPE is landfilled, it does not biodegrade significantly and the biogenic carbon is essentially stored. The landfilling processes resulted in 0.15 kg CO_2-eq (sanitary)–0.23 kg CO_2-eq (unsanitary) emissions for 1 kg HDPE (bio-based or petrochemical-based), or a 1.0%–13.7% increase in GWP100. If energy is recovered from incineration, this energy replaces the local electricity mix, resulting in avoided emissions that reduce GWP100 outcomes.

The impact of the mechanical recycling process itself was always the same because it always concerned the mechanical recycling of HDPE with the same energy requirements and avoiding the production of virgin petro-HDPE. Combined with production emissions, GWP100 from mechanical recycling led to a lower impact compared to virgin HDPE. Mechanical recycling was always the end-of-life option resulting in the lowest GWP100 for petro-HDPE. For bio-HDPE, the biogenic carbon accounting method led to remarkable differences: mechanical recycling seemed to have the lowest impact when biogenic carbon is accounted for in production (Figure 5b), whereas incineration with energy recovery appeared to result in the lowest impact in the case of accounting for biogenic carbon at end-of-life (Figure 5a).

For land use, terrestrial acidification, and water use, the different end-of-life scenarios did not significantly affect the environmental impact of bio-HDPE. This could be attributed to the relatively low environmental impact of both electricity and petro-HDPE in these impact categories. For particulate matter formation, electricity with energy recovery resulted in a reduction in environmental impact, because electricity (especially from fossil fuels) is a known source of particulate matter formation (Zhang et al., 2015). This effect was also observed for fossil fuel depletion. However, mechanical recycling of both bio-HDPE and petro-HDPE resulted in the lowest fossil fuel potential, because it avoided the production of virgin petro-HDPE.

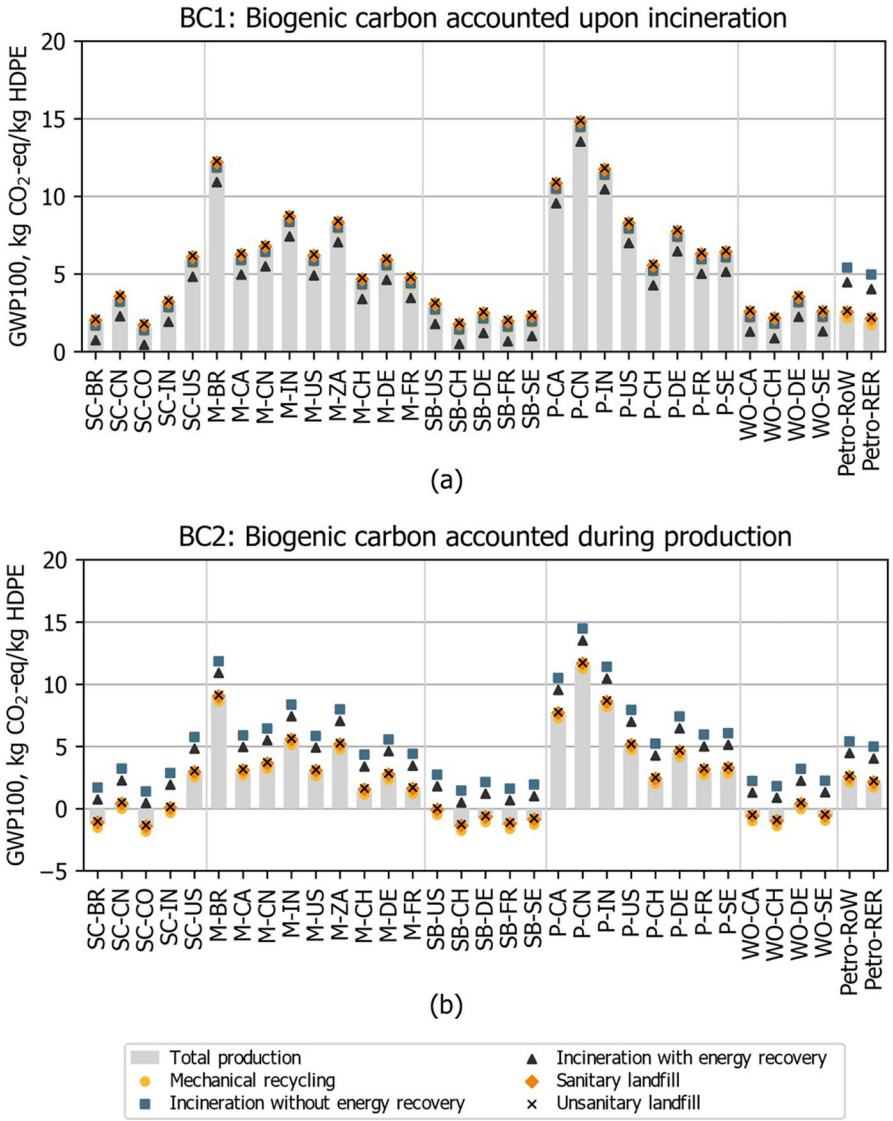

FIGURE 5 Comparison of five end-of-life scenarios for 1 kg bio-based high-density polyethylene (bio-HDPE) and petrochemical-based HDPE (petro-HDPE): mechanical recycling, incineration with energy recovery, and incineration without energy recovery. Carbon accounting was considered in two ways: (a) biogenic carbon was accounted for upon incineration of the plastic, and (b) biogenic carbon was accounted for during production. The gray bar represents the total production global warming potential (GWP100); the markers represent the total GWP100 after the different end-of-life scenarios. Underlying data can be found in the supplementary information (S6).

3.3 | Sensitivity analysis

Table 3 displays the results of the sensitivity analysis. The highest sensitivity was found for the ethanol conversion efficiency. Reducing the conversion rate of ethanol into ethylene by 10% led to an increase in GWP100 of 3.3%–9.2%, depending on the scenario. Changing the ethanol conversion efficiency affected the amount of ethanol needed and, by extension, the amount of biomass needed. Therefore, sensitivity to ethanol conversion efficiency depended on the part of the environmental impact caused by biomass and ethanol production combined. As such, the sensitivity to the ethanol conversion efficiency of the sugarcane- and sugar-beet-based scenarios was relatively low compared to wood-, maize-, and potato-based scenarios.

The sensitivity to the amount of electricity needed for the conversion of ethanol into ethylene was less significant. Increasing the amount of electricity by 10% resulted in GWP100 increases ranging from 0.01% to 2.05%, depending on the scenario. The sensitivity depended on the GWP100 of the local electricity mixture and the fraction of the GWP100 of bio-HDPE production attributed to ethylene production. The scenario sugarcane in India (SC-IN) had the highest sensitivity with 2.46%. Ethylene conversion caused 24% of the GWP100 in that scenario, and the GWP100 of elec-

TABLE 3 Outcomes of the sensitivity analysis for ethanol conversion efficiency and ethylene production efficiency.

	Ethanol conversion efficiency		Ethylene production energy	
Scenario	+10%	−10%	+10%	−10%
SC-BR	+5.21%	−5.21%	+0.45%	−0.45%
SC-CN	+4.89%	−4.89%	+1.36%	−1.36%
SC-CO	+4.04%	−4.04%	+0.75%	−0.75%
SC-IN	+3.31%	−3.31%	+2.05%	−2.05%
SC-US	+8.12%	−8.12%	+0.41%	−0.41%
M-BR	+9.22%	−9.22%	+0.07%	−0.07%
M-CA	+8.45%	−8.45%	+0.02%	−0.02%
M-CN	+7.55%	−7.55%	+0.70%	−0.70%
M-IN	+7.97%	−7.97%	+0.73%	−0.73%
M-US	+7.91%	−7.91%	+0.40%	−0.40%
M-ZA	+7.82%	−7.82%	+0.62%	−0.62%
M-CH	+8.37%	−8.37%	+0.03%	−0.03%
M-DE	+8.30%	−8.30%	+0.39%	−0.39%
M-FR	+8.24%	−8.24%	+0.08%	−0.08%
SB-US	+5.29%	−5.29%	+0.83%	−0.83%
SB-CH	+5.23%	−5.23%	+0.09%	−0.09%
SB-DE	+5.62%	−5.62%	+0.97%	−0.97%
SB-FR	+5.28%	−5.28%	+0.21%	−0.21%
SB-SE	+6.35%	−6.35%	+0.08%	−0.08%
P-CA	+9.00%	−9.00%	+0.01%	−0.01%
P-CN	+8.81%	−8.81%	+0.32%	−0.32%
P-IN	+8.39%	−8.39%	+0.54%	−0.54%
P-US	+8.14%	−8.14%	+0.30%	−0.30%
P-CH	+8.40%	−8.40%	+0.03%	−0.03%
P-DE	+8.54%	−8.54%	+0.30%	−0.30%
P-FR	+8.48%	−8.48%	+0.06%	−0.06%
P-SE	+8.62%	−8.62%	+0.03%	−0.03%
WO-CA	+6.06%	−6.06%	+0.05%	−0.05%
WO-CH	+6.27%	−6.27%	+0.08%	−0.08%
WO-DE	+7.06%	−7.06%	+0.67%	−0.67%
WO-SE	+6.93%	−6.93%	+0.07%	−0.07%

Abbreviations: SC, sugarcane; M, maize; SB, sugar beet; P, potato; WO, wood. BR, Brazil; CA, Canada; CH, Switzerland; CN, China; CO, Colombia; DE, Germany; FR, France; IN, India; SE, Sweden; US, United States; ZA, South Africa.

tricity in India is also the highest of all locations studied (nearly three times as high as for electricity in Brazil). These two factors combined caused the relatively high sensitivity to the electricity needed for ethylene production in this case.

4 | DISCUSSION

Bio-based plastics hold the potential to yield lower GWP100 and reduce fossil fuel dependency compared to petrochemical-based plastics. However, sourcing and end-of-life decisions significantly affect the environmental impact of bio-based plastics. Moreover, the results of LCA studies on bio-based plastics can vary greatly depending on the methodologies employed. In this study, we introduce a comprehensive analysis by developing 31 sourcing scenarios and 5 end-of-life scenarios for bio-based HDPE, comparing them to their petrochemical-based counterparts. While our sourcing results align with preexisting research, the extensive range of scenarios considered in this study and the consistent methodology employed

contribute to the current body of literature by assessing a much wider array of possibilities. Additionally, we examine the influence of various biogenic carbon accounting methods on LCA outcomes, showing that these methods significantly affect end-of-life preferences and their alignment with circular economy principles. This research not only provides a nuanced understanding of the environmental implications of bio-HDPE but also offers critical insights into the methodological choices that shape LCA results and their alignment with circular economy principles.

Selecting a bio-based plastic for products should carefully consider three aspects: biomass type, production location, and end-of-life. Biomass type had the biggest effect on environmental impact outcomes. Sugar-based biomass such as sugarcane or sugar beet was preferred over other biomass types, based on its lower environmental impacts across impact categories. This could be attributed to relatively high yields and the direct availability of fermentable sugars. Although the yield of cellulose-based biomass (wood) was high, associated land use was also high as well as the energy required to convert cellulose into fermentable sugars. These outcomes were in agreement with previous work in biofuels (Devi et al., 2023; Muñoz et al., 2014).

The location of biomass cultivation also affected the environmental impact. This could be attributed to three factors: energy mix, climate, and local agricultural practice. The energy mix affected the impact of the chemical processes. The climate affected the need for agricultural operations such as irrigation and pesticide use. Local agricultural practice involved activities such as the method of cultivation (manual or machinal) and fuels used in machinery (Tsiropoulos et al., 2014). The location with the lowest environmental impact therefore also depended on the biomass type. In most cases, the environmental impact correlated with the GWP100 of the local electricity mix. Locations in Europe (Germany, Sweden, France, and Switzerland) typically resulted in the lowest environmental impact.

Bio-HDPE has a relatively simple production process with few chemical conversion steps (Lee et al., 2019). At the same time, the molecular structure of PE ($(CH_2)_n$) means that most of the molecular weight consists of carbon atoms, and therefore, the biogenic carbon storage of this polymer is relatively high. If more extensive chemical conversion is needed (such as for ethanol-based bio-based polypropylene (Machado et al., 2016), or bio-PET (Gursel et al., 2021)), the effect of energy mix may become even more pronounced.

The results with respect to the different end-of-life scenarios show an apparent preference for incineration with energy recovery over mechanical recycling of biobased plastics if biogenic carbon is only accounted for upon molecular decomposition (i.e., following the JRC guidelines). This is because of not accounting for the carbon stored in the polymer when produced. The obtained results contradict the circular economy principle that products and materials should be kept at high value for as long as possible (The Ellen MacArthur Foundation, 2013). It is also counterintuitive if we consider that mechanically recycled material still holds the potential of either sequestration or avoided emissions upon incineration after the next lifecycle. Furthermore, in some impact categories, such as land use and water use, bio-HDPE always yielded a higher environmental impact than petro-HDPE. Therefore, circularity and sustainability in the plastics industry do not just require the use of renewable feedstocks, but also retaining the value of plastics at a high level for as long as possible (The Ellen MacArthur Foundation, 2013).

It is therefore important to already account for biogenic carbon during production, even though this is not supported by the current JRC guidelines. This has two major advantages: first, the potential impact reduction of bio-based plastics in all stages of the lifecycle is compared more fairly with the impact of petrochemical plastics. Twenty bio-HDPE production scenarios resulted in lower GWP100 than petro-HDPE when accounting for biogenic carbon uptake during production, compared to six when biogenic carbon was accounted for upon incineration. Second, value-retaining end-of-life options are shown to have a lower impact compared to incineration with energy recovery: Mechanical recycling is the end-of-life option resulting in the lowest cradle-to-grave GWP100 in all scenarios, followed by landfilling (i.e., the sequestration of biogenic carbon). Incineration without energy recovery is the least favorable end-of-life option.

The scope of this study has imposed several limitations, which also present opportunities for future research. The results presented in this article are likely accurate for bio-based low-density polyethylene since it has the same production process until the polymerization step, which did not account for a large share of the environmental impact in this study. However, these results are not necessarily valid for other polymers. A similar study could be conducted with different polymer types to study if the same principles described here hold. However, data availability for chemical conversion processes involved in bio-based plastic production is notoriously poor (Bishop et al., 2021). A better understanding of the environmental impact of bio-based plastics under different sourcing conditions ultimately requires improved access to bio-based polymer production data.

CONFLICT OF INTEREST STATEMENT

The authors declare no conflict of interest.

DATA AVAILABILITY STATEMENT

The data that supports the findings of this study are available in the supporting information of this article.

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SUPPORTING INFORMATION

Additional supporting information can be found online in the Supporting Information section at the end of this article.

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