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Comparative Life Cycle Assessment in the plastic sector: A systematic literature review

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ABSTRACT

European institutions have recently introduced new environmental policies relevant to the plastics sector. This could accelerate the transition to alternative materials. However, this transition needs to be based on reliable environmental assessments. A systematic review of the literature is presented with the aim of investigating the diffusion and the methodological setup of Life Cycle Assessment (LCA) in a comparative context for plastics and their alternatives. Emphasis was placed on the ability to support decision making, and a new procedure was proposed to assess whether the studies can support in the selection among alternatives. For this purpose, 79 articles were analyzed. The analysis showed that LCA is by far the most widely used environmental sustainability tool, although there is a lack of homogeneity in its application. Subjectivity remains in the definition of the methodological approaches, which are rarely discussed in the interpretation. The need for a comprehensive analysis often leads to trade-offs between environmental aspects which should be handled quantitatively and transparently, but weighting is rarely applied. However, weighting procedures may not be conclusive if different methodological approaches lead to conflicting findings. Many of the recommendations formulated involve testing several methodological combinations, requiring a multi-criteria analysis tool to aggregate the results.

1. Introduction

Plastics are ubiquitous in our production system due to their many benefits, but at the same time they are recognized as a source of serious environmental problems throughout their life cycle (Nielsen et al., 2020). Despite this, the global use of plastics between 2019 and 2060 is set to triple, according to OECD forecasts (OECD, 2022). European institutions have recently introduced new regulations to deal with these issues, such as Single Use Plastic ban (European Union, 2019), Circular Economy Action Plan and Plastic Strategy (European Commission, 2018, 2020), and the contribution system 'Plastic own resource' (European Union, 2020). Emblematic in this direction is the statement contained in the European strategy for plastic in a circular economy: 'Innovative materials and alternative feedstocks for plastic production are developed and used where evidence clearly shows that they are more sustainable compared to the non-renewable alternatives' (European Commission, 2018). This highlights two fundamental aspects. The first is that it is not assumed that plastic material from a non-renewable source is necessarily less sustainable than other materials from renewable raw materials (e.g. biomass). The second is the centrality and importance of metrics and tools for assessing environmental sustainability. These assessment tools have evolved rapidly. Green Chemistry metrics, closely related to resource consumption and waste generation, have led the way to a more comprehensive and complex approach, i.e., Life Cycle Thinking (LCT) (Sheldon, 2018). At the same time, the adoption of an effective communication strategy becomes increasingly important. The European Integrated Product Policy (European Commission, 2003) has already recognized the key role of consumers and the importance of raising their awareness and responsibility toward sustainability, but this concept can be extended to all decision-makers along the supply chain. Therefore, environmental sustainability tools (ESTs) must satisfy the need to provide as the most accurate and comprehensive measure, to enable effective and transparent communication, and to support the decision-making process. The balance between these needs is still an open challenge. In this context, there has been a proliferation of methodologies and standards that attempt to address these needs in different

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and effective ways (Martinez et al., 2019). Among these ESTs, Life Cycle Assessment (LCA) is of central because of its life cycle perspective and its comprehensive, quantitative, and science-based approach (Bjørn et al., 2018). The relevance of LCT and LCA is reflected in the increasing implementation in European policies (Sala et al., 2021). Although LCA is widely applied, there remain methodological issues that could undermine its robustness as a decision support tool. Other authors such as Bishop et al. (2021) and Tonini et al. (2021) have in the past investigated the methodological approaches used in comparative LCA and Carbon Footprint studies among plastics from different feedstocks, revealing a lack of homogeneity. These two studies, however, have some limitations. Bishop et al. (2021) focuses on comparisons between bio- and fossil-plastics, while Tonini et al. (2021) also considers recycled ones, but with a focus on the Carbon Footprint. In both cases, moreover, the capability of the reviewed studies to support decision-makers is not investigated.

Based on the identified gaps, this article presents a systematic review of peer-reviewed comparative studies applying LCA to plastic materials and their alternatives. This analysis focuses only on studies with a life cycle perspective in the packaging, building & construction and plastic in primary form sectors. These sectors were selected because they are the ones most discussed in the literature, and packaging and building materials are also the two sectors with the highest demand for plastics in Europe (Plastic Europe, 2021).Given the challenges in comparing environmental sustainability of different systems, the aim of this review is to gain insights on the ESTs used (to confirm that LCA is the most widely used tool in this sector), the methodological setup adopted in the studies and their ability to support decision makers. The research questions that will be addressed in this study can be summarized as follows:

- RQ1: Is LCA the most used EST in the literature to perform comparative studies among plastic materials or among plastic materials and non-plastic alternatives?
- RQ2: What are the methodological choices made in comparative LCA studies of plastics and their main alternatives?
- RQ3: Do the reviewed LCA studies support decision makers in selecting the preferable alternative from the environmental sustainability perspective?

The novelty of this research with respect to previous reviews (Bishop et al., 2021; Tonini et al., 2021) is the extension of the analysis to comparisons between plastics and non-plastic materials, as well as the evaluation of the ability of the studies to support decision-making. For the latter point, a new evaluation procedure has been proposed that considers a set of criteria based on the requirements of ISO 14040 and ISO 14044 and the characteristics of the study (both in terms of methodological approach and results obtained).

2. Material and methods

A systematic review includes four phases (Maestrini et al., 2017): source identification, source selection, source evaluation and data analysis.

2.1. Source identification

The source identification phase was conducted using the peer-reviewed academic databases Scopus and Web of Science.

The following string of keywords was used:

TITLE-ABS-KEY((environmental W/1 (performance OR impact OR impacts OR sustainability)) AND (comparative OR comparison) AND (plastic OR plastics OR polymer OR polymers))

To delineate the boundaries of the analysis, the following filters were applied:

- Only peer-reviewed papers were included (exclusion of book chapters and conference papers).
- Papers published before 2013 were excluded. As LCA methodology continues to evolve, this time criterion was added to provide a state-of-the-art view (Bishop et al., 2021).
- Only articles in English language have been included.

Through the software Mendeley Reference Manager (Elsevier, 2021) duplicate results were eliminated. The results of the search are described in Table 1.

2.2. Source selection

Once the subset of unique and potentially relevant articles was identified, a first selection process was performed on the titles and abstracts, to exclude contributions that, although they fell within the research criteria, were not related to the topic of interest. Therefore, the exclusion of studies was made according to the following criteria:

- Publications that do not use environmental sustainability metrics were excluded.
- Publications that do not present a comparison of alternatives were excluded.
- Publications not dealing with plastic materials or plastic goods were excluded.
- Publications that do not present case studies (e.g., reviews) were excluded.

As a result of this selection, 542 elements were excluded, resulting in 146 selected articles.

2.3. Source evaluation

The resulting 146 articles were evaluated according to the criteria described in Table SI-1.1 (Calzolari et al., 2022; Maestrini et al., 2017). During the evaluation, 11 articles out of 146 were excluded because they did not fall within the scope of the investigation. First, these 135 remaining publications were classified according to perspective. It was possible to identify two macro-groups: articles with a life-cycle perspective (cradle-to-grave or cradle-to-gate) and articles focused only on the end of life. The latter were excluded from the analysis because their focus is on the evaluation of different end-of-life waste management systems, and thus they cannot be traced back to a product or material perspective, which is the focus of this research. For publications with a life-cycle perspective, a further classification is proposed according to the sector: packaging, building & construction, plastic in primary form, medical devices, energy, textile, automotive, and others. Table 2 summarizes the results of the classification of publications. This analysis focuses only on studies with a life cycle perspective in the packaging (49), building & construction (15) and plastic in primary form (15) sectors. These sectors were selected because they are the ones most discussed in the literature, and packaging and building materials are also the two sectors with the highest demand for plastics (Plastic Europe, 2021). The chosen sectors cover 67.5% of studies with a life cycle perspective.

2.4. Methodology of analysis

To answer RQ1, all publications were analyzed (according to the criteria described in Table SI-1.1) by identifying the EST(s) used. In addition to the EST used (LCA, LCA-based Footprint, or others), it was highlighted whether the ESTs were applied alone or in combination. For LCA and LCA-based footprint, the approach used, i.e., attributional or consequential, was also examined.

To address RQ2, the main methodological aspects of the articles that used LCA or an LCA-based footprint were analyzed. The considered

Table 1
Criteria for the search for articles and resources identified.

Database	Fields of search	Language	Subject Area	Document Types	Years	Total	Total Both	Duplicate	Remaining
Scopus Web of Science	Article title; Abstract, Keywords	English	No restriction	Article, Review	2013–2022	535 439	974	286	688

Table 2
Classification by perspective and area of application of selected publications.

Perspective	N	Area of application	N	Relevance for the analysis
End-of-life	18	Waste	18	Excluded
Life cycle	117	Packaging	49	Included
		Building & Construction	15	Included
		Plastic in primary form	15	Included
		Medical devices	8	Excluded
		Energy	5	Excluded
		Textile	4	Excluded
		Automotive	3	Excluded
		Others	18	Excluded

methodological aspects, grouped by the four stages of LCA described by ISO 14044, are: Goal and Scope definition (system boundary, functional unit), Life Cycle Inventory (multifunctionality management, both for co-products and end-of-life), Life Cycle Impact Assessment (choice of impact categories and methods, normalization, and weighting), and Life Cycle Interpretation (sensitivity, and uncertainty analysis). Although the aspects listed do not cover all the methodological variables of an LCA study (e.g., cut-off is not addressed), it was felt that the most relevant ones were selected in light of the research objectives. The methodological aspects covered are in line with others reviews in the field (Bishop et al., 2021; Deviatkin et al., 2019; Moretti et al., 2021a; Tonini et al., 2021). Finally, to address RQ3, a new procedure was developed to assess whether the study can assist decision makers in selecting the preferred alternatives from an environmental sustainability perspective. Indeed, according to ISO 14040, there is no single solution on how to best apply LCA in the context of decision making. The proposed procedure involves the analysis of the articles content according to two sets of criteria. The first set of criteria (CS-*i*) was selected based on the requirements contained in the ISO 14040 and ISO 14044 standards, while the second set (CR-*i*) is closely related to the results obtained in the studies.

For the CS-*i* set, the focus was on requirements more related to methodological aspects (investigated in RQ2) than to inventory and data quality, for which it is assumed that the minimum requirements are met by the reviewed articles. It is also emphasized that the analysis assumes that a scientific article does not necessarily have to meet all reporting requirements for external communication defined in the ISO 14044 standard. In addition, it is assumed that the decision makers are internal stakeholders (e.g. the LCA commissioner) and therefore the limitations of ISO 14044 on the use of weighting procedures and MCDA tools for comparative assertions to be disclosed to the public are not applicable. The CS-*i* criteria and their relationship to the requirements of the standards ISO 14040 and ISO 14044 are shown in Table 3(more details on the requirements of the ISO standards considered can be found in Table SI-1.2). Most of the criteria analyzed (CS-1, CS-6, CR-1, CR-2, and C”-3) were based on objective evaluations related to whether or not certain aspects were addressed within the text of the article and the analysis of the results presented. For other criteria (CS-2, CS-3, CS-4, and CS-5), however, the evaluation required a process of critical analysis by the authors, potentially introducing aspects of subjectivity. For CS-2 and CS-3, the subjectivity relates to the need to assess whether the functional unit and system boundaries are consistent with the goal of the study, while for CS-4 the minimum number of impact categories to meet the comprehensiveness requirement was arbitrarily set at three. Finally, criterion CS-5 was considered satisfied if a sensitivity analysis on a

Table 3
Description of the selected criteria to assess the capability to support decision making.

Criteria	Description	Reference to ISO standard
CS-1	Are all the information needed to answer the following criteria available in the text?	ISO 14040 (4.1.6), ISO 14044 (4.2.2)
CS-2	Is the chosen functional unit representative of the function and in line with the goal of the study?	ISO 14044 (4.2.3.2)
CS-3	Are the system boundaries 'cradle-to-grave' or, if different, are they in line with the goal of the study?	ISO 14040 (4.1.2), ISO 14044 (4.2.3.3.1)
CS-4	Has a comprehensive and coherent with the goal set (arbitrarily $\geq$3) of impact categories been assessed?	ISO 14040 (4.1.7), ISO 14044 (4.2.3.4, 4.4.2.2, 4.4.5)
CS-5	Were the effects of the main methodological choices for which alternative approaches are common (e.g., functional unit, EoL allocation) on the study's conclusions assessed?	ISO 14044 (4.3.4.1, 4.4.5, 4.5.1.1, 4.5.3.3)
CS-6	Was the uncertainty of the results assessed?	ISO 14044 (4.4.5, 4.5.3.3)
CR-1	Is there an alternative that is better (or statistically equivalent) in all mid-point indicators analyzed, including sensitivity analysis effects on methodological aspects?	–
CR-2	If assessed, is there an alternative that is better (or statistically equivalent) in all end-point indicators analyzed, including sensitivity analysis effects on methodological aspects?	–
CR-3	If weighted results are presented, is there a better alternative also considering the effects of sensitivity analyses on methodological aspects?	–

methodological aspect was presented within the study. The diagram in Fig. 1 describes the classification of the articles into six different situations based on the evaluation of the above-mentioned criteria. In detail, the situations can be described as follows:

- Situation 1. The article cannot support decision makers due to lack in the goal and scope definition phase. A study may fall into this situation if there is a mismatch between the objectives and the scope definition, or if the minimum information to assess the criteria are not available within the article (lack of transparency). A prime example of this casuistry is when the study uses cradle-to-gate boundaries despite the compared alternatives have differences in end-of-life management.
- Situation 2. The article cannot support decision makers due to lack in the interpretation phase. This situation includes those studies where aspects of uncertainty related to methodological choices or aspects of the inventory (at least of the background data) have not been considered and tested. In these cases, decision makers do not know the degree of robustness of the information they receive and its sensitivity to the choices made by the LCA practitioner.
- Situation 3. The article can support decision makers because one of the alternatives analyzed is better (or statistically equivalent according to the uncertainty analysis performed) in all the mid-point indicators analyzed (and thus deemed consistent with the objective of the study) regardless of possible (subjective) methodological choices.

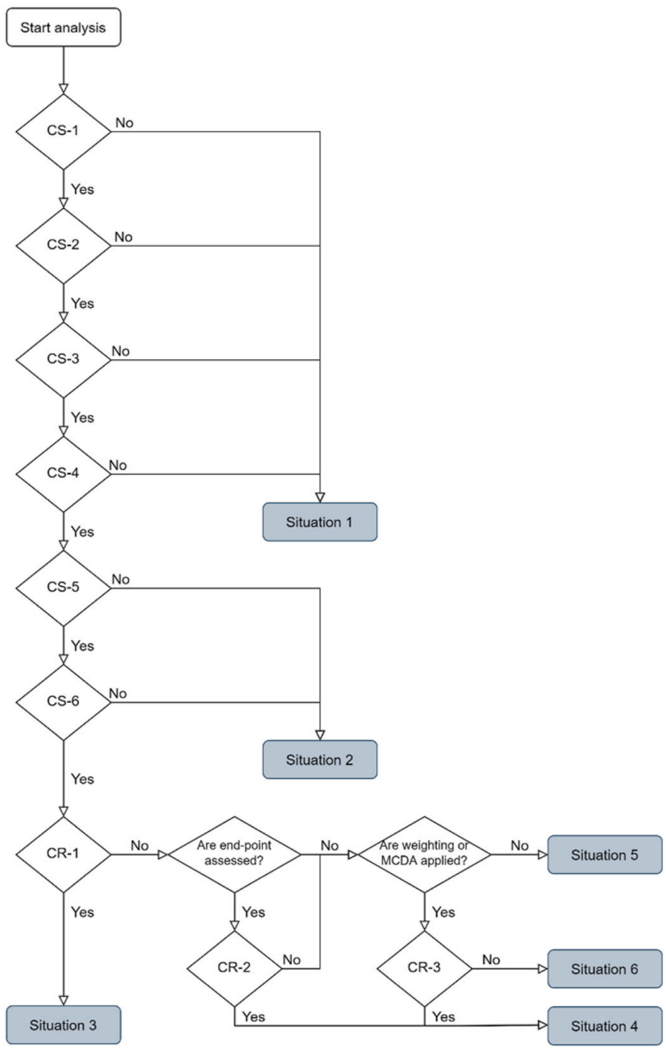

Fig. 1. Diagram of the criteria evaluation process and allocation to different situations.

- Situation 4. The article can support decision makers in a similar way to Situation 3, but the elements of subjectivity introduced in the end-point analysis and normalization and weighting or multi criteria decision analysis (MCDA) must be considered.
- Situation 5. The article cannot directly support decision makers as there are trade-offs that have to be managed using MCDA tools (including weighting procedures) at LCIA level and/or goal and scope definition (to manage different results under different methodological conditions).
- Situation 6. The article cannot directly support decision makers, as the study reaches different conclusions by making different methodological choices (e.g., the choice of allocation principle). In these cases, therefore, the decision maker does not get unambiguous results but will have to apply MCDA tools (including weighting procedures) at the goal and scope definition stage (to manage different results under different methodological conditions).

3. Results and discussion

In this section, the main results from the analysis are reported and discussed. In the introductory phase, the distribution of the articles in terms of sources (journals) and year of publication is presented, as well as a general contextualisation of studies, and then space is given to the three research questions. It is emphasized that all the results reported below refer only to the 79 articles falling in the packaging, building & construction, and plastic in primary form sectors. The summary of the analysis for all the sources is shown in Table SI-2.1.

The 79 articles were derived from 27 sources with different research areas. The most represented journals are Journal of Cleaner Production (19), Science of The Total Environment (11), International Journal of Life Cycle Assessment (10), Resources, Conservation and Recycling (6) and Sustainability (Switzerland) (5) (Fig. 2). Seven journals are present in the sample with two contributions, while fourteen are present with one. The other journals are present with two (7) or only one article (14). All the most represented journals have environmental science as one of their focus areas.

In the chosen time range, there is fairly constant activity between 2013 and 2018 with a significant increase in the number of articles published from 2019 (for 2022, the number is lower because the analysis was conducted on January 22, 2022.) (Fig. 3).

There are several reasons for the comparisons found in the studies analyzed. The most common situation is the comparison of several equivalent alternatives currently on the market to identify the less impactful. This may be done by considering only material variations (e.g., PET vs. PLA bottle) (Chen et al., 2016; Del Borghi et al., 2021; Desole et al., 2022; Maga et al., 2019; Moretti et al., 2021b; Piao et al., 2022; Ros-Dosda et al., 2019) or by considering different systems that provide the same function (e.g., PET vs. reusable glass bottle) (Ferrara et al., 2021; Koskela et al., 2014; Stefanini et al., 2021). In such cases, full scale production is almost always considered. For the analysis of plastics in primary form, the goal is to compare the performance of a polymer developed in the laboratory or on the pilot scale with the main alternatives on the market (e.g. (Ang et al., 2021; Nitkiewicz et al., 2020; Righi et al., 2017; Samer et al., 2021; Suriano et al., 2021),). The aims may be different: to provide a preliminary estimate of the impacts of the new material compared to the state of the art (Günkaya and Banar, 2016; Suriano et al., 2021), to identify environmental hot spots from an eco-design perspective (Righi et al., 2017), to optimise formulations and processes (Ang et al., 2021; Righi et al., 2017; Samer et al., 2021). Finally, a third motivation that leads to the comparison may be the evaluation of the effects of legislative actions such as the ban on single use plastics (e.g. (Chitaka et al., 2020; Gao and Wan, 2022; Zanghelini et al., 2020),). For example, Gao and Wan (2022) assessed the environmental consequences related to the substitution of PP straws with biodegradable alternatives in the United States, highlighting the environmental trade-offs between marine litter and LCA results. Thus, the motivations of the reviewed articles are manifold.

Although the analysis is limited to the three main sectors (packaging, building & construction, and plastic in primary form), a great variability is observed in terms of specific applications and materials. Among packaging, the most studied applications are bottles (27%), crates

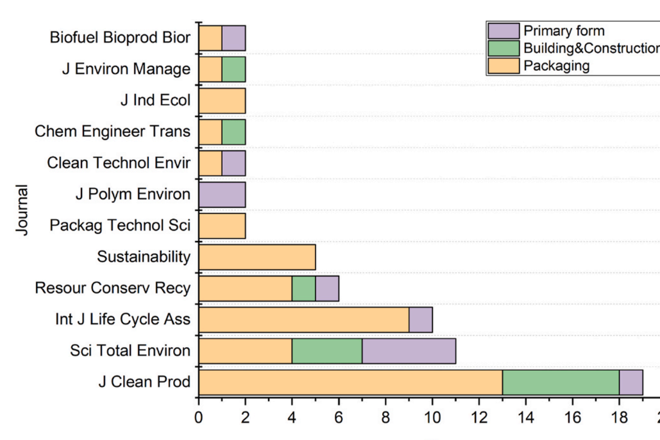

Fig. 2. Journals with at least two articles belonging to the analyzed sample.

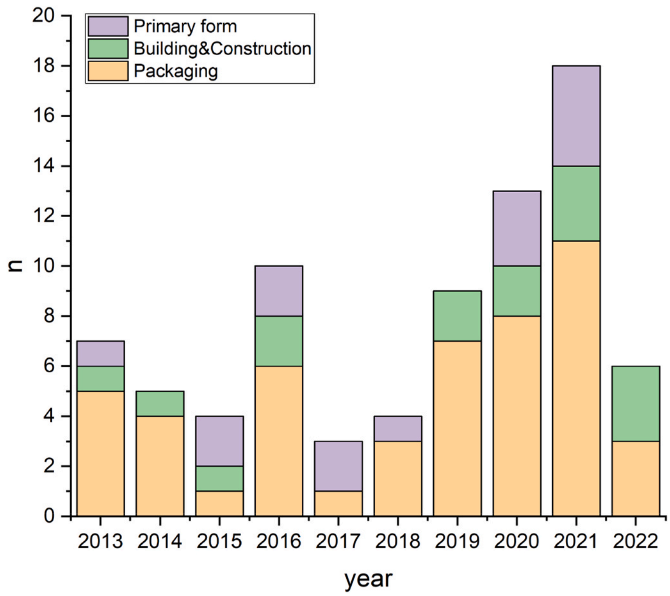

Fig. 3. Historical series of published papers.

(16%), films (12%), trays (10%), straw, cup, pallets (8%), and tableware (6%). The other applications in the packaging sector are investigated only in one study. The Building & Construction sector shows a more homogeneous situation, with asphalts and flooring systems with three studies each, followed by composites, insulation, piping, and screens (2).

On the contrary, the distribution of the types of materials investigated is more interesting and with sharper patterns. The most popular comparisons are those between traditional polymers and non-plastic materials (27), followed by those with biobased polymers (20). Only 11% of the studies do not present a traditional plastic (fossil-based and virgin) as an alternative. The different combinations of analysis are reported in Fig. 4. These combinations are evenly distributed among the three sectors analyzed. In fact, in the Building & Construction sector, only one study analyzed bio-based alternatives (La Rosa et al., 2014). In studies of plastics in primary form, bio-based alternatives are studied in 13 out of 15 contributions, while comparisons with recycled polymers (2) or non-plastic materials (1) are scarce.

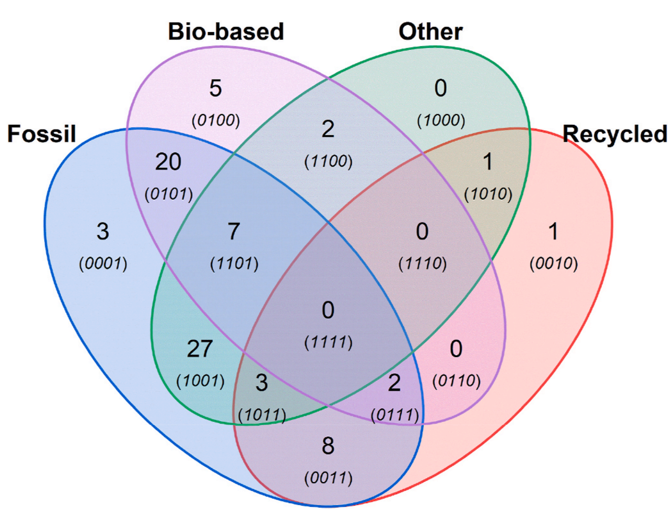

Fig. 4. Material sources classification according to the analyzed alternatives. The values indicate the number of studies comparing products made from plastic (fossil, bio-based, or recycled) or non-plastic materials. The graph then describes how the comparisons are distributed according to the material type of the alternatives considered.

3.1. RQ1: Diffusion of LCA as EST for comparative purpose

In line with RQ1, the considered articles were classified according to the EST used. EST has evolved over the years, from mass-based Green Chemistry indicators (e.g. E factor) to more comprehensive tools such as LCA (Sheldon, 2018). It is evident that LCA has gained a central role as a decision making tool in the field of environmental sustainability in recent decades. This is mainly due to its ability to provide an overview of the entire life cycle of a product or system with respect to a wide range of potential environmental impacts (Hauschild et al., 2018). LCA's multi-indicator approach can be particularly complex to interpret and communicate, which is why frameworks focused on a single area of concern (e.g., carbon and water footprint) have become popular (Ridoutt et al., 2016; Weidema et al., 2008). These LCA-based footprints have the advantage of being more immediate in understanding but have obvious limitations in comparative applications between different systems, i.e., they are not able to highlight a potential burden shifting.

The 79 considered articles confirm these trends. In fact, 75 articles use LCA as an EST (in 2 cases in combination with other metrics/tools, such as emergy analysis (de Souza Junior et al., 2020) and marine litter index (Stefanini et al., 2021)), while carbon footprint is used in alone or in combination with water and ecological footprint in 4 cases. Only in one case is the comparison based solely on a non-LCA-based metric, i.e. embodied CO_2 (Valente et al., 2022). The complete list of ESTs used in the analyzed articles is shown in Table SI-2.1.

Therefore, almost all articles (78 out of 79) present LCA or LCA-based footprint results. Among these, only in one case is the consequential approach used (the rarity of this approach in the plastic sector is in line with the findings of the review by Bishop et al. (2021)). In the other 77 articles, the attributional approach is not always stated directly by the authors (trend confirmed by Moretti et al. (2020)), but was deduced from the methodological setup of the goal and scope definition (e.g., system boundary and multifunctionality).

Theoretically, the consequential approach seems to be more suitable for assessing the effects of future decisions (Ekvall et al., 2016), such as the transition to a new material, particularly for emerging technologies (Bishop et al., 2021; Thonemann and Pizzol, 2019), but it is still a debated issue (Brander et al., 2019; Weidema et al., 2018). In addition to the different theoretical viewpoints, other elements that limit the application of the consequential approach in favor of the attributional are: difficulties in dissemination, higher complexity of the models (Bishop et al., 2021), lack of specific secondary data (Weidema, 2017) and uncertainty in building scenarios for replaced by-products.

3.2. RQ2: LCA methodological setup

This section provides an analysis of the methodological decisions made in the 78 studies using an LCA-based EST.

3.2.1. System boundary

Concerning the life cycle stages considered, the following cases were found: Cradle to Grave, Cradle to Gate, Gate to Gate, Cradle to Cradle. In this analysis, Cradle to Grave is also considered a study in which intermediate stages, such as the use phase, are omitted. Overall, the Cradle to Grave approach is the most widely used, but there is a clear difference between packaging and other sectors (Table 4). For the packaging sector, the number of studies considering the entire life cycle is close to 90%. On the contrary, this number drops around 30% for studies dealing with Building & Construction sector and plastics in primary form where the Cradle to Gate approach is the most used.

The use of the Cradle to Gate approach is usually justified because the subsequent stages are very uncertain (Resalati et al., 2021), the product life is very long, the subsequent stages are equivalent for all alternatives and therefore can be excluded (Ang et al., 2021; Lv et al., 2021; Piao et al., 2022; Wäger and Hischier, 2015), or the goal of the study is limited to the analysis up to the production of the

Table 4
System boundary considered in the reviewed articles. The percentages are calculated with respect to the column.

System Boundary	Packaging	Building&Construction	Primary form	Total
Cradle to Grave	90% (44)	29% (4)	33% (5)	68% (53)
Cradle to Gate	10% (5)	64% (9)	60% (9)	29% (23)
Gate to Gate	0% (0)	0% (0)	7% (1)	1% (1)
Cradle to Cradle	0% (0)	7% (1)	0% (0)	1% (1)

product/material (Bos et al., 2016; La Rosa et al., 2014; Salehi et al., 2022; Santos et al., 2021). In some cases, the decision is not justified in the article (Alvarenga et al., 2013; de Souza Junior et al., 2020; Kamau-Devers and Miller, 2020; Marcinkowski and Gralewski, 2020; Marson et al., 2021; Nitkiewicz et al., 2020; Samer et al., 2021; Suriano et al., 2021; Vahidi et al., 2016). This approach can be particularly controversial in comparisons between fossil-based and (partially) bio-based materials. In the comparison between bio-based PVC and fossil-based PVC proposed by Alvarenga et al. (2013), the exclusion of the end-of-life does not allow the different impacts of the biogenic CO_2 share to be appreciated. Nitkiewicz et al. (2020) recognize that their comparison among different PHB production scenario could be very different applying a Cradle to Grave assessment. In other articles, the extension of the system boundary to the end of life is seen as a possible future development (Kamau-Devers and Miller, 2020).

Among the considered articles, there are 2 isolated cases of the use of other system boundaries. A Gate-to-Gate system was considered by Righi et al. (2017) to analyze the environmental impacts of two protocols for PHB production starting from the microbial biomass to the pre-formed polymer. Although this approach allows comparison of the impacts of the production process, it is unable to quantify the effects of variations in raw material inputs related to different process yields. Instead, Ros-Dosdà et al. (2019) define their study on floor coverings as Cradle to Cradle due to integration into the system boundary of the recycling process. In fact, the approach is not particularly different from a Cradle to Grave study that uses a multifunctionality management approach that includes end-of-life recycling impacts ("*0:100, no credits*" EoL formulas according to Allacker et al.'s (2017) nomenclature).

3.2.2. Functional unit

The ISO 14044 standard defines the functional unit as the "quantified performance of a product system for use as a reference unit" (ISO, 2020). The ISO standard highlights that 'comparison between systems shall be made on the basis of the same functions quantified by the same functional unit' and that 'shall be consistent with the goal and the scope of the study' (ISO, 2020). The literature confirms that the choice of functional unit in a comparative study can heavily influence the results (Manzardo et al., 2019).

The choice of functional unit can be guided by sectoral guidelines, such as product category rules. For the Building&Construction sector the main European reference is the EN 15804:2012+A2:2019 standard, which defines two options by means of which the function(s) of the system can be expressed: functional and declared unit (CEN, 2019). The definition of functional unit is consistent with that of ISO 14044. Instead, a declared unit shall be applied when there is a wide range of possible uses and functions of the product, and is typically expressed in terms of number of items, mass, length, area or volume (CEN, 2019). A similar classification is suggested by the Packaging category rules developed by EPD International EPD System (2022). According to this guideline, only studies based on Cradle to Grave system boundary can adopt a functional unit, which shall be expressed as one packaging product unit.

The functional/declared units used in the revised studies were grouped into six categories: mass based (e.g., 1 kg), amount of contained products (e.g., 1 liter of milk), number of items (e.g., 1 tray), number of items with the same carrying capacity (e.g., 1 tray with a volume of 1 liter), number of uses (e.g., 1 use of cup) and functional unit related to the performance during the use phase (e.g., thermal resistance for an insulation panel).

Table 5 summarizes the types of functional/declared units used in the revised articles, depending on the application field and system boundaries. The segmentation by sector allows for two decisive patterns to be identified. All comparative analyzes evaluating plastics in primary form use a mass-based functional unit. This is understandable given the impossibility of traceability of the material to a specific application (and thus to a function), but certainly limits the study's conclusions. The situation was the opposite for the Building&Construction sector. In this case, the most frequent functional units are intended to be representative of the actual application of the product, trying to improve possible alternatives with different technical performance (Cherubini et al., 2019; Feifel et al., 2015; Ingrao et al., 2016; Marcinkowski and Gralewski, 2020; Piao et al., 2022; Resalati et al., 2021; Ros-Dosda et al., 2019; Vahidi et al., 2016). A clear example of this approach is the comparison among different insulation panels proposed by Resalati et al. (2021), which use the "amount of insulating core material needed to achieve a U-value of 0.27 W/m^2K in a 1 m^2 of panel" as functional unit. Other similar examples include selecting a functional unit for comparing pavement systems by considering area and reference service life (Feifel et al., 2015; Ros-Dosda et al., 2019), or for piping systems, again by defining operating conditions, a distance, and a time horizon (Marcinkowski and Gralewski, 2020; Vahidi et al., 2016).

For the packaging sector, the situation is more complex and deserves to be evaluated on a case-by-case basis. A more detailed overview of the functional units used for the ten categories of packaging is shown in Fig SI-2.1.

For bottles and crates, the most common approach is to compare alternatives with respect to the amount of product contained (or to compare packaging with the same available volume). Only two deviations from this approach were found: the use of 1 kg as functional unit for comparing bio-based and fossil-based PET bottles was used by Chen et al. (2016), while Abejón et al. (2020) considers the number of necessary uses to fulfill the distribution of 1000 metric tons of fruits and vegetables with single use and reusable crates. In general, also for other applications such as cups and pallets, the use of the functional unit based on the number of uses is used in comparisons considering single use and reusables alternatives, or otherwise significantly different life expectancies (Abejón et al., 2020; Anil et al., 2020; Civancik-Uslu et al., 2019; Woods and Bakshi, 2014). For tray studies, the comparison is always made between alternatives with similar dimensions, which may, however, have some differences in available volume due to the mechanical properties of the materials (e.g., XPS and PET trays for meat described by Maga et al. (2019)). The three observed scenarios are therefore: comparison among trays with the same available volume (David et al., 2021; Toniolo et al., 2013, 2017), comparison according to the weight of contained products (Blanc et al., 2019) or comparison among trays with similar geometry disregarding small differences in available volume (Maga et al., 2019). Even in film comparisons, the situation is fragmented, as functional units based on: surface (Deng et al., 2013; Günkaya and Banar, 2016; Leceta et al., 2013), total production over ten year from a single plant (expressed in mass) (Joachimiak-Lechman et al., 2019), and number of uses (Civancik-Uslu et al., 2019). More homogeneous is the choice of functional unit for tableware and straw, where the comparison is made according to a defined number of items.

3.2.3. Multifunctionality

To describe the lifecycle of plastics, whether fossil, recycled or bio-based, the issue of multifunctionality must be addressed. The ISO 14044 standard (incorporating Amendment 2) provides a hierarchy of approaches to deal with multifunctionality (ISO, 2020): avoid allocation

Table 5
Functional unit considered in the reviewed articles. The percentages are calculated with respect to the column. One case of 'gate to gate' system boundary is omitted from the table.

Functional unit	Sector			System Boundary		Total
	Packaging	Building& Construction	Primary form	Cradle to Grave	Cradle to Gate	
Mass	4% (2)	29% (4)	100% (15)	9% (5)	65% (15)	27% (21)
Amount of contained products	35% (17)	0% (0)	0% (0)	32% (17)	0% (0)	22% (17)
Number of items (with same carrying capacity)	29% (14)	0% (0)	0% (0)	25% (13)	4% (1)	18% (14)
Number of items	18% (9)	14% (2)	0% (0)	17% (9)	9% (2)	14% (11)
Performance in use	0% (0)	57% (8)	0% (0)	6% (3)	17% (4)	10% (8)
Number of uses	8% (4)	0% (0)	0% (0)	8% (4)	0% (0)	5% (4)
Surface	6% (3)	0% (0)	0% (0)	4% (2)	4% (1)	4% (3)

by dividing the unit process into single-functionality sub-processes or expanding the product system to include the additional functions to the co-products; perform allocation according to underlying physical relationship between co-products; perform allocation according to different relationships, e.g., economic value of the co-products. Although the latest version of the standard may lead to misunderstandings (Heijungs et al., 2021), system expansion can be applied in two ways: by adding functions (enlargement) or by subtracting additional functions (substitution) (Finkbeiner, 2021). In a comparative perspective, the option of adding functions to the system could lead to inconsistency between the systems being compared, which is why the substitution approach seems more suitable. The appropriateness of using system expansions approach in an attributional LCA is widely debated in the literature (Moretti et al., 2020), but Amendment 2 of ISO 14044 confirms that it can be applied in all types of LCAs (and not only in consequential ones) (Finkbeiner, 2021; ISO, 2020, p. 14).

The issue of multifunctionality arises at two stages: for the management of co-products during production processes and for end-of-life management, due to the presence of material flows for recycling and/or energy recovery. Of the articles considered, 74% and 51% did not clearly state the approach used for managing, respectively, co-products and end-of-life multifunctionality. The end-of-life value drops to 40% for Cradle to Grave studies only. This trend was confirmed by Bishop et al. (2021). Considering the great influence the allocation approach can have on the results (Toniolo et al., 2017), not discussing this aspect leads to a loss of transparency. It is also important to emphasize that ISO 14044 prescribes evaluating the effects of different allocation approaches where applicable (ISO, 2020).

3.2.3.1. Multifunctionality for co-products. Multi-output processes can often be only in the background system, e.g., for fossil-based polymers, whereas in the foreground system there is no need to manage multifunctionality between co-products. For example, in Maga et al. (2019) the allocation approach between co-products is not discussed as it is not necessary for the foreground system (trays production from plastic granulates). Implicitly, the allocation approach adopted in the secondary datasets used is therefore inherited, without any discussion of consistency with the goals of the study. A similar situation has been found, for example, in (Desole et al., 2022; Ferrara et al., 2021; Koskela et al., 2014; Moy et al., 2021). The choices made in the construction of the database therefore play a fundamental role. The Ecoinvent database, which is by far the most used source of secondary data among the reviewed articles, uses economic allocation when subdivision is not feasible (except for energy, for which an exergy-based allocation is used) (Ecoinvent, 2022). Instead, Ecoinvent applies substitution only in the consequential system model (Ecoinvent, 2022).

Among the studies that stated how they approached multifunctionality between co-products, the most frequently used approach is the allocation based on physical relationships. Mass-based allocation approach is used in seven articles dealing with bio-based plastics (Alvarenga et al., 2013; Ang et al., 2021; Chen et al., 2016; Chitaka et al., 2020; Deng et al., 2013; La Rosa et al., 2014; Moretti et al., 2021b) and in one focusing on fossil-based PP manufacturing (Lv et al., 2021). For example, a mass-based allocation for PLA was adopted by Moretti et al. (2021b) in a comparison among bio- and fossil-based cups derived from corn starch and highlight the impossibility of deriving the allocation approach used in the secondary data for fossil-based plastics. In the same article, the surplus energy generated during the production of PLA from sugarcane was managed by substituting grid electricity. In a similar situation, an exergetic allocation among bagasse and electricity was used by Alvarenga et al. (2013). The use of energy as an allocation criterion was used by Belboom and Léonard (2016) to separate impacts between bio-based ethanol and pulps and by Hansen et al. (2015) to deal with the crude oil supply chain. In Hansen et al.'s article (2015), economic allocation is also used to split the impacts between ethanol and the electricity surplus. The starch content was used by Papong et al. (2014) to allocate burdens between cassava starch and pulp, while in the same situation Changwichan et al. (2018) used an economic-based approach. Economic allocation was applied in other four studies (Amasawa et al., 2021; Kočí, 2019; Rodríguez et al., 2020; Wäger and Hischier, 2015), e.g., to deal with wood pallets and wood chips (Kočí, 2019) or with banana and banana fiber (Rodríguez et al., 2020). Wäger and Hischier (2015) present a comparison between virgin and recycled plastic from WEEE. As a base case, they consider an economic allocation to share the burdens of WEEE separation between plastic-containing fractions and other parts, but tested also another approach (mass-based allocation) throughout the sensitivity analysis (as suggested by ISO standard) (Wäger and Hischier, 2015). In their case, the results were not significantly affected by the different approaches.

The substitution approach was found in six articles, five of which related to the packaging sector (Abejón et al., 2020; Chen et al., 2016; Moretti et al., 2021b; Van der Harst et al., 2014; Vural Gursel et al., 2021) and one to plastics in primary form (Bos et al., 2016). Vural Gursel et al. (2021), comparing a fossil- and bio-based route for PET, applied substitution both for energy surplus (e.g., derived from bagasse burning) and for digestate, beet pulp, and carbolime that are able of avoiding mineral fertilizer production. Bos et al. (2016) discuss extensively different methods (substitution and mass-based allocation) to deal with multifunctionality in the life cycle of five vegetable oils for bio-based products. It should be emphasized that in the article is reported that: 'LCA prescribes the use of system expansion (substitution method or consequential LCA) whenever possible, and use allocation methods (attributional LCA) only when absolutely inevitable' (Bos et al., 2016). Thanks to the latest amendments to the ISO standard, it has been clarified that there are no limitations in the use of system expansion in attributional LCA (Finkbeiner, 2021; ISO, 2020, p. 14044). The article then presents a comparison between substitution and mass allocation for the management of the multifunctionality of crops cultivation stage (oil, meal, straw, bunch, and kernels), but, it is a comparison between consequential and attributional results (in the first case, also marginal productions related to surplus demand for vegetable oils are taken into account).

The analysis thus showed how a multitude of approaches can be used, with significant implications for the study's conclusions.

Furthermore, there appear to be no criteria according to which one approach is more relevant than others in an absolute sense, unless their prevalence is found in the literature for similar cases. In fact, not even PCRs such as EN 15804 (CEN, 2019) or the one for packaging by International EPD System (2022) take a position on this aspect, reproposing the hierarchy of ISO 14044 without the system expansion option.

3.2.3.2. Multifunctionality for end-of-life. As mentioned above, 51% of the reviewed articles do not declare the multifunctionality management approach used for end-of-life. This value drops to 40% for Cradle to Grave articles only, where end-of-life management of the product/material is required. Bishop et al. (2021) presented a list of waste management operations with multifunctionality: 'second life from recycling materials; compost from composting; biogas; electricity, and heat from incineration; and energy from landfill gas'. All the processes listed are potentially relevant to the materials covered by the articles considered in this review. For the management of multifunctionality at this stage, the hierarchy of ISO 14044 presented above is still applicable. Due to the complexity of the issue, other standards (e.g., ISO 14067 (ISO, 2018) and PAS 2050 (British Standards Institution, 2011)), guidelines (e.g., PEF method (Zampori and Pant, 2019)) or category rules (e.g., EN 15804 (CEN, 2019)) suggest specific allocation procedures, in some cases as mandatory (CEN, 2019; Zampori and Pant, 2019), in others as optional (British Standards Institution, 2011; ISO, 2018). Allacker et al. (2017), as part of their definition of the end-of-life allocation approach for PEF, present an extensive literature analysis and formalize 11 different formulas, demonstrating the lack of consensus (Toniolo et al., 2017). An assessment of 12 different allocation procedures according to 10 criteria (e.g., easy to use, understandable, relevant to decision makers) was proposed by Ekvall et al. (2020), concluding that none could fully satisfy all of them.

In the reviewed articles, excluding those in which the approach is not declared, the most common procedures are substitution (Abejón et al., 2020; Albrecht et al., 2013; Amasawa et al., 2021; Belboom and Léonard, 2016; Bertolini et al., 2016; Cleary, 2013; Deng et al., 2013; Ferrara et al., 2021; Ferrara and De Feo, 2020; Fieschi and Pretato, 2018; Hottle et al., 2017; Kočí, 2019; Korbelyiova et al., 2021; Koskela et al., 2014; Lo-Iacono-ferreira et al., 2021; Maga et al., 2019; Moretti et al., 2021b; Papong et al., 2014; Piao et al., 2022; Potting and van der Harst, 2015; Rybaczewska-Blazejowska and Mena-Nieto, 2020; Toniolo et al., 2017; Van der Harst et al., 2014; Vural Gursel et al., 2021; Woods and Bakshi, 2014), cut-off approach (Chitaka et al., 2020; David et al., 2021; de Souza Junior et al., 2020; Del Borghi et al., 2021; Marson et al., 2021; Rodríguez et al., 2020; Salehi et al., 2022; Toniolo et al., 2013, 2017; Wäger and Hischier, 2015) (or "100:0, no credit" according to the nomenclature proposed by Allacker et al. (2017)), and Circular Footprint Formula (CFF) suggested by PEF method (Boesen et al., 2019). Besides the main ones, other approaches were found: '100:100' (in which recycling impacts are allocated to both the upstream and downstream system) (Civancik-Uslu et al., 2019; Santos et al., 2021) and '50:50' (equal share of recycling impacts among the upstream and downstream system) (Cherubini et al., 2019; Zanghelini et al., 2020). The circular footprint formula (CFF) recommended by PEF method is the European Commission's attempt to provide a comprehensive procedure that can handle all aspects: recycled material, material recycling and its quality, allocation of the burdens and benefits between upstream and downstream product systems, incineration, energy recovery and landfilling. The effort to make it more comprehensive and able to reflect decisive characteristics (e.g., secondary material's quality) results in greater complexity of use and understanding (Ekvall et al., 2020). Among the articles reviewed, CFF is applied only by Boesen et al. (2019).

An extensive description of the different allocation procedures can be found in (Allacker et al., 2017; Ekvall et al., 2020), the following is a quick introduction aimed at clarifying the choices made in the reviewed articles.

The substitution approach can be applied to provide credits related to output flows from waste management operations (materials or energy). Different approaches can be described for materials destined for mechanical recycling: substitution of the same virgin material (closed-loop recycling) or substitution of other virgin material (open-loop) (Ekvall and Tillman, 1997). Correction factor can be applied in relation to the efficiency of the recycling process and to the quality ratio of primary and secondary material (Rigamonti et al., 2009). For example, Ferrara and De Feo (2020), considered a recycling efficiency of 76% and a substitution ratio of 0.81 for the closed loop recycling of PET bottle replacing PET granulate. The definition of avoided materials can be more complex for compostable plastics (Bishop et al., 2021). Moretti et al. (2021b), in defining an end-of-life scenario for PLA cups, considered both mechanical recycling and composting. For mechanical recycling, a process efficiency of 70% and a substitution rate (compared to granulated PET) of 0.81 were considered. For the fraction intended for composting, the boundaries include all processes up to field application, taking into account credits due to the non-use of synthetic fertilizers (the N, P, and K content derives from organic contamination, and not from PLA, for which a zero content of the three elements was assumed) (Moretti et al., 2021b). In a similar situation, Van der Harst et al. (2014), Papong et al. (2014) (for PLA) and Amasawa et al. (2021) (for PHBH) ignored the organic contamination, therefore, compost has not been credited as an avoided fertiliser. In contrast, Deng et al. (2013), dealing with a wheat gluten film, considered the avoided impacts associated with compost in relation to the N content of the material.

Energy flows (electricity and heat) may result from incineration processes or from the combustion of biogas generated through anaerobic digestion or collected from landfills. The main distinguishing element between the reviewed articles is the energy mix considered as substituted: average grid mix (e.g. (Belboom and Léonard, 2016; Maga et al., 2019; Papong et al., 2014),) or marginal grid mix (e.g. (Moretti et al., 2021b; Vural Gursel et al., 2021),).

A homogeneous approach is not always used for the substitution of end-of-life material and energy flows. For example, Maga et al. (2019) recognize the credits due to energy recovery, but adopt the cut-off approach for the burdens and benefits of mechanical recycling. According to the cut-off approach (sometimes referred to as the approach of recycled content or '100:0' (Allacker et al., 2017; Ekvall et al., 2020)), each product should be assigned all environmental impacts caused by the product, specifically allocating the recycling processes to the system using recycled material (Ekvall and Tillman, 1997). This approach is widely applied in the context of EPDs, being recommended by EN 15804 (CEN, 2019) (with the exception of module D) and from the International EPD System (2022). According to the assessment carried out by Ekvall et al. (2020), cut-off method is probably the easiest to apply, but does not differentiate the output of recycling processes according to their quality. Toniolo et al. (2017), investigated the influences of the choice of allocation approach in a comparative study of PET trays with the same recycled content but different recyclability, obtaining homogeneous results by applying cut-off and substitution. As the authors recognized, the result is closely linked to the particular conditions of the case study, in fact, Van der Harst et al. (2014) and Hottle et al. (2017) obtained significant differences when applying the two approaches.

Again, there is no possibility of convergence towards a single approach, making it imperative to test the most common options in the literature in order to establish a degree of robustness of the conclusions of the comparative analysis.

3.2.4. Life Cycle Impact Assessment methods

Life Cycle Impact Assessment (LCIA) is the LCA phase aimed to assess the magnitude of the contribution of each elementary flow to a potential impact on the environment (Hauschild et al., 2018). According to ISO 14044 (ISO, 2020) LCIA shall include three mandatory elements: selection of impact categories, indicators, and characterization models; classification (assignment of LCI results to the selected impact categories); and characterization (calculation of category indicator results). The results of the category indicator can be expressed at two different levels of the environmental mechanism (Hauschild et al., 2018): midpoint (more measurable but less representative of the concerns observed in the environment (Rosenbaum et al., 2018), e.g., Eutrophication, Acidification) or endpoint (more relevant but less transparent (Rosenbaum et al., 2018), e.g., ecosystem quality, Human health). The aim of this section is to explore the choices of impact assessment methods (i.e., collection of characterization models (Hauschild et al., 2013)) and level of analysis (midpoint or endpoint) in the reviewed articles. A summary of the methods used and their level of analysis can be found in Table SI-2.2.

The most used impact assessment method is by far ReCiPe (Huijbregts et al., 2016) (considering both the 2008 and 2016 version), followed by CML-IA (de Bruijn et al., 2002), IMPACT 2002+ (Jolliet et al., 2003), TRACI (Bare, 2002) and the EF method (Zampori and Pant, 2019). The Eco-indicator 99 (Goedkoop and Spriensma, 2001) method records two applications in the first years of the analyzed interval, while in more recently the characterisation models section of the ILCD (EC-JRC, 2011) has been applied in three articles. In some cases, the authors selected individual indicators (or a set of indicators related to an area of concern), such as the GWP 100 indicator based on the IPCC model to assess the impacts of GHG emissions, the cumulative energy demand (CED) (Frischknecht and Jungbluth, 2004) to have an insight of the different energy resources consumed, UseTox (Rosenbaum et al., 2008) for indicators of human and aquatic toxicity, Ecological scarcity (Frischknecht, rolf et al., 2006) and the water footprint model proposed by Hoekstra et al. (2011). Among these models, only the IPCC was applied alone in four studies (Korbelyiova et al., 2021; Leejarkpai et al., 2016; Lo-Iacono-ferreira et al., 2021; Papong et al., 2014), while in the other cases a combination of indicators or indicators and methods is always observed. For example, Piao et al. (2022) defined its own set of indicators using IPCC, CED, USEtox and Ecological scarcity, while Zanghelini et al. (2020) took single indicators from different methods (e.g., ReCiPe, CML) and created a set of 13 impact categories. The use of indicators from different methods may be a way to increase the comprehensiveness of the evaluation, but could lead to inconsistencies in perspective (Bare and Gloria, 2008).

The adoption of multiple indicators (and a variety of methods) is essential from a comparative perspective. According to ISO 14044 an evaluation of the completeness of the LCIA shall be done (ISO, 2020), and Bare and Gloria (2008) proposed a taxonomy for assessing the completeness of the relevant impact categories. The reviewed articles range from considering only one impact category to a whole set of 18, for example, ReCiPe. If only a limited number of impact indicators are used, this must be stated as a limitation of the study and conclusions must be formulated accordingly. For example, Leejarkpai et al. (2016), comparing PS, PET and PLA boxes, state that PLA "showed the greatest environmental benefit" although the only environmental indicator considered was GWP. On the contrary, using a very broad and comprehensive set of indicators allows the identification of burden-shifting phenomena between environmental categories but makes the interpretation of the results challenging. For this reason, midpoint indicator analysis can also be integrated with endpoint analysis, as recommended by Rosenbaum et al. (2018). Indeed, an endpoint level analysis based on a smaller number of indicators can facilitate decision making by providing more condensed information (Rosenbaum et al., 2018). However, limitations associated with endpoint modelling must be recognized, whereby some pathways remain uncovered, losing completeness (Laurent et al., 2020). It can therefore be concluded that to maximise the information that can be derived from the methods used, where possible, a combined assessment and interpretation of midpoint and endpoint indicators may be a solution. Approximately 20% of the reviewed articles evaluate both midpoint and endpoint indicators, with ReCiPe being by far the most widely used method. It combines the 18 midpoint indicators into three 'endpoint areas of protection'': damage to human health, damage to the ecosystem, and damage to resource availability (Huijbregts et al., 2016). Similar endpoint indicators are considered by the IMPACT 2002+ method, which, however, treats climate change as a separate damage category (Jolliet et al., 2003). Although the use of the endpoint perspective leads to a reduction in the number of indicators, it is common in a comparison not to obtain univocal results. For example, in the comparison among beverage packaging provided by De Feo et al. (2022), no alternative (aluminium, glass, PET, aseptic carton) performs better in the three endpoint indicators. Similar trends are reported in (Ferrara et al., 2021; Nitkiewicz et al., 2020). It follows, therefore, that if the goal of the analysis is to support decision making, weighting procedures or other multi-criteria approaches may be necessary.

3.2.5. *Normalization and weighting*

The strength of LCA is the ability to assess impacts on multiple environmental aspects throughout the life cycle of the system considered. If the aim of the study is to analyze the environmental profiles of a material as to support well as the decision in the comparison and choice between different alternatives, tools to achieve a synthesis of multi-criteria results become necessary. In this regard, ISO 14044 (ISO, 2020, p. 14044) defines the optional normalization (calculating the magnitude of the indicators of categories relative to the reference information) and weighting procedures (converting and possibly aggregating the indicators' results in impact categories using numerical factors based on value-choices). Normalization approaches can be classified into internal approaches (referring to impacts of an alternative under study) or external approaches (referring and external reference independently of the object of the LCA study) (Hélias et al., 2020). An example of external normalization can be found in the PEF methodology, in which impacts are referred to those of an average European citizen in 2010 (EC-JRC, 2014). Typical problems of external normalization are related to uncertainty linked to the data of the reference system, discrepancies between the substances of the two life cycle inventories (analyzed and reference system), inverse proportionality and the large amount of data required (Hélias et al., 2020; Kim et al., 2013; Pizzol et al., 2017; White and Carty, 2010).

Weighting, that is a form of Multi Criteria Decision Analysis (MCDA) (Laurin et al., 2016), can be used to facilitate decision making representing an evaluation of the relative importance of the impacts (Pizzol et al., 2017). Nowadays the most common method adopted in LCA is Simple Additive Weighting (Laurin et al., 2016), but the use of more advanced MCDA approaches is growing (Dias et al., 2019; Pizzol et al., 2017; Prado et al., 2012). Even if practitioners perceive normalization and weighting processes negatively due to uncertainty and loss of robustness (Pizzol et al., 2017), their contribution to the interpretation of results and decision making is recognized (Kim et al., 2013; Laurent et al., 2020).

Among the reviewed studies, in 3 articles (Burek et al., 2018; Fieschi and Pretato, 2018; Koskela et al., 2014) only normalization is applied, while in the other 13 cases both normalization and weighting procedures are used (see Table SI-2.1 for complete references). In applying normalization alone, in two cases, reference was made to an external set referring to a European (Koskela et al., 2014) or global scale (Fieschi and Pretato, 2018). An internal reference related to the annual impact of the reference system (fresh milk delivery system) on the market of interest (US) was used by Burek et al. (2018). As underlined by the authors, normalization alone is not a measure of relative importance of the impact categories, but provides a useful element for the interpretation of their magnitude.

In 9 out of the 13 articles applying weighting, the default set of the considered LCIA method (e.g. IMPACT, 2002+, ReCiPe, EF method) was used in combination with an external (European or global) normalization references (Alvarenga et al., 2013; Changwichan et al., 2018; Deng et al., 2013; Ingrao et al., 2016; Joachimiak-Lechman et al., 2019; L. Simões et al., 2013; Ros-Dosda et al., 2019; Vural Gursel et al., 2021). In the other four studies, ad hoc normalization and weighting sets were developed and/or MCDA tools were applied. A weighting score system (from 0 to 10) based on anonymous survey, combining impact categories and inventory indicators (e.g. solid waste disposed, water consumption) normalized to Palestinian per capita value was developed by Saleh (2016). The procedure presented involves multiplying the value of the non-normalized indicators and then summing them up. Therefore, the normalization and weighting procedures were not integrated but applied separately for different purposes. A weighting system combining an inventory indicator (marine litter) and impact categories was presented also by Zanghelini et al. (2020), but in this case there was a direct integration with the normalization procedure. The authors defined a hybrid LCIA method combining ILCD recommendations, Company Materiality matrix and the coverage of areas of protection (Zanghelini et al., 2020), thus recreating a normalization set from different sources with different geographical scope (Brazilian, European, Global). The weights were recalculated from the original method to consider the additional indicators while also assessing the variability of the rankings between alternatives with respect to the weight given to marine litter. An example of relationship between LCA results and national objectives was presented by Lv et al. (2021) applying a normalization and weighting system based on the Chinese energy conservation and emission reduction targets (emission intensity per unit GDP). Finally, Gao and Wan (2022) used an MCDA tool, the relative environmental impact index (REI), which involves normalizing against the maximum value in each impact category and then directly summing the values. It can thus be defined as an internal normalization system with equal weights for all indicators.

3.2.6. Sensitivity check

The sensitivity of a LCA model describes the extent to which the variation of a LCI parameter or a choice in scope definition leads to results variation (Hauschild et al., 2018). According to ISO 14044 a sensitivity check (that includes sensitivity and uncertainty analysis) shall be considered in order to assess the reliability of the final results and conclusions (ISO, 2020). Laurent et al. expand the sensitivity check toolbox by including scenario and breakeven analysis (Laurent et al., 2020). The line between sensitivity and scenario/breakeven analysis can be subtle and they are often not differentiated, although the aims may be different. Among the reviewed articles, a multitude of investigated aspects and methodologies were identified.

The topics subjected to sensitivity analysis are summarized in Table 6. The analysis shows that LCI is more investigated than the methodological choices in scope definition. Sensitivity of LCI has been investigated for several reasons: to identify the most significant parameters of the model (e.g. through systematic perturbation (Abejón et al., 2020; Albrecht et al., 2013; Lv et al., 2021)), to assess the influence of aspects over which there is less control or availability of data (e.g., secondary data (Amasawa et al., 2021; Chitaka et al., 2020), transport distances (Anil et al., 2020; Chitaka et al., 2020; Cleary, 2013; Del Borghi et al., 2021; Ferrara and De Feo, 2020; Marcinkowski and Gralewski, 2020; Salehi et al., 2022; Santos et al., 2021; Wäger and Hischier, 2015), use (Abejón et al., 2020; Albrecht et al., 2013; Anil et al., 2020; Chitaka et al., 2020; Civancik-Uslu et al., 2019; Cleary, 2013; Kočí, 2019; Koskela et al., 2014; Lo-Iacono-ferreira et al., 2021; Marcinkowski and Gralewski, 2020; Ros-Dosda et al., 2019; Zanghelini et al., 2020)), to pinpoint product or process features that could be eco-designed (e.g., recycled content, physical properties, source of raw material (Abejón et al., 2020; Albrecht et al., 2013; Desole et al., 2022; Ferrara and De Feo, 2020; Korbelyiova et al., 2021; Lo-Iacono-ferreira et al., 2021; Marcinkowski and Gralewski, 2020; Moretti et al., 2021b; Resalati et al., 2021; Rodríguez et al., 2020; Salehi et al., 2022; Toniolo et al., 2013; Van der Harst et al., 2014)), and to evaluate future or alternative scenarios for production or use (e.g., electricity mix (Ang et al., 2021; Konstantinidis et al., 2021; Moretti et al., 2021b; Potting and van der Harst, 2015; Woods and Bakshi, 2014), end of life scenario (Abejón et al., 2020; Albrecht et al., 2013; Changwichan et al., 2018; Cherubini et al., 2019; Del Borghi et al., 2021; Desole et al., 2022; Feifel et al., 2015; Ferrara and De Feo, 2020; Fieschi and Pretato, 2018; Gao and Wan, 2022; Haylock and Rosentrater, 2018; Hottle et al., 2017; Konstantinidis et al., 2021; Leceta et al., 2013; Leejarkpai et al., 2016; Maga et al., 2019; Moretti et al., 2021b; Papong et al., 2014; Potting and van der Harst, 2015; Rodríguez et al., 2020; Saleh, 2016; Toniolo et al., 2013; Van der Harst et al., 2014; Wäger and Hischier, 2015; Woods and Bakshi, 2014; Zanghelini et al., 2020)). Of the choices in the scope definition, the management of multifunctionality is by far the aspect most investigated through sensitivity analysis (Abejón et al., 2020; Albrecht et al., 2013; Belboom and Léonard, 2016; Bos et al., 2016; Cherubini et al., 2019; Civancik-Uslu et al., 2019; Deng et al., 2013; Hansen et al., 2015; Hottle et al., 2017; Koskela et al., 2014; Piao et al., 2022; Potting and van der Harst, 2015; Rodríguez et al., 2020; Toniolo et al., 2017, 2013; Van der Harst et al., 2014; Wäger and Hischier, 2015; Zanghelini et al., 2020). However, to a very limited extent, sensitivity analysis performed with different LCIA methods. Toniolo et al. (2013) and Cleary (2013) both performed an alternative analysis by replacing ReCiPe with IMPACT 2002+ (and TRACI for Cleary). Toniolo et al. (2013) found substantial differences only in the ecotoxicity impact category, while Cleary (2013), who performed the assessment at the midpoint and endpoint level, observed only one variation in ranking between the alternatives. Some aspects suggested by the standard, such as cut-off criteria, normalization and weighting, functional units and setting of system boundary, were not investigated in the reviewed articles.

In a comparative LCA, the discussion of the results of a sensitivity analysis should also be extended to the effects on the study's conclusions i.e., ranking between alternatives. Among the articles that apply weighting procedures, only seven also discuss sensitivity analyzes (Gao and Wan, 2022; Lv et al., 2021; Zanghelini et al., 2020; Ros-Dosda et al., 2019; Changwichan et al., 2018; Saleh, 2016; Deng et al., 2013). In all these cases, the effects of sensitivity are presented and discussed for both characterized and weighted results. The way in which the results of the sensitivity analysis are considered in the conclusions of the study is discussed in the next section (RQ3).

As said above, uncertainty analysis is part of the sensitivity check.

Table 6
Percentage of studies presenting a sensitivity analysis for the different aspects according to the sector of application or material.

Sector/Material	Life cycle inventory								Scope definition	
	Property of products	Source of raw materials	Production process	Energy mix	Transports	Use scenario	EoL scenario	Secondary data	Multi functionality	LCIA methods
Packaging	20%	2%	16%	8%	10%	20%	43%	4%	22%	6%
Building&Construction	20%	0%	13%	0%	20%	13%	13%	0%	13%	0%
Primary form	0%	13%	13%	7%	7%	0%	20%	0%	33%	0%
Fossil	19%	4%	14%	6%	13%	17%	36%	3%	26%	4%
Recycled	13%	0%	7%	0%	20%	13%	20%	0%	33%	13%
Bio-based	11%	6%	14%	8%	3%	3%	36%	6%	22%	0%
Other materials	18%	3%	18%	8%	15%	30%	35%	3%	20%	5%

Heijungs (2021) presented a review of the main approaches for managing quantifiable uncertainty in comparative LCA studies, but as Schaubroeck et al. (2020) pointed out, sustainability assessments are also affected by non-quantified (or non-quantifiable) forms of uncertainty (impractical, unknown and accuracy uncertainty). As Heijungs' review points out, in most cases (quantifiable) uncertainty is reported in terms of statistics values (e.g., mean, standard deviation, percentile) and is qualitatively discussed (Heijungs, 2021). This kind of treatment of uncertainty can be a limitation in drawing conclusions from a comparative study. It is worth mentioning that the ISO 14044 standard requires that uncertainty be addressed in comparative studies (ISO, 2020), but in the reviewed articles only 22 of 78 (28%) discuss the uncertainty of the results. It is also confirmed for this sample of articles that Monte Carlo analysis is the most widely used approach (13 out of 22), as pointed out by Heijungs (2021).

Although the basic approach is Monte Carlo, this has been applied very differently. For example, Ang et al. (2021) conducted a Monte Carlo analysis considering the variability of only one foreground inventory parameter (electricity consumption), while in other cases background data were considered. A further level of differentiation concerns the subsequent analysis of simulation results. In most cases, a overlapping analysis (Günkaya and Banar, 2016; Toniolo et al., 2013) (or a qualitative discussion) of confidence intervals is presented, while the application of statistical significance analysis is limited. In fact, only Lv et al. (2021) presented the results of a null hypothesis significance test based on Monte Carlo simulation results. Another statistical approach, not based on Monte Carlo but on Cox method and Z-test, has been presented by Woods and Bakshi (2014). In eight of the remaining cases, uncertainty is dealt only with through sensitivity analyses, thus assessing the spread of the deterministic results that may arise from methodological and/or inventory choices (e.g., secondary data sources). In half of these cases (Chen et al., 2016; Cleary, 2013; Del Borghi et al., 2021; La Rosa et al., 2014; Moy et al., 2021), the evaluation is limited to inventory aspects, while in the others (Albrecht et al., 2013; Hottle et al., 2017; Potting and van der Harst, 2015; Van der Harst et al., 2014) multi-functionality management is also taken into account. In all other cases, uncertainty is not discussed or only mentioned in general terms as a limitation of the study.

3.3. RQ3: Support to decision makers

This section evaluated the conclusions of the studies, in particular their ability to support decision makers in selecting the preferred alternatives from an environmental sustainability perspective. LCA, as a multi-indicator metric, can lead to conflicting results when comparing different alternatives which show trade-offs between environmental aspects. Furthermore, the lack of transparency or comprehensiveness on methodological and interpretation choices may undermine the basis for an informed decision. The application of the criteria described in section 2.4 allowed the classification of all 79 articles according to the different situations (Table 7).

Three out of four studies failed to support the decision making process due to shortcomings in the goal and scope definition and interpretation phase (Situations 1 and 2). The main reason for classifying studies in Situation 1 is the adoption of system boundaries not in line with the goal of the study (CS-3), followed by the adoption of a limited number of impact categories (CS-4). Regarding the interpretation phase, the lack of uncertainty assessment (CS-6) and the investigation of the effects of different methodological approaches (CS-5) are the main reasons for classifying a study as not able to support the decision maker. Given the high level of subjectivity in the choice of functional unit, it was decided to consider CS-2 satisfied for all studies. An in-depth analysis of all these aspects can be found in RQ2.

The remaining 19 studies thus demonstrated the appropriateness of the methodological approach with respect to the objectives of the study and completeness in the subsequent interpretation phase. This does not

Table 7
Classification of the reviewed article in the different situations according to their ability to support decision making.

Situation	Number of articles	Support to decision making	Overall environmental sustainability declared
Situation 1	21	No	3
Situation 2	39	No	12
Situation 3	6	Yes	4
Situation 4	0	Yes (considering subjectivity of end-point and weighting)	0
Situation 5	11	No, but MCDA could be applied	0
Situation 6	2	No, but MCDA could be applied	1

automatically guarantee the possibility of supporting decision makers, as the specific results achieved must be taken into account. In fact, only 6 studies (less than 8%) present comparisons in which an alternative is better (or statistically equivalent) in all the considered mid-point indicators (Situation 3). For the latter claim, it is considered that the set of mid-point indicators chosen by the authors is reasonably comprehensive and in line with the goal of the study, as well as the main methodological assumptions have been tested. Five of these six studies (Abejón et al., 2020; Albrecht et al., 2013; Koskela et al., 2014; Toniolo et al., 2013, 2017) fall in the area of packaging (trays and crates), while Cherubini et al. (2019) analyzed a construction product. In all these cases, the complete set of mid-point indicators featured by ReCiPe and/or CML IA impact assessment methods were used. In these cases, the different methodological approaches tested didn't show variations in the rankings in the different impact categories.

Even without questioning the completeness of the sensitivity check performed, mid-point analysis alone is conclusive only in a very limited number of cases. In 11 studies, the comparison of environmental impact profiles shows trade-offs that are not managed, thus preventing direct decision support (Situation 5). In these cases, however, the environmental data made available can serve as the basis for subsequent applications of MCDA tools.

Finally, the last two studies (Deng et al., 2013; Zanghelini et al., 2020) fall into Situation 6. Although the weighting procedure is used, both studies show that the comparison is significantly affected by methodological choices. Deng et al. (2013) obtained contradictory results when comparing fossil-based and bio-based films depending on the evaluation method used (ReCiPe, IMPACT, 2002+ and Ecoindicator 99), whereas, for example, the testing of different allocation approaches, although very relevant in absolute terms, did not vary the ranking between alternatives. In the other study, Zanghelini et al. (2020) observed a variation in rankings depending on the weight assigned to the marine litter indicator.

Extending the discussion to studies that stop at mid-point indicators, the key role of the choice of methodological options to be considered in the goal and scope phase and tested in the interpretation phase thus emerges. In the event of conflicting results, it will therefore be necessary to proceed with the adoption of MCDA tools also to manage the goal and scope definition phase, but as remarked by Zanghelini et al. (2018) MCDA is rarely applied in supporting LCA methodological choices.

Considering the limitations on the conclusions that can be drawn from the analyzed studies, it is interesting to investigate the presence of statements of overall environmental sustainability in the articles. In 20 of the reviewed studies (Table 7), the conclusions state the superiority of one alternative over the others, but it must be investigated whether this superiority is supported or derives from an incorrect or incomplete interpretation of the results. For the following examples extracted from reviewed articles, the source has not been reported, as this is not intended to be a direct criticism of the authors' work. The 14% of the studies in Situation 1 and 31% of the studies in Situation 2 report claims of overall environmental sustainability within the conclusions. These claims can arise from various shortcomings in the interpretation phase. For example, one study states that 'X has lower environmental impact on the most relevant and recognized environmental issues […]. It can thus be concluded that from an environmental life cycle viewpoint, the use of X is the preferred option'. In this example, therefore, a weight was, in a non-transparent and qualitative manner, given to the different impact categories, drawing an unsupported conclusion. Taking instead as an example a study of the Situation 6, the following claim can be found: '[…] it can be concluded that X offers a better environmental performance compared to Y'. In this case, therefore, in addition to not emphasizing how the conclusion could be different with an alternative methodological approach (as demonstrated within the study), it is not pointed out that the conclusion is confined to the weighting criteria adopted (and thus to the relevant stakeholders). Therefore, the presence of these claims can cause further confusion and lead decision makers to erroneous conclusions.

4. Conclusion

Knowing the environmental performance of a product or material is a key element to making an informed decision. The first purpose of this article was to investigate through a literature review if LCA is the most used EST to compare plastic materials and the main alternatives in the packaging, construction products, and primary form sectors (RQ1). Once the analysis of the 79 selected articles showed that LCA is by far the most used EST, the methodological setup of these studies (RQ2) and their ability to support decision makers in selecting the preferable alternative from the environmental sustainability perspective (RQ3) were explored. Table 8 summarized the main lessons learned and recommendations. The analysis of the choices made with respect to the main LCA methodological aspects revealed a lack of homogeneity in the goal and scope definition and frequent gaps in the interpretation phase. Therefore, a new procedure was proposed to assess the ability of LCA studies to support decision makers, based on criteria of transparency and comprehensiveness according to ISO 14044, as well as the conclusiveness of the study. The procedure proved effective and showed that only a limited proportion of studies met the considered requirements. Furthermore, when assessing a large set of impact categories, it is common to come across trade-offs between environmental aspects, making it necessary to have weighting tools/MCDAs to manage them in a quantitative and transparent manner. Nevertheless, even with the adoption of these tools, there is no guarantee of unambiguously identifying an alternative, because different methodological choices (e.g., dealing with multifunctionality) can lead to different results. There is therefore a need to systematically test different methodological approaches and manage the results of the different combinations with MCDAs, e.g., based on the spread of the different approaches in the literature. This type of approach is rarely applied (Zanghelini et al., 2018) and represents the main possibility of future development identified in this research.

Three main limitations of this research have been identified. The analysis was limited to three application sectors (Packaging, Building&Construction and primary form). These are the most investigated in the literature and coincide with the sectors of largest plastics application worldwide. In any case, extending the analysis to other sectors might lead to different conclusions or highlight other critical issues. The second limitation is related to the exclusion of inventory aspects from the point analysis (both in RQ2 and in the criteria of RQ3). This choice was made because potentially the uncertainty connected with these aspects can be overcome with the collection of more and more punctual and precise data, while for the purely methodological aspects dealt with in the article, there will remain aspects of subjectivity that cannot be overcome. Finally, the search criteria used for the selection of sources (database, keywords, time span) may have led to the exclusion of relevant material. Comparison with review articles based on similar sources but with a different perspective did not reveal any shortcomings.

Table 8
Lessons learned and recommendations.

Lessons learned	Recommendations
The attributional approach is by far the most widely used in the reviewed studies but is not always specified within the study.	Stating the approach used is essential to allow a correct interpretation of the presented results.
Cradle-to-gate system boundaries are the majority option for studies in the Building&Construction (64%) and Primary form (60%) sectors	The exclusion of downstream stages, if not justified by particular conditions, may lead to erroneous conclusions. The exclusion of end-of-life management, for example, does not allow to appreciate differences in emission profiles (e.g., fossil or biogenic) and/or different quality of final outputs (e.g., recycled feedstock or compost). It is therefore recommended to use Cradle to Grave boundaries, testing different scenarios if necessary.
74% and 51% of the studies did not clearly state the approach used to manage, respectively, co-product and end-of-life multifunctionality	Given the level of subjectivity in the choice of multifunctionality management approach, in addition to clarifying which one is used, the main alternatives should be systematically tested and discussed.
Normalization and weighting procedures are used only in 17% of the studies	Considering that it is rare to find alternatives that perform better in all mid-point indicators, if the objective of the study was to identify the alternative that best meets the selected environmental sustainability criteria, not adopting weighting procedure could lead to qualitative and non-transparent evaluations.
Sensitivity analyses on methodological aspects and uncertainty analysis are used in less than one study out of three.	The lack of these two steps of the sensitivity check makes it impossible to know the robustness of the results obtained with respect to subjective choices and the uncertainty of the data used. The selection of methodological approaches to be tested with sensitivity analysis can be based on their diffusion in the literature.
Even with the adoption of weighting procedure, there is no guarantee of unambiguously identifying an alternative, because different methodological choices can lead to different rankings.	It is necessary to use weighting or MCDA to handle different scenarios related to goal and scope definition choices, based, for example, on literature reviews or the judgement of a panel of experts.
25% of the studies contain overall sustainability statement in their conclusions, which are not always supported by the results presented.	Authors are recommended not to overstate the conclusions of their study and reviewers are advised to discourage the use of this kind of statements.

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Declaration of competing interest

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Data availability

Data will be made available on request.

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.cesys.2023.100119.

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