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Life cycle sustainability assessment of substituting fossil based with biogenic materials

A German case study on drinking cups and insulation boxes

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Abstract

Bioeconomy is often cited as one pathway toward sustainable materials and a circular economy in an urban–rural context. This study conducts a life cycle sustainability assessment (LCSA)—life cycle assessment, social life cycle assessment, and life cycle costing (LCC)—to assess the benefits and impacts of substituting fossil polymer-based products with biogenic alternatives through two product systems: drinking cups and insulation boxes. In detail, we assess the environmental impacts, social hotspots, and societal costs subject to various product characteristics. The latter comprises, among others, different materials (fossil-based polymers, first-generation and second-generation biomass), allocation scenarios, electricity mixes, use cycles, and end-of-life (EoL) quotas. The LCSA is conducted with primary data provided by industry partners and secondary data from ecoinvent, the social hotspots database, and the literature. The results show that the drinking cup from second-generation bio-polyethylene (bio-PE) performs best in most environmental impact categories, followed by the fossil-based polypropylene (PP) cup. When substituting PP cups with bio-PE cups, 32% of CO_2 eq. emissions and 37% of water can be saved, while land use and particulate matter emissions increase by 37% and 7%, respectively. Due to low recycling rates in the status quo, cups made of polylactide acid—a first-generation bio-based polymer—often have higher environmental impacts than fossil-based ones. Governance and health and safety are the most prominent social categories and are especially linked with raw materials transportation. Similar trends are observed for the insulation box product system. The study identifies improvements in EoL practices, using biomass as-is, and regional sourcing as essential for enhancing bio-based materials' sustainability.

KEYWORDS
bioeconomy, environmental impacts, industrial ecology, plastic substitution, social hotspots, societal costs

1 INTRODUCTION

Plastic, now an integral part of everyday life, has seen global production reach around 8.3 billion metric tons between 1950 and 2015. With 9% being recycled, 12% incinerated, and 79% being landfilled or directly exposed to nature during that period, the plastic industry remains a linear economy (Geyer et al., 2017). About 8% of global oil production serves as feedstock and energy for plastic manufacturing, while over a third is allocated to the production of quickly discarded packaging (Thompson et al., 2009). The global export of plastic waste and the implementation of waste picking (Morais et al., 2022) intensifies the general trend of impact displacement from developed to developing countries (Wiedmann & Lenzen, 2018), further contributing to social issues such as child labor.

To reduce waste generated by single-use plastics, the city of Augsburg, Germany, has established a reusable drinking cup system called "Augsburger Becher," which is one outcome of the project *reGIOcycle*.[1] Another aspect of single-use packaging is the online food retail sector, which is heavily dependent on disposable insulation boxes made of expanded polystyrene (EPS) (Barnes et al., 2011; Wagner, 2020), with an average annual revenue growth of 27.5% from 2015 to 2022 in Germany (bevh, 2023). One challenge with EPS packaging is managing the material's end-of-life (EoL). Even though recycling is possible, it is often not economically viable, that is, due to high transportation costs. In the European Union, 50% of EPS waste is thus landfilled, and the remaining portion is thermally treated (Gallego-Schmid et al., 2019). In the region of Augsburg, 100% is incinerated (cf. Table 1 of Supporting Information S1), leading to a permanent removal from the material cycle. A product or process that does not align with the principles of the circular economy is leading to environmental challenges and can arguably thus not be considered sustainable (Gallego-Schmid et al., 2019).

In response to many of the contemporary challenges, the United Nations (2015) established the sustainable development goals (SDGs) as a shared foundation for achieving peace, prosperity, well-being, education, diminished inequality, and adequate climate action, many of which can, in one way or another, be linked with plastics. Since plastic production and disposal contribute to greenhouse gas emissions, reducing plastics promotes SDG13 "Climate Action" (Nasrollahi et al., 2020). Plastic packaging and single-use products contribute to overconsumption, wherefore sustainable alternatives and reducing plastic waste align with SDG12 "Responsible Consumption and Production" (Miller, 2020). Harmful chemicals in plastics can affect human and animal health, which is why reducing plastic consumption creates healthier living conditions (SDG3, "Good Health and Well-Being") (Alabi et al., 2019; Thompson et al., 2009). Lastly, plastic waste can contaminate water resources, compromising SDG6 "Clean Water and Sanitation" (Kumar et al., 2021). Minimizing plastic pollution is crucial for advancing sustainable development by addressing environmental, social, and economic challenges.

While the SDGs function as the acknowledged "high-level shared blueprint" (Valdivia et al., 2021), the life cycle sustainability assessment (LCSA) method aims to enhance the application of sustainable development principles by assessing products and services by considering environmental, social, and economic impacts throughout their life cycle. The LCSA framework, advocated by the Life Cycle Initiative since 2011 (UNEP, 2011), combines environmental life cycle assessment (LCA), social life cycle assessment (S-LCA), and life cycle costing (LCC). The framework intends to guide more effective LCSA applications, fostering alignment with ISO 14040 and promoting a clear interpretation of results to inform decision-making toward more sustainable product supply chains, from raw material acquisition to EoL strategies. Despite increasing interest, LCSA is still in an early stage and the community has acknowledged the limitations of S-LCA regarding data availability, metric standardization, and practical implementation. The recently published ISO 14075:2024 (DIN EN ISO, 2024) addresses these gaps and provides a structured framework to enhance the credibility and applicability of S-LCA (Type II, impact pathway approach) and social life cycle performance assessment (S-LCPA; Type I, reference scale approach) aiming at a better integration with environmental and economic evaluations for a more comprehensive approach to sustainability.

While the challenges and impacts of plastic consumption are primarily associated with fossil-based polymers, bio-based polymers are often seen as a sustainable alternative if avoiding plastics is regarded as infeasible (Hottle et al., 2013; Pellis et al., 2021; Walker & Rothman, 2020). For example, drinking cups made of bio-based polylactide acid (PLA) could substitute polypropylene (PP) cups to minimize the use of fossil resources while exhibiting similar thermal properties (Naser et al., 2021). Polystyrene boxes can be replaced by bio-based alternatives, such as straw-, hemp-, or recycled cardboard-based boxes, to avoid the (fossil or biogenic) polymer altogether. This is in line with the EU's plan to emphasize the utilization of biological resources to produce nutritious food, animal feed, fuels, and materials derived from natural sources to transition away from petroleum-based products (European Commission, 2012). However, regarding biogenic products, a significant distinction needs to be made between different biomass "generations," namely *first-generation* (1G) and *second-generation* (2G) materials. The 1G bio-based products are based on edible, starch- or oil-based biomass (Dutta et al., 2014; Wietschel, 2022). While biogenic products, in general, contribute to the goal of reducing fossil-based plastics, the production of 1G-based biopolymers comes with its drawbacks, including, among other things, higher land use and water consumption compared to petroleum-based plastics (Brizga et al., 2020) and a conflict with food production (Lewandowski, 2015; Messmann et al., 2023). In contrast, 2G products are based on non-edible, for example, lignocellulosic feedstock, such as agricultural residues or biomass as a by-product (Dutta et al., 2014; Wietschel, 2022), which is why second-generation bio-based products generally promise lower impacts than first-generation products (Hauschild et al., 2018). With 3.2 million metric tons of agricultural residues in Bavaria (Germany) in 2022, many municipalities could, in theory, substitute fossil-based products by using this unemployed, second-generation bioeconomic potential[2] as feedstock for bio-based products (Wietschel et al., 2021a).

To address challenges linked with raw materials, such as unsustainable sourcing, linear instead of circular economies (Paletta et al., 2019), littering (Wurm et al., 2020), downcycling (Helbig et al., 2022), no cascading use (Hildebrandt et al., 2017), and raw material dissipation (Charpentier Poncelet et al., 2022a, 2022b; Helbig et al., 2020) academia discusses the topic with increasing interest, but the results still need to be clarified, indicating a need for further research. The study of Gallego-Schmid et al. (2019) focuses on the environmental impacts of widely used takeaway containers, including aluminum, polypropylene, and extruded polystyrene, and comparisons to reusable polypropylene containers. The results indicate that single-use polypropylene containers have the highest environmental impacts, particularly regarding global warming potential (GWP), while extruded polystyrene containers exhibit the lowest implications due to lower material and electricity requirements. Barrio et al. (2021) discuss the development of a bio-based multi-layer panel using residual coniferous bark, aiming to contribute to environmental protection while maintaining competitiveness and high social standards. These include benefits in terms of climate change mitigation when compared to standard panels. Van der Harst and Potting (2013) compared 10 LCA studies of disposable cups, focusing on their GWP. The analysis revealed that none of the different cup types—conventional fossil plastics, bioplastics, or paperboard—consistently performed better than others in terms of GWP, and variations were attributed to factors such as cup material, weight, production processes, waste processes, allocation options, and data used. The study suggests that a more robust evaluation could be achieved in future research through the systematic and simultaneous use of sensitivity and scenario analyses. Miller (2020) emphasizes the necessity for comprehensive LCSA to measure plastics' sustainability accurately.

The study at hand examines the environmental, social, and economic sustainability of substituting two exemplary product systems—(i) drinking cups and (ii) insulation boxes—with biogenic alternatives. With previous meta-analyses in this field being ambiguous about the advantageousness of different types of biogenic products (Barrio et al., 2021; UNEP, 2021; Van der Harst & Potting, 2013), the main contribution of this work to the academic discussion is a cradle-to-grave LCSA of two product systems based on real case studies. Therefore, our study explores the conditions and characteristics, such as different biomass "generations" and recycling scenarios, under which diverse fossil, 1G, or 2G products are environmentally, socially, and economically sustainable. In detail, we set out to answer the following research questions:

RQ1: What are the environmental, social, and economic benefits or impacts of substituting conventional, fossil-based products with biogenic ones?
RQ2: How do characteristics (e.g., biomass "generation," recycling quotas, electricity mixes, lorry exhaust emission standards, use cycles, and raw material origins) affect the results of the assessed value chains?

Section 2 defines the goal and scope (Section 2.1), life cycle inventory analysis (Section 2.2), and impact assessment (Section 2.3). Section 3 presents the study's findings, while the discussion and conclusion (Section 4) explores their implications, significance, and limitations (cf. Figure 1 of Supporting Information S1).

2 | METHODS

This work considers two product systems: (i) drinking cups and (ii) insulation boxes. In each product system, we consider four products relating to fossil- as well as 1G-, 2G- and composite 1G/2G-based materials. In the case of the drinking cups, all products are made of (fossil or biogenic) polymers, while for the different insulation boxes, the biomass is used as-is. The two product systems and the respective materials are chosen based on their practicability within the industry-partnered reGIOcycle project. In the case of straw (used in the 2G box), the practicability was proven by its local availability as well as real-life examples of straw-based box production in the region. The reference products and their materials are based on the most abundant materials on the market (i.e., polystyrene for insulation boxes or PP for drinking cups). In detail, we conduct an LCSA of the following products, materials, and biomass "generations":

i. Drinking cups:
a. Polypropylene (**PP**) from fossil crude oil. Thermoplastic polypropylene is lightweight, durable, and heat and chemical resistant.
b. **PLA** from 1G sugar cane. PLA is a biodegradable polymer. However, sugar cane cultivation competes for land and water, requiring energy-intensive processing.
c. **Composite** of **PLA** from 1G sugar cane + 2G **cellulose fibers**. This biocomposite blends PLA with cellulose fiber from non-food sources, enhancing mechanical and thermal properties. Challenges include 2G biomass availability and compatibility.
d. Bio-polyethylene (**bio-PE**) from starch from 2G maize stover serves as a "drop-in" solution, mirroring the properties and applications of fossil-based PE cups.
ii. Insulation boxes:
a. **EPS** from fossil crude oil. EPS foam boxes offer good thermal insulation, shock absorption, and moisture resistance. However, they are challenging to recycle or biodegrade.

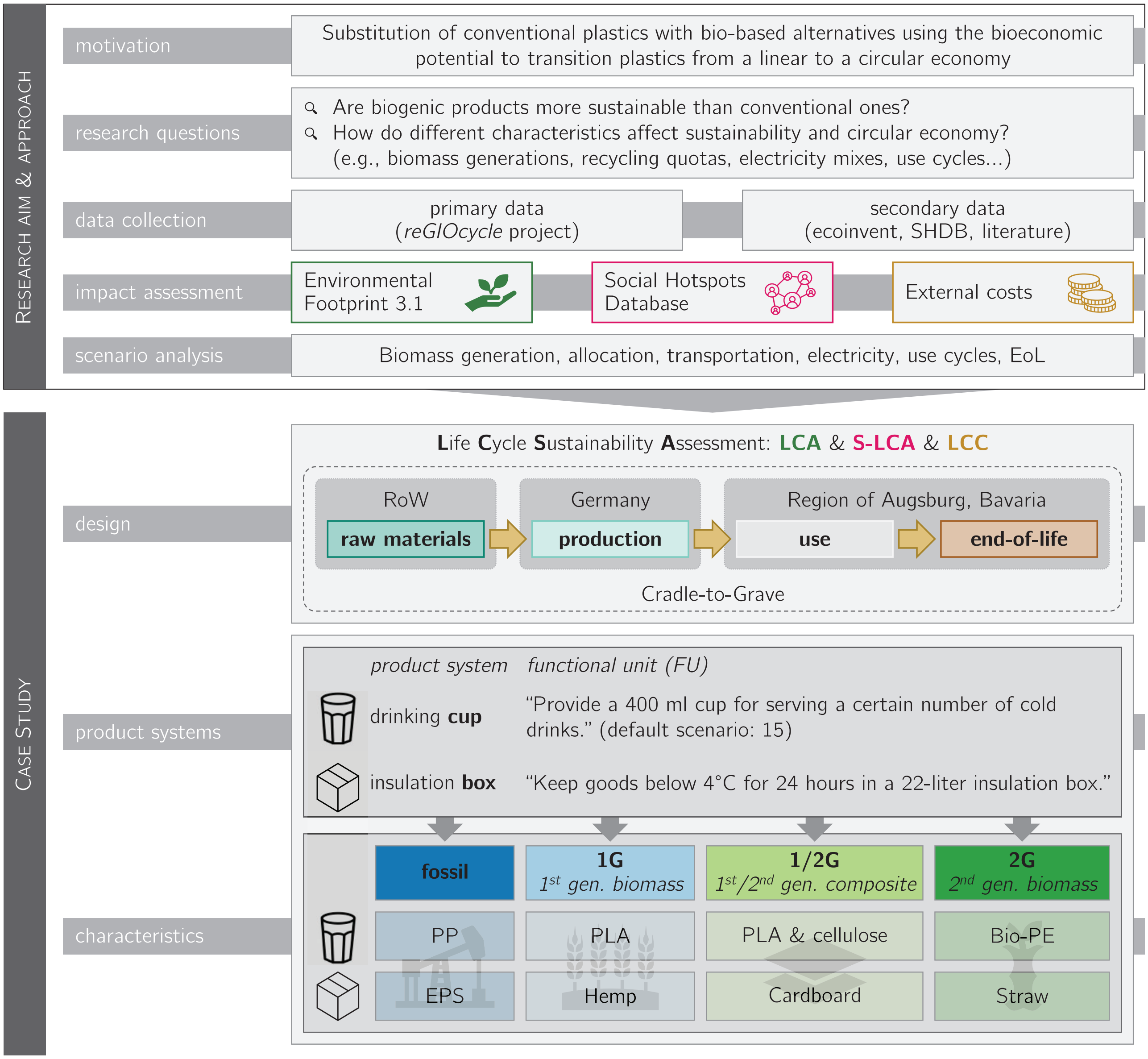

FIGURE 1 Conceptual design of research and case study.

b. 1G **hemp**-based insulation mats or boards shaped into boxes provide thermal and acoustic insulation and biodegradability.
c. 1/2G (virgin and recycled) **cardboard** boxes made from virgin or recycled pulp maintain thermal insulation and mechanical strength. They are recyclable but susceptible to moisture and mold.
d. 2G barley **straw**. Utilizing agricultural residue, barley straw boxes offer good thermal and acoustic insulation but face supply challenges due to seasonality.

The data for the different products were either gathered in collaboration with industry partners as part of the *reGIOcycle* project (for the more innovative, i.e., the 1G, 1G/2G, and 2G ones) in the German city of Augsburg as well as other south German cities and regions (cf. Table 1 of Supporting Information S1) or retrieved from secondary sources such as literature and the ecoinvent database (for the more established, i.e., the fossil ones, cf. Table 2). Figure 1 summarizes this work's study design, data sources, impact assessment methods, focal products, and functional units. Goal and scope, life cycle inventories, life cycle impact assessment, and all the different product characteristics that may influence the advantageousness of one product vis-à-vis the others are detailed in the following subsections. All inventory models are created analogously to each other to meet the goal of comparability. All assumptions made and all data used are presented in Supporting Information S1 and S2.

TABLE 1 Product characteristics, methodological choices, and possible modeling combinations.

Value chain	Product characteristics of drinking cups and insulation boxes
Raw materials: biomass generation	Fossil | 1G | 1/2G | 2G
Raw materials: zero-allocation assumption (cf. Section 4.14 of Supporting Information S1)	No | ecoinvent default | yes
Transportation: raw mat. → prod.	Globally | Thailand | Brazil | Europe | Germany or proxy CH | Augsburg region
Production: electricity mix	German electricity mix (2022) | renewable energy only
Transportation: within Germany	German lorry mix (2022) | only EURO 6 lorries
Use: number of usages	1 | 10 | 15 | 20 | 100 | 300 | 500
EoL: recycling quota [%]	0 | 43 | 50 | 78 | 90 | 100
LCA aspect	**Methodological choices**
LCIA: biogenic CO_2	EF 3.1 (0/0) | IPCC 2021 GWP100 (0/0) | IPCC 2021 GWP100 (+1/-1)
Allocation: ecoinvent database	Allocation at point of substitution (APOS) | allocation, cut-off by classification

2.1 | Goal and scope definition

The goal and scope of attributional LCA, S-LCA, and LCC are identical to ensure comparability. Cradle-to-grave system boundaries are chosen with a geographic focus on the region of Augsburg. This covers the following stages, for all methods (LCA, S-LCA, and LCC): (i) raw materials, (ii) transportation (of raw materials, final products, and waste), (iii) production, (iv) use (only for cups), and (v) EoL. In the first step, the LCSA aims to assess the influence of biomass "generation" on the products' overall environmental, social, and economic impact on the cradle-to-grave life cycle. In the second step, we investigate other potentially relevant conditions and characteristics, such as different allocation methods for second-generation material, electricity mixes, lorry exhaust emissions, use cycles (for the cups), and recycling quotas. Table 1 illustrates the resulting theoretical 24,192 model combinations per product system. Since not all are realistic or worthwhile investigating, we select the most promising variants detailed in Section 3.4. In the default case, we use the ecoinvent default allocation factor (except for 2G raw materials, where a waste product is assumed and is allocated 0% of the product system's impacts), transportation as given in Germany in 2020, and the German electricity mix for final product manufacturing. Furthermore, we assume 15 whole-number use cycles, based on the average (14.8) observed in the related literature from Table 2, to maintain realism and align with established practices in the literature (Almeida et al., 2018; Althammer et al., 2017; Cottafava et al., 2021; Garrido & Alvarez del Castillo, 2007; Kauertz et al., 2019), and material-specific recycling quotas based on local municipal waste treatment data (cf. Table 1 of Supporting Information S1).

2.2 | Life cycle inventory

The products were modeled in SimaPro 9.5.0.0, using the ecoinvent 3.9.1 and SHDB (2019) databases and market-based data for the LCC. Our LCA, S-LCA, and LCC models are parameter based, allowing for diverse scenarios and sensitivity analyses. While modeled in SimaPro, the inventories provided in Supporting Information S2 are structured in a brightway2-compatible format, promoting adaptability for future scenarios and inviting further research contributions.

Two categories of data need differentiation: (i) generic background data and (ii) product-specific foreground data detailing the flows linked to the examined process. Primary data provided by industry and governmental partners within the project are used whenever possible and prioritized over secondary data. Relevant literature sources are given in Table 2 to verify and complete the dataset. They were identified on Web of Science with the following search string:

TS = ((cup* OR box*) AND (LCA OR "life cycle assessment" OR LCC OR "life cycle costing" OR SLCA OR "social life cycle assessment" OR LCSA OR "life cycle sustainability assessment"))

2.2.1 | Life cycle assessment

The cradle-to-grave approach in this LCA, from raw material extraction to manufacturing, distribution, use, and eventual disposal or recycling, embraces a holistic perspective that facilitates a nuanced understanding of the product's environmental footprint (DIN EN ISO, 2021; Hauschild, 2017; Hauschild et al., 2018). We use the ecoinvent allocation at the point of substitution (APOS) database to model background processes. The

TABLE 2 Relevant literature regarding life cycle inventory analysis.

	Method			Fossil		1G		1/2G		2G		
Article	LCA	LCC	S-LCA	PP	EPS	PLA	Hemp	PLA	Paper	Bio-PE	Straw	System boundaries
Almeida et al. (2018)	x	–	–	X	–	x	–	–	–	–	–	c2gr
Althammer et al. (2017)	x	–	–	X	–	–	–	–	–	–	–	c2gr
Changwichan and Gheewala (2020)	x	–	–	X	–	x	–	–	–	–	–	c2gr
Cheroennet et al. (2017)	x	x	–	–	x[b]	x	–	–	–	–	–	c2use
Cottafava et al. (2021)	x	–	–	X	–	x	–	–	–	–	–	c2gr
Gallego-Schmid et al. (2019)	x	–	–	X	x	–	–	–	–	–	–	c2gr
Garrido and Alvarez del Castillo (2007)	x	–	–	X	–	–	–	–	–	–	–	c2use
Hansen et al. (2015)	x	–	–	–	x[b]	–	–	–	–	–	–	c2ga
Ingrao et al. (2015)	x	–	–	–	x[b]	–	–	–	–	–	–	c2gr-use
Kauertz et al. (2019)	x	–	–	X	x	x	–	–	–	–	–	c2gr
Leejarkpai et al. (2016)	x	x	–	–	x[b]	x	–	–	–	–	–	c2gr
Maga et al. (2019)	x	–	–	X	x[a], x[b]	x	–	–	–	–	–	c2gr-use
Moretti et al. (2021)	x	–	–	X	–	x	–	–	–	–	–	c2gr-use
Razza et al. (2015)	x	–	–	–	x	–	–	–	–	–	–	c2gr
Suwanmanee et al. (2013)	x	–	–	–	x[b]	x	–	–	–	–	–	c2cga
Taengwathananukool et al. (2013)	x	–	–	–	–	x	–	–	–	–	–	c2gr
Uihlein et al. (2008)	x	–	–	–	x[b]	x	–	–	–	–	–	c2gr-use
Van der Harst and Potting (2014)	x	–	–	–	x[b]	–	–	–	–	–	–	c2gr
Van der Harst et al. (2016)	x	–	–	–	x[b]	–	–	–	–	–	–	rm + EoL
Study at hand	x	x	x	x	x	x	x	x	x	x	x	c2gr

Abbreviations: c2cga: cradle-to-consumer gate; c2ga: cradle-to-gate; c2gr: cradle-to-grave; c2gr-use: cradle-to-grave without use; c2use: cradle-to-use; rm + EoL: raw materials and end of life.
[a]Extruded polystyrene (XPS).
[b]Polystyrene (PS).

APOS method allocates environmental impacts based on market prices at the point where products or by-products substitute each other, reflecting real-world economic dynamics. Furthermore, employing the system expansion approach emphasizes the burden that is potentially avoided through recycling and the substitution of primary resources within the products' first usage. Vice versa, the recycled material encompasses environmental impacts associated with the entire life cycle of the initial product. Meanwhile, the cut-off approach excludes impacts beyond the product's initial life cycle, meaning impacts from recycled materials or waste recovery are not included.

Additionally, we adjusted the background data used to the needs of our study, for example, by considering that the harvesting process requires less diesel since the straw stalks do not need to be chopped for use in insulation boxes (in contrast to being chopped for composting). Lastly, the absence of explicit consequential modeling is justified by the relatively low global significance of the product systems, underscoring the pragmatic and context-specific nature of the chosen methodology. This approach enables the identification of hotspots and areas for improvement across the entire life cycle, providing valuable insights for more informed decision-making to the ongoing development of a more sustainable and circular economy.

2.2.2 | Social life cycle assessment

Currently, product-focused S-LCA faces challenges because it requires quantitative metrics, while many social effects are qualitative. In contrast, qualitative, organization-oriented approaches struggle to connect an organization's social performance to specific products and are thus primarily used for due diligence on suppliers and address only a limited part of the product's life cycle (van Dulmen et al., 2025). In the work at hand, we evaluate social risks along global upstream and downstream (cf. Figure 2) value chains of a generic version of the focal products as suggested by Kühnen and Hahn (2017) and Sousa-Zomer and Cauchick Miguel (2018), as well as on site (analogous to direct emissions in environmental LCIs). This evaluation is conducted as a social life cycle performance assessment (S-LCPA, formerly Type I) introduced by DIN EN ISO (2024) using the reference scale approach of the social hotspots database (SHDB, 2019) to assess social risks in 25 subcategories, ranging from labor conditions and human rights to community well-being (Sala et al., 2015). We map the sectors of the underlying Global Trade Analysis Project (GTAP9) input–output model to processes of the environmental model (see Figure 2) and use monetary values from actual market prices, detailed in Supporting Information S2. The SHDB (2019) enables a systematic identification of social "hotspots" in global value chains throughout each stage of a product's life cycle, pinpointing areas where social risks are most pronounced. It empowers practitioners to assess and address social issues in a targeted manner, fostering sustainable and socially responsible practices.

2.2.3 | Life cycle costing

Traditionally, LCC examines the total expenses incurred throughout a product's life cycle, often by assuming the consumer's or producer's viewpoint. It encompasses all costs associated with the entire life span of the product (Swarr et al., 2011). In contrast, our approach shifts the focus to the broader societal context by calculating the external costs borne by society, induced by the product's environmental impacts. By adopting this perspective, we highlight the connection between a product's environmental and monetary consequences for society. We underscore that the environmental burdens associated with a product or service extend beyond ecological concerns and have significant economic implications (Pieper et al., 2020). Currently, the often advocated "polluter-pays principle" is not being implemented, meaning that society bears the costs (e.g., from damages, for restoration, or of avoidance) of environmental impacts induced by consumers and/or producers (Michalke et al., 2023). Moreover, companies may pass on the costs to consumers or engage in regulatory arbitrage. In this study, analogously to Michalke et al. (2023), we monetize the environmental midpoints using the framework by the European Commission et al. (2020). In this framework, cost factors are gathered through literature, existing frameworks like the Environmental Prices Handbook (de Bruyn et al., 2018), as well as experts. In current approaches, monetization factors are not spatially specific. Different cost approaches (avoidance, damage, and resource depletion) are used for the 16 impact categories of the EF 3.1 method. The monetization factors are presented with a "low," "central," and "high" value in $\frac{€_{2018}}{\textit{Impact unit}}$ (cf. Table 2 of Supporting Information S1). The "low" values represent minimum estimates, serving as a conservative lower bound. "Central" values are considered the most probable estimates, while "high" values indicate the maximum estimates, functioning as an upper bound and providing a more stringent perspective. This approach also facilitates effective communication of our results to a broader audience, making the results more accessible to addressees outside academia, contributing a greater understanding of the interconnectedness between environmental sustainability, economic well-being, and encouraging informed decision-making and responsible consumption practices in society.

2.3 | Life cycle impact assessment

To be consistent with the geographical focus on Germany in this study, the environmental impact assessment uses the European-focused Environmental Footprint 3.1 (EF) method (Andreasi Bassi et al., 2023). As recommended by the IPCC (2023), the EF3.1 method does not account for CO_2

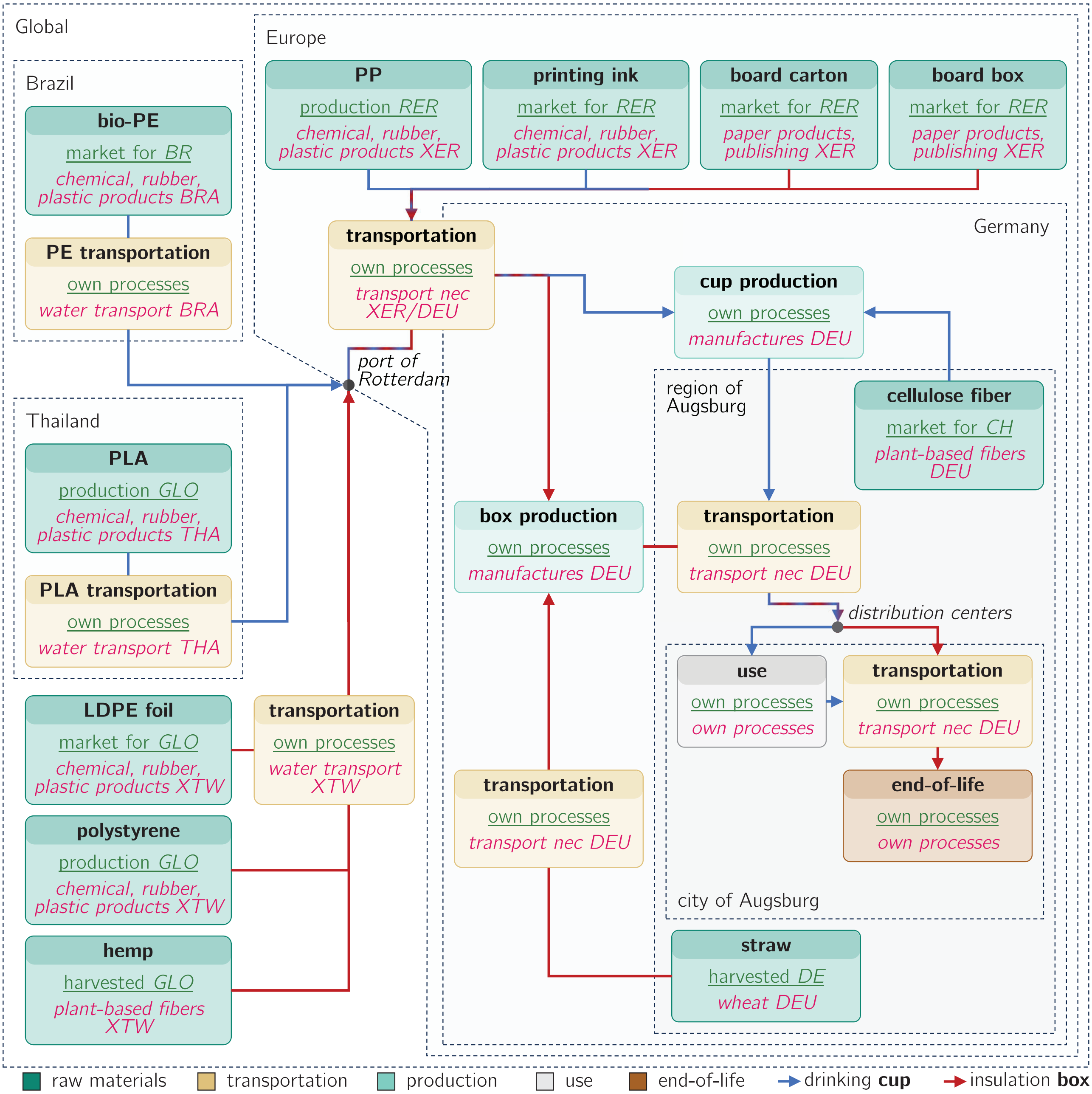

FIGURE 2 Geographical resolution for life cycle assessment (underlined, green) and social life cycle assessment data (italic, *pink*) and the supply chain of the drinking cups in blue and the insulation box in red for default product characteristics.

uptake and biogenic CO_2 emissions ("0/0 approach") as these are part of a short carbon cycle and have no net impact (European Commission et al., 2023). While the "IPCC 2021 GWP100 (including CO_2 uptake)" method does include biogenic CO_2 uptake and emissions ("+1/−1 approach"), it is important to note that even with this method, biogenic CO_2 uptake and emissions still result in a net-zero impact due to the short carbon cycle. Any variations over a full life cycle will only occur if other biogenic carbon substances (such as methane) are released or if biogenic carbon is temporarily stored in a material that is used in the next product life cycle. Therefore, in our case study, the 0/0 approach is used as the default to reflect the net-zero impact of the short carbon cycle, while the +1/−1 approach is applied in the scenario analysis (cf. Section 3.4). Given the bioeconomic perspective of this study, we focus our results on the impact categories related to challenges associated with the local bioeconomy (climate change, land use, and water consumption), as well as those identified to be most relevant in terms of their normalized values (here: particulate matter) (Ingrao et al., 2018; Yang et al., 2023). Nevertheless, results for all 16 midpoints of the EF3.1 method are provided in Supporting Information S1 and S2.

For the S-LC(P)A, we use the "Social Hotspot 2018 All Factors" impact assessment method provided by the SHDB (2019) and detailed by Benoit Norris et al. (2019). Similar to its competitor, the Product Social Impact Life Cycle Assessment (PSILCA) database, it compiles indicators for various social topics (e.g., labor practices, health and safety, and community engagement) at both national and sector-specific levels, assigns a risk level (i.e., a characterization factor; based on international standards, specifying severity and likelihood), weights them with labor intensity data, and aggregates them with data from GTAP9. In this way, databases like SHDB and PSILCA help identify areas with the greatest risk for adverse social conditions, enabling users to pinpoint "hotspots" where stakeholders (e.g., workers, local communities, and consumers) may be most vulnerable on average. While using S-LCPA databases remains relatively novel compared to well-established environmental LCA approaches, this framework systematically integrates social dimensions into sustainability assessments, allowing for targeted interventions in regions or supply chain segments most at risk of adverse effects.

The external costs (EC) framework published by the European Commission et al. (2020) is used for the LCC. Although the monetization of environmental impacts in the EC framework is based on EF2.0, we assume, as reassured by personal communication with the authors, that their monetization factors can also be applied to LCIA results retrieved with EF3.1. To quantify the monetary pressure on society, we multiply the environmental impacts, for example, in CO_2 eq., with the corresponding monetization values from Table 2 of Supporting Information S1. Here, we employ the range between the central, upper, and lower cost factors to deal with the uncertainties of the monetization approach.

3 | RESULTS

In the following, we present key findings of the applied LCSA by LCA, S-LCA, and LCC methods. All results (e.g., the complete set of midpoints cf. Figures 2 to 32 of Supporting Information S1) can be drawn from Supporting Information S1 and S2. Section 3.1 focuses on environmental impacts, Section 3.2 presents the social hotspots, and Section 3.3 discusses the calculated societal costs for the default scenario. Section 3.4 explores other combinations of characteristics and conditions (cf. Table 1).

3.1 | Environmental impacts (LCA)

Figure 3 presents the environmental impacts of the different products of the two product systems for four midpoints (climate change [kg CO_2 eq.], land use [Pt], particulate matter [disease inc.], and water use [m^3 depriv.]; cf. Section 2.3). Employing the system expansion methodology in this LCA study, outcomes featuring negative values can be understood as "avoided impacts" or favorable environmental results. In contrast, positive values indicate detrimental effects (with 100% representing the impacts of the conventional, fossil-based product).

Among the cups, the 1G variant causes the highest *climate change* impact of 0.47 kg CO_2 eq., while the 2G cup performs significantly better with 0.10 kg CO_2 eq. The EoL stage highly influences the result of both cups—while EoL alone contributes 0.15 kg CO_2 eq. to the overall impact of the 1G cup, the avoided burden dominates in the EoL stage of the 2G cup, improving this cup's result by −0.07 kg CO_2 eq. The latter is also the case for the fossil cup, which profits from relatively good recycling quotas, while the 1/2G cup's performance suffers from the same problems as the 1G cup. For the insulation box, the picture is quite different. Here, all biogenic alternatives lead to a CO_2 eq. emissions reduction up to 80% or 3.99 kg CO_2 eq. per box.

The midpoint of *land use* holds significant importance when discussing biomass "generations" or different allocation methods. It becomes evident that all biogenic products (both cups and boxes) exert a notably higher land use impact than their fossil counterparts (except for the 2G box). This underscores the interplay between agricultural practices and their ecological land use impacts. The intense refinery processes in converting biomass to polymers contribute to the higher environmental impacts of the 2G cup, despite the 2G box and the 2G cup being modeled with zero allocation for raw materials. The 1/2G box has a more than 10 times higher land use impact than the fossil one because, on a global scale, virgin paper is often sourced from tropical woods. Overall, the midpoint analysis illuminates the multifaceted nature of land use considerations in biomass generation and underscores the imperative of holistic assessment frameworks for sustainable resource management. Here, including ecosystem services (out-of-scope in the study at hand) could add to this assessment.

The lowest impact in terms of *particulate matter* among the drinking cups is observed in the fossil-based variant. Even the 2G variant, while only slightly worse by 7%, underscores the general trend of biogenic cups leading to higher emissions (mainly from raw materials procurement and transportation). Conversely, in the case of insulation boxes, biogenic alternatives exhibit lower impacts compared to fossil ones. The 2G variant substantially reduces particulate matter emissions by 40%. This significant reduction highlights the potential of second-generation biomass materials that are not transformed into plastics but are used as biomass as-is solutions.

The findings reveal a clear correlation between biomass cultivation/allocation and *water* impacts. Water impacts are significantly lower when reducing cultivation amounts and in scenarios with low-impact allocation to biogenic residues. For example, the 2G cup (bio-PE) saves about 0.52 m^3 depriv. (−37%) compared to fossil PP cups. Conversely, PLA-based cups exhibit impacts that are 240% higher. In contrast to the cups, the 1G box already shows an 18% reduction in water use compared to the fossil box, while the 2G variant demonstrates a further reduction of up to 31% (2.46 m^3 depriv.). Moreover, the analysis underscores the broader trend that first-generation biomass cultivation is water intensive, emphasizing

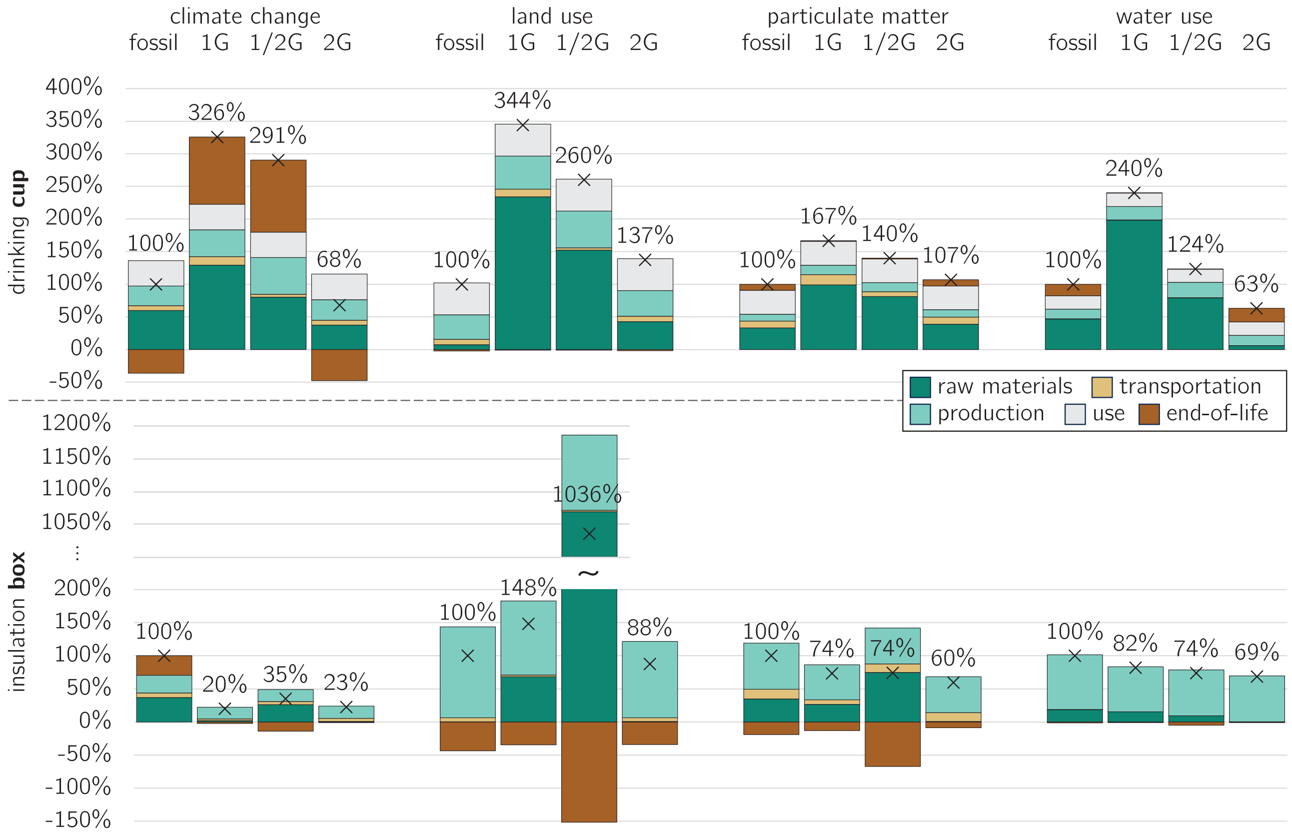

FIGURE 3 Environmental impacts of the different products in the two product systems. The 100% refer to the impacts of the conventional fossil product; negative impacts can be understood as avoided burdens; the color scheme is analogous to the previous figures. The depicted products are: fossil polyethylene (PP) cup, 1G polylactide acid (PLA) cup, 1G PLA + 2G cellulose fiber composite cup (1/2G), 2G bio-polyethylene (bio-PE) cup; fossil expanded polystyrene (EPS) box, 1G hemp box, virgin + recycled cardboard box (1/2G), and 2G barley straw box. The underlying data for Figure 3 can be found in Sheet A5(a) of Supporting Information S2.

the favorable attributes of second-generation biomass in mitigating water usage within product systems. These insights underscore the importance of biomass practice strategies for enhancing water use efficiency and overall sustainability.

Contrary to common assumptions, the results show that utilizing bio-based materials does not automatically translate into higher environmental sustainability. Our investigation emphasizes the critical significance of the EoL phase, highlighting that circular solutions have a stronger lever for sustainability than linear (fossil or 1G bio-based) approaches. The transportation of raw materials, even for global shipping, is not a major driver for most impact categories. As the importance of the use phase (of the cups) highly depends on the number of use cycles, Section 3.4 explores this in more detail. Identified hotspots in our analysis include providing raw materials, particularly bio-based, due to intense agricultural processes. Notably, 2G bio-based products emerge as the most promising, particularly when assuming 0% allocation (further explored in Section 3.4) or even the only broadly environmentally beneficial alternative.

3.2 | Social hotspots (S-LCA)

Since the selected S-LCA approach is based on monetary cost data, higher product costs lead to seemingly higher social risks. The study's heterogeneous products should not be compared through absolute risk values (in mrheq) to avoid suggesting that a product associated with lower risks is inherently "better" than one with higher risks. Therefore, social risks are presented only in relative (Figure 4). The percentual distribution of risks in the five primary categories of the SHDB can draw attention to hotspots, that is, activities and/or social aspects of the supply chain that should be examined more closely. While these categories—namely "labor rights & decent work," "health & safety," "human rights," "governance," and "community"—are more thematically focused than, for example, the stakeholder categories of the Guidelines for Social Life Cycle Assessment of Products and Organizations, they are still indicative of the various stakeholders (workers, society, local community, value chain actors, and consumers) of the evaluated value chains (cf. Table 9 of Supporting Information S1). Figures 36 to 51 of Supporting Information S1 provide analogous information on the level of subcategories. Even if transportation only accounts for 26% of the total costs occurring over the cups' life cycles, the

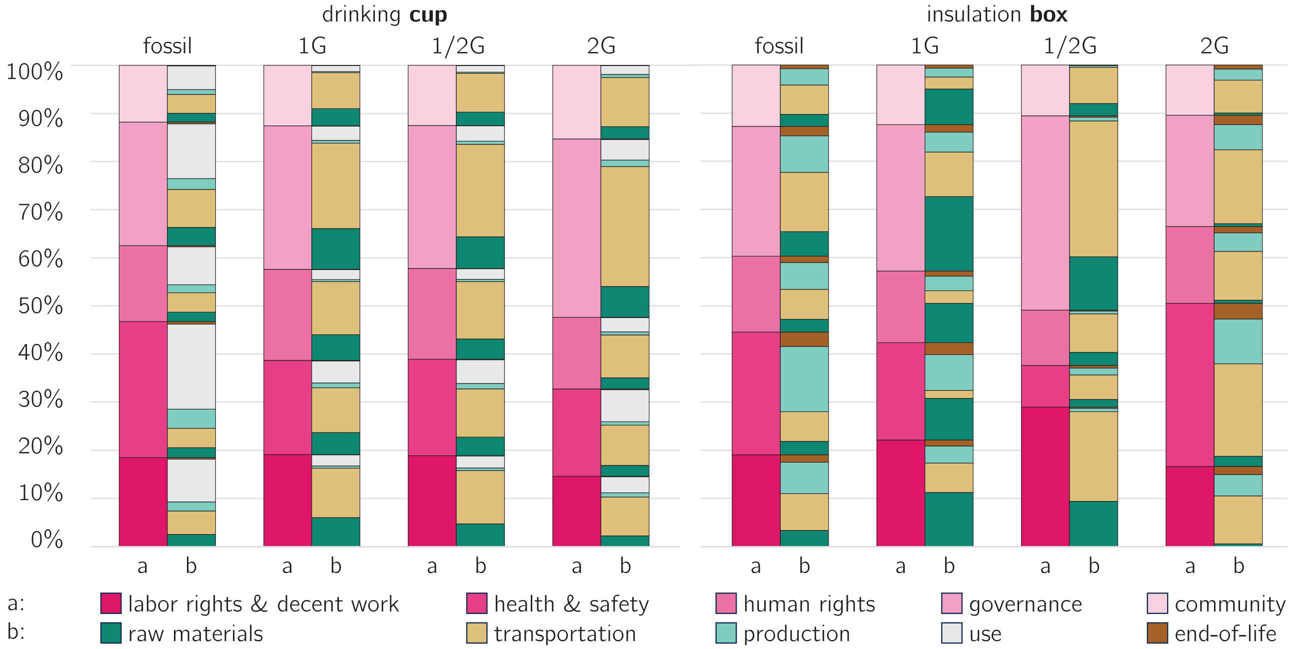

FIGURE 4 Social hotspots for the different products in the two product systems. The color scheme is analogous to the previous figures for the activities, with colors representing social categories (a) and life cycle stages (b). Figures 36 to 51 of Supporting Information S1 display the information on the level of the 25 subcategories. The underlying data for Figure 4 can be found in Sheet A5(a) of Supporting Information S2.

transportation phase poses a potential social hotspot, contrasting with the LCA and LCC results (cf. Supporting Information S2). Cardboard boxes are heavier than those made of other materials, resulting in increased transportation costs and, therefore, higher risks associated with the transportation sector. Products whose raw materials are sourced from outside of Europe score high for governance. Notably, although cup cleaning (taking place in the city of Augsburg) is responsible for 50% to 56% of the total costs, it still plays a minor role socially (cf. Supporting Information S2). Figures 34 and 35 of Supporting Information S1 highlight the social hotspots geographically, showing that even though most processes (cf. Figure 2) take place in Germany, the risks associated with raw materials dominate.

3.3 | Societal costs (LCC)

The variants of each product system exert comparable societal, environmentally caused costs over their life cycle. Notably, the PLA-based cup stands out with the highest external costs, with values reaching up to 0.28€ per cup at the upper bound. An analysis of cup variants reveals that raw material sourcing significantly influences overall external costs. Conversely, the 2G box demonstrates the lowest costs, while the fossil variant records the highest. Here, emphasis is placed on the raw material selection, which has a substantial cost impact for both fossil and 1/2G boxes due to the elevated CO_2 emissions associated with these materials. Furthermore, the societal costs for a cup deposit system (35,000 cups in the city of Augsburg) can range from 1096€ (bio-PE) up to 2968€ (PLA). The boxplots in Figure 5 display elongated whiskers toward higher costs, the central and lower costs are relatively close, suggesting greater certainty regarding the minimum external costs compared to the maximum. Figures 52 to 57 of Supporting Information S1 highlight the difference between upper, lower, and central values.

3.4 | Scenario analyses

Given the parameters and the modular structure of the conducted LCSA, multiple scenarios can be investigated. In the following, we focus on the use phase for the cups, EoL quotas, and impact allocation for 2G raw material for the insulation boxes to explicitly address circular economy aspects and the robustness of certain assumptions and methodological choices.

The number of usages is the decisive factor in determining the importance of the cup's use phase. It varies strongly between different cup materials and midpoints (Table 3). For fossil and 2G cups, the use phase dominates between 12 and 35 usages. Since PLA-based cups entail higher environmental impacts due to raw material extraction and the absence of efficient recycling pathways, the use phase becomes dominant only after 34 to 147 uses. This underscores the importance of promoting multi-use products, particularly in the status quo recycling quotas.

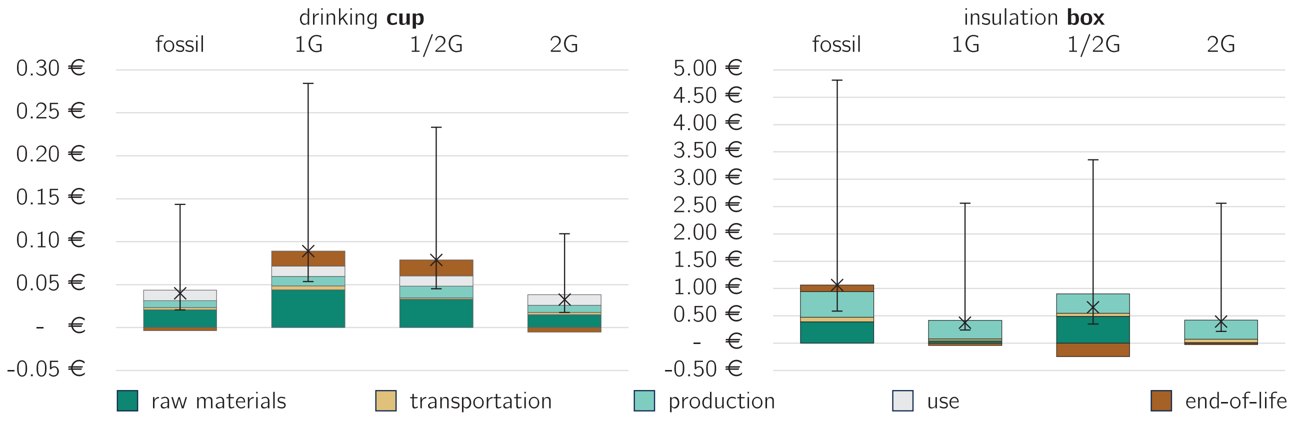

FIGURE 5 Societal environmental costs of the different products in the two product systems. The whiskers show the range between the lower, central, and upper monetization values. Negative impacts can be understood as avoided burdens. The color scheme is analogous to the previous figures. The depicted products are: fossil polyethylene (PP) cup, 1G polylactide acid (PLA) cup, 1G PLA + 2G cellulose fiber composite cup (1/2G), 2G bio-polyethylene (bio-PE) cup; fossil expanded polystyrene (EPS) box, 1G hemp box, virgin + recycled cardboard box (1/2G), and 2G barley straw box. The underlying data for Figure 5 can be found in Sheet A5(a) of Supporting Information S2.

TABLE 3 Critical count of usages at which the use phase becomes the dominant life cycle assessment phase for specific midpoints.

Midpoint	Fossil	1G	1/2G	2G	All cups
Climate change [kg CO_2 eq.]	23	**50**	43	*15*	50
Land use [Pt]	*12*	**72**	46	14	72
Particulate matter [disease inc.]	*14*	**41**	34	16	41
Water use [m^3 depriv.]	35	**147**	59	*16*	147
Maximum of all 16 from EF3.1	*70*	147	**252**	78	252

Bold numbers represent the highest values per midpoint, while italic ones are the smallest.

Concerning the insulation boxes, in the scenario of 100% recycling, no waste incineration takes place. This scenario is most significant for the EPS box and serves to identify a theoretical lower bound. Each of the four discussed midpoints improves by at least 8%, climate change even by 56%. Since the components of the straw box are already incinerated less frequently than the EPS box (since they can be disposed of as organic waste), the reduction is less pronounced (1% to 12%, depending on the midpoint).

For the 2G raw materials, we investigate replacing the allocation factor of zero with the ecoinvent default, discarding the supposed key selling point—and potential lever for greenwashing—of 2G products as substitutes for fossil or 1G ones. Compared to the base case without economic allocation, reducing environmental impacts by substituting EPS boxes with straw boxes decreases from 3.84 to 3.11 kg CO_2 eq. (−19%) and from 2.46 to 2.24 m^3 depriv. (−9%). Regarding land use, where the 2G box performed already worse than the EPS box, the allocation scenario means an increase of 9% compared to the default.

We also examine the impact of methodological choices in terms of biogenic CO_2. As discussed in Section 2.3, there are various ways to handle biogenic CO_2, with the most common being the "0/0" and "+1/−1" approaches. For an accurate comparison, we present the CO_2 impacts for the EF3.1 method (0/0) as well as "IPCC 2021 GWP100" without (0/0) and with CO_2 uptake (+1/−1) in Figures 58 and 59 of Supporting Information S1. The main finding is that the IPCC method with CO_2 uptake shows 33% lower climate change impacts than without CO_2 uptake and 45% lower impacts than EF3.1. Except for the 2G box, the +1/−1 method consistently results in lower impacts than EF3.1, suggesting that EF3.1 may overestimate impacts. However, the general trend of which product is least harmful remains unchanged. Additional details on the different LCIA methods can be found in Supporting Information S1.

Lastly, to examine the effects of different allocation methods, we analyze both the APOS default and the cut-off approach, detailed in Section 6.2 of Supporting Information S1. The results reveal that, in most cases, the differences between the two methods are small, especially when incineration rates at the EoL stage are high (such as for PLA and EPS, which have an incineration rate of 100%).

4 | DISCUSSION AND CONCLUSION

The results emphasize the environmental significance of many usages per product and closed material cycles, especially for petroleum-based ones. Regarding the environmental, social, or economic advantages of biogenic over fossil cups and insulation boxes (RQ1), we can conclude that the results do not support a universally valid statement, as this depends heavily on the product's characteristics and thus needs to be evaluated for each application case individually. 2G products show good potential to be promising sustainable alternatives across various impact categories compared to their fossil-based counterparts. However, products purely or partially based on first-generation materials face challenges primarily due to the significant environmental footprint of agricultural production and the constraints posed in their case of limited recyclability. Furthermore, high usage cycles are an essential lever overall. Regarding potential social hotspots, the regional sourcing of raw materials is a key driver. Lastly, the results show that biogenic raw materials have the lowest environmental and social risks if used as-is (i.e., without being used as a precursor for polymers).

Despite these obstacles, advancements in technology and innovative practices offer opportunities to enhance the sustainability of these products. Fostering collaborations between industries and investing in research and development can further propel the transition toward more environmentally friendly alternatives.

4.1 | Insights for a circular economy

Industrial ecology, focusing on material optimization, waste reduction, and life cycle assessment, is crucial in shaping a sustainable circular economy. In their white paper, Van Ewijk et al. (2023) provide a comprehensive knowledge synthesis, benefiting science-policy exchanges, education, and collaborative efforts across academia, policymakers, and industry. In the following, we address how our study positions itself regarding seven of the "ten insights from industrial ecology for the circular economy" stated in the study, namely:

(i) "Nature offers a model for the industry": The results show that the bioeconomy, particularly if the biogenic matter is only transformed moderately, can significantly reduce environmental pressure.
(ii) "Environmental impacts are inevitable": We show that a bio-based product does not automatically indicate a sustainable alternative, and even the highest recycling quotas and avoided burdens cannot outweigh the overall impacts from raw materials, production, transportation, and use. This is the case even when 0% of agricultural environmental impacts are allocated to 2G biomass. In contrast, in the case of PLA (1G cup), we see that with the lack of EoL treatment (e.g., due to lack of sortability or economically inviably small waste streams), bio-based products can have far higher environmental impacts than their fossil-based but well recyclable counterparts. Hence, no product possesses inherent sustainability; instead, one should focus on identifying products with reduced environmental and social risk to avoid greenwashing.
(iii) "A life cycle perspective avoids burden shifting": Given the diversity of the results, conducting a cradle-to-grave LCSA has proven to be a critical factor for avoiding burden shifting, be it across the environmental, social, and economic dimensions of sustainability or regarding the products' EoL vis-à-vis their cradle-to-gate impacts.
(iv) "Early systemic intervention prevents waste": As seen in Section 3.4, after 72 use cycles, the use phase dominates the other life cycle phases for all relevant midpoints for all types of cups, including the cup's primary material. This highlights the need to avoid single-use products first and push the number of usages as high as possible.
(v) "Location shapes environmental impacts": This is valid for all investigated products and sustainability dimensions and most markedly for raw materials acquisition.
(vi) "Technology is not a panacea": As exemplified with the 1G/2G cup, the technological advancement of being technically able to blend cellulose with PLA through advanced additives may seem like a sustainable solution for lowering the impacts of the cup's polymers but renders incineration instead of recycling almost inevitable. Furthermore, the externalities from raw material extraction, transportation, and EoL treatment far outweigh the impacts of electricity consumption during the production stage, strongly limiting the environmental savings potential even in the case of an electricity mix with 100% from renewables. Lastly, by investigating different ways of using biomass, we can conclude that it may be advantageous to consider "simpler" solutions, for example, creating insulation boxes directly from biomass without the detour of a bio-based polymer.
(vii) "The future is unknown but may be anticipated": Therefore, when devising the different characteristics (with a theoretical number of 24,192 possible scenario combinations), we often included improbable minimum and maximum values (e.g., 0% and 100% recycling quotas, and so on) to identify possible lower and upper bounds.

4.2 | Limitations

First, with the industry landscape in the region of Augsburg, Germany, and the limited number of companies participating in bioeconomic value chains, only the 2G (straw) insulation box could be realized with the locally available bioeconomic potential in the short term. Second, studies such as the one by Valdivia et al. (2021) highlight the need to establish standards for LCSA and its subordinate method of LCC. Based on the assumptions made and the availability of primary data, we deem the LCA and LCC models to be an overall accurate representation of the assessed products.

Due to the lower data availability compared to the LCA, the quantitative results of the S-LCA are primarily suited to hint at possible hotspots in the respective value chains but are less suited to allow for general statements on the product's overall social sustainability. On a qualitative level, we deem the presented LCSA and the conclusions that can be drawn from it to be valid overall. Naturally, however, despite employing reasonable methods and making firm assumptions (cf. Supporting Information S1), this study's findings are subject to uncertainty due to (Section 4.2.1) methodological constraints, (Section 4.2.2) limitations in available data, and (Section 4.2.3) reliance on assumptions.

4.2.1 | Methodological constraints

As demonstrated by the example of biogenic CO_2 emissions (see Section 3.4 and Section 6.1 of Supporting Information S1), the choice of the LCIA method and methodological assumptions can greatly affect the results. For example, the APOS and the cut-off approaches lead to different outcomes, making it important to consider how the chosen method influences the study's conclusions—even though we show (cf. Supporting Information S1) that in the application case at hand, the APOS and the cut-off models show only minor differences and lead to the same overall conclusions. This underscores the importance of the initial phase of an LCSA, which is defining the goal and scope of the study.

The S-LCA method, in contrast, has not yet been developed to the same level of methodological maturity as LCA. S-LCAs based on databases such as the SHDB rely on monetary values and are better suited for comparing the same products across different countries rather than different products within the same region. This is because the sectors of the underlying input–output models are not detailed enough to distinguish between, for example, renewable energy and coal power—both grouped under a single “electricity” sector. As a result, higher costs would always suggest higher social risks, which is questionable at best. Although the chosen methodology and the generic perspective do not allow for determining actual, materialized social risks, the assessment of the risk for socially adverse circumstances along upstream and downstream supply chains is a valuable tool for decision-makers, allowing them to identify possible hotspots in their supply chains and efficiently direct sustainability actions to the most critical (geographical or thematical) areas (Messmann et al., 2023).

Lastly, regarding the LCC, the monetization values listed in Table 2 of Supporting Information S1 warrant critical examination, as they need to be constantly updated and only provide robust results for a short period. Specifically, the cost of 1 ton of CO_2 (here: 102.50€ as central and 193.60€ as upper value) seems extraordinarily low, considering that existing estimates range from 195€ in 2020 (or 250€ in 2050; Umweltbundesamt, 2020), over 375€ (Ricke et al. 2018) up to 832€ (or 908€ in 2050; Umweltbundesamt, 2024).

4.2.2 | Limitations in available data

As for many LCSA studies, some background data are relatively dated and may not accurately represent the current practice. The statement that S-LCA studies suffer from less available data than LCA studies (Valdivia et al. (2021) is also true for the study at hand. However, owing to the primary data from industry partners, we deem our data quality to be good overall. Nevertheless, manufacturing *costs* could not be disclosed; the S-LCA is based on general market prices instead of primary cost data. Despite these choices, we deem our results to be robust and valid. The study's scope could be expanded to include factors like different waste compositions, demand shifts, cost dynamics, industry trends, and a consequential modeling perspective.

4.2.3 | Reliance on assumptions

The reliance on assumptions is often rooted in the limitations of available data. We use primary data whenever possible and make assumptions as accurate as possible based on the literature. However, there are natural limitations to how accurately the assumptions taken can mirror real-world applications or be transferred to other exemplary applications.

4.3 | Outlook

Due to the existing constraints, further investigations are required in various geographical areas and with different primary product systems. These studies should assess the sustainability of substituting fossil-based products with biogenic alternatives (1G and 2G) across diverse conditions and include, inter alia, ecosystem services for nature-based solutions. Based on existing studies in the field of, for example, second-generation bioethanol (Wietschel et al., 2021b), ecosystem services (Almenar et al., 2021, 2023), the raw material database developed by Wietschel et al. (2021a), and the present work, future research could develop a comprehensive framework for the utilization and prioritization of the locally existing bioeconomic potential. Based on the concept of planetary boundaries, such a framework for regional boundaries could include, for example, an optimization approach that operationalizes the potential multi-criteria and multi-stakeholder decision-making in the local bioeconomy.

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DATA AVAILABILITY STATEMENT

The data supporting this study's findings are available in this article's Supporting Information S1 and S2.

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ENDNOTES

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[2] The bioeconomic potential is the amount of second-generation biomass that can be withdrawn, considering legal, sustainability, and technological issues (Thorenz et al. 2018).

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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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