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# Environmental assessment of reusable plastic packaging systems: A systematic review of LCA research across industrial sectors

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A B S T R A C T

This paper presents a systematic review of life cycle assessment (LCA) studies evaluating the environmental performance of reusable plastic packaging compared with single-use alternatives. A structured search in Web of Science and Scopus identified 245 publications from 2000 to 2025, of which 55 met relevance, quality, and transparency criteria. These were grouped into original LCA studies and LCA review papers to support an unbiased assessment, and then analyzed to identify key application areas, methodological trends, and factors influencing environmental outcomes.

The review shows that polypropylene (PP), high-density polyethylene (HDPE), and polyethylene terephthalate (PET) are the most commonly assessed polymers in reusable systems. Although reusable packaging has been studied across several sectors, notable gaps persist in cosmetics, beverages, and delivery applications. In contrast, studies on food and fresh produce generally report environmental benefits when sufficient reuse cycles are applied. However, the comparability of existing findings is constrained by methodological inconsistencies, including narrow selections of impact categories, inconsistent end-of-life (EOL) modelling methodology, and omission of important life cycle stages such as backhaul transport and secondary packaging.

Two parameters emerge as central to determining the environmental performance of reusable packaging systems: the number of reuse cycles and the configuration of reconditioning activities, including washing, refilling, and return logistics. Reported reuse cycle assumptions vary widely across studies, highlighting the need for more realistic and application-specific definitions. Overall, the review underscores the potential environmental benefits of reusable plastic packaging while emphasizing the need for more consistent methodological choices to support reliable and comparable LCA outcomes.

## 1. Introduction

Packaging –as a critical component across all industries (Ait-Oubahou et al., 2019) – is the essential fusion of science, technology, and art. It serves a critical role in protecting products, extending shelf life, simplifying logistics, driving sales, and ensuring consumer satisfaction (Verma et al., 2021). The scale of this global market was demonstrated with a value of USD 1.08 trillion in 2024 and projections to grow to USD 1.45 trillion by 2032, representing a compound annual growth rate of 3.93%. In 2024, the Asia Pacific region was the dominant force, accounting for 38.43% of the global market share (Fortune Business Insights, 2025).

A packaging system usually consists of three layers. Primary packaging is in direct contact with the product and protects it from contamination and leakage. Secondary packaging helps with retail handling and bundling and has expanded rapidly with the rise of e-commerce, making it the largest packaging segment today (Fortune Business Insights, 2025). While tertiary packaging ensures that goods can be transported and stored efficiently, at the global scale, plastic packaging represented the largest material share, exceeding 590 million metric tons of consumption. This was followed by paper and paperboard at around 460 million metric tons, while glass and metal reached

*Abbreviations:* EOL, End-of-Life; LCA, Life Cycle Assessment; WOS, Web of Science; ABS, Acrylonitrile Butadiene Styrene; PET, Polyethylene Terephthalate; HDPE, High-Density Polyethylene; PP, Polypropylene; PBT, Polybutylene Terephthalate; EPE, Expanded Polyethylene; rPET, Recycled Polyethylene Terephthalate; rHDPE, Recycled High-Density Polyethylene; EPS, Expanded Polystyrene; HIPS, High-impact Polystyrene; PE, Polyethylene; rPE, Recycled Polyethylene; PUR, Polyurethane; PS, Polystyrene.

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roughly 300 million and 150 million metric tons, respectively (MG Market Growth Reports, 2026).

Consequently, this industry generates a significant waste stream. Reliable global figures are limited, but EU data give an indication of the scale: in 2023, the EU produced 79.7 million tonnes of packaging waste. Paper and cardboard were the largest fraction (40.4%), followed by plastics (19.8%), glass (18.8%), and wood (15.8%) (Eurostat, 2025).

To address these challenges, the EU has gradually built a regulatory framework around the waste hierarchy. While the concept dates back to the 1970s, the most influential policies emerged in the 1990s, especially the Packaging and Packaging Waste (PPW) Directive. It aims to reduce the environmental impact of packaging and ensure smooth functioning of the European market (Da Cruz et al., 2014). According to its targets, at least 70% of all packaging waste must be recycled by 2030 (eurostat, 2025). At the European level, this transition is being reinforced through policies such as the Waste Framework Directive(Waste Framework Directive, 2024) and the proposed Packaging and Packaging Waste Regulation (PPWR) (THE EUROPEAN PARLIAMENT, 2025). The PPWR further integrates sustainability criteria based on life cycle thinking by requiring packaging to be recyclable by design and at scale by 2035, introducing minimum recycled content targets, and supporting reuse and refill systems as part of the broader objective of achieving climate neutrality by 2050. In this context, reusable packaging systems are attracting increasing attention (Budhiraja et al., 2025).

Efforts to reduce the impacts of packaging have focused on several strategies, including material reduction, the use of recycled content, substitution with renewable materials, and improvements in design for recycling (Mudgal et al., 2024). These approaches align with the three central principles of the circular economy: reduce, reuse, and recycle (See Fig. 1).

Yet, recycling of plastic packaging is still constrained by high costs, technical limitations, and consumer behavior (Chen et al., 2021). Consequently, reusable packaging has gained increasing attention as a more effective route to reduce material consumption, energy use, and emissions (Coelho et al., 2020). Studies estimate that at least 20% of plastic packaging could be replaced by reusable systems (Lendal and Wingstrand, 2019; World Economic Forum, 2017). Policy actions have followed this trend: the Single-Use Plastics Directive targets various disposable items, including cutlery, coffee cups, stirrers, and straws, signaling a shift away from throw-away formats (European Parliament, 2019).

Reusable packaging is already used in various applications and forms, as indicated in Fig. 2. However, switching from single-use to reusable packaging often requires redesigning the supply chain to include reverse logistics (except in consumer-refill systems). This shift can change economic optima, stakeholder roles, logistics processes, and organizational structures. In this model, retailers act as a funnel, managing both the forward flow of products and the return flow of reusable packaging (Coelho et al., 2020).

Despite growing interest, academic research on reusable packaging remains uneven. Much of the literature focuses on takeaway food containers, cups, and boxes, with an emphasis on waste reduction more than on comprehensive life cycle assessments (Shokri et al., 2014). Furthermore, research beyond the food and beverage sector is scarce, leaving significant knowledge gaps for cosmetics, household products, agriculture, logistics, and e-commerce applications (Yadav et al., 2024).

This review aims to address these gaps by examining reusable plastic packaging across a broader range of industries. It provides an updated overview of the environmental assessment methods applied, the indicators used, and the findings reported to date. By identifying areas where evidence remains limited, the paper highlights key directions for future research.

This review is particularly relevant now, as the PPW Directive becomes more demanding and industries are under growing pressure to rethink packaging systems. The findings can help researchers identify open questions, and they can support policymakers and companies in making more informed decisions about product design, regulation, and investment in reuse systems. Ultimately, this work contributes to building a stronger evidence base for the transition toward more sustainable and effective reusable packaging.

## 2. Materials and methods

A systematic literature review was carried out using the Web of Science (WOS) and Scopus databases. The search was conducted in several stages to ensure that all relevant studies at the intersection of LCA and reusable plastic packaging were captured. The search strategy combined keywords reflecting both the methodology and the topic of interest. In WOS, a combined Semantic and Boolean search was used with the following search strings: TS = (life cycle assessment OR LCA OR environmental assessment OR environmental impacts) AND (reusable plastic packaging OR returnable plastic packaging OR refillable plastic packaging). Moreover, the literature search in Scopus was conducted using the following search strings: TITLE-ABS KEY ((reusable plastic packaging OR returnable plastic packaging OR refillable plastic packaging) OR (life cycle assessment OR LCA)).

As LCA has evolved substantially over past years, this paper focuses

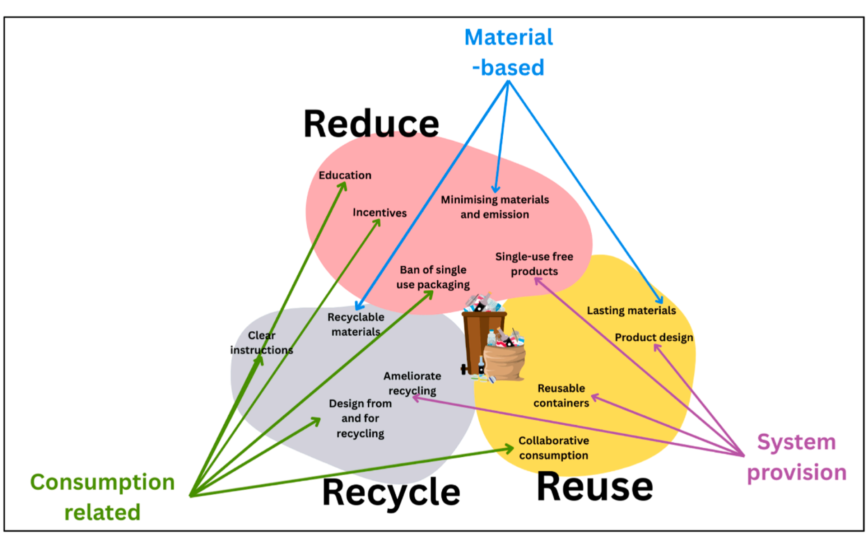



**Fig. 1.** Intervention in plastic and associated basic CE principle. The illustration is based on information from (Rabiu and Jaeger-Erben, 2024).


**Fig. 2.** Current developments of reusable packaging systems ((Coelho et al., 2020)).

on papers published between 2000 and 2025 in order to ensure methodological consistency and relevance. Earlier studies were excluded because LCA practice prior to 2000 often relied on non-standardized frameworks, limited databases, and less mature impact assessment methods, which may compromise comparability with more recent studies. Therefore, to avoid analysis of very early studies whose methodological limitations are likely already criticized and improved in subsequent research, the journal papers before 2000 are not considered in this review study. In this study, all the selected studies are peer-reviewed journal papers, published in English and the latest search to collect the related papers was conducted in August 2025. Fig. 3

The initial search yielded a total of 245 peer-reviewed publications: 198 from Web of Science and 47 from Scopus. After removing 36 duplicates, 209 papers remained. These publications were screened based on title, abstract, author keywords, and source. Papers were marked as relevant if they addressed at least one of the following themes:

- Types and applications of reusable plastic packaging used internationally.
- EOL treatment options for reusable plastic packaging.
- Environmental assessments of reusable plastic packaging.

Using these criteria, 141 papers were excluded. Most of which focused on topics outside the scope of this review, such as:

- Recycling of plastic packaging instead of reuse systems.
- Studies on single-use packaging rather than reusable systems.
- Refillable systems made from non-plastic materials (e.g., glass).
- Analyses of polymer chemistry without consideration of full packaging systems.
- Consumer behavior or social science-oriented studies.

The remaining 68 papers underwent a full-text review. During this more detailed analysis, an additional 13 papers were excluded due to the following reasons:

- Poor data quality: Studies which did not explain about life cycle inventory step, and did not mention the source of data used in the modeling clearly were excluded.
- Lack of transparency or failed to justify the basic assumption (e.g., unjustified modelling choices such as unclear system boundaries, impact assessment methods, or functional units).
- Unrelated scope of the study (e.g., focusing more on social aspects of reusable packaging or using a material type other than plastic for reusable packaging). It should be mentioned that this reason was also a main factor to exclude the paper in previous screening process, however, it was not obvious in title or abstract section therefor the authors overlooked them till this step.

Following all screening steps, 55 papers were identified as relevant and were included in the final analysis. These publications were used to address the following research questions:

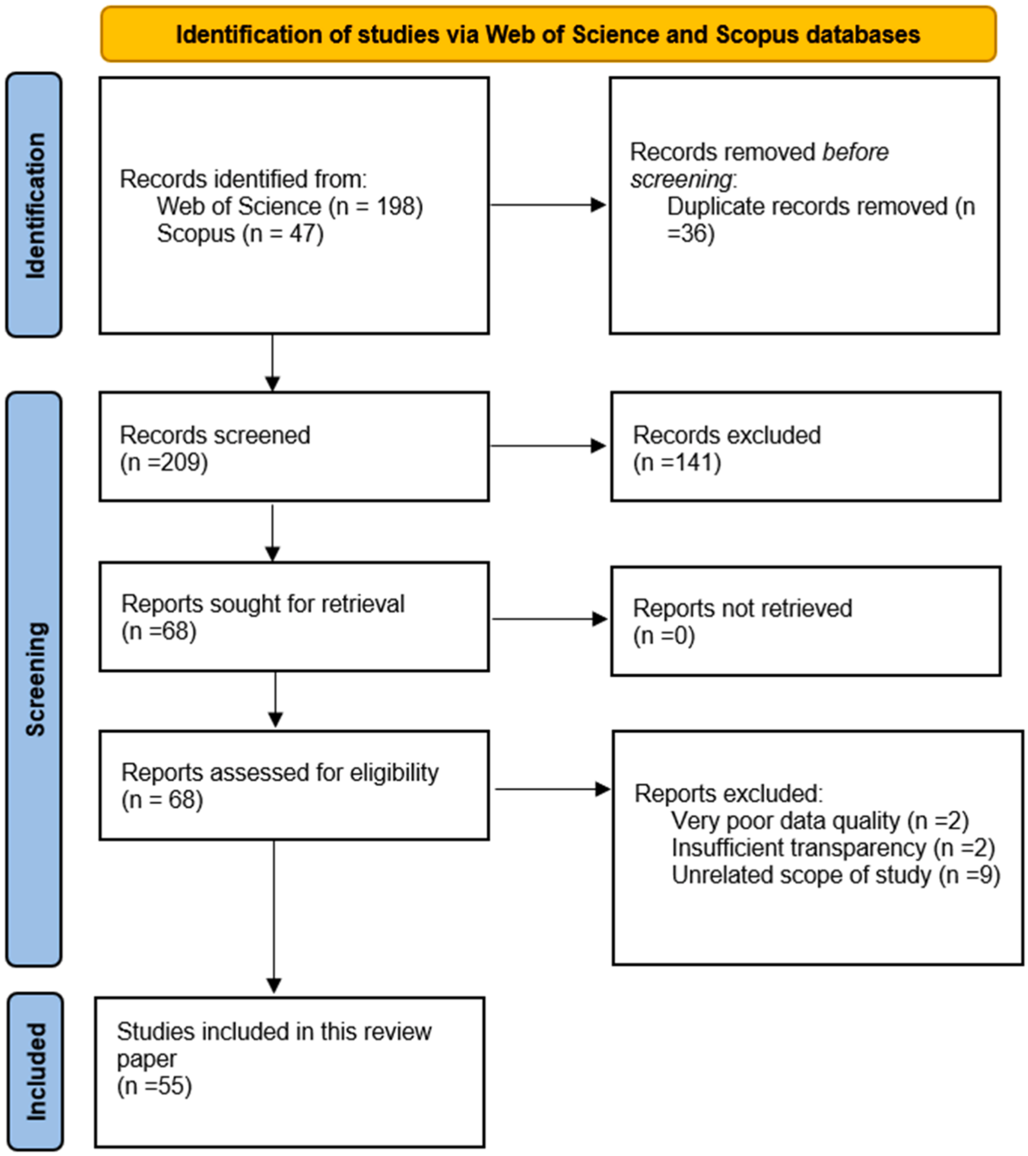
RQ1: What is the current state of environmental assessment for reusable plastic packaging?
RQ2: Which sectors or applications have been overlooked in existing environmental studies?
RQ3: What methodological choices are used in LCA studies of reusable plastic packaging?
RQ4: How comprehensive, reliable, and comparable are the existing LCA studies?
RQ5: How does the environmental performance of reusable plastic packaging compare with the representative single-use alternatives in different sectors?

To address the research questions, each of the 55 selected publications was analyzed in detail. The review examined the publication year, sectoral focus, and type of reusable packaging system. Methodological aspects were also assessed, including the scope of each LCA, the standards followed, the modeling tools used, and the technical features defined under ISO 14,040/44. These elements were evaluated to determine the overall robustness, transparency, and comparability of the studies.

To avoid bias in the assessment, the publications were first categorized into two groups:

(i) original LCA studies (34 publications).
(ii) LCA review papers (21 publications).

The first group of publications was used to address research questions 1, 2, 3, 4, and 5. These studies were evaluated based on the methodological quality of their LCA modeling in accordance with ISO 14,040/44. The assessment covered: (1) the reliability and transparency of the LCA, (2) the quality of inventory data sources, (3) the defined




**. 3.** PRISMA flow diagram of literature selection process in this study (Diagram template from (Page et al., 2021).

system boundaries and covered life cycle stages, (4) the impact assessment methods and selected impact categories, and (5) the presentation and interpretation of results.

The second group was used to address research questions 1 and 2 by examining the aim and scope of each review and identifying their main focus areas. This also helped avoid repetition by ensuring that the present review emphasized findings that were not covered in these existing review papers.

Finally, the findings were synthesized to evaluate the environmental advantages and drawbacks of reusable packaging across different industries, identify potential areas for improvement, and ultimately answer the research questions Fig. 4.

## 3. Environmental assessment studies on reusable plastic packaging

Publications assessing the environmental impact of reusable packaging systems in different industrial sectors and applications gained momentum only in 2020 onwards (see Fig. 5). The number of publications increased through years to reach 10 publications in 2024, with the expectation of obtaining more reliable publications in the current year (2025) due to growing interest and also establishment of new prohibitive laws in plastic packaging application. in this field. Analyzing the number of papers published in each specific industry reveals that the environmental analysis of packaging started to be recognized by researchers considerably after year 2019, with a stronger focus on food packaging and also packaging for fruit and vegetable products. Furthermore, there is a consequential increase in the number of review papers. This rise after 2019 can be explained by the growing political, industrial, and societal pressure related to plastic pollution, including new EU regulations, bans on single-use plastics, and corporate commitments to recycled content. These developments increased the urgency of conducting LCAs on packaging, especially food-related applications where material changes directly affect safety and shelf life.

To characterize the current state of scientific development on the environmental analysis of reusable packaging, the main industrial sectors represented in the 55 reviewed publications were examined. Nearly one-third of the studies (27%, 15 papers) focused on packaging for fruits and vegetables, making it the most extensively analyzed sector. Food product packaging emerged as the second most frequently studied category, accounting for 22% of the publications. Within this group, 11 studies assessed reusable packaging for general food products, while

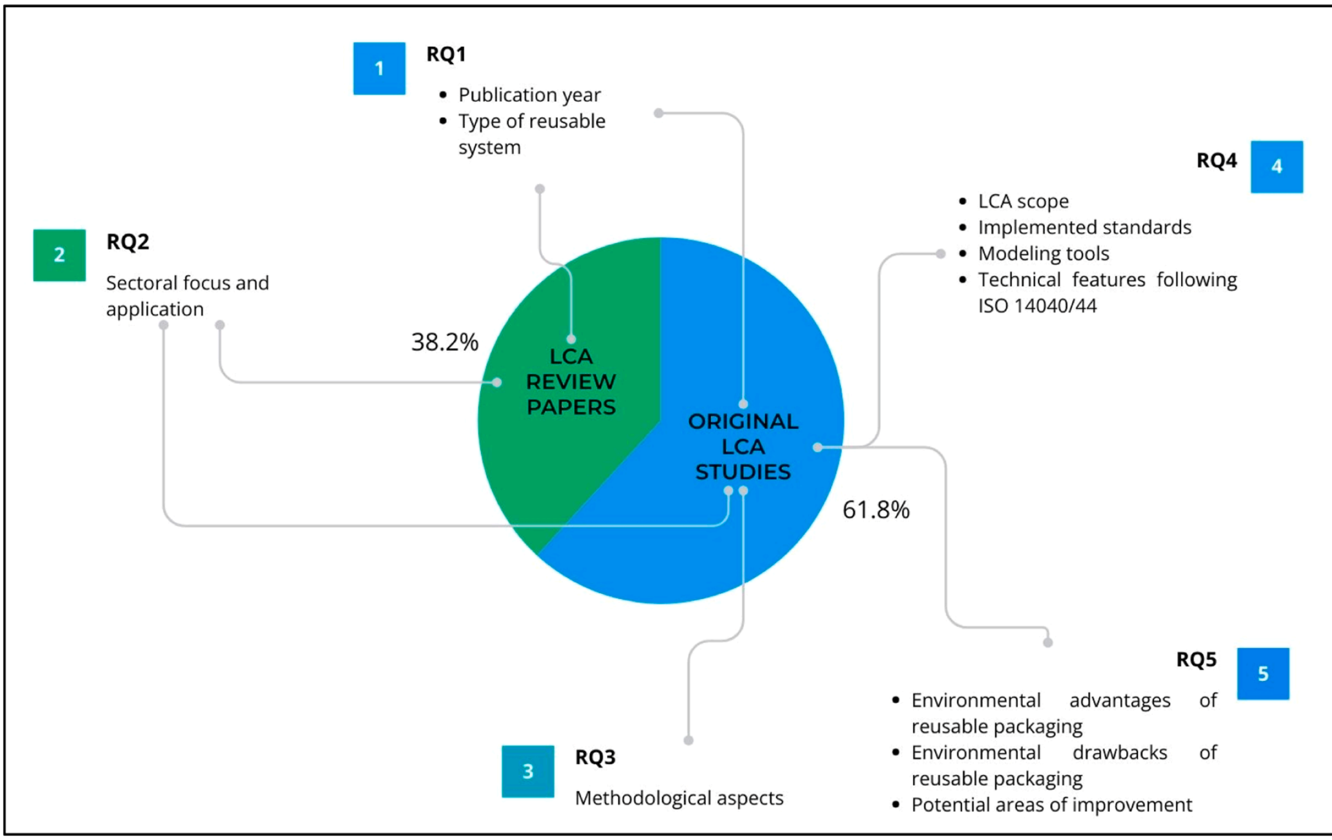
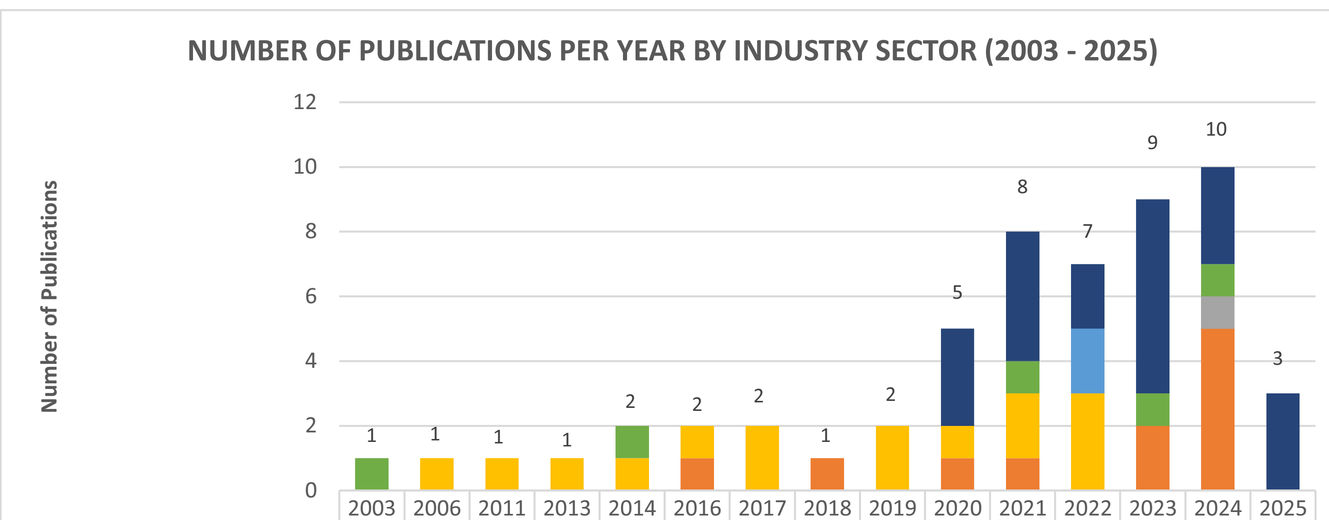



**Fig. 4.** Distribution of reviewed papers based on their alignment with the defined research questions.




| | 2003 | 2006 | 2011 | 2013 | 2014 | 2016 | 2017 | 2018 | 2019 | 2020 | 2021 | 2022 | 2023 | 2024 | 2025 |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Review papers | | | | | | | | | | 3 | 4 | 2 | 6 | 3 | 3 |
| Delivery packaging | 1 | | | | 1 | | | | | | 1 | | 1 | 1 | |
| Cosmetics products | | | | | | | | | | | | 2 | | | |
| Fruit & vegetable products | | 1 | 1 | 1 | 1 | 1 | 2 | | 2 | 1 | 2 | 3 | | | |
| Beverage products | | | | | | | | | | | | | | 1 | |
| Food Products | | | | | | 1 | | 1 | | 1 | 1 | | 2 | 5 | |

**Fig. 5.** Distribution of publications per year for the 55 reviewed studies (Note: the data for 2025 represent a partial year, as the literature search was conducted in August 2025).

only one study examined reusable packaging for beverages.

In contrast, significantly fewer studies addressed reusable packaging for delivery services (Camps-Posino et al., 2021; Koskela et al., 2014; Ross and Evans, 2003; Tan et al., 2023; Zhou et al., 2024) and cosmetics (Gatt and Refalo, 2022; Helmes et al., 2022). This limited attention highlights a clear gap in the existing literature and underscores the need for further research to address unresolved environmental questions within these underexplored sectors.

Publications under the second category (LCA review papers) were 21 papers in total. These were analyzed based on their focus. Among which, 12 publications aimed to review the current status of scientific development on LCA studies and presented the outcomes (Ahamed et al., 2021; Alhazmi et al., 2021; Deeney et al., 2023; Dolci et al., 2025; Espinoza-Orias and Lundquist, 2025; Gao et al., 2025; Gómez and Escobar, 2022; Kousemaker et al., 2021; Marson et al., 2023; Miller, 2022; Pålsson and Olsson, 2023; Sabate and Kendall, 2024), another 4 publications discussed the technical aspects of LCA modeling and methods to improve the modeling process (Fan et al., 2023; Nordahl and Scown, 2024; Pellengahr et al., 2023; Spierling et al., 2020), and 5 studies focused on studying the current status of circularity in plastic packaging field (Karayılan et al., 2021; Mahmoudi and Parviziomran, 2020; Mastellone, 2020; Schmidt and Laner, 2023; Shamsuddoha and Kashem, 2024).

**Table 1**
A summary of information extracted from reviewed papers.

| References | Software | Database | Data quality[a] | Standards | System boundary |
|---|---|---|---|---|---|
| (Gatt and Refalo, 2022) | SimaPro | Ecoinvent | **Medium** | ISO 14,040 and 14,044 | cradle-to-grave |
| (Helmes et al., 2022) | SimaPro | Ecoinvent | **Low** | ISO 14,040 and 14,044 | use-to-grave |
| (Thomassen et al., 2024) | SimaPro | Ecoinvent, Agribalyse | **Medium** | ISO 14,040 and 14,044 | cradle-to-grave |
| (Greenwood et al., 2021) | SimaPro | Ecoinvent, Industry data 2.0 | **Low** | ISO 14,040 and 14,044 | cradle-to-grave |
| (Ceballos-Santos et al., 2024) | SimaPro | Ecoinvent | **High** | ISO 14,040 and 14,044 | cradle-to-grave |
| (Kim et al., 2023) | SimaPro | Ecoinvent | **High** | Not mentioned | cradle-to-cradle |
| (Snyder and Park, 2024) | SimaPro | Ecoinvent | **Low** | ISO 14,040 and 14,044 | cradle-to-grave |
| (Hitt et al., 2023) | Self-developed model (using SimaPro database) | Ecoinvent | **Low** | Not mentioned | cradle-to-grave |
| (Battini et al., 2016) | Self-developed model | Ecoinvent | **Low** | Not mentioned | cradle-to-grave |
| (Yadav et al., 2024) | SimaPro | Ecoinvent | **Medium** | ISO 14,040 and 14,044 | cradle-to-grave |
| (Camps-Posino et al., 2021) | GaBi | GaBi | **Low** | ISO 14,040 and 14,044 | cradle-to-grave |
| (Tan et al., 2023) | GaBi | Ecoinvent | **Medium** | ISO 14,040 and 14,044 | cradle-to-grave |
| (Rasines et al., 2024) | SimaPro | Ecoinvent, ELCD | **Medium** | ISO 14,040 and 14,044 | cradle-to-grave |
| (Singh et al., 2006) | Self-developed model | Self-developed database | **Medium** | ISO 14,040 and 14,041, SETAC 15technical framework | cradle-to-grave |
| (Boschiero et al., 2019) | SimaPro | Ecoinvent | **High** | ISO 14,040 and 14,044 | post-harvest gate-to-gate |
| (Tua et al., 2019) | SimaPro | Ecoinvent | **High** | ISO 14,040 and 14,044, Product Environmental Footprint (PEF) Guide | cradle-to-grave |
| (Levi et al., 2011) | Self-developed model | Ecoinvent | **Medium** | ISO 14,040 and 14,044 | cradle-to-grave |
| (Tua et al., 2017) | SimaPro | Ecoinvent | **Medium** | ISO 14,040 and 14,044 | cradle-to-grave |
| (Accorsi et al., 2022) | SimaPro | Ecoinvent, ELCD | **High** | ISO 14,040 and 14,044 | cradle-to-grave |
| (Albrecht et al., 2013) | GaBi | GaBi | **Medium** | ISO 14,040 and 14,044 | cradle-to-grave |
| (Silva et al., 2024) | SimaPro | Ecoinvent | **High** | ISO 14,040 and 14,044 | cradle-to-gate |
| (Accorsi et al., 2014) | Manual calculation based on ISO 14,040 | Ecoinvent | **Medium** | ISO 14,040 and 14,044 | cradle-to-grave |
| (Zhou et al., 2024) | Manual calculation based on ISO 14,040 | Ecoinvent | **Medium** | ISO 14,040 and 14,044 | cradle-to-grave |
| (Alzubi et al., 2022) | OpenLCA | ELCD | **Medium** | ISO 14,040 and 14,044 | cradle-to-grave |
| (Abejón et al., 2020) | GaBi | GaBi | **Medium** | ISO 14,040 and 14,044 | cradle-to-grave |
| (Ross and Evans, 2003) | Manual calculation based on ISO 14,041 | Self-collected database | **High** | *ISO 14,040<br>*ISO 14,041<br>*ISO 14,042<br>*ISO 14,043 | cradle-to-grave |
| (López-Gálvez et al., 2021) | SimaPro | Ecoinvent | **Medium** | ISO 14,040 and 14,044 | cradle-to-grave |
| (Blanca-Alcubilla et al., 2020) | GaBi | GaBi | **Medium** | Not mentioned | cradle-to-grave |
| (Hilmarsdóttir et al., 2024) | SimaPro | Ecoinvent | **High** | ISO 14,040 and 14,044 | 1.Cradle-to-grave (for base case)<br>2.Cradle-to-cradle (for other scenarios) |
| (Koskela et al., 2014) | Manual calculation based on ISO 14,040 | Ecoinvent | **Medium** | ISO 14,040 and 14,044 | cradle-to-grave |
| (Sasaki et al., 2022) | MiLCA v2.0 | IDEA v2.0 | **Low** | Not mentioned | cradle-to-grave |
| (Bernstad Saraiva et al., 2016) | SimaPro | Ecoinvent | **Medium** | ISO 14,040 and 14,044 | cradle-to-grave |
| (Gallego-Schmid et al., 2018) | GaBi | GaBi<br>Ecoinvent | **Medium** | ISO 14,040 and 14,044 | cradle-to-grave |
| (Lo-Iacono-Ferreira et al., 2021) | SimaPro | Ecoinvent | **Medium** | *ISO 14,067<br>*ISO 14,044<br>*ISO 14,026<br>*ISO 14,071 | cradle-to-grave |

[a] Data quality was based on the source of collected data: High: primary data sources; Medium: mix between primary and secondary data sources; Low: only based on secondary data sources (e.g., ecoinvent database, another study and/or literature review).

## 4. Review of structural completeness and credibility

The quality and reliability of the implemented LCA model were assessed by first evaluating the general modeling structure for its comprehensiveness, accuracy, and transparency. Subsequently, key data indicative of the LCA study's overall status were collected. This process provided an unbiased perspective on the reliability of the studies and established an equitable basis for comparing the findings across different LCAs.

Table 1 summarizes the key features in all reviewed papers from the first category (original LCA studies). More than half of the studies (52%) used SimaPro to model the LCA and 80% of the studies selected ecoinvent database as their secondary data source. However, there were a few papers that either chose different software (e.g., Gabi or OpenLCA) and databases (e.g., Gabi, ELCD), or define their own calculation methods and database.

The quality of data was evaluated based on the type of data sources used. It should be emphasized that data quality is not representing the technical quality of LCA modeling but only assessing the quality of collected data based on data sources. Therefore, data were classified as high quality when primarily derived from primary data sources, medium quality when based on a combination of primary and secondary sources, and low quality when relying predominantly on secondary data sources. Using this classification, 24% of the analyzed original LCA studies were identified as high-quality data, while 56% were categorized as medium quality. Given the holistic nature of LCA and the requirement to collect data across multiple processes, life cycle stages, and scenarios, studies using high and medium quality data sources were considered to provide an acceptable level of data quality for this review. In contrast, 21% of the studies relied mainly on secondary data sources (e.g., Econinvent, other LCA study and/or another literature review). These were therefore categorized as low quality, introducing a greater degree of uncertainty into their results. This study characterization is included in Table 1.

85% of the original LCA studies (34 publications) clearly communicate the employed standards to be ISO 14,040 and 14,044. However, five studies apply an LCA framework that is consistent with ISO 14,040/14,044, but does not explicitly mention adherence to these standards.

The last column of Table 1 shows an overall view of the system boundaries for the reviewed studies to illustrate the level of comprehensiveness of the conducted study. 80% of the papers are defining a cradle-to-grave system boundary. This represents a good level of completeness because based on official definition, a cradle-to-grave system boundary covers the whole processes from production of a product to the EOL phase (e.g., disposal, reuse, or recycling) (Circular Ecology, 2023). Only one paper extends this system boundary to cradle-to-cradle (Kim et al., 2023). According to (Circular Ecology, 2023) in cradle-to-cradle system boundary, the product can be readily reused or recycled at the end of the initial life. There is also one study that limits its scope to the use-to-grave phase (with a main focus on EOL) (Helmes et al., 2022), and another that considers the gate-to-gate phase (focusing primarily on packaging and product storage) (Boschiero et al., 2019).

## 5. Critical analysis to the applied methods

### 5.1. Methodological characteristics

Table 2 presents the key technical features of the reviewed original LCA studies. In some cases, the applied approaches and selected impact categories were not explicitly reported; however, they could be inferred from the descriptions provided in the papers.

Several studies applied more than one impact assessment method, primarily because their research objectives require the evaluation of specific environmental impacts that are not fully covered by a single assessment method. For example, (Hitt et al., 2023) and (Boschiero et al., 2019) combined the IMPACT 2002 and CED 1.8 methods to

**Table 2**
Methodological characteristics of the reviewed papers: CC: climate change, OD: ozone depletion, IR: ionizing radiation, PHOF: photochemical ozone formation, PM: particulate matter, NCHT: non cancer human toxicity, CHT: cancer human toxicity, AC: acidification, FE: freshwater eutrophication, ME: marine eutrophication, TA: terrestrial acidification, TE: terrestrial eutrophication, FEC: freshwater ecotoxicity, TEC: terrestrial ecotoxicity, MEC: marine ecotoxicity, LU: land use, WU: water use, FR: fossil resource use, MMRU: mineral and material resource use, CED: Cumulative energy demand, HH: human health, ES: Ecosystems, TSW: Total solid waste, UAP: Urban air pollution.

| References | Impact assessment method | Impact assessment categories | EOL modelling approach |
|---|---|---|---|
| (Gatt and Refalo, 2022) | ReCiPe 2016-endpoint (H) | HH, ES | Cut-off |
| (Helmes et al., 2022) | ReCiPe 2016-midpoint (H) | 3 categories (CC, FR, MMRU) | Avoided burden |
| (Thomassen et al., 2024) | EF 3.0 [1] | 16 categories (CC, IR, OD, PHOF, PM, NCHT, CHT, AC, FE, ME, TE, FEC, LU, WU, FR, MMRU) | Avoided burden |
| (Greenwood et al., 2021) | ReCiPe 2016-midpoint (H) | 18 categories (CC, OD, IR, PM, PHOF (terrestrial ecosystems & human health), TA, FE, ME, TEC, FEC, MEC, NCHT, CHT, LU, MMRU, FR, WU) | Avoided burden |
| (Ceballos-Santos et al., 2024) | EF 3.1 | 6 categories (CC, AC, EF, ME, WU, FR) | Avoided burden |
| (Kim et al., 2023) | ReCiPe 2016-midpoint (H) | 18 categories (CC, OD, IR, PM, PHOF (terrestrial ecosystems & human health), TA, FE, ME, TEC, FEC, MEC, NCHT, CHT, LU, MMRU, FR, WU) | Cut-off (with partial energy-recovery credit for incineration) |
| (Snyder and Park, 2024) | TRACI 2.1 | 10 categories (OD, CC, AC, FE & ME, NCHT, CHT, FR, PHOF, FEC &MEC & TEC, PM) | Cut-off |
| (Hitt et al., 2023) | 1.IMPACT 2002+v2.15 2.IPCC 2021 3. ReCiPe 2016-midpoint (H) | 3 categories (CED, CC, WU) | Cut-off |
| (Battini et al., 2016) | EPD (SEMC)[2] | 1 category (CC) | Cut-off |
| (Yadav et al., 2024) | ReCiPe 2016-midpoint (H) | 18 categories (CC, OD, IR, PM, PHOF (terrestrial ecosystems & human health), TA, FE, ME, TEC, FEC, MEC, NCHT, CHT, LU, MMRU, FR, WU) | Avoided burden |
| (Camps-Posino et al., 2021) | EF 3.0 | 1 category (CC) | Avoided burden |
| (Tan et al., 2023) | CML 2001 | 9 categories (CED, PM, FE & TE, ME, AC, CC, EP, FEC, MEC) | Avoided burden |
| (Rasines et al., 2024) | EF 3.1 | 8 categories (CC, PHOF, PM, AC, FE, LU, FR, MMRU) | Hybrid (50/50 allocation) |
| (Singh et al., 2006) | Not clarified | 3 categories (CC, CED, TSW) | Cut-off (with partial energy- |

(continued on next page)

**Table 2** (*continued*)

| References | Impact assessment method | Impact assessment categories | EOL modelling approach |
|---|---|---|---|
| | | | recovery credit for incineration) |
| (Boschiero et al., 2019) | 1.IPCC 2007 2. CED 1.8 | 2 categories (CC, CED) | Out of scope of system boundary |
| (Tua et al., 2019) | 1. ILCD[3] 2. CED 1.8 | 15 categories (CC, OD, NCHT, CHT, PM, PHOF, AC, TE, FE, ME, FEC, MMRU, FR, WU, CED) | Avoided burden |
| (Levi et al., 2011) | EPD (SEMC) | 6 categories (CC, OD, AC, FE& ME & TE, FR & MMRU, PHOF) | Avoided burden |
| (Tua et al., 2017) | ILCD | 14 categories (CC, AC, OD, PHOF, TE, FE, ME, FEC, NCHT, CHT, PM, WU, MMRU, CED) | Avoided burden |
| (Accorsi et al., 2022) | 1.CML 2002 2.Eco-indicatr 99 (H) 3. ReCiPe 2016-midpoint (H) | 11 categories (CC, OD, NCHT& CHT, FEC& MEC& TEC, PHOF, AC, FE & ME & TE, FR, MMRU, LU, WU) | Cut-off |
| (Albrecht et al., 2013) | CML 2002 | 6 categories (CC, AC, PHOF, MMRU & FR, CED, FE & ME) | Avoided burden |
| (Silva et al., 2024) | 1. CML-IA baseline, 2. the AWARE 3. ReCiPe 2008 midpoint | 8 categories (CC, AC, FE & ME, PHOF, WU, MMRU & FR, OD) | Not modelled (Out of scope) |
| (Accorsi et al., 2014) | IPCC 2007 | 1 category (CC) | Avoided burden |
| (Zhou et al., 2024) | ReCiPe 2016-midpoint (H) | 11 categories (CC, FE, FEC, MEC, TEC, CHT & NCHT, PHOF, OD, LU, AC, PM) | Avoided burden |
| (Alzubi et al., 2022) | Not specified | 9 categories (AC, CC, FEC, FE, LU, OD, PM, WU, MMRU) | Not modelled (Out of scope) |
| (Abejón et al., 2020) | 1.CML 2001 2.IMPACT 2002+v2.1 | 7 categories (CC, OD, AC, FE& ME& TE, PHOF, CED, WU) | Avoided burden |
| (Ross and Evans, 2003) | Manual calculation | 3 categories (FR, CC, PHOF) | Avoided burden |
| (López-Gálvez et al., 2021) | 1.CML 2.CED | 11 categories (CC, OD, PHOF, MMRU, AC, FE & ME & TE, FEC, MEC, TEC, CHT, CED) | Avoided burden |
| (Blanca-Alcubilla et al., 2020) | EF | 1 category (CC) | Only landfill emissions |
| (Hilmarsdóttir et al., 2024) | CML | 11 categories (MMRU, FR, CC, OD, CHT, FEC, MEC, TEC, PHOF, AC, FE & ME & TE) | EOL Recycling |
| (Koskela et al., 2014) | ReCiPe 2011-midpoint (H) | 7 categories (CC, AC, FE, PHOF, PM, LU, FR) | Avoided burden |
| (Sasaki et al., 2022) | LIME2 (endpoint method) | 13 categories (CC, PHOF, AC, OD, FE & ME & TE, LU, CHT, FEC & MEC & TEC, WU, CED, MMRU, TSW, UAP) | Only landfill and incineration emissions |
| (Bernstad Saraiva et al., 2016) | ILCD | 11 categories (CC, OD, PHOF, AC, FE, | Avoided burden |
| | | ME, FEC, MEC, TEC, CHT, NCHT) | |
| (Gallego-Schmid et al., 2018) | CML 2001 | 12 categories (CED, MMRU, FR, AC, FE & ME & TE, CC, CHT, FEC, MEC, TEC, OD, PHOF) | Avoided burden |
| (Lo-Iacono-Ferreira et al., 2021) | IPCC 2007 | 1 category (CC) | Avoided burden |

[1] The Environmental Footprint indicator set.
[2] Environmental Product Declaration (EPD) method recommended by the Swedish Environmental Management Council (SEMC).
[3] International Reference Life Cycle Data System.

quantify the cumulative energy demand (CED), as their main impact assessment methods did not include this category. The IMPACT 2002 apply a combined midpoint and damage approach to link all LCA results to four damage categories via 14 midpoint impact categories (Jolliet et al., 2003). The CED 1.8 methods, on the other hand, calculates the direct and indirect energy used during the life stages of a product and categorizes into renewable and non-renewable energy sources (Boschiero et al., 2019). The consistency of applying multiple methods was evaluated in accordance with ISO 14,044 (International Organization for Standardization, 2006b), which states that if the selected characterization model is insufficient to meet the defined goal and scope, additional methodologies may be adopted. Therefore, the use of multiple impact assessment methods is fully aligned with ISO 14,044.

Except for the study by Singh et al. (2006) – which is a life cycle inventory study and only quantifies the $CO_{2\text{-eq}}$ emissions – and the study by Alzubi et al. (2022) – which did not clearly mention the impact assessment method – all other papers explained the selected impact assessment method, the chosen impact categories and the EOL modelling approach. The review shows that these methodological choices have a substantial influence on the reliability and accuracy of the LCA results. For example, Yadav et al. (2024) demonstrated that the interpretation of LCA outcomes strongly depends on the selected impact categories. In their study, although some categories—such as climate change—decreased for reusable PP food packaging, other categories increased. Specifically, the reusable container, when used 10, 30, or 100 times under different EOL scenarios (recycling and incineration), showed higher impacts for marine eutrophication, freshwater eutrophication, ionizing radiation, human carcinogenic toxicity, mineral resource scarcity, and human non-carcinogenic toxicity compared with single-use alternatives. These increases were primarily attributed to emissions from the washing stage.

In line with ISO 14,044 (International Organization for Standardization, 2006b), the selection of impact categories must be justified and consistent with the study's goal. The methodology used by Rasines et al. (2024) provides a good example of a transparent and scientifically grounded selection process, where the authors applied a weighting analysis and retained only those impact categories with a weighting score above 80%.

Kim et al. (2023) reported LCA results for 18 impact categories; however, the interpretation focused primarily on climate change. Their comparison of environmental performance across categories revealed that water consumption increased substantially in one of the reusable packaging scenarios, offering no environmental advantage over single-use packaging. Notably, the study showed that replacing recycled aggregates with virgin aggregates significantly reduce the water consumption of the reusable option. This finding underscores not only the importance of selecting impact categories fairly but also the need for a holistic perspective in LCA studies to capture diverse environmental dimensions of a product or service. In this study (Kim et al., 2023), considering different impact categories through a holistic system boundary allowed highlighting the negative aspects of recycling processes and also recognized that "material production" and "reconditioning transportation" processes cause the most considerable environmental impacts for respectively 1 and 300 cycles. This shows that 1. recycling practice is not always the most preferable waste management strategy for all impact categories, 2. the selected life cycle stages in the system boundary plays an important role on the calculated environmental impacts.

Overall, a considerable number of the reviewed papers restricted their analysis to a limited range of impact categories, most often climate change (see Appendix B). Although some studies reported results for a wider set of categories, they did not provide further interpretation or discussion of these results (Ceballos-Santos et al., 2024; Greenwood et al., 2021; Helmes et al., 2022).

It is important to note that several authors acknowledged these limitations. For instance, Boschiero et al. (2019) clarified that their conclusions were based solely on the selected impact categories and did not represent the full set available within the chosen impact assessment method. In contrast, other studies justified their selection of impact categories based on the stated goals of the research, following ISO 14, 044. A representative example is the study by Camps-Posino et al. (2021), focusing exclusively on $CO_{2\text{-eq}}$ emission of the modelled product. Although aligned with the study's goal, this work is more accurately characterized as a carbon footprint analysis rather than a full LCA, as it considered only one impact category Fig. 6.

Moreover, the selection of the EOL modeling approach represents an additional source of methodological inconsistency among the reviewed studies. Before evaluating how EOL modeling was applied in the analyzed papers, a brief explanation of the two most commonly used EOL approaches is provided in this section. To calculate the environmental impacts of a EOL stage of a product or service, cut-off and avoided burden approach can be employed which are common and well-developed in LCA studies. Cut-off approach considers no burden from the activities beyond the life cycle of the product or service. This means the burden of processes like raw material extraction, transportation, or use phase are attributed to the product or service, but burden of recycling activities is assigned to the product that used the recycled aggregates in some other system boundaries. On the other hand, avoided burden approach allocates the burden of recycling activities to the product generating the recycled materials (Hermansson et al., 2022).

A good practice for EOL modeling can be described as follow:

- First, the selected EOL approach (cut-off or avoided burden) should be explicitly justified based on the goal of study.
- Studies are encouraged to report results for both EOL approaches whenever possible, in order to evaluate the sensitivity of comparative conclusions to allocation choices.
- Assumptions regarding reuse cycles and the practices related to recycling at the end of the final use cycle should be transparently documented and consistently applied.
- System boundaries should be clearly defined to ensure that all relevant reuse-related processes, such as reverse logistics, washing, and redistribution, are included.
- The implications of recycling credits and waste management modelling on comparative results should be explicitly discussed to avoid biased interpretations in favor of either reusable or single-use systems.

The overview of the EOL modeling approaches presented in Table 2 shows that most of the reviewed studies adapted the avoided burden approach. Under this approach, the burdens and credits associated with waste management processes—such as recycling, landfilling, and incineration—are included within the system boundary of the LCA. In contrast, some studies applied a cut-off approach, in which no burdens or credits from waste management processes are assigned to the

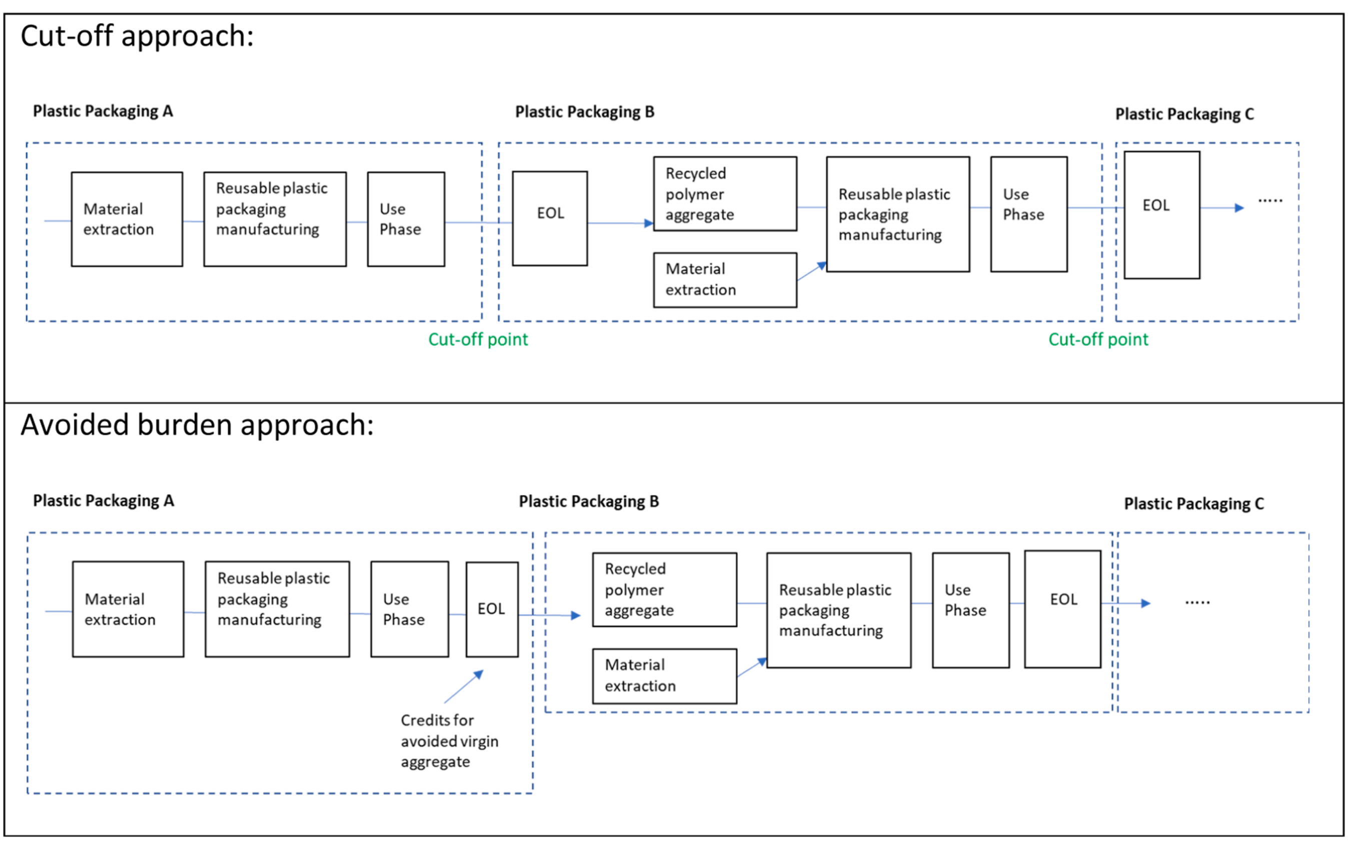



**Fig. 6.** Visualized Illustration of cut-off and avoided burden approach (Insight from(Husmann et al., 2025)).

modeled packaging. This fundamental difference in modeling the EOL stage, which is inherently a multi-output process, can lead to substantial variation in LCA results and, consequently, may result in divergent or even misleading interpretations and conclusions.

ISO 14,040 (International Organization for Standardization, 2006a) provides clear guidance for modeling multi-output processes. Section 4.3.4.1 of this standard states that "whenever several alternative allocation procedures seem applicable, a sensitivity analysis shall be conducted to illustrate the consequences of the departure from the selected approach." The same section further recommends avoiding allocation altogether by applying system expansion (i.e., the avoided burden approach) or subdivision. Additionally, Section 4.3.4.3 specifies that these allocation procedures apply to EOL processes, particularly recycling and reuse.

Therefore, when modeling the EOL stage of a product, sensitivity analyses are essential to assess whether the choice of EOL modeling approach significantly influences the LCA results. Despite this requirement, as shown in Table 2, this analysis was not identified in the reviewed studies. However, some studies applied scenario analyses by varying EOL parameters such as recycling or disposal shares (e.g., Camps-Posino et al., 2021; Yadav et al., 2024). none of the reviewed papers performed such analyses. In most cases, the selected EOL methodology was merely stated, without strong justification for the choice and without conducting sensitivity analyses to verify that the selected approach did not substantially affect the final results.

### 5.2. Coverage of the life cycle stages

Because the inclusion or omission of life cycle stages in reusable plastic packaging systems can significantly influence environmental performance outcomes, Table A. 1 in appendix A reviews the extent to which the analyzed studies covered the relevant phases of the life cycle. The considered stages were: raw materials extraction, packaging production, secondary and tertiary packaging production, logistics, reconditioning, and EOL treatment methodology. Most of the analyzed studies did not consider all the key life cycle stages in their assessment. In some cases, however, excluded stages were justified appropriately. For instance, Gatt & Refalo (2022) omitted backhaul transportation because the reusable blush powder case they modeled allows the consumer to replace the powder pan at home, eliminating the need for return transport. Similarly, Snyder & Park (2024) excluded backhaul transportation in their study of reusable food packaging on a university campus. In this context, students are already present on campus for reasons unrelated to dining, so additional return trips for packaging are not part of the system under study.

However, several publications (29 out of 34 studies, 85% of the reviewed studies) excluded life cycle stages that are relevant and should not have been omitted—such as the production of secondary and tertiary packaging—thereby reducing the accuracy of the results. Koskela et al. (2014) demonstrated that secondary packaging, together with packaging weight and transportation distance, can significantly influence the environmental performance of packaging systems. Excluding such stages therefore risks misrepresenting the true environmental impacts of reusable packaging.

As indicated in Fig. 7, 85% of the reviewed studies (29 out of 34) included the raw material extraction stage of the primary packaging in the LCA models. The five studies that excluded this stage did not provide a scientific justification. In contrast, the packaging manufacturing stage was included in 100% of the reviewed studies, while the inclusion rate drops to 24% for secondary or tertiary packaging production and 82% for reconditioning stages (e.g., washing, filling).

Logistics were modelled in all reviewed papers except in Tua et al. (2019), Levi et al. (2011), and Silva et al. (2024), where transportation was only partially included (e.g., limited to transport from manufacturer to warehouse), due to data limitations or without clear justification. This is notable given that several studies emphasized the significant contribution of transportation to overall environmental performance (Accorsi et al., 2014; Battini et al., 2016; Hitt et al., 2023; Rasines et al., 2024; Snyder and Park, 2024; Tan et al., 2023).

While 76% of reviewed studies are considering backhaul transportation stage, its importance is explicitly demonstrated by Hitt et al. (2023) study, who demonstrated that the environmental advantages of reusable plastic packaging are highly sensitive to assumptions regarding backhaul transport. Their analysis showed that if as little as 5% of customers make dedicated trips solely to return containers, the reusable system can exceed the life cycle greenhouse gas emissions and primary energy use of single-use alternatives.

The EOL stage of the packaging is predominantly included and clearly described across the reviewed papers. However, Boschiero et al., 2019 and Silva et al., 2024 excluded this stage from the study as it was

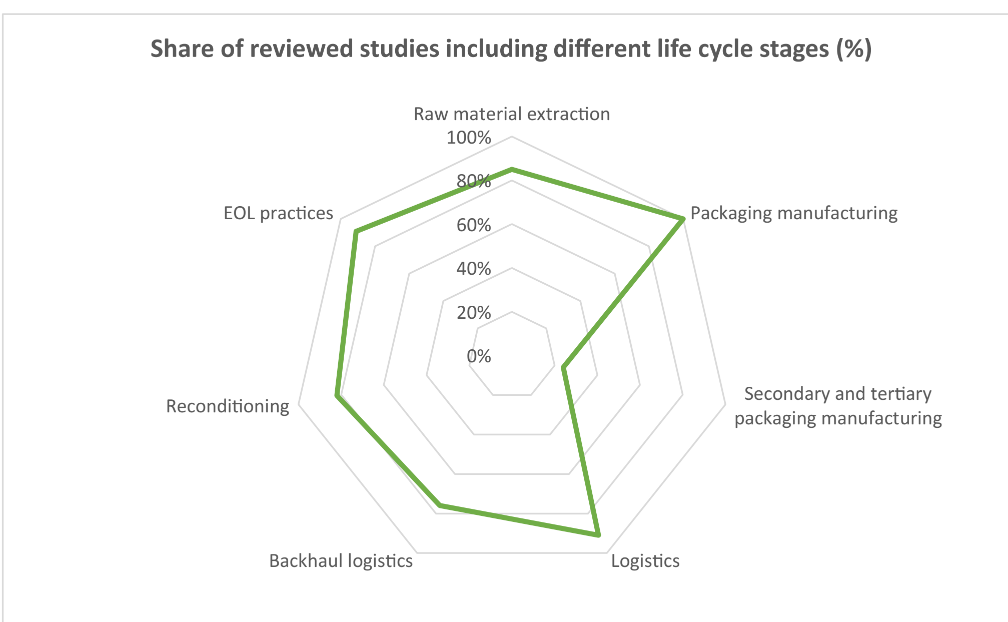



**Fig. 7.** Percentage of reviewed studies including different life cycle stages in the system boundary.

out of the study's scope. While the study by Alzubi et al., 2022 did not consider this stage, despite its significance. Nevertheless, the methodological approaches used to model this stage are often insufficiently applied. The main scenarios for EOL stage are often landfilling, incineration, or recycling which is usually justified through national statistics (Kim et al., 2023; Snyder and Park, 2024) or reference to existing literature (Thomassen et al., 2024). Several studies considered the distribution of waste flows as a varying parameter in the scenarios to

**Table 3**
Overview of findings from reviewed papers comparing reusable and single-use plastic packaging.

| References | Material type | Application | R>S[1] | R≈S[2] | R<S[3] | Reuse cycles | Break-even reuse cycles |
|---|---|---|---|---|---|---|---|
| (Helmes et al., 2022) | PET + rPET<br>HDPE + rHDPE | Cosmetic packaging | | | ✓<br>✓ | 1 time | Not specified<br>Not specified |
| (Thomassen et al., 2024) | PP | Food packaging | ✓ | | | 5 times | *22 times for FR<br>*16 times for CC |
| (Greenwood et al., 2021) | PBT<br>PP | Food packaging | | | ✓<br>✓ | 50 times | 4<br>4 |
| (Ceballos-Santos et al., 2024) | HDPE | Food packaging | | ✓ | | 1260 times | Not specified |
| (Kim et al., 2023) | rPET<br>EPE | Food packaging | | | ✓<br>✓ | 300 times | 7<br>12 |
| (Snyder and Park, 2024) | PP | Food packaging | | | ✓ | 300 times | 17 |
| (Hitt et al., 2023) | PP | Food packaging | | | ✓ | 20 times | 4 for CC<br>13 for CED |
| (Battini et al., 2016) | PP | Food packaging | ✓ | | | various (15, 30, 45 and 60 times) | *for short distance 30 times<br>*for long distance not reached within the study |
| (Yadav et al., 2024) | PP | Food packaging | | | ✓ | various (10, 30 and 100 times) | 6 times |
| (Camps-Posino et al., 2021) | PP | Delivery packaging | | | ✓ | 50 times | Not specified |
| (Tan et al., 2023) | PP | Delivery packaging | | | ✓ | 50 times | Not specified |
| (Rasines et al., 2024) | PP | Fruit & vegetables packaging | | | ✓ | 140 times | *5 times compared to single-use wooden crate<br>*29 times compared to single-use cardboard crates |
| (Singh et al., 2006) | PP + rPP | Fruit & vegetables packaging | | | ✓ | Not specified | Not specified |
| (Boschiero et al., 2019) | HDPE | Fruit & vegetables packaging | | | ✓ | Not specified | Not specified |
| (Tua et al., 2019) | PP + rPP | Fruit & vegetables packaging | | | ✓ | various (1–125) | 3 times |
| (Levi et al., 2011) | PP | Fruit & vegetables packaging | ✓ | | | 200 times | Not specified |
| (Tua et al., 2017) | PP + rPP | Fruit & vegetables packaging | ✓ | | | 50 times | Not specified |
| (Accorsi et al., 2022) | PP + rPP | Fruit & vegetables packaging | | | ✓ | 100 times | 15 times |
| (Albrecht et al., 2013) | PP | Fruit & vegetables packaging | | | ✓ | 50 times | *40–60 times compared to single-use wooden d box<br>*5–15 times compared to single-use cardboard box |
| (Silva et al., 2024) | PET + rPET | Beverage packaging | | | ✓ | Not specified | 2 times |
| (Accorsi et al., 2014) | PP | Fruit & vegetables packaging | | | ✓ | various (30, 50, and 70 times) | Not specified |
| (Zhou et al., 2024) | *PET+PE for returnable bags<br>*PP for returnable boxes | Delivery packaging | | | ✓ | 30 times | *15 times for returnable bags<br>*7 times for returnable boxes |
| (Alzubi et al., 2022) | PP + rPP | Fruit & vegetables packaging | | | ✓ | Not specified | Not specified |
| (Abejón et al., 2020) | PP + rPP + HDPE + rHDPE | Fruit & vegetables packaging | | | ✓ | 100 times (conservative scenario)<br>150 times (Technical scenario) | Not specified |
| (Ross and Evans, 2003) | EPS+ HIPS+PE | Delivery packaging | | | ✓ | Not specified | Not specified |
| (López-Gálvez et al., 2021) | PP | Fruit & vegetables packaging | | | ✓ | 150 times | 15 times |
| (Blanca-Alcubilla et al., 2020) | ABS + PS + PP | Food packaging | ✓ | | | 10 times | Not specified |
| (Hilmarsdóttir et al., 2024) | HDPE + rHDPE + PUR | Food packaging | | | ✓ | 96 times | Not specified |
| (Koskela et al., 2014) | HDPE | Delivery packaging | ✓ | | | 700 times | Not specified |
| (Sasaki et al., 2022) | PP | Fruit & vegetables packaging | ✓ | | | 100 times | Not specified |
| (Bernstad Saraiva et al., 2016) | rHDPE + PS | Fruit & vegetables packaging | ✓ | | | Not specified | *Min: 4 times (short distance) for CC<br>*Max: 35 times (long distance) for CC |
| (Lo-Iacono-Ferreira et al., 2021) | PP | Fruit & vegetables packaging | ✓ | | | various (20, 50, and 100 times) | Not specified |

[1] Reusable > Single-use.
[2] Reusable ≈ Single-use.
[3] Reusable < Single-use.

study the impact of different EOL scenarios on the environmental performance of the packaging (Camps-Posino et al., 2021; Gatt and Refalo, 2022; Yadav et al., 2024). Nevertheless, a few studies only assume the distribution of waste flow among different scenarios of recycling, incineration or landfilling without providing scientific justification (Tan et al., 2023).

Given the substantial contribution of processes such as transportation and EOL to overall LCA results, and their strong influence on the interpretation of environmental performance, the exclusion of specific processes may systematically bias results in favor of either reusable or single-use packaging systems. For example, excluding raw material extraction or primary packaging production tends to favor reusable packaging, as reusable containers are typically heavier and more material-intensive due to design requirements for durability and multiple use cycles.

Similarly, the omission or partial modelling of transportation processes may bias results in favor of reusable systems, since additional transport activities (e.g., transport to washing or reconditioning facilities and return logistics) are specific to reuse systems. Although these additional transport burdens may be partly offset by transportation processes associated with single-use packaging, such compensation depends on an optimally designed and efficiently operated reuse system.

Moreover, simplified or insufficiently justified EOL modelling may bias results in either direction, depending on whether recycling credits and disposal burdens are over- or underestimated under avoided burden or cut-off approaches. Overall, incomplete system boundaries can lead to systematic distortions in comparative LCA results, highlighting the importance of comprehensive and transparent modelling to avoid favoring either reusable or single-use systems by construction.

## 6. Environmental analysis of reusable plastic packaging in comparison to single-use plastic packaging

### *6.1. Most common polymers used in reusable plastic packaging*

Across the reviewed literature, a range of material types has been used to manufacture reusable plastic packaging. This section evaluates the environmental performance of reusable plastic packaging produced from the three most commonly assessed polymers: PP, HDPE, and PET., consequently the results of other polymer materials (e.g., EPC, HIPS and PE from (Ross and Evans, 2003)) are not discussed in this section. It should be noted that studies assessing hybrid common polymer systems are considered in all relevant sections. For example, the study by (Helmes et al., 2022), which evaluates the environmental impacts of both PET and HDPE reusable packaging, is discussed in both the PET and HDPE sections (double-mentioned). This approach allows the environmental performance of each polymer to be analyzed separately. Table 3 summarizes the comparative environmental impacts of these materials relative to their single-use counterparts. The table was compiled by drawing on the dominant conclusions for the selected impact categories presented in Table 2 to determine whether reusable packaging outperformed single-use packaging overall. While certain studies identified scenarios in which a specific packaging type may perform better, such cases were not representative of the majority of modeled conditions and were therefore treated as exceptions.

For consistency and comparability, two studies (Gallego-Schmid et al., 2018; Gatt and Refalo, 2022) were excluded from this assessment. The former was omitted because its LCA results focused primarily on endpoint impact categories (human health and ecosystems), which are not aligned with the midpoint-focused assessments in the remaining studies. The latter was excluded because it compared two reusable systems (plastic and glass) rather than comparing reusable plastic packaging with single-use alternatives, and therefore did not provide a compatible basis for analysis.

a. PP: This polymer type is the most commonly used material for reusable packaging production (Accorsi et al., 2014). Among the 32 reviewed studies, 22 considered PP or a mix of PP with other polymers as the primary material for reusable plastic packaging and compared their environmental performance with single-use alternatives ((Abejón et al., 2020; Accorsi et al., 2014, 2022; Albrecht et al., 2013; Alzubi et al., 2022; Battini et al., 2016; Blanca-Alcubilla et al., 2020; Camps-Posino et al., 2021; Greenwood et al., 2021; Hitt et al., 2023; Levi et al., 2011; Lo-Iacono-Ferreira et al., 2021; López-Gálvez et al., 2021; Rasines et al., 2024; Sasaki et al., 2022; Singh et al., 2006; Snyder and Park, 2024; Tan et al., 2023; Thomassen et al., 2024; Tua et al., 2017, 2019; Yadav et al., 2024; Zhou et al., 2024)). The overview of the environmental performance of reusable PP plastic packaging, provided in Table 3, shows that this material generally performs better than single-use packaging from an environmental perspective. This advantage can be attributed to the lower emissions during the raw material production stage, as reusable packaging can be utilized multiple times (Zhou et al., 2024).

However, some studies reported a deterioration in the environmental performance of reusable pp packaging compared with single-use alternatives. For example, Thomassen et al. (2024) who modelled a reusable PP packaging for rice found that for the defined functional unit, the climate change and fossil resource depletion impact are, respectively, two and three times higher than those of a conventional single-use multilayer PET + LDPE rice packaging. The study identified reusability and reuse cycle as the most influential parameters in these impact categories, showing that increasing the reuse cycles from 5 to 16 and 22 uses allows the reusable pp packaging to outperform its single-use counterpart in climate change and fossil resources use, respectively.

Similarly, Battini et al. (2016), Levi et al. (2011) and Tua et al. (2017) also reported that single-use packaging can environmentally perform better for all or part of the selected impact categories. Battini et al. (2016) proposed an improved form of single-use corrugated fiberboard boxes with removable films and compared its environmental impacts with a reusable pp crate to transfer fruit and vegetables under different scenarios for supply chain and transportation distance. The study showed that reusable crate outperforms single-use ones at short transport distance. However, increasing the transportation distance, the environmental benefits of reusable crate diminish and eventually reverse, primarily due to the higher weight of reusable crates. Levi et al. (2011) and Tua et al. (2017) similarly emphasized on the critical role of transportation distance on the environmental performance of packaging; for example, Levi et al. (2011) found that reusable PP packaging performs better than single-use packaging only when the transportation distance is below 1200 km.

b. HDPE: seven out of the 32 reviewed studies considered HDPE or a blend of HDPE with other polymers, as the primary material for reusable (Abejón et al., 2020; Bernstad Saraiva et al., 2016; Boschiero et al., 2019; Ceballos-Santos et al., 2024; Helmes et al., 2022; Koskela et al., 2014; Hilmarsdóttir et al., 2024). In Helmes et al. (2022), despite a very low reuse cycles (i.e., only one reuse cycle), HDPE reusable shampoo bottles outperformed single-use HDPE shampoo bottle in terms of environmental performance. In contrast, Ceballos-Santos et al. (2024) reported that HDPE reusable crates used for fish distribution have a similar environmental performance compared to single-use Expanded polystyrene (EPS) crates. Nonetheless, several parameters were identified as influential in this comparison. For instance, by considering a washing facility located on-site, the reusable crates show a substantial improvement in environmental impacts, enabling them to outperform single-use crates. On the other hand, since transportation distances were kept constant across scenarios in this study, the weight of the crates became a decisive factor, favoring single-use EPS, which weigh only

0.20 kg compared to 1.20 kg for reusable crates. Overall, the study concluded that for distances exceeding 160 km, single-use and reusable crates perform similarly in environmental terms.

c. PET: Across the reviewed studies, four focused on PET as the primary material for reusable plastic packaging (Helmes et al., 2022; Kim et al., 2023; Silva et al., 2024; Zhou et al., 2024). Each of these studies included a proportion of recycled polyethylene terephthalate (rPET) in the packaging composition and consistently reported that PET-based reusable systems outperform comparable single-use options (see Table 3). This enhanced environmental performance is largely driven by replacing virgin PET with rPET, which substantially lowers the impacts associated with PET aggregate production—a major hotspot in packaging life cycles (Silva et al., 2024). Silva et al. (2024) assessed the environmental performance of production of a novel recycled PET (called rPET flake), a virgin PET and a rPET which was produced conventionally (called rPET pellets). The results showed that except for water consumption and Eutrophication impact categories, production of virgin PET presents the highest environmental impacts. Comparison of the LCA results for the three PET production pathways shows that, relative to virgin PET production, rPET flakes achieve an average reduction of 79% of all impact categories, while production of rPET pellets provide only about a 10% reduction.

Notably, these environmental benefits remain strong even when number of reuse cycles are low. PET reusable packaging can outperform single-use equivalents with only two reuse cycles for beverage packaging (Silva et al., 2024), one reuse for cosmetic packaging (Helmes et al., 2022), and seven reuse cycles for food packaging (Kim et al., 2023).

### 6.2. *Reusable plastic packaging versus single-use plastic packaging across different sectors*

Fig. 8 presents a comparative overview of the environmental performance of reusable plastic packaging and single-use packaging across different sectors for the selected impact categories listed in Table 2. The bar chart in the lower section illustrates the share of studies reporting whether reusable plastic packaging performs better (green), worse (red), or equal (yellow) compared to single-use packaging. However, this comparison is indicative rather than exact, because as presented in Table 2, the selected impact categories differ among studies. Consequently, the interpretation of the results is limited to the selected impact categories and may not represent the full environmental profile. On the other hand, several studies investigate different parameters or scenarios and the results are presented based on these defined parameters or scenarios. Only the most representative findings for each study were included in Fig. 8.

The pie charts above each bar illustrate how the 32 reviewed studies are distributed across different sectors. Fruit and vegetable packaging accounts for the largest portion (15 studies), followed by food packaging (10 studies). In contrast, cosmetics, delivery, and beverage packaging represent much smaller shares—with 1, 5, and 1 study respectively. This distribution shows that comparative LCAs of reusable versus single-use packaging have been heavily concentrated in the food and fruit-and-vegetable sectors, while applications in cosmetics and beverages remain relatively understudied. Overall, the findings point to a clear environmental benefit of reusable systems, yet also expose a notable research imbalance across packaging types. However, the limited number of available studies on cosmetic, beverage, and delivery packaging highlight this important fact that the interpretations presented in this section should be treated with caution, as they are based on a small

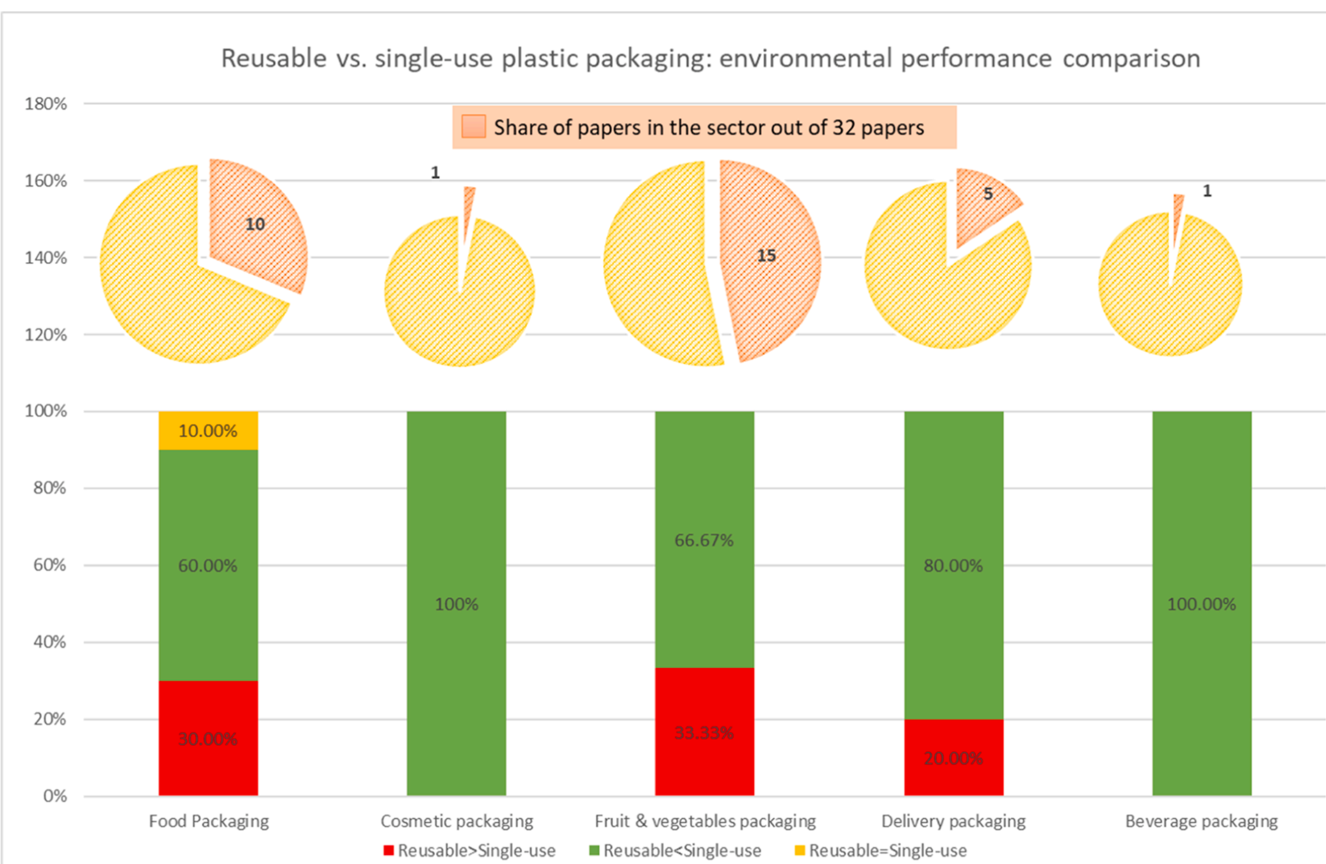



**Fig. 8.** Reusable vs single-use plastic packaging: environmental comparison across application categories.

sample size.

The results indicate that reusable packaging most of the time has a lower environmental impact. Specifically, in cosmetic and beverage packaging, analyzed studies conclude that reusable plastic packaging options outperform single-use ones. However, this depends on the number of minimum reuse cycles needed to have the reusable plastic packaging system outperforming single-use ones. In beverage packaging, the minimum required reuse cycles were calculated as 2 times (Silva et al., 2024), while for cosmetics packaging, no minimum reuse cycle was specified.

Similarly, 63.64% of food packaging studies (minimum reuse cycles ranging from 4 (Greenwood et al., 2021) to 30(Battini et al., 2016) times) and 66.67% of fruit and vegetable packaging studies (minimum reuse cycles ranging from 3 (Tua et al., 2019) to 40 (Albrecht et al., 2013) times) show better performance for reusable packaging systems. This trend is even stronger for delivery packaging, where 80% of studies (with minimum reuse cycle of 7 times (Zhou et al., 2024)) show improved performance. However, only one study on food packaging (or 9% of reviewed studies on food packaging sector) finds no significant differences between the two systems (Ceballos-Santos et al., 2024).

It should be noted that the results discussed above were not filtered according to study quality, which may influence the robustness of their interpretation. Therefore, the findings were re-evaluated using only the highly ranked studies (overall quality score >85%) based on the assessment presented in Appendix B. When the analysis is restricted to these studies, the overall trend remains consistent: reusable systems outperform single-use packaging only after a minimum number of rotations is achieved. For example, (Yadav et al., 2024) reports lower impacts for reusable takeaway food containers after approximately six reuse cycles, while (Tua et al., 2019) shows improved performance of reusable plastic crates after three cycles. Similarly, (Accorsi et al., 2022) demonstrates that reusable crate logistics networks become environmentally preferable over time as manufacturing burdens are distributed across multiple rotations, (Hilmarsdóttir et al., 2024) identifies lower impacts for multi-use fish tubs compared with single-use EPS boxes, and (Gatt and Refalo, 2022) observes substantially reduced impacts for reusable cosmetic packaging. However, these studies also emphasize that the benefit is conditional and depends strongly on transport distance, return efficiency, and system performance (Accorsi et al., 2022). It should be clarified that (Gallego-Schmid et al., 2018), although showing the same general trend of improved performance for reuse, compares reusable plastic containers with single-use glass packaging rather than single-use plastic packaging. In line with these findings, (López-Gálvez et al., 2021) shows that reusable plastic crates exhibit lower environmental impacts than single-use cardboard and wooden boxes across all assessed impact categories, provided that a minimum of approximately 15 reuse cycles is achieved.

The highly ranked literature further indicates that operational parameters, rather than packaging material alone, determine environmental performance. (Accorsi et al., 2022) highlights the sensitivity of results to reverse logistics and distribution transport, and (Lo-Iacono-Ferreira et al., 2021) shows that reusable systems may exhibit higher impacts when transport burdens outweigh manufacturing savings. Consequently, reusable packaging cannot be considered environmentally superior by default; it becomes preferable only when sufficient rotations are achieved within an efficient collection and reprocessing system. The conclusions derived from the full dataset therefore remain valid but should be interpreted as conditional: reuse generally reduces impacts only under well-functioning operational conditions.

### 6.3. Critical factors and their impacts on LCA results

#### 6.3.1. EOL approach

It should be noted that while interpreting LCA results, the selection of the EOL approach has a direct influence on LCA outcomes in favor of reusable plastic packaging. Although a comparison of LCA results of reviewed papers across different EOL methodologies is not possible due to the lack of sensitivity analyses, it is expected that the application of the cut-off approach tends to underestimate total environmental burdens, as it does not account for the environmental impacts of waste-management activities. As also shown in Table 3, the studies that conclude that reusable plastic packaging performs worse than single-use packaging (all applying cradle-to-grave system boundaries) predominantly use the avoided burden approach, with the exception of (Battini et al., 2016; Blanca-Alcubilla et al., 2020; Sasaki et al., 2022).

This is because reusable plastic packaging involves additional processes related to reuse activities, such as reverse transportation, washing operations, and redistribution. In contrast, single-use packaging benefits from recycling credits after one use cycle, whereas reusable packaging mainly relies on reuse and receives recycling credits only once, at the end of the final use cycle. Although the recycling burden increases for single-use plastic packaging and the avoidance of recycling due to reuse plays a role in the final LCA results for reusable plastic packaging the associated credits can outweigh the burdens in some studies and improve the environmental performance of single-use systems relative to reusable plastic packaging (Thomassen et al., 2024;Tua et al., 2017; Levi et al., 2011; Koskela et al., 2014; Bernstad Saraiva et al., 2016; Lo-Iacono-Ferreira et al., 2021).

#### 6.3.2. Reuse phase and factors impacting it

The review of LCA findings shows that the number of reuse cycles is a key parameter influencing environmental outcomes. Yadav et al. (2024), for instance, evaluated multiple reuse scenarios and found that, under EOL incineration, a 10-cycle scenario reduced global warming potential (GWP) by 46%, while extending reuse to 100 cycles led to an 83% reduction relative to a single-use system per functional unit. Their study also demonstrated that the effectiveness of the return logistics for empty containers plays a critical role in determining overall impacts. These results underscore the need to apply realistic and achievable reuse-cycle assumptions to ensure that modeled LCA outcomes accurately reflect the true environmental performance of reusable packaging systems. In other words, the definition of realistic reuse cycles ensures that production and transportation burdens are effectively distributed across multiple reuse cycles without underestimating the impact by reusable packaging systems.

However, the number of reuse cycles can be significantly impacted by the reuse performance of the material. Material performance is highly sensitive to factors such as return rates, washing temperatures (which may accelerate material degradation), application context, weathering, and handling intensity (Alassali et al., 2021).

For a systematic evaluation, the number of reuse cycles for the 34 studies were assessed in relation to defined reuse cycles in operational reuse plastic packaging systems (see appendix B). Studies that assume reuse cycles within empirically supported ranges and provide plausible operational contexts were highly ranked as realistic (Accorsi et al., 2014; Helmes et al., 2022; Kim et al., 2023; Rasines et al., 2024; Yadav et al., 2024; Zhou et al., 2024)). Conversely, studies that either assume unrealistically high reuse numbers or do not sufficiently justify reuse cycles obtained low score for this assumption, as it reflects on study's credibility (e.g., Singh et al., 2006; Boschiero et al., 2019; Silva et al., 2024; Alzubi et al., 2022). Analysis of reuse cycles in appendix B, shows Studies that assume reuse cycles within empirically supported ranges and provide plausible operational contexts often rank among the top-performing studies whose results are reliable for further interpretation (e.g., Gatt and Refalo, 2022; Yadav et al., 2024; Tua et al., 2019; Lo-Iacono-Ferreira et al., 2021).

On the other hand, even studies employing realistic reuse assumptions are recommended to include alternative scenarios to improve robustness and reliability, where plastic packaging under different recovery and reconditioning systems can be impacted and results in lowering the lab-approved number of reuse cycles.

Achievable reuse cycles are determined by system performance rather than material durability. In practice, the realized number of rotations depends on recovery probability across each cycle, which is constrained by both technical design and user behavior.

System design strongly influences survival in circulation. Efficient reprocessing activities, such as locating centralized washing facilities near or within distribution centers, reduce handling losses and premature discarding (Ceballos-Santos et al., 2024; Hitt et al., 2023; Yadav et al., 2024). Similarly, integrating reverse logistics with forward distribution transport minimizes additional transport steps and improves return rates (Accorsi et al., 2022; Hitt et al., 2023; Tan et al., 2023; Yadav et al., 2024). In contrast, fragmented collection infrastructure, long return distances, or complex sorting requirements shorten operational lifetimes regardless of technical durability.

Across most studies, washing operations, virgin granulates production, packaging manufacturing, and transportation processes were identified as the main contributors in environmental impacts (Accorsi et al., 2022; Camps-Posino et al., 2021; Hitt et al., 2023; Rasines et al., 2024; Snyder and Park, 2024). However, Snyder & Park (2024) indicate detergent use as the dominant contributor, noting that other washing steps had limited influence due to efficient water and energy use. In some sectors, such as fruit and vegetable packaging, washing was not a major contributor, primarily due to lower energy requirement or the absence of specific detergent use. Therefore, some studies in this sector introduced the washing process as a minor contributor to the overall life cycle impact of a reusable plastic packaging (Tua et al., 2017). These findings highlight that in modeling of the washing process, factors such as scale of the washing activity, used detergent, or energy sources can significantly affect the environmental impacts of reusable plastic packaging.

Single-use plastic packaging sometimes performed environmentally better than reusable plastic packaging as a result of the lower weight of the packaging and the exclusion of washing and reverse logistics phases (Levi et al., 2011; Thomassen et al., 2024; Tua et al., 2017). However, these factors can be shifted in favor of reusable plastic packaging by defining optimal reuse cycles, and efficient management of reverse logistics and washing operations.

## 7. Conclusions and recommendations

In this review, a clear research gap becomes visible when looking at how reusable plastic packaging is assessed in comparison to single-use systems, especially in sectors like cosmetics, delivery services, and beverages. Studies on food and fresh product packaging generally agree that reusable plastic packaging options can lead to notable environmental improvements—assuming they are reused enough times. However, the analysis of published LCA studies reveals notable methodological inconsistencies—including differences in system boundaries, assumptions, and impact assessment methods—which limit the comparability of results and weaken the reliability of overarching conclusions.

85% of the papers state that they follow the relevant LCA standards, but many do not fully apply the methodological steps required for transparent and comprehensive modeling. This is particularly noticeable in the choice of impact categories and EOL approach. Several studies rely on only a small set of impact categories without explaining why others were excluded. Yet literature has shown that results can be highly sensitive to which categories are chosen (López-Gálvez et al., 2021; Snyder and Park, 2024; Yadav et al., 2024). Climate change is by far the most commonly assessed category, likely because it is widely recognized and easier to interpret. In contrast, categories related to marine and aquatic pollution appear far less often, despite the growing attention on plastic litter and microplastic contamination in marine ecosystems. It is therefore recommended that authors provide a more rigorous scientific justification for the selection of impact categories in LCA studies. In addition, normalization can support the identification of the most relevant impact categories which should be considered in the interpretation of the results.

Another recurring issue is the lack of sensitivity analysis related to EOL modeling. According to ISO 14,044, such analysis is required when multiple methodological options are possible, yet none of reviewed papers neither justify their choice nor explore how using a different approach would affect the results. Given the strong influence of EOL modelling on comparative results, future LCA studies should evaluate both avoided burden and cut-off approaches and explicitly justify the chosen method based on the aim of the study and the intended representation of reuse and recycling processes within the system.

However, even when EOL modelling approaches are clearly described, the assumptions behind them need careful consideration because they often rely on idealized recycling conditions that may not reflect real-world systems. Recycling rates are frequently based on general values and may overlook regional differences in collection, sorting, and recycling infrastructure. In addition, reusable plastic packaging can experience material degradation, contamination, or quality loss after repeated use and washing, which may limit closed-loop recycling and lead to downcycling. As a result, avoided burden models that assume full substitution of virgin material may overestimate EOL credits. Therefore, EOL benefits in reusable packaging LCAs should be interpreted cautiously, and studies should transparently report recycling pathways, substitution factors, material quality, and regional infrastructure conditions.

Furthermore, the review has shown that many studies omit relevant life cycle stages. Secondary and tertiary packaging, as well as return transportation, are many times excluded, despite the major role they have in determining the environmental performance of reusable plastic packaging. For instance, return transport distances and structure of distribution systems can determine whether a reusable system performs better or worse than its single-use counterpart. Depending on the defined system boundary, it is recommended that an LCA study includes specific processes to ensure realistic results and robust interpretation. For cradle-to-grave system boundaries, it is essential to assess (i) packaging production (including raw material extraction) due to its potentially high energy intensity; (ii) transportation activities (e.g., packaging transport and product distribution) given their relevance for facility location and reuse system design; (iii) reconditioning processes, because of their decisive influence on the comparative environmental performance of reusable versus single-use systems, and (iv) EOL processes due to the potential credits and burdens associated with different modelling approaches.

Across the reviewed literature two other parameters that stand out as especially influential are a) the number of reuse cycles and b) the design of reconditioning activities. More reuse cycles generally lead to better environmental performance, but only if the assumed numbers are realistic for the given application. The fact that reported reuse cycles in literature range from 1 to >1200 highlights the need for clearer and more practical guidance. Similarly, the environmental benefits of reusable systems depend heavily on how reconditioning is organized—washing, refilling, and return logistics all need to be efficient to obtain a feasible reusable packaging system. However, it should be noted that reconditioning practices, particularly washing operations, are highly sensitive to sector-specific characteristics. Variations in cleaning requirements can substantially alter the environmental performance of reusable plastic packaging systems. For instance, in the cosmetics and personal care sector, washing often represents a dominant contributor to life cycle impacts. Stricter hygiene standards lead to higher energy demand and more intensive detergent use. In contrast, in the fruit and vegetable sector, washing typically remains a minor contributor, as cleaning requirements are less stringent, consequently, associated energy and detergent inputs are lower. Hence, the interpretation of LCA results for reconditioning processes must explicitly account for sectoral specifications.

Beyond these parameters, and irrespective of the application sector, packaging weight and logistics consistently emerge as influential determinants of environmental performance. Reductions in packaging mass, the use of centralized or on-site washing facilities, and the integration of reverse logistics with forward distribution flows can substantially improve the overall environmental performance of reusable packaging systems.

Generally, the evidence indicates that the environmental performance of reusable packaging systems is highly dependent on system design and operational conditions, which can deviate based on geographical location and the implemented strategies and policies. Favorable outcomes are associated with optimum reuse cycles, efficient return logistics, and optimized washing processes. Where these conditions are not achieved, environmental benefits may diminish, and in some cases, reuse systems may not outperform single-use alternatives.

These findings have clear implications for both policymakers and industry actors. Reuse should not be treated as inherently superior in every situation. When return rates are low, transport distances are extended, washing processes require high energy inputs, or the necessary infrastructure is lacking, the anticipated environmental benefits may not be achieved. Under such conditions, broad or uniform reuse mandates risk creating unintended trade-offs rather than improvements. Policy measures and industry strategies should therefore be based on case-specific environmental assessments and supported by reliable operational data.

## 8. Limitations

Despite the comprehensive and systematic approach applied in this review, several limitations should be acknowledged. First, a geographic bias is expected, as the geographical location of the modelled reuse systems was not within the focus of this study, therefore if most of the reviewed studies originate from European countries, this can reflect strong policy support for circular economy strategies and reuse infrastructure within the EU. Consequently, the findings may not be fully transferable to regions with different waste management systems, return infrastructures, consumer behavior patterns, or living habits (e.g., Asia or Latin America).

Second, the reviewed studies focused mostly on conventional plastics, resulting in underrepresentation to emerging materials and innovative system designs. Most studies focus on conventional polymers such as PET, PP, and HDPE. As a result of the market relevance of emerging materials and advanced systems (e.g., bio-based plastics, biodegradable polymers, or hybrid refill systems combining reusable and single-use components), their environmental assessment requires further investigation.

Moreover, microplastic emissions and environmental littering were generally not considered in the reviewed studies, although these aspects may be particularly relevant when comparing reusable and single-use plastic packaging systems.

Additionally, methodological heterogeneity across studies (e.g., system boundaries, EOL modeling approaches, reuse cycle assumptions, and allocation methods) limits direct comparability. Also, due to lack of primary data in some cases, some studies used secondary or a mix between primary and secondary data sources. Although this does not mean a lower quality in LCA modelling per se, it can significantly impact the studies' specificity. However, this review critically analyzed methodological heterogeneity, data sources, and modeling approaches, to develop a fair basis for comparison.

Furthermore, the reviewed literature may be affected by publication bias, as studies reporting favorable environmental performance of reuse systems may be more likely to be published and disseminated than studies showing limited, context-dependent, or negative environmental benefits. As a result, the overall trends identified in this review should be interpreted with caution, particularly when drawing broader conclusions regarding the environmental superiority of reuse systems across different sectors and applications.

## Declaration of generative AI and AI-assisted technologies in the manuscript preparation process

During the preparation of this work the author(s) used ChatGPT 5.1 in order to improve the text's readability and reduce the wording in some places. After using this tool, the author(s) reviewed and edited the content as needed, taking full responsibility for the content of the published article.

## CRediT authorship contribution statement

**Mahsa Doostdar:** Writing – original draft, Visualization, Validation, Resources, Methodology, Investigation, Formal analysis, Data curation, Conceptualization. **Ayah Alassali:** Writing – original draft, Visualization, Validation, Supervision, Conceptualization. **Kerstin Kuchta:** Validation, Supervision.

## Declaration of competing interest

The authors declare that they have no conflicts of interest. Study screening and full-text eligibility assessment were performed independently by two authors. Any discrepancies were resolved through discussion and consensus to avoid biased selection.

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## Appendix A

Table A1 presented an overview of life cycle stages which are included in reviewed papers.

**Table A1**
Overview of included life cycle stages in the reviewed papers.

| References | RME[1] | PM[2] | Logistics | S and/or TP[3] | Backhaul logistics | Preparation for reuse[4] | EOL |
|---|---|---|---|---|---|---|---|
| (Gatt and Refalo, 2022) | ✓ | ✓ | ✓ | × | Not needed | Not needed | ✓ (Landfill & recycling) |
| (Helmes et al., 2022) | × | ✓ | ✓ | × | ✓ | ✓ | ✓ (Incineration & recycling |
| (Thomassen et al., 2024) | × | ✓ | ✓ | ✓ | × | ✓ | ✓ (Landfill & recycling & incineration) |
| (Greenwood et al., 2021) | × | ✓ | ✓ | × | ✓ | ✓ | ✓ (Landfill & recycling |
| (Ceballos-Santos et al., 2024) | ✓ | ✓ | ✓ | ✓ | ✓ | ✓ | ✓ Recycling |
| (Kim et al., 2023) | ✓ | ✓ | ✓ | × | ✓ | ✓ | ✓ (Landfill & recycling & incineration) |
| (Snyder and Park, 2024) | ✓ | ✓ | ✓ | × | Not needed | ✓ | ✓ (Landfill & recycling & incineration) |

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**Table A1** (*continued*)

| References | RME[1] | PM[2] | Logistics | S and/or TP[3] | Backhaul logistics | Preparation for reuse[4] | EOL |
|---|---|---|---|---|---|---|---|
| (Hitt et al., 2023) | ✓ | ✓ | ✓ | ✓ | ✓ | ✓ | ✓ (Landfill & recycling & incineration |
| (Battini et al., 2016) | × | ✓ | ✓ | × | ✓ | ✓ | ✓ (Landfill & recycling |
| (Yadav et al., 2024) | ✓ | ✓ | ✓ | × | ✓ | ✓ | ✓ (Incineration & recycling |
| (Camps-Posino et al., 2021) | ✓ | ✓ | ✓ | × | ✓ | ✓ | ✓ (Landfill & recycling & incineration) |
| (Tan et al., 2023) | ✓ | ✓ | ✓ | × | ✓ | × | ✓ (Incineration & recycling |
| (Rasines et al., 2024) | ✓ | ✓ | ✓ | × | ✓ | ✓ | ✓ (Landfill & recycling & incineration |
| (Singh et al., 2006) | ✓ | ✓ | ✓ | × | ✓ | ✓ | ✓ (Landfill & recycling & incineration) |
| (Boschiero et al., 2019) | ✓ | ✓ | ✓ | × | × | × | Out of scope |
| (Tua et al., 2019) | ✓ | ✓ | Partially included | × | ✓ | ✓ | ✓ (Incineration & recycling) |
| (Levi et al., 2011) | ✓ | ✓ | Partially included | × | ✓ | ✓ | ✓ (Landfill & recycling |
| (Tua et al., 2017) | × | ✓ | ✓ | ✓ | ✓ | ✓ | ✓ (Incineration & recycling) |
| (Accorsi et al., 2022) | ✓ | ✓ | ✓ | ✓ | ✓ | ✓ | ✓ (Incineration & recycling) |
| (Albrecht et al., 2013) | ✓ | ✓ | ✓ | × | ✓ | ✓ | ✓ (Recycling) |
| (Silva et al., 2024) | ✓ | ✓ | Partially included | × | Not needed | ✓ | Out of Scope |
| (Accorsi et al., 2014) | ✓ | ✓ | ✓ | × | ✓ | ✓ | ✓ (Landfilling & Incineration & recycling) |
| (Zhou et al., 2024) | ✓ | ✓ | ✓ | × | ✓ | ✓ | ✓ (Incineration & recycling) |
| (Alzubi et al., 2022) | ✓ | ✓ | ✓ | × | ✓ | × | × |
| (Abejón et al., 2020) | ✓ | ✓ | ✓ | × | ✓ | ✓ | ✓ (Incineration & recycling) |
| (Ross and Evans, 2003) | ✓ | ✓ | ✓ | ✓ | × | Not needed | ✓ (Landfill & recycling) |
| (López-Gálvez et al., 2021) | ✓ | ✓ | ✓ | × | ✓ | ✓ | ✓ (Landfill & recycling & incineration |
| (Blanca-Alcubilla et al., 2020) | ✓ | ✓ | ✓ | × | ✓ | ✓ | ✓ (Landfill) |
| (Hilmarsdóttir et al., 2024) | ✓ | ✓ | ✓ | × | ✓ | ✓ | ✓ (Landfill & recycling & incineration |
| (Koskela et al., 2014) | ✓ | ✓ | ✓ | ✓ | ✓ | ✓ | ✓ (Incineration & recycling) |
| (Sasaki et al., 2022) | ✓ | ✓ | ✓ | × | × | × | ✓ (Landfill & incineration |
| (Bernstad Saraiva et al., 2016) | ✓ | ✓ | ✓ | × | ✓ | ✓ | ✓ (Landfill & recycling & incineration |
| (Gallego-Schmid et al., 2018) | ✓ | ✓ | ✓ | ✓ | Not needed | ✓ | ✓ (Landfill & recycling & incineration |
| (Lo-Iacono-Ferreira et al., 2021) | ✓ | ✓ | ✓ | × | ✓ | ✓ | ✓ (Incineration & recycling |

[1] Raw Material Extraction.
[2] Packaging Manufacturing.
[3] Secondary and/or tertiary packaging.
[4] For example, washing, filling.

## Appendix B

Appendix B presents a semi-quantitative assessment of methodological completeness across the included studies. This assessment is intended to provide a structured overview of key methodological choices (e.g., system boundaries, impact category coverage, EOL modelling), rather than a formal data quality or risk-of-bias evaluation. Established tools for systematic reviews (e.g., AMSTAR-2 or ROBIS) were not applied, as the objective of this study is not to assess bias but to evaluate comparability and methodological consistency across LCA studies.

This structured three-step assessment framework was applied enable a transparent and consistent comparison of studies addressing reuse plastic packaging systems. The framework combines qualitative data extraction with semi-quantitative scoring, allowing both methodological range and assumption realism to be evaluated.

**1. Data Quality and Source Assessment**

Data quality was assessed based on the origin of the data used in each study, recognizing the strong influence of data sources on the robustness of environmental assessments. A three-level scoring scheme was applied:

- Score 2: Predominant use of primary data.
- Score 1: Mixed use of primary and secondary data (e.g., ecoinvent database, another study and/or literature review).
- Score 0: Reliance on secondary databases only.

This approach reflects the increasing uncertainty associated with more generic data sources and supports differentiation between empirical and assumption-driven studies.

**2. Impact Category Coverage**

Studies were further evaluated based on the percentage of mid-point impact categories included, following the ReCiPe methodology. A higher share of covered mid-point categories was interpreted as an indicator of analytical completeness. For studies that exclusively applied endpoint indicators, this criterion was considered not applicable and therefore excluded from their assessment, as endpoint approaches inherently aggregate mid-point information.

**3. EOL Modelling Approach**

The treatment EOL was initially evaluated based on whether studies applied avoided burden approaches, cut-off approaches, or a combination of both. The original scoring logic assigned:

- Score 2 when more than one approach was applied.
- Score 1 when only one approach was used.
- Score 0 when no EOL approach was defined.

However, during the review it became evident that except (Alzubi et al., 2022), all studies explicitly defined one EOL modelling approach within their system boundaries but none of them considered two or more EOL approaches in their LCA model. As a result, this indicator did not provide meaningful differentiation across studies and was therefore excluded from the final quality assessment.

**4. Use of Standards**

Compliance with recognized methodological standards was assessed as a binary criterion:

- Score 1 when a study explicitly followed any recognized standard (e.g., ISO 14,040/14,044 or related standards).
- Score 0 when no standard was referenced/defined.

This indicator was retained to reflect transparency and methodological alignment with established assessment practices.

**5. Life Cycle Stage Coverage**

The inclusion of relevant life cycle stages was assessed to capture system completeness. The following stages were evaluated individually:

- Raw material extraction.
- Packaging manufacturing.
- Secondary and tertiary packaging manufacturing.
- Logistics.
- Reconditioning.
- EOL practice.

Each stage was assigned 1 if included and 0 if excluded. The overall life cycle stage score was then calculated as the average of these individual scores, allowing partial coverage to be reflected without disproportionately penalizing studies with narrow but well-defined scopes.

**6. Assessment of Reuse Cycles**

Reuse cycle assumptions were assessed individually, recognizing that realistic reuse potential depends strongly on sector, product type, and packaging format. Sector-specific reference values were extracted from (Rateau and Forbicini, 2025) and applied as follows:

- Reusable takeaway and food containers: typically designed for 100–200 wash cycles.
- Refillable PET bottles: generally, 10–25 reuse cycles before degradation.
- Reusable food containers/takeaway bowls: maximum 200 cycles.
- Beverage crates: often remain in use for 10–15 years, corresponding to several hundred rotations, with some industry sources indicating lifetimes exceeding 500 uses under controlled, closed-loop conditions.

Where specific data on the number of reuse cycles was unavailable, estimates were based on comparable applications. For example, values for refillable PET bottles were used for cosmetic products, given their similar use conditions and safety requirements. In cases where reuse numbers exceeded reference values only marginally (e.g., Kim et al., 2023; Snyder and Park, 2024), a partial score of 30% was retained, acknowledging that such assumptions may be valid under specific operational conditions.

Table B1 summarizes the comparative evaluation of the reviewed studies across data quality, impact category coverage, life cycle stage inclusion, realism of reuse assumptions, and the resulting overall score. The overall evaluation reflects methodological robustness and decision-readiness, rather than serving as a strict ranking.

Data quality shows substantial variability across the reviewed studies. Only a limited number of studies achieve the highest score (100%), indicating predominant use of primary data or well-documented mixed datasets (e.g., Ceballos-Santos et al., 2024; Kim et al., 2023; Tua et al., 2019; Accorsi et al., 2022; Hilmarsdóttir et al., 2024). In contrast, several studies rely heavily on secondary or tertiary data sources, reflected in scores of 0%, which substantially lowers their overall evaluation despite strengths in other categories (e.g., Helmes et al., 2022; Snyder and Park, 2024; Sasaki et al., 2022).

This highlights that data origin remains a key differentiating factor in the robustness of reuse-related environmental assessments.

Coverage of mid-point impact categories varies widely. Some studies include nearly the full set of ReCiPe mid-point indicators (100%), contributing positively to analytical completeness (e.g. Gatt and Refalo, 2022; Greenwood et al., 2021; Kim et al., 2023; Gallego-Schmid et al., 2018). Others include only a limited subset, with values below 20%, which constrains the interpretability of their results (Helmes et al., 2022; Battini et al., 2016; Camps-Posino et al., 2021).

Most studies achieve relatively high scores (typically 67–100%) for life cycle stage inclusion, indicating that raw material extraction, manufacturing, logistics, reconditioning, and end-of-life processes are generally considered. The inclusion of secondary and tertiary packaging system was the most neglected among all other stages. Studies with full coverage (100%) tend to achieve higher overall scores, provided that other assumptions are also realistic (e.g., Accorsi et al., 2022; Gallego-Schmid et al., 2018).

The assessment of reuse cycle realism emerges as one of the most influential factors in the overall evaluation. Studies within empirically supported ranges achieve high scores (90–100%). Conversely, studies that either assume unrealistically high reuse numbers or do not sufficiently justify reuse cycles receive low or zero scores in this category, significantly reducing their overall evaluation (e.g., Singh et al., 2006; Boschiero et al., 2019; Silva et al., 2024; Alzubi et al., 2022). This confirms that reuse cycle assumptions are a critical driver of result credibility. Table B2

**Table B1**

Quality Assessment of the critical factors in reviewed papers.

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**Table B1** (*continued*)

| | Study | (Gatt & Refalo, 2022) | (Helmes et al., 2022) | (Thomassen et al., 2024) | (Greenwood et al., 2021) |
|---|---|---|---|---|---|
| | Data quality (out of 3) | Medium | Low | Medium | Low |
| | Nr. Of studied impact categories (out of total nr. Of impact categories per | 2 endpoint impact categories, 0 out of 18 mid-point impact categories | 3 out of 18 mid-point impact categories | 6 out of 16 mid-point impact categories | 18 out of 18 mid-point impact categories |
| | Percentage of mid-point impact categories included based on recipe | Not relevant | 17% | 38% | 100% |
| | Applied EOL approaches | Cut-off | Avoided burden | Avoided burden | Avoided burden |
| | System boundaries scope | cradle-to-grave | use-to-grave | cradle-to-grave | cradle-to-grave |
| | Applied Standard | ISO 14040 and 14044 | ISO 14040 and 14044 | ISO 14040 and 14044 | ISO 14040 and 14044 |
| Lifecycle stages | Raw material extraction | ✓ | × | × | × |
| | Packaging manufacturing | ✓ | ✓ | ✓ | ✓ |
| | sec. and ter. Packaging manufacturing | × | × | ✓ | × |
| | Logistics | ✓ | ✓ | ✓ | ✓ |
| | Reconditioning | Not needed | ✓ | ✓ | ✓ |
| | EOL practice | ✓ | ✓ | ✓ | ✓ |
| | Assumed reuse cycles (times) | Max 3 | 1 | 5 | 50 |
| | Application sector | Cosmetics- Blush compact (a case + a pan) | shampoo bottles | rice packaging (polymer type: PP) | Reusable Food container (PP) |
| | Realistic or not? | Yes | Yes | Yes | Yes |

(*continued on next page*)

**Table B1** (*continued*)

| | | |
|---|---|---|
| (Ceballos-Santos et al., 2024) | (Kim et al., 2023) | (Snyder & Park, 2024) |
| High | High | Low |
| 6 out of 16 mid-point impact categories | 18 out of 18 mid-point impact categories | 10 out of 10 mid-point impact categories |
| 38% | 100% | 100% |
| | | |
| Avoided burden | Cut-off (with partial energy-recovery credit for incineration) | Cut-off |
| cradle-to-grave | cradle-to-cradle | cradle-to-grave |
| ISO 14040 and 14044 | Not mentioned | ISO 14040 and 14044 |
| ✓ | ✓ | ✓ |
| ✓ | ✓ | ✓ |
| ✓ | × | × |
| ✓ | ✓ | ✓ |
| ✓ | ✓ | ✓ |
| ✓ | ✓ | ✓ |
| 1260 | 300 | 300 |
| Reusable plastic crates for fish (HDPE) | Reusable Food container (PP) | Reusable Food container (PP) |
| No | No | No |

(continued on next page)

**Table B1** (*continued*)

| | | | | |
|---|---|---|---|---|
| (Hitt et al., 2023) | (Battini et al., 2016) | (Yadav et al., 2024) | (Camps-Posino et al., 2021) | (Tan et al., 2023) |
| Low | Low | Medium | Low | Medium |
| 3 out of a mix list of impact | 1 out of 13 mid-point impact categories | 18 out of 18 mid-point impact categories | 1 out of 16 mid-point impact categories | 9 out of 11 mid-point impact categories |
| 17-21% | 8% | 100% | 6% | 82% |
| Cut-off | Cut-off | Avoided burden | Avoided burden | Avoided burden |
| cradle-to-grave | cradle-to-grave | cradle-to-grave | cradle-to-grave | cradle-to-grave |
| Not mentioned | Not mentioned | ISO 14040 and 14044 | ISO 14040 and 14044 | ISO 14040 and 14044 |
| ✓ | ✗ | ✓ | ✓ | ✓ |
| ✓ | ✓ | ✓ | ✓ | ✓ |
| ✓ | ✗ | ✗ | ✗ | ✗ |
| ✓ | ✓ | ✓ | ✓ | ✓ |
| ✓ | ✓ | ✓ | ✓ | ✗ |
| ✓ | ✓ | ✓ | ✓ | ✓ |
| 20 | various (15, 30, 45 and 60) | various (10, 30 and 100) | 50 | 50 |
| Reusable Food container (PP) | Reusable PP crates (to transport either fruits, vegetables or pre-packed meat) | Take a way food container (PP) | a reusable food container (PP) and a | reusable plastic delivery boxes (PP) |
| Yes | Yes | Yes | Yes | Borderline |

(continued on next page)

**Table B1** (*continued*)

| (Rasines et al., 2024) | (Singh et al., 2006) |
|---|---|
| Medium | Medium |
| 8 out of 16 mid-point impact categories | 3 out of unknown list of impact categories (impact assessment method is not mentioned) |
| 50% | only 3 which makes the range below 40% no matter which impact assessment method is used |
| Hybrid (50/50 allocation) | Cut-off (with partial energy-recovery credit for incineration) |
| cradle-to-grave | cradle-to-grave |
| ISO 14040 and 14044 | ISO 14040 and 14041, SETAC 15technical framework |
| ✓ | ✓ |
| ✓ | ✓ |
| ✕ | ✕ |
| ✓ | ✓ |
| ✓ | ✓ |
| ✓ | ✓ |
| 140 | Not specified |
| Reusable PP crates | Reusable PP crates |
| Yes | No |

(*continued on next page*)

**Table B1** (*continued*)

| (Boschiero et al., 2019) | (Tua et al., 2019) | (Levi et al., 2011) |
|---|---|---|
| High | High | Medium |
| 2 out of 2 midpoint impact categories | 15 out of 17 midpoint impact categories | 6 out of 13 mid-point impact categories |
| 100% | 88% | 46% |
| Out of scope of system boundary | Avoided burden | Avoided burden |
| post-harvest gate-to-gate | cradle-to-grave | cradle-to-grave |
| ISO 14040 and 14044 | ISO 14040 and 14044, Product Environmental Footprint (PEF) Guide | ISO 14040 and 14044 |
| ✓ | ✓ | ✓ |
| ✓ | ✓ | ✓ |
| × | × | × |
| ✓ | Partially included | Partially included |
| × | ✓ | ✓ |
| Out of scope | ✓ | ✓ |
| Not specified | various (1-125) | 200 |
| Reusable HDPE bins | Reusable PP crates | Reusable PP crates |
| No | Yes | Yes |

(continued on next page)

**Table B1** (*continued*)

| | | | | |
|---|---|---|---|---|
| (Tua et al., 2017) | (Accorsi et al., 2022) | (Albrecht et al., 2013) | (Silva et al., 2024) | (Accorsi et al., 2014) |
| Medium | High | Medium | High | Medium |
| 6 out of 16 mid-point impact categories | 1 out of 1 mid-point impact category | 6 out of 11 mid-point impact categories | 8 out of a mix list of impact categories (min | 1 out of 1 mid-point impact category |
| 38% | 100% | 55% | 44%- 73% | 100% |
| Avoided burden | Cut-off | Avoided burden | Not modelled (Out of | Avoided burden |
| cradle-to-grave | cradle-to-grave | cradle-to-grave | cradle-to-gate | cradle-to-grave |
| ISO 14040 and 14044 | ISO 14040 and 14044 | ISO 14040 and 14044 | ISO 14040 and 14044 | ISO 14040 and 14044 |
| × | ✓ | ✓ | ✓ | ✓ |
| ✓ | ✓ | ✓ | ✓ | ✓ |
| ✓ | ✓ | × | × | × |
| ✓ | ✓ | ✓ | Partially included | ✓ |
| ✓ | ✓ | ✓ | ✓ | ✓ |
| ✓ | ✓ | ✓ | Out of Scope | ✓ |
| 50 | 100 | 50 | Not specified | various (30, 50, and 70) |
| Reusable PP crates | Reusable PP crates | Reusable PP crates and reusable PE crates | Reusable water bottles | Reusable Food container (PP) |
| Yes | Yes | Yes | No | Yes |

(continued on next page)

**Table B1** (*continued*)

| | | |
|---|---|---|
| (Zhou et al., 2024) | (Alzubi et al., 2022) | (Abejón et al., 2020) |
| Medium | Medium | Medium |
| 11 out of 18 mid-point impact | 9 out of unknown list of impact categories | 7 out of a mix list of impact categories (min 11 impact categories, max 14) |
| 61% | 50% | 50% - 64% |
| Avoided burden | Not modelled | Avoided burden |
| cradle-to-grave | cradle-to-grave | cradle-to-grave |
| ISO 14040 and 14044 | ISO 14040 and 14044 | ISO 14040 and 14044 |
| ✓ | ✓ | ✓ |
| ✓ | ✓ | ✓ |
| × | × | × |
| ✓ | ✓ | ✓ |
| ✓ | × | ✓ |
| ✓ | × | ✓ |
| 30 | Not specified | *100 (conservative scenario) *150 (Technical scenario) |
| returnable express bag (PE or PET) and | Reusable PP crates | Reusable PP crates and HDPE crates |
| Yes | No | Yes |

(continued on next page)

**Table B1** (*continued*)

| | | |
|---|---|---|
| (Ross & Evans, 2003) | (López-Gálvez et al., 2021) | (Blanca-Alcubilla et al., 2020) |
| High | Medium | Medium |
| 3 out of unknown list of impact categories (no impact assessment method is selected because the assessment is done manually) | 11 out of 12 mid-point impact categories | 1 out of 16 mid-point impact categories |
| 17% | 92% | 6% |
| | | |
| Avoided burden | Avoided burden | Only landfill emissions |
| cradle-to-grave | cradle-to-grave | cradle-to-grave |
| *ISO 14040, *ISO 14041, *ISO 14042, *ISO 14043 | ISO 14040 and 14044 | Not mentioned |
| ✓ | ✓ | ✓ |
| ✓ | ✓ | ✓ |
| ✓ | × | × |
| ✓ | ✓ | ✓ |
| Not needed | ✓ | ✓ |
| ✓ | ✓ | ✓ |
| Not specified | 150 | 10 |
| It is a reusable protective case for delivery of refrigerators (big households) polymer type: EPS | Reusable PP crates | Reusable Food container (ABS, PS, PP) for flight meals |
| No | Yes | Yes |

(*continued on next page*)

**Table B1** (*continued*)

| | |
|---|---|
| (Hilmarsdóttir et al., 2024) | (Koskela et al., 2014) |
| High | Medium |
| 11 out of 11 mid-point impact categories | 7 out of 18 mid-point impact categories |
| 100% | 39% |
| EOL Recycling | Avoided burden |
| 1.Cradle-to-grave (for base case) & 2. Cradle-to-cradle (for other scenarios) | cradle-to-grave |
| ISO 14040 and 14044 | ISO 14040 and 14044 |
| ✓ | ✓ |
| ✓ | ✓ |
| × | ✓ |
| ✓ | ✓ |
| ✓ | ✓ |
| ✓ | ✓ |
| 96 | 700 |
| Reusable fish tubs (PE) | Reusable HDPE crates (for bread delivery) - washing is also included |
| Yes | Borderline |

(*continued on next page*)

**Table B1** (*continued*)

| | | | |
|---|---|---|---|
| (Sasaki et al., 2022) | (Bernstad Saraiva et al., 2016) | (Gallego-Schmid et al., 2018) | (Lo-Iacono-Ferreira et al., 2021) |
| Low | Medium | Medium | Medium |
| 3 endpoint impact categories, 13 out of 13 mid-point impact categories | 11 out of 16 mid-point impact categories | 12 out of 11 mid-point impact categories | 1 out of 1 mid-point impact category |
| 100% | 69% | 100% | 100% |
| Only landfill and incineration emissions | Avoided burden | Avoided burden | Avoided burden |
| cradle-to-grave | cradle-to-grave | cradle-to-grave | cradle-to-grave |
| Not mentioned | ISO 14040 and 14044 | ISO 14040 and 14044 | *ISO 14067, *ISO 14044, *ISO 14026, *ISO 14071 |
| ✓ | ✓ | ✓ | ✓ |
| ✓ | ✓ | ✓ | ✓ |
| × | × | ✓ | × |
| ✓ | ✓ | ✓ | ✓ |
| × | ✓ | ✓ | ✓ |
| ✓ | ✓ | ✓ | ✓ |
| 100 | Not specified | 50 | various (20, 50, and 100) |
| Reusable PP crates | Reusable HDPE crates | Reusable Food container (PP) | Reusable PP foldable box |
| Yes | No | Yes | Yes |

**Table B2**
Overall evaluation score of the reviewed papers.

| Study | Data quality (out of 3) | Percentage of mid-point impact categories included based on ReCiPe 2016 | Overall coverage of life cycle stages | Standard (yes/no) | Percentage of Realistic consideration of no of reuse cycles | Overall evaluation |
|---|---|---|---|---|---|---|
| (Gatt and Refalo, 2022) | 50% | 100% | 83% | 100% | 100% | 87% |
| (Helmes et al., 2022) | 0% | 17% | 67% | 100% | 100% | 57% |
| (Thomassen et al., 2024) | 50% | 38% | 83% | 100% | 100% | 74% |
| (Greenwood et al., 2021) | 0% | 100% | 67% | 100% | 100% | 73% |
| (Ceballos-Santos et al., 2024) | 100% | 38% | 100% | 100% | 10% | 70% |
| (Kim et al., 2023) | 100% | 100% | 83% | 0% | 30% | 63% |
| (Snyder and Park, 2024) | 0% | 100% | 83% | 100% | 30% | 63% |
| (Hitt et al., 2023) | 0% | 20% | 100% | 0% | 100% | 44% |
| (Battini et al., 2016) | 0% | 8% | 67% | 0% | 100% | 35% |
| (Yadav et al., 2024) | 50% | 100% | 83% | 100% | 100% | 87% |
| (Camps-Posino et al., 2021) | 0% | 6% | 83% | 100% | 100% | 58% |
| (Tan et al., 2023) | 50% | 82% | 67% | 100% | 100% | 80% |
| (Rasines et al., 2024) | 50% | 50% | 83% | 100% | 100% | 77% |
| (Singh et al., 2006) | 50% | 30% | 83% | 100% | 0% | 53% |
| (Boschiero et al., 2019) | 100% | 100% | 67% | 100% | 0% | 73% |
| (Tua et al., 2019) | 100% | 88% | 67% | 100% | 100% | 91% |
| (Levi et al., 2011) | 50% | 46% | 67% | 100% | 100% | 73% |
| (Tua et al., 2017) | 50% | 38% | 83% | 100% | 100% | 74% |
| (Accorsi et al., 2022) | 100% | 100% | 100% | 100% | 100% | 100% |
| (Albrecht et al., 2013) | 50% | 55% | 83% | 100% | 100% | 78% |
| (Silva et al., 2024) | 100% | 50% | 67% | 100% | 0% | 63% |
| (Accorsi et al., 2014) | 50% | 100% | 83% | 100% | 100% | 87% |
| (Zhou et al., 2024) | 50% | 61% | 83% | 100% | 100% | 79% |
| (Alzubi et al., 2022) | 50% | 50% | 50% | 100% | 0% | 50% |
| (Abejón et al., 2020) | 50% | 55% | 83% | 100% | 100% | 78% |
| (Ross and Evans, 2003) | 100% | 17% | 83% | 100% | 0% | 60% |
| (López-Gálvez et al., 2021) | 50% | 92% | 83% | 100% | 100% | 85% |
| (Blanca-Alcubilla et al., 2020) | 50% | 6% | 83% | 0% | 100% | 48% |
| (Hilmarsdóttir et al., 2024) | 100% | 100% | 83% | 100% | 100% | 97% |
| (Koskela et al., 2014) | 50% | 39% | 100% | 100% | 50% | 68% |
| (Sasaki et al., 2022) | 0% | 100% | 67% | 0% | 100% | 53% |
| (Bernstad Saraiva et al., 2016) | 50% | 69% | 83% | 100% | 0% | 60% |
| (Gallego-Schmid et al., 2018) | 50% | 100% | 100% | 100% | 100% | 90% |
| (Lo-Iacono-Ferreira et al., 2021) | 50% | 100% | 83% | 100% | 100% | 87% |

Overall evaluation scores range from 38% to 98%, indicating a broad spectrum of methodological quality within the reviewed literature. High-performing studies (above ~85%) combine strong data quality, comprehensive impact and life cycle coverage, and realistic reuse assumptions (e.g., Accorsi et al., 2022; Hilmarsdóttir et al., 2024; Tua et al., 2019).

Mid-range scores (60–75%) typically reflect trade-offs, such as good system boundaries but weaker data quality or partially optimistic reuse assumptions (e.g., (Tan et al., 2023; Rasines et al., 2024;(Zhou et al., 2024). Low scores (below ~50%) are primarily associated with limited data quality and/or unrealistic reuse modelling, even when impact coverage or system boundaries appear adequate.

To improve readability and provide a quick reference for readers, the overall quality evaluation of the reviewed studies is additionally presented in Fig. B1. The evaluation was based on several methodological criteria, including data quality, coverage of midpoint impact categories, inclusion of relevant life cycle stages, application of recognized standards, and the realism of assumed reuse cycles.

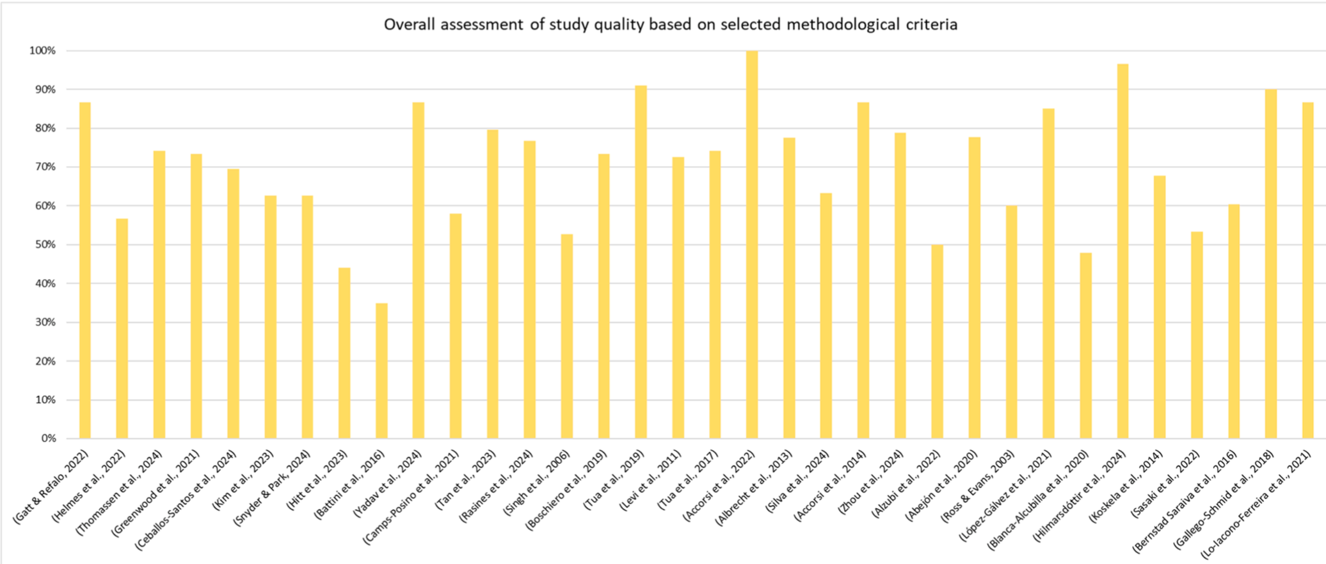


**Fig. B1.** Overall quality evaluation of the reviewed LCA studies based on selected assessment criteria.

## Data availability

Data will be made available on request.

## References


Abejón, R., Bala, A., Vázquez-Rowe, I., Aldaco, R., Fullana-i-Palmer, P., 2020. When plastic packaging should be preferred: life cycle analysis of packages for fruit and vegetable distribution in the Spanish peninsular market. Resour. Conserv. Recycl. 155, 104666. https://doi.org/10.1016/J.RESCONREC.2019.104666.

Accorsi, R., Battarra, I., Guidani, B., Manzini, R., Ronzoni, M., Volpe, L., 2022. Augmented spatial LCA for comparing reusable and recyclable food packaging containers networks. J. Clean. Prod. 375, 134027. https://doi.org/10.1016/J.JCLEPRO.2022.134027.

Accorsi, R., Cascini, A., Cholette, S., Manzini, R., Mora, C., 2014. Economic and environmental assessment of reusable plastic containers: a food catering supply chain case study. Int. J. Prod. Econ. 152, 88–101. https://doi.org/10.1016/J.IJPE.2013.12.014.

Ahamed, A., Vallam, P., Iyer, N.S., Veksha, A., Bobacka, J., Lisak, G., 2021. Life cycle assessment of plastic grocery bags and their alternatives in cities with confined waste management structure: a Singapore case study. J. Clean. Prod. 278, 123956. https://doi.org/10.1016/j.jclepro.2020.123956.

Ait-Oubahou, A., Nur Hanani, Z.A., Jamilah, B., 2019. Packaging. Postharvest Technology of Perishable Horticultural Commodities. Elsevier, pp. 375–399. https://doi.org/10.1016/B978-0-12-813276-0.00011-0.

Alassali, A., Picuno, C., Chong, Z.K., Guo, J., Maletz, R., Kuchta, K., 2021. Towards higher quality of recycled plastics: limitations from the material's perspective. Sustainability 13 (23), 13266. https://doi.org/10.3390/su132313266.

Albrecht, S., Brandstetter, P., Beck, T., Fullana-i-Palmer, P., Grönman, K., Baitz, M., Deimling, S., Sandilands, J., Fischer, M., 2013. An extended life cycle analysis of packaging systems for fruit and vegetable transport in Europe. Int. J. Life Cycle Assess. 18 (8), 1549–1567. https://doi.org/10.1007/s11367-013-0590-4.

Alhazmi, H., Almansour, F.H., Aldhafeeri, Z., 2021. Plastic Waste Management: a review of existing life cycle assessment studies. Sustainability 13 (10), 5340. https://doi.org/10.3390/su13105340.

Alzubi, E., Kassem, A., Noche, B., 2022. A comparative life cycle assessment: polystyrene or polypropylene packaging crates to reduce citrus loss and waste in transportation? Sustainability 14 (19), 12644. https://doi.org/10.3390/su141912644.

Battini, D., Calzavara, M., Persona, A., Sgarbossa, F., 2016. Sustainable packaging development for fresh food supply chains. Packag. Technol. Sci. 29 (1), 25–43. https://doi.org/10.1002/pts.2185.

Bernstad Saraiva, A., Pacheco, E.B.A.V., Gomes, G.M., Visconte, L.L.Y., Bernardo, C.A., Simões, C.L., Soares, A.G., 2016. Comparative lifecycle assessment of mango packaging made from a polyethylene/natural fiber-composite and from cardboard material. J. Clean. Prod. 139, 1168–1180. https://doi.org/10.1016/J.JCLEPRO.2016.08.135.

Blanca-Alcubilla, G., Bala, A., de Castro, N., Colomé, R., Fullana-i-Palmer, P., 2020. Is the reusable tableware the best option? Analysis of the aviation catering sector with a life cycle approach. Sci. Total. Environ. 708, 135121. https://doi.org/10.1016/J.SCITOTENV.2019.135121.

Boschiero, M., Zanotelli, D., Ciarapica, F.E., Fadanelli, L., Tagliavini, M., 2019. Greenhouse gas emissions and energy consumption during the post-harvest life of apples as affected by storage type, packaging and transport. J. Clean. Prod. 220, 45–56. https://doi.org/10.1016/J.JCLEPRO.2019.01.300.

Budhiraja, V., Haslebacher, U., Rabanizada, N., Taveau, M., Kržan, A., 2025. Work package 2 deliverable D2.1: waste and litter generation by plastic food packaging strategies. https://stopp-project.eu/wp-content/uploads/2025/10/V1-D2.1-WASTE-AND-LITTER-GENERATION.pdf.

Camps-Posino, L., Batlle-Bayer, L., Bala, A., Song, G., Qian, H., Aldaco, R., Xifré, R., Fullana-i-Palmer, P., 2021. Potential climate benefits of reusable packaging in food delivery services. A Chinese case study. Sci. Total. Environ. 794, 148570. https://doi.org/10.1016/J.SCITOTENV.2021.148570.

Ceballos-Santos, S., de Sousa, D.B., García, P.G., Laso, J., Margallo, M., Aldaco, R., 2024. Exploring the environmental impacts of plastic packaging: a comprehensive life cycle analysis for seafood distribution crates. Sci. Total. Environ. 951, 175452. https://doi.org/10.1016/J.SCITOTENV.2024.175452.

Chen, H.L., Nath, T.K., Chong, S., Foo, V., Gibbins, C., Lechner, A.M., 2021. The plastic waste problem in Malaysia: management, recycling and disposal of local and global plastic waste. SN. Appl. Sci. 3 (4), 437. https://doi.org/10.1007/s42452-021-04234-y.

Circular Ecology. (2023, December 5). *Environmental glossary of terms and definitions*.

Coelho, P.M., Corona, B., ten Klooster, R., Worrell, E., 2020. Sustainability of reusable packaging–Current situation and trends. Resour. Conserv. Recycl.: X 6, 100037. https://doi.org/10.1016/J.RCRX.2020.100037.

Da Cruz, N.F., Ferreira, S., Cabral, M., Simões, P., Marques, R.C., 2014. Packaging waste recycling in Europe: is the industry paying for it? Waste Manag. 34 (2), 298–308. https://doi.org/10.1016/J.WASMAN.2013.10.035.

Deeney, M., Green, R., Yan, X., Dooley, C., Yates, J., Rolker, H.B., Kadiyala, S., 2023. Human health effects of recycling and reusing food sector consumer plastics: a systematic review and meta-analysis of life cycle assessments. J. Clean. Prod. 397, 136567. https://doi.org/10.1016/J.JCLEPRO.2023.136567.

Dolci, G., Puricelli, S., Cecere, G., Tua, C., Fava, F., Rigamonti, L., Grosso, M., 2025. How does plastic compare with alternative materials in the packaging sector? A systematic review of LCA studies. Waste Manag. Res.: J. Sustain. Circ. Econ. 43 (3), 339–357. https://doi.org/10.1177/0734242X241241606.

Espinoza-Orias, N., Lundquist, L., 2025. Life cycle assessment of reusable food and beverage packaging systems: a proposal of good practice. J. Clean. Prod. 499, 145207. https://doi.org/10.1016/J.JCLEPRO.2025.145207.

European Parliament. (2019). *Directive (EU) 2019/904 on the reduction of the impact of certain plastic products on the environment.*

eurostat. (2025). *Packaging waste statistics*.

Fan, Y.V., Čuček, L., Krajnc, D., Klemeš, J.J., Lee, C.T., 2023. Life cycle assessment of plastic packaging recycling embedded with responsibility distribution as driver for environmental mitigation. Sustain. Chem. Pharm. 31, 100946. https://doi.org/10.1016/J.SCP.2022.100946.

Fortune Business Insights. (2025). *Packaging market size, share & industry analysis, by material (Plastic, Paper & Paperboard, Metal, Glass, Wood, and Others), by product type (Rigid Packaging and Flexible Packaging), by Packaging type (Primary Packaging, Secondary Packaging, and Tertiary Packaging), by end-use industry (Food, Beverages, Cosmetics & Personal Care, Chemicals & Lubricants, Healthcare, Consumer Products, Building & Construction, Electronics, Automotive, and Others), and regional forecast, 2025-2032.*

Gallego-Schmid, A., Mendoza, J.M.F., Azapagic, A., 2018. Improving the environmental sustainability of reusable food containers in Europe. Sci. Total. Environ. 628-629, 979–989. https://doi.org/10.1016/J.SCITOTENV.2018.02.128.

Gao, S., Yao, J., Zhao, X., Ren, P., Gustavsson, M., Wu, C., 2025. Review on life cycle analysis (LCA) studies of reusable plastic crates for fruit and vegetables. Int. J. Sustain. Eng. 18 (1). https://doi.org/10.1080/19397038.2025.2457345.

Gatt, I.J., Refalo, P., 2022. Reusability and recyclability of plastic cosmetic packaging: a life cycle assessment. Resour. Conserv. Recycl. Adv. 15, 200098. https://doi.org/10.1016/J.RCRADV.2022.200098.

Gómez, I.D.L., Escobar, A.S., 2022. The dilemma of plastic bags and their substitutes: a review on LCA studies. Sustain. Prod. Consum. 30, 107–116. https://doi.org/10.1016/J.SPC.2021.11.021.

Greenwood, S.C., Walker, S., Baird, H.M., Parsons, R., Mehl, S., Webb, T.L., Slark, A.T., Ryan, A.J., Rothman, R.H., 2021. Many happy returns: combining insights from the environmental and behavioural sciences to understand what is required to make reusable packaging mainstream. Sustain. Prod. Consum. 27, 1688–1702. https://doi.org/10.1016/J.SPC.2021.03.022.

Helmes, R.J.K., Goglio, P., Salomoni, S., van Es, D.S., Vural Gursel, I., Aramyan, L., 2022. Environmental impacts of end-of-life options of biobased and fossil-based polyethylene terephthalate and high-density polyethylene packaging. Sustainability 14 (18), 11550. https://doi.org/10.3390/su141811550.

Hermansson, F., Ekvall, T., Janssen, M., Svanström, M., 2022. Allocation in recycling of composites - the case of life cycle assessment of products from carbon fiber composites. Int. J. Life Cycle Assess. 27 (3), 419–432. https://doi.org/10.1007/s11367-022-02039-8.

Hilmarsdóttir, G.S., Margeirsson, B., Spierling, S., Ögmundarson, O., 2024. Environmental impacts of different single-use and multi-use packaging systems for fresh fish export. J. Clean. Prod. 447, 141427. https://doi.org/10.1016/J.JCLEPRO.2024.141427.

Hitt, C., Douglas, J., Keoleian, G., 2023. Parametric life cycle assessment modeling of reusable and single-use restaurant food container systems. Resour. Conserv. Recycl. 190, 106862. https://doi.org/10.1016/J.RESCONREC.2022.106862.

Husmann, J., Northey, S., Beylot, A., Blömeke, S., Herrmann, C., 2025. Inconsistencies in handling of multifunctionality in the environmental footprint of electric vehicle batteries: a cross-industry analysis. Int. J. Life Cycle Assess. 30 (7), 1560–1578. https://doi.org/10.1007/s11367-025-02506-y.

International Organization for Standardization. (2006a). *Environmental management — Life cycle assessment — Principles and framework (ISO 14040:2006)*.

International Organization for Standardization. (2006b). *Environmental management — Life cycle assessment — Requirements and guidelines (ISO 14044:2006)*.

Jolliet, O., Margni, M., Charles, R., Humbert, S., Payet, J., Rebitzer, G., Rosenbaum, R., 2003. IMPACT 2002+: a new life cycle impact assessment methodology. Int. J. Life Cycle Assess. 8 (6), 324. https://doi.org/10.1007/BF02978505.

Karayılan, S., Yılmaz, Ö., Uysal, Ç., Naneci, S., 2021. Prospective evaluation of circular economy practices within plastic packaging value chain through optimization of life cycle impacts and circularity. Resour. Conserv. Recycl. 173, 105691. https://doi.org/10.1016/J.RESCONREC.2021.105691.

Kim, S.Y., Kang, D.H., Charoensri, K., Ryu, J.R., Shin, Y.J., Park, H.J., 2023. Comparative life cycle assessment of reusable and disposable distribution packaging for fresh food. Sustainability 15 (23), 16448. https://doi.org/10.3390/su152316448.

Koskela, S., Dahlbo, H., Judl, J., Korhonen, M.R., Niininen, M., 2014. Reusable plastic crate or recyclable cardboard box? A comparison of two delivery systems. J. Clean. Prod. 69, 83–90. https://doi.org/10.1016/J.JCLEPRO.2014.01.045.

Kousemaker, T.M., Jonker, G.H., Vakis, A.I., 2021. LCA Practices of plastics and their recycling: a critical review. Appl. Sci. 11 (8), 3305. https://doi.org/10.3390/app11083305.

Lendal, A., & Wingstrand, S.L. (.2019). *Reuse rethinking packaging*.

Levi, M., Cortesi, S., Vezzoli, C., Salvia, G., 2011. A comparative life cycle assessment of disposable and reusable packaging for the distribution of Italian fruit and vegetables. Packag. Technol. Sci. 24 (7), 387–400. https://doi.org/10.1002/pts.946.

Lo-Iacono-Ferreira, V.G., Viñoles-Cebolla, R., Bastante-Ceca, M.J., Capuz-Rizo, S.F., 2021. Carbon footprint comparative analysis of cardboard and plastic containers used for the international transport of Spanish tomatoes. Sustainability 13 (5), 2552. https://doi.org/10.3390/su13052552.

López-Gálvez, F., Rasines, L., Conesa, E., Gómez, P.A., Artés-Hernández, F., Aguayo, E., 2021. Reusable plastic crates (RPCs) for fresh produce (Case Study on Cauliflowers): sustainable packaging but potential Salmonella survival and risk of cross-contamination. Foods. 10 (6), 1254. https://doi.org/10.3390/foods10061254.

Mahmoudi, M., Parviziomran, I., 2020. Reusable packaging in supply chains: a review of environmental and economic impacts, logistics system designs, and operations management. Int. J. Prod. Econ. 228, 107730. https://doi.org/10.1016/J.IJPE.2020.107730.

Marson, A., Piron, M., Zuliani, F., Fedele, A., Manzardo, A., 2023. Comparative Life Cycle Assessment in the plastic sector: a systematic literature review. Clean Environ. Syst. 9, 100119. https://doi.org/10.1016/J.CESYS.2023.100119.

Mastellone, M.L., 2020. Technical description and performance evaluation of different packaging plastic waste management's systems in a circular economy perspective. Sci. Total. Environ. 718, 137233. https://doi.org/10.1016/J.SCITOTENV.2020.137233.

MG Market Growth Reports. (2026). *Packaging Materials Market size, share, growth, and industry analysis, by type (Plastic, Paper & Paperboard, Metal, Glass), by application (Food & Beverage, Healthcare, Personal Care, Industrial), regional insights and forecast to 2035*.

Miller, S.A., 2022. The capabilities and deficiencies of life cycle assessment to address the plastic problem. Front Sustain. 3. https://doi.org/10.3389/frsus.2022.1007060.

Mudgal, D., Pagone, E., Salonitis, K., 2024. Selecting sustainable packaging materials and strategies: a holistic approach considering whole life cycle and customer preferences. J. Clean. Prod. 481, 144133. https://doi.org/10.1016/J.JCLEPRO.2024.144133.

Nordahl, S.L., Scown, C.D., 2024. Recommendations for life-cycle assessment of recyclable plastics in a circular economy. Chem. Sci. 15 (25), 9397–9407. https://doi.org/10.1039/D4SC01340A.

Page, M.J., McKenzie, J.E., Bossuyt, P.M., Boutron, I., Hoffmann, T.C., Mulrow, C.D., Shamseer, L., Tetzlaff, J.M., Akl, E.A., Brennan, S.E., Chou, R., Glanville, J., Grimshaw, J.M., Hróbjartsson, A., Lalu, M.M., Li, T., Loder, E.W., Mayo-Wilson, E., McDonald, S., Moher, D., 2021. The PRISMA 2020 statement: an updated guideline for reporting systematic reviews. BMJ n71. https://doi.org/10.1136/bmj.n71.

Pålsson, H., Olsson, J., 2023. Current state and research directions for disposable versus reusable packaging: a systematic literature review of comparative studies. Packag. Technol. Sci. 36 (6), 391–409. https://doi.org/10.1002/pts.2722.

Pellengahr, F., Ghannadzadeh, A., van der Meer, Y., 2023. How accurate is plastic end-of-life modeling in LCA? Investigating the main assumptions and deviations for the end-of-life management of plastic packaging. Sustain. Prod. Consum. 42, 170–182. https://doi.org/10.1016/J.SPC.2023.09.014.

Rabiu, M.K., Jaeger-Erben, M., 2024. Reducing single-use plastic in everyday social practices: insights from a living lab experiment. Resour. Conserv. Recycl. 200, 107303. https://doi.org/10.1016/J.RESCONREC.2023.107303.

Rasines, L., San Miguel, G., Corona, B., Aguayo, E., 2024. Addressing the circularity and sustainability of different single-use and reusable crates used for fresh fruit and vegetables packaging. Food Packag. Shelf. Life 46, 101391. https://doi.org/10.1016/J.FPSL.2024.101391.

Rateau, F., Forbicini, M., 2025. Durability of reusable packaging. https://ecostandard.org/wp-content/uploads/2025/07/2025-07-25_ECOS_Durability-reusable-packaging-technical-paper.pdf.

Ross, S., Evans, D., 2003. The environmental effect of reusing and recycling a plastic-based packaging system. J. Clean. Prod. 11 (5), 561–571. https://doi.org/10.1016/S0959-6526(02)00089-6.

Sabate, K., Kendall, A., 2024. A review evaluating the gaps in plastic impacts in life cycle assessment. Clean Environ. Syst. 14, 100205. https://doi.org/10.1016/J.CESYS.2024.100205.

Sasaki, Y., Orikasa, T., Nakamura, N., Hayashi, K., Yasaka, Y., Makino, N., Shobatake, K., Koide, S., Shiina, T., 2022. Determination of the most environmentally friendly packaging for peach during transportation by modeling the relationship between food loss reduction and environmental impact. J. Food Eng. 331, 111120. https://doi.org/10.1016/J.JFOODENG.2022.111120.

Schmidt, S., Laner, D., 2023. The environmental performance of plastic packaging waste management in Germany: current and future key factors. J. Ind. Ecol. 27 (6), 1447–1460. https://doi.org/10.1111/jiec.13411.

Shamsuddoha, M., Kashem, M.A., 2024. Zero Plastic drive: a comprehensive review on unveiling innovative sustainable solutions for a circular plastics economy. Sustainability 16 (23), 10329. https://doi.org/10.3390/su162310329.

Shokri, A., Oglethorpe, D., Nabhani, F., 2014. Evaluating sustainability in the UK fast food supply chain. J. Manuf. Technol. Manag. 25 (8), 1224–1244. https://doi.org/10.1108/JMTM-04-2013-0031.

Silva, B., Costa, I., Santana, P., Zacarias, M.E., Machado, B., Silva, P., Carvalho, S., Faria, F., Basto-Silva, C., 2024. Environmental performance of different water bottles with different compositions: a cradle to gate approach. Clean Prod. Lett. 6, 100061. https://doi.org/10.1016/j.clpl.2024.100061.

Singh, S.P., Chonhenchob, V., Singh, J., 2006. Life cycle inventory and analysis of re-usable plastic containers and display-ready corrugated containers used for packaging fresh fruits and vegetables. Packag. Technol. Sci. 19 (5), 279–293. https://doi.org/10.1002/pts.731.

Snyder, W.R., Park, J., 2024. Environmental and economic analysis of reusable and single-use food packaging formats in University campus food services. Packag. Technol. Sci. 37 (9), 841–855. https://doi.org/10.1002/pts.2826.

Spierling, S., Venkatachalam, V., Mudersbach, M., Becker, N., Herrmann, C., Endres, H.-J., 2020. End-of-life options for bio-based plastics in a circular economy—Status quo and potential from a life cycle assessment perspective. Resources 9 (7), 90. https://doi.org/10.3390/resources9070090.

Tan, Q., Yang, L., Wei, F., Chen, Y., Li, J., 2023. Is reusable packaging an environmentally friendly alternative to the single-use plastic bag? A case study of express delivery packaging in China. Resour. Conserv. Recycl. 190, 106863. https://doi.org/10.1016/J.RESCONREC.2022.106863.

THE EUROPEAN PARLIAMENT, 2025. Packaging and Packaging Waste Regulation 2025/40 (PPWR). https://eur-lex.europa.eu/eli/reg/2025/40/oj.

Thomassen, G., Peeters, E., Van Hee, N., Noëth, E., Du Bois, E., Boone, L., Compernolle, T., 2024. The environmental impacts of reusable rice packaging: an extended comparative life cycle assessment. Sustain. Prod. Consum. 45, 333–347. https://doi.org/10.1016/J.SPC.2024.01.014.

Tua, C., Biganzoli, L., Grosso, M., Rigamonti, L., 2019. Life cycle assessment of reusable plastic crates (RPCs). Resources 8 (2), 110. https://doi.org/10.3390/resources8020110.

Tua, C., Nessi, S., Rigamonti, L., Dolci, G., Grosso, M., 2017. Packaging waste prevention in the distribution of fruit and vegetables: an assessment based on the life cycle perspective. Waste Manag. Res.: J. Sustain. Circ. Econ. 35 (4), 400–415. https://doi.org/10.1177/0734242X16688259.

Verma, M.K., Shakya, S., Kumar, P., Madhavi, J., Murugaiyan, J., Rao, M.V.R., 2021. Trends in packaging material for food products: historical background, current scenario, and future prospects. J. Food Sci. Technol. 58 (11), 4069–4082. https://doi.org/10.1007/s13197-021-04964-2.

Waste Framework Directive, 2024. Directive 2008/98/EC on waste and repealing certain directives. https://www.bundesumweltministerium.de/en/law/directive-2008-98-ec-on-waste-and-repealing-certain-directives.
World Economic Forum. (2017). *The new plastics economy catalysing action*.
Yadav, P., Silvenius, F., Katajajuuri, J.M., Leinonen, I., 2024. Life cycle assessment of reusable plastic food packaging. J. Clean. Prod. 448, 141529. https://doi.org/10.1016/J.JCLEPRO.2024.141529.
Zhou, K., Song, Y., Xian, X., Liu, Y., Fang, W., Liu, D., Jia, Y., Liu, J., 2024. A comprehensive comparison of the life-cycle environmental impacts of traditional and returnable express delivery packaging in China. J. Clean. Prod. 434, 140017. https://doi.org/10.1016/J.JCLEPRO.2023.140017.