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Integrating circularity indicators with life cycle assessment for plastic packaging: A refill-and-reuse system case study

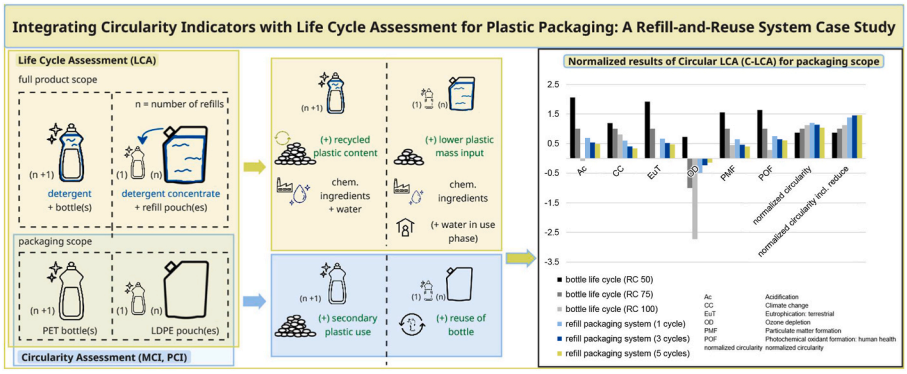
Annalisa Karin Nolte *, Milad Goudarzi, Kathrin Greiff

ANTS - Department of Anthropogenic Material Cycles, RWTH Aachen University, Wüllnerstraße 2, Aachen, 52062, Germany

HIGHLIGHTS

- Integrated Circular-LCA combining LCA with MCI and PCI for nano-level plastic packaging assessment.
- Five refills cut packaging-related impacts by ~ 60% and boost circularity (with adapted MCI considering mass reduction).
- Original MCI peaks at one reuse due to the virgin-plastic-based refill pouch – recycled content would mitigate trade-off.
- Reuse and Recycle deliver clear net gains with limited trade-offs while recover offers more constrained advantages.
- Dualscope approach reveals environmental gains through new packaging design are unremarkable next to use & production phase.

GRAPHICAL ABSTRACT



ARTICLE INFO

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Circular economy assessment of packaging systems
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Life cycle assessment of reuse and refill system
Circularity assessment using material and product circularity indicator (MCI & PCI)

ABSTRACT

This study addresses the lack of methodological consensus for nano-level Circular Economy (CE) assessments of plastic packaging. Existing approaches are reviewed to identify relevant methods. The outcome is the application of an ISO 59020 oriented, integrated Circular Life Cycle Assessment (Circular-LCA) that combines LCA with Material Circularity Indicator (MCI) and Product Circularity Indicator (PCI). Applied to a use case comparing a conventional single-use detergent bottle with a concentrate pouch in a refill-and-reuse system, indicator informativeness, system boundaries, and sensitive parameters are critically examined. Three complementary scopes are applied: packaging-only, a packaging + product transport scope to consider the packaging's volumes transported, and a full-product scope including detergent production and use. Results show mostly gains from transitioning to refill-and-reuse: even one refill mostly reduces packaging-related impacts by on average 35% and substantially increases circularity. The original MCI peaks at one reuse, while its adapted version integrating reduced mass peaks at five reuses and then declines due to the pouch's dominant virgin feedstock. Introducing recyclate content for refill packaging would mitigate this trade-off. Reduce, Reuse, and Recycle emerge as primary levers with net benefits and limited trade-offs, while Recover offers supplementary advantages. In the full-product scope, packaging improvements are overshadowed by impacts from detergent production and use, underscoring the need for holistic evaluation. The study demonstrates the value of combining life-cycle methods

* Corresponding author. Department of Anthropogenic Material Cycles, RWTH Aachen University, Wüllnerstr. 2, 52062, Aachen, Germany. www.ants.rwth-aachen.de
E-mail address: annalisa.nolte@ants.rwth-aachen.de (A.K. Nolte).

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with circularity indicators, identifies sensitive parameters, and thereby offers guidance for practitioners navigating the implementation of Packaging & Packaging Waste Regulation, ISO 59000, and Digital Product Passports.

1. Introduction

The concept of Circular Economy (CE) emerged to foster a transformation to an economic model maintaining the value of materials as long as possible while minimizing generation of waste (EMF, 2013). Plastics illustrate the challenge: 400 million tons are produced worldwide in 2022, while only 35.5 million tons are from mechanically recycled feedstock. Packaging is the main application field of plastic in Europe (PlasticsEurope, 2023) – predominantly a linear single-use product with a short lifespan (OECD, 2022; Thomassen et al., 2024). This sector faces substantial transformation imposed by the European Packaging and Packaging Waste Regulation (PPWR), setting out ambitious targets, e.g., to make all packaging on the EU market recyclable in an economically viable way by 2030 (European Commission, 2025b). Therefore, it appears crucial for the transition towards CE to rethink and optimize plastic packaging product systems and life cycles.

At the nano (product) level, CE is operationalized through product systems that embed R-strategies (e.g., Reduce, Repair, Recycle) deriving from the waste hierarchy (Circular Economy Act, 2012; Potting et al., 2017; Saidani et al., 2017). For packaging, this ideally means refusing packaging. Where this is not feasible, reducing packaging material or "design for reuse", should be preferred over continuous upgrading of recycling facilities (Network for circular plastic packaging, 2023). Companies and governmental organizations are increasingly adopting strategies rooted in the CE concept to reduce global plastic production (Fliaster and Bähr, 2025). Yet R-strategies are often equated with sustainability without robust evidence (Papo and Corona, 2022). Several studies reveal that R-strategies may entail unintended environmental trade-offs when evaluated across the full life cycle of a product (Geisendorf and Pietrulla, 2018; Rigamonti and Mancini, 2021). Hence, their effects need to be holistically evaluated using scientific methodology (Papo and Corona, 2022).

A variety of methods exists to assess CE at product-level. However, there is no methodological consensus on nano-level CE assessment for plastic packaging (Matos et al., 2024). Existing studies often either apply LCA or circularity indicators, focus on isolated R-strategies and employ system boundaries without thoroughly discussing their implications for their results' meaningfulness (Bher and Auras, 2024; Cottafava et al., 2021; Ferrara and Feo, 2020). Scarcely, previous studies analyze trade-offs between circularity and environmental impacts for packaging systems bundling multiple R-strategies while differentiating between packaging and full-product-scope results in view of PPWR, ISO 59020, and emerging Digital Product Passport (DPP) requirements (Bher and Auras, 2024; Thomassen et al., 2024).

This gap is addressed by applying parts of an existing Circular Life Cycle Assessment (C-LCA) framework on a recent plastic packaging use case, following ISO 59020 principles but combining LCA with the Material Circularity Indicator (MCI) and Product Circularity Indicator (PCI) (Bracquené et al., 2020; EMF and Granta Design, 2019 ; Luthin et al., 2023). A conventional single-use detergent bottle is compared with a concentrate pouch in a refill-and-reuse system to operationalize and critically examine the framework. The overarching aim of this study is to answer these research questions:

- Is existing methodology for CE assessment sufficient to capture all major differences between the compared packaging systems?
- Which of the packaging systems compared achieves higher performance in terms of transition towards CE?
- Are there any trade-offs between circularity and environmental impact savings? What are the key parameters, and how can they be analyzed efficiently?

First, relevant previous CE assessment approaches are identified and reviewed. The selected methodology is then systematically applied and critically reflected to disclose strengths and weaknesses showcased on the examined packaging designs. Lastly, practical implications can be drawn from the results, especially pertinent to stakeholders in the liquid household product packaging sector. Beyond that, implications for related standards (PPWR, ISO 59020, and DPP) (Ferrante et al., 2025) are discussed.

2. Related work

2.1. Methodological development of CE assessment

Despite the increasing number of frameworks and methods for CE assessment, development, and consistent application of robust CE indicators remains an area of active research and debate (Ferrante et al., 2025; Luthin et al., 2023). Selection of suitable indicators depends fundamentally on specific application and objective of assessment, i.e., research questions to be answered (Ahmed et al., 2022; Pauliuk, 2018; Saidani et al., 2019). Consequently, it is important to conduct case studies to test applicability of new assessment methods for different sectors and to optimize them further (Ferrante et al., 2025).

2.2. CE indicators and assessment frameworks on nano (product) level

Instead of treating CE, circularity, and environmental sustainability as separate goals, integration enables identification of potential trade-offs or synergies between closing material cycles and minimizing environmental impacts, and can be referred to as Circular-LCA (Luthin et al., 2023). Luthin et al. (2023) suggest a framework for sector-unspecific CE assessment of products (nano level) and recommend using specific CE indicators complementarily to LCA, life cycle costing (LCC), and social LCA (S-LCA). This is particularly important because an increasing CE does not automatically guarantee lower environmental impacts (Anastasiades et al., 2020; Hagedorn et al., 2024; Rigamonti and Mancini, 2021; Vadoudi et al., 2022). Although sustainability performance is integrated into some CE indicators, it is usually only superficial and insufficient (Oliveira et al., 2021). The other way around, Cilleruelo Palomero et al. (2024) introduce to integrate circularity indicators into LCA by incorporating them into *openLCA* and adapting the *ecoinvent* database to trace circularity throughout whole value chains.

In 2024, ISO 59020 was published, a sector-unspecific standardized framework for CE assessment of organizations, regions, or products, harmonizing the most prevalent circularity metrics, such as the Material Circularity Indicator (MCI) by (ISO, 2024; Munonye, 2025;EMF and Granta Design, 2019). The standard provides general principles for circularity measurement, guidance on system boundary setting, data acquisition, indicator selection and their aggregation. Though, while being methodologically straightforward, ISO 59020 lacks specification on meaningful indicator aggregation (ISO, 2024). The avoided dimensionality constrains their interpretive depth and comparative power (Toivanen, 2024).

Despite availability of a multitude of product-level CE indicators, only a few are compatible with the unique sectoral characteristics of consumer goods and plastic packaging. Matos et al. (2024) identified relevant circularity indicators for plastic packaging products and their

relation to CE principles and design tools. Next to MCI, they name the Value-based Resource Efficiency (VRE) by Di Maio et al. (2017), and Circular Economy Indicator Prototype (CEIP) by Cayzer et al. (2017) as most relevant indicators aligned with CE principles and "design-for-X" strategies. A review of relevant case studies suggests that MCI is arguably the most widely applied indicator among researchers, possibly due to its comparably long existence (Table 6 in Annex B). This observation is supported by several references (Bracquené et al., 2020; Lonca et al., 2020; Rigamonti and Mancini, 2021; Sazdovski et al., 2024). In line with this, Rasines et al. (2024) identify MCI as the most suitable material-based indicator for packaging applications. Although there are notable limitations when applying the MCI, such as oversimplified modeling of End of Life (EoL) processes and *50:50* waste-allocation rule (Figueirinhas et al., 2026). The Product Circularity Indicator (PCI) can be seen as an advancement aiming to address several limitations of the original MCI while reinforcing its strengths. One major enhancement is the inclusion of quality, which is essential in CE contexts (Bayram et al., 2024; Bracquené et al., 2020). Nevertheless, both indicators are restricted in the amount of R-strategies incorporated (Sazdovski et al., 2024). In addition, their aggregated formulation as well as specific allocation methods in the methodological approach can influence the results and reduce informative value, hindering insights on the effects of single R-strategies (Hagedorn et al., 2024). If aggregation is applied, it is therefore crucial to present all input parameters transparently to allow meaningful interpretation and discussion of results (ISO, 2024).

2.3. Previous CE assessments of plastic packaging

There are multiple comparative case studies on different plastic packaging designs, usually focusing on conducting LCAs to obtain information on their environmental performance (Ferrara and Feo, 2020; Toniolo et al., 2013; Yadav et al., 2024). Cottafava et al. (2021) provide critical insights into the trade-offs across impact categories, but do not integrate any dedicated circularity indicators, leaving their CE assessment incomplete. Thomassen et al. (2024) supplemented their LCA with MCI and material flow analysis, thus offering a more holistic view of circularity. Their reusable system significantly outperforms the single-use option in terms of circularity. Nonetheless, environmental impacts per functional unit (FU) remain considerably higher, largely due to heavier packaging design and added burdens of reverse logistics. Papo and Corona (2022) integrate all three pillars of sustainability – environmental, economic, and social – into their packaging system assessment, but neglect the inclusion of circularity indicators.

Only few LCA studies particularly incorporate multiple loops and circularity indicators while considering whole product systems and packaging's desired functionalities or performances (e.g., containment, protection, communication, and convenience to users) highly dependent on application cases (Bertolini et al., 2016; Jagoda et al., 2021). Another limitation of most product-based LCA studies is that they often simplify recycling processes and waste management practices, creating a gap between real-world and modelled EoL scenarios (Pellengahr et al., 2023).

3. Materials & methodology

3.1. Goal & scope

This work aims to contribute to methodological discussions by conducting and critically reflecting on a product-level CE assessment (focus on ecological and circularity aspects) oriented towards principles of ISO 59020 (2024) for a recent use case. This case study involves two specific product "systems in focus". The first product (Fig. 1 a) represents a conventional manual dishwashing liquid (conv) which is delivered in a transparent plastic bottle made from 100% recycled PET (rPET), according to the producer, with a PP label and HDPE cap. Divergently, to increase generalizability, this study makes a more conservative assumption of 75% rPET content as a reference scenario (for further explanation, see section 3.4). In contrast, the second product (Fig. 1 b) is a concentrated liquid packaged (conc) in a flexible virgin LDPE pouch (Henkel, 2024a, 2024b).



Fig. 1. Images of the examined products a) bottle of conventional detergent, and b) refill pouch of detergent concentrate.

The concentrate is meant to refill a regular bottle, creating a detergent mix like the original, same formula and function (see Fig. 2). The main contrast lies in packaging and when the detergent is diluted. Considering all resource inputs and outputs (ignoring time), the only difference left is the packaging design. Thus, for circularity assessment, only the two packaging designs are evaluated.

To enable a holistic LCA, it is essential to consider packaging's functions linked to its content. Hence, three complementary scopes for the LCA are employed. Firstly, the **packaging-only scope** narrows the focus solely on packaging-related processes by excluding two almost identical environmental-impact-intensive phases (detergent production and use phase). Secondly, we add the volume difference transported (as the major difference for LCA) – **packaging + product transport scope**. Thirdly, the **full-product scope** includes all phases of conventional detergent and concentrate. These distinctions allow targeted insights. Fig. 3 depicts the cradle-to-cradle system examined (embracing both product and packaging systems). Secondary and tertiary packaging for logistics is not included due to lack of data. However, they are assumed to be similar for both systems, and their environmental impacts are expected to be relatively low (Kang et al., 2013; Molina-Besch et al., 2019).

The study applies system expansion in combination with an avoided burden approach in which bottle production (partially using recycled material as input) and potential subsequent recycling (producing recycled material as output, substituting primary materials) is included. This avoids allocation as recommended by the ISO 14040-44 standards (ISO, 2006). For a fair comparison, the **FU** is defined as "manual cleaning of 180 place settings (PS) in a German household". A place setting is a complete set of crockery, cutlery, and glasses for one person at a meal (Golsteijn et al., 2015; Stamminger et al., 2007). To clean 180 PS, the **reference flow** of either 450 ml conventional detergent or 120 ml detergent concentrate (diluted with 330 ml water) is needed, which equals the bottle's and pouch's content volumes. Table 1 outlines enclosed circular strategies (Bocken et al., 2016) for clarification.

This study implements a CE assessment to evaluate the effectiveness of the employed R-strategies. To do so, a comparative LCA in accordance with ISO 14040/44 is conducted using the software OpenLCA, complemented with relevant CE metrics.

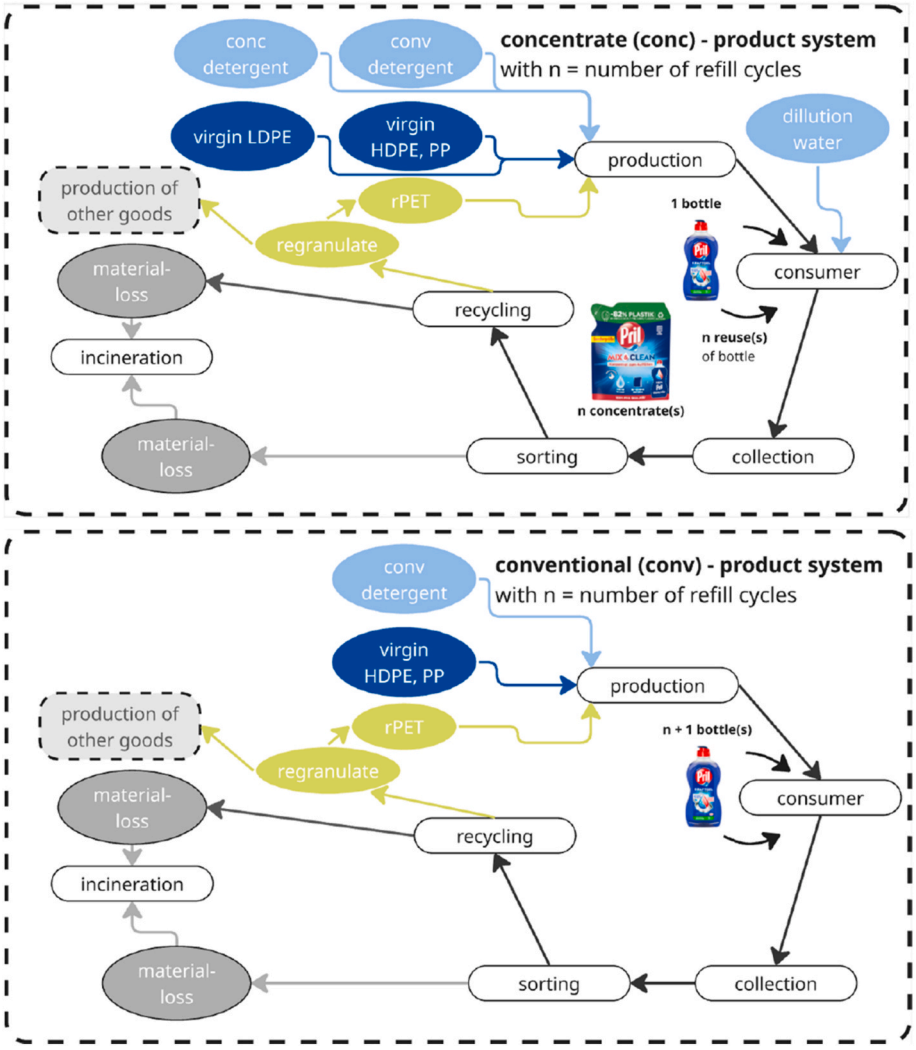


Fig. 2. Concentrate and conventional product system.

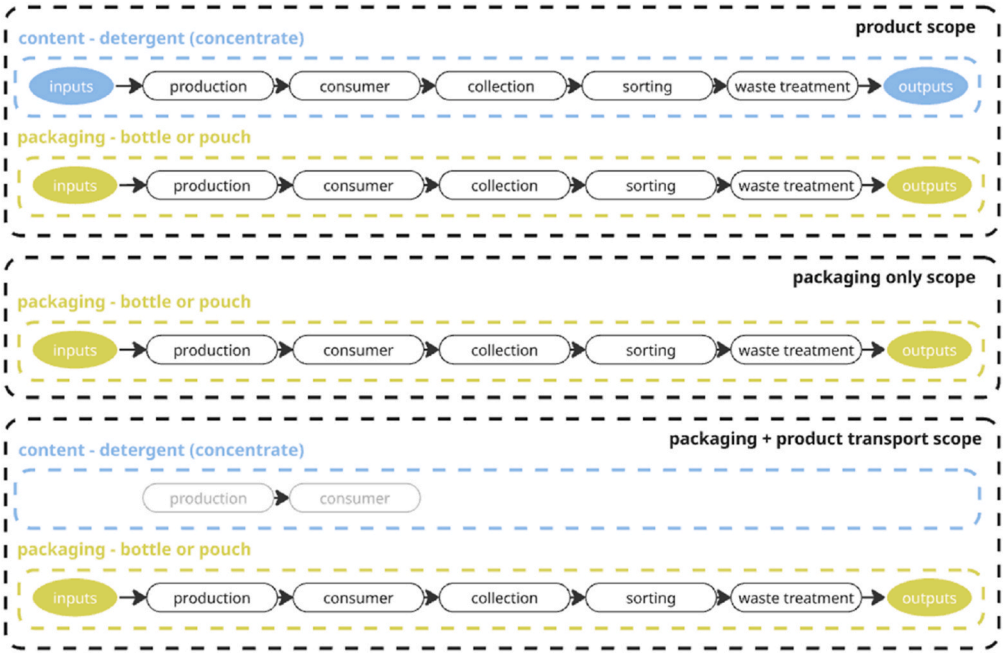


Fig. 3. Simplified visualization of the three applied scopes and their system boundaries examined in LCA.

Table 1
Case study relevant R-concepts and strategies based on Potting et al. (2017) and Bocken et al. (2016).

R-Concept	Circular Strategy	Conv	Conc	Contextual fit
R1-Rethink	Narrowing and Slowing		X	Change in product system design and delivery; more intensive use of container; user engagement to self-mix
R2-Reduce	Narrowing		X	Less material use in packaging reduces plastic consumption; concentrated detergent liquid minimizes transportation emissions through weight reduction
R3-Reuse	Slowing		X	Extending the lifespan of the bottle during the use phase by enabling content refill
R8-Recycle	Closing	X	X	Use of recycled content in bottles and high recyclability of both bottle and pouch materials
R9-Recover	Closing	X	X	Use of residuals from recycling processes in energy recover plants

3.2. Life cycle inventory (LCI)

Data needed for this analysis is sourced from various channels: scientific literature, technical reports, databases like ecoinvent 3.10, and some reasoned assumptions. This work focuses on EoL modeling of packaging designs described in detail in the following sections. Remaining LCI information can be retrieved from Annex C.

3.2.1. Light-weight packaging (LWP) waste sorting (part I)

LWP waste comprises a collection of mixed packaging materials, including e.g. metals, composites, and the largest post-consumer plastic waste flow in Germany (Conversio, 2024b; Wagner and Günther, 2018). The efficiency of waste sorting is variable and case-specific due to general heterogeneity of packaging waste streams and applied sorting technology (Jakobs and Kroell, 2024; Kranert, 2017). Probabilities for efficient sorting of the two packaging designs in Germany are derived from literature as 81% for PET bottles and 50% for flexible small-sized PE packaging (Central Agency Packaging Register and Federal Environment Agency, 2026) with 19 - 50% material losses directed to municipal waste incineration plants. Operational waste sorting energy expenses are derived from a combination of previous literature (Bilitewski, 2013; Knappe et al., 2021), machine specification sheets, and expert interviews based on the premise of a “prevailingly average German light-weight packaging sorting facility”.

3.2.2. Plastic waste treatment

After sorting, there are three pathways for pre-sorted and baled LWPW in Germany: energy recovery, mechanical recycling, or less common chemical recycling (Conversio, 2024b). This study considers the two main waste treatments (incineration and mechanical recycling). The modeling of expenses for plastic recycling is based on relevant literature (Bilitewski, 2013; Christiani and Beckamp, 2020; Dorn and Fabian, 2024), machine specification sheets, and expert interviews. Average yields for plastic recycling plants (PET 61%, HDPE 75%, and films 70%) are derived from Bulach et al. (2022).

3.2.3. Material and energy recovery

Recyclability of both packaging designs is assessed by predicting their possible fates in average German sorting and recycling facilities. It is assumed that cap and label remain attached to the bottle until separation at PET recycling plants and are fed into respective recycling processes as recoverable PO-flakes. The PET bottle is assumed to be recyclable into prime quality (bottle-grade). The LDPE pouch is assumed to be recyclable into PE granulates (Bulach et al., 2022; Christiani and Beckamp, 2020; Kusch et al., 2021).

Plastic type-specific default substitution factors are retrieved from previous literature, being 0.95 for PO-flakes, 0.44 for rLDPE granulates, and 1 for r-PET to give credit for recyclate production following the avoided burden approach (Bulach et al., 2022; Christiani and Beckamp, 2020; Kusch et al., 2021; Schulte et al., 2023b). Section C.7 in Annex C provides more details on how double counting is avoided.

Any material losses are assumed to be energetically recovered in a municipal waste incineration plant, as common waste residue treatment in Germany (Ragaert et al., 2017). The resulting net energy production of 3.93 MJ/kg electric energy and 7.67 MJ/kg thermal energy is credited and retrieved from an ecoinvent 3.10 process for mixed plastic waste treatment.

3.3. Environmental impact assessment

The environmental impact of the product systems is calculated using the Environmental Footprint 3.1 (EF 3.1) characterization model. The EF 3.1 is adopted because it includes the latest IPCC characterization factors (Jungbluth, 2025). This study generally considers all EF 3.1 impact categories, though for climate change (CC) and freshwater ecotoxicity, only aggregated impacts are considered. In addition, the impact category of non-renewable energy resources is replaced by the cumulative energy demand (CED) to consider both renewable and non-renewable energy demand (Frischknecht et al., 2015). Similarly, the impact category metal/mineral material resources is replaced by the total raw material input (RMI) to measure raw material intensity of products (Mostert and Bringezu, 2019). Deeper discussion of implications of the chosen impact category set can be found in Annex F. All LCIA results are displayed as percentages relative to the reference scenario of the conventional full-product-system or the bottle life cycle (RC of 75%).

3.4. Sensitivity analysis

The sensitivity analysis focuses on key subjects: number of reuse cycles, RC, the efficiency of EoL processes, primary material substitution coefficients, transports, and consumer behavior.

3.4.1. Consumer behavior

The producer states the bottle is theoretically infinitely reusable. Besides the default scenario of 1 reuse cycle ($n = 1$), the scenarios of 3 and 5 reuse cycles are explored. The consumer behavior's impact is further tested by varying the dilution and resource consumption behavior in terms of water and energy usage during manual dishwashing, assuming a variation of $\pm 20\%$ for environmentally conscious (eco) to inefficient use patterns, as depicted in Table 2.

3.4.2. RC of PET bottle

The RC of the examined PET bottle reaches 100% according to the producer. However, multiple references conclude that 75% represents the optimal RC threshold for PET (beverage bottles) that can be maintained indefinitely without significant loss of quality (PETCYCLE, 2020; Pinter et al., 2021). Ideal RC for non-beverage PET bottles might be even lower due to higher contamination potential and material losses. Sensitivity analysis includes RC from 50% to 100%, with 75% as default for the PET bottle.

3.4.3. Efficiency of EoL processes

In addition to generic material losses (see section 3.2.1), a worst-case material recovery scenario is investigated by employing a packaging-specific sorting efficiency of 0 for both designs. In this case, 100% of the packaging is energetically recovered after being processed through

Table 2
Resource use per FU of different user patterns.

User pattern	Water usage [l/FU]	Energy usage [kWh/FU]	Volume of detergent used [ml/FU]
eco	540	7.82	360
average	675	9.77	450
inefficient	810	11.72	540

Table 3
Recycling yields and substitution factors for different types of plastics retrieved from Bulach et al. (2022)[1], Meys et al. (2020)[(2)], Kusch et al. (2021)[(3)], Gala et al. (2015)[(4)], Schulte et al. (2023b)[(5)] complemented by assumption.[(6)].

Variable	PET	PP	HDPE	LDPE
Recycling yield (state of the art)	63 % [(1)]	74 % [(1)]	75 % [(1)]	70 % [(1)]
Recycling yield (conservative)	61 % [(1)]	68 % [(1)]	70 % [(1)]	65 % [(1)]
Substitution factor (best case)	1.00 [(2)]	0.95 [(1)]	0.90 [(3)]	0.90 [(3)]
Substitution factor (default/ average)	0.95 [(1)]	0.90 [(3)]	0.75 [(4)]	0.70 [(2)]
Substitution factor (worst-case)	0.90 [(6)]	0.70 [(2)]	0.70 [(2)]	0.44 [(5)]

non-plastic-specific sorting. Furthermore, effects of conservative assumptions for recycling yields and substitution factors presented in Table 3 are explored.

3.5. Uncertainty assessment

To account for considerable uncertainty associated with LCI data and modelling assumptions, results are interpreted in light of their robustness as required by ISO 14044 and ILCD Handbook (ILCD, 2010; ISO, 2006). Empirical studies indicate that LCA data and results frequently exhibit uncertainties of 20% or more (Frischknecht et al., 2005; Huijbregts, 1998; Weidema and Wesnæs, 1996). In line with this, we adopt a pragmatic relevance threshold of 20%. Relative differences below this value are not interpreted as qualitatively meaningful within this study.

3.6. Circularity indicator selection

Based on the presented literature review, it is concluded to apply both MCI and PCI to the investigated use case: MCI by (EMF and Granta Design, 2019) is the most prevalent circularity metric (Rasines et al., 2024; Rigamonti and Mancini, 2021). It appears important to deliver MCI results for high comparability to previous studies. In addition, the MCI is recommended for product circularity assessments by Luthin et al. (2023) and for plastic packaging by Matos et al. (2024) (more details on the selection process in Annex D).

Furthermore, Luthin et al. (2023) identify the resource inflow and outflow indicators from ISO 59020 as minimum requirement for circularity assessment. However, our analysis reveals these indicators largely overlap with input parameters of MCI and do not provide additional insights for our case study. Therefore, their results are only included and discussed in Annex E. The PCI as MCI's upgrade is more robust and recommended for sensitivity analysis, especially when clearly differentiating between sorting and recycling, as intended for this use case (Bracquené et al., 2020; Luthin et al., 2023). Both indicator compositions are briefly described to clarify interpretation, while more details can be found in Annex D.

3.6.1. Material circularity indicator (MCI)

The (product) material circularity indicator MCI_p ranges from 0.1 for fully linear products to 1 for fully circular products. It is a quantitative indicator measuring the degree of product circularity by considering use of virgin, recycled, and reused materials or components, EoL recycling, product life span, and product utility. The total MCI is derived by a weighted average approach based on the components' masses (m_i) and their contribution to the FU (U_i), depending on the number of reuse cycles (n), following formulas 3.1-3.3 (EMF and Granta Design, 2019):

$$MCI_{tot} = MCI_i * m_i * U_i \tag{3.1}$$

with $U_i = n - 1$ for the pouch and (3.2)

$$U_i = \frac{1}{n} \text{ for any bottle components} \tag{3.3}$$

Additionally, to account for the reduced mass needed per FU, an adjusted MCI_{red} is developed based on the following approach: integrating the reduced mass share (C_{red}) into the MCI formula as an optional extension, comparing the reference packaging mass (M_{ref}) to the refill scenario specific reduced mass (M_{red}).

$$C_{red} = 1 - \frac{M_{red}}{M_{ref}} \tag{3.4}$$

$$MCI_{red.i} = 1 - LFI \times F(X) \times (1 - C_{red}) \tag{3.5}$$

3.6.2. Product Circularity Indicator (PCI)

PCI's interpretation is analogous to MCI. In contrast to MCI, the recycling efficiency is two-fold via differentiation between efficiency of material separation and recycled feedstock production. This is highly relevant to enable a more precise analysis of potential material losses. In the context of net exchange of recycled material, a quality factor of material entering and leaving the product system is introduced (Bracquené et al., 2020).

Generally, the same input data is used for PCI and MCI. Though for PCI, the efficiency of material separation is included (Central Agency Packaging Register and Federal Environment Agency, 2026). For any production efficiencies, 100% are assumed. For effective analysis, LCA and circularity assessment examine the same scenarios regarding RC and number of reuses.

3.7. Integrated analysis of LCA and CE indicator results

This case study strives to explore key parameters and how they can be analyzed efficiently. There is a great variety of Multi-Criteria Decision-Making (MCDM) methodologies to facilitate decision-making, aiming to simplify the complexity of CE assessment (Dos Santos Gonçalves and Campos, 2022; Niero and Kalbar, 2019; Romero-Perdomo and González-Curbelo, 2023). However, the reduced complexity and informativeness of results (Tzeng and Huang, 2011) is not beneficial for this study's purpose. Thus, the integrated analysis of LCA and CE indicators is implemented by placing the identified most relevant impact categories and circularity indicator side-by-side to detect potential target conflicts and draw conclusions. Next to CC, the most relevant impacts are identified by filtering with an empirical threshold of 40% for the standard deviation of relativized results. Although no explicit weighting is performed, any selection and presentation of a limited set of impact categories entails value choices and thus implies indirect weighting (ISO, 2021). To overcome unequal presentation, MCI results are relativized to the reference scenario.

4. Results of CE assessment

4.1. LCA results of the two packaging designs and product systems

As the main research focus lies on plastic packaging assessment, the results firstly showcase impacts of the examined packaging systems. This is followed by presenting results for the packaging + product transport scope and lastly full-product-scope, including packaging and its content needed to fulfill the FU of manual dishwashing of 180 PS. This helps to put results into perspective, allowing holistic assessment. Relative differences below a pragmatic relevance threshold of 20% are not

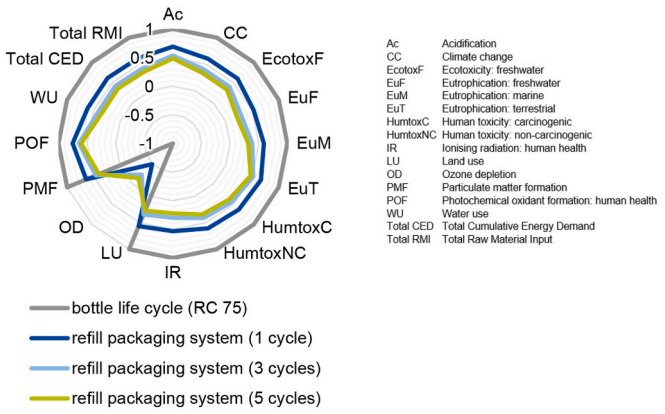


Fig. 4. Relativized LCIA results for conventional bottle (reference) and refill packaging system, including effect of multiple refill cycles.

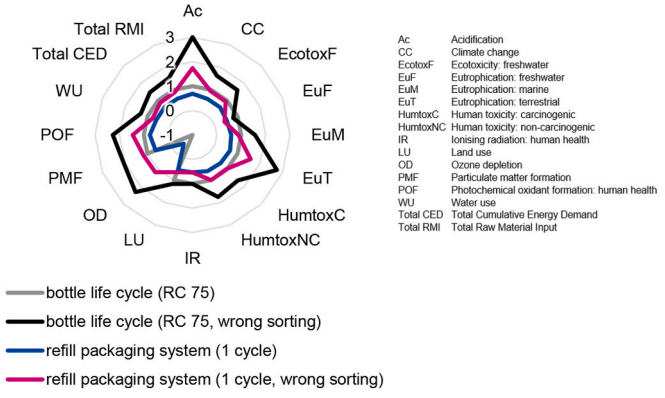


Fig. 5. Relativized LCIA results for the conventional bottle and the refill packaging system, including effect of wrong sorting leading to energetic recovery only.

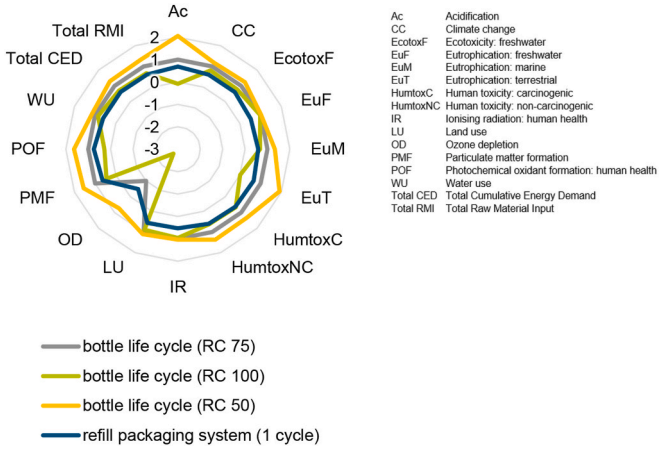


Fig. 6. Relativized LCIA results for conventional bottle and refill packaging system, including different RC shares.

interpreted as qualitatively meaningful (see section 3.5).

4.1.1. LCIA results for the different packaging designs

Studying relative (to reference *bottle life cycle (RC 75)*) environmental impacts of the two investigated packaging systems, it is evident the refill pouch is mostly more eco-friendly (see Fig. 4). The higher the number of refills before disposal, the higher the share of the LDPE pouch contributing to the FU, lowering environmental impacts. Exception is ozone depletion (OD), which is sensitive to terephthalic acid needed for PET production. Since more virgin PET can be substituted at the EoL than virgin PET is used for bottle production, benefits arise.

The increase from one to three refill cycles has a larger implication than from three to five, which is due to the more leveraged share of the pouch to the FU. Refill packaging attains lower impacts than the wrongly sorted bottle (see Fig. 5). Though both wrong sorting scenarios appear partly (especially for freshwater eutrophication) more favorable in comparison to correct sorting, which is due to their energetic recovery achieving larger benefits than their material recovery in certain categories.

Missing material recovery affects multiple impacts, e.g., acidification (Ac), terrestrial eutrophication (EuT), OD, and photochemical oxidant formulation (POF). In contrast, Fig. 6 illustrates consequences of varying the PET bottle's RC.

The differences between RCs do not all reach the significance threshold of 20% in comparison to the reference scenario. It is evident that RC is mostly beneficial and especially sensitive to Ac, OD, and EuT. For some impact categories, PET bottle (RC 100%) accomplishes lower impacts than refill packaging system (1 cycle). In addition, sensitivity of recycling yields and substitution factors is analyzed in Annex G showing differences not exceeding the significance threshold of 20%.

Environmental impact contributions of refill packaging system (1 cycle) are exemplarily outlined in Fig. 7 as they comprise impacts of both packaging designs with equal share to the FU. Negative impacts, such as for bottle recycling, indicate the avoided burden for generated recyclate exceeds expenses caused by material processing.

Waste incineration is characterized by mostly small influence on overall results except for CC and freshwater ecotoxicity. Impacts of bottle manufacturing are highest, while transport remains largely below 5% contribution. Although significant contributions to transport and waste collection emerge for particulate matter formation (PMF). To test whether the small relevance of transport in LCIA results persists, the scope is extended, considering reduced transported weight.

4.1.2. LCIA results for packaging + product transport scope

As one of the major product and packaging differences is the reduced volume of packaging, Fig. 8 demonstrates relativized LCIA results for both packaging systems, including their product transport. In principle, the same trend can be perceived, as refill system achieves lower impacts than conventional bottle.

The exception of OD is erased in comparison to Fig. 4. If transport distances increase, benefits for the refill packaging system would further develop. After learning the new refill packaging system is mostly more environmentally friendly, the analysis's scope is widened to entire product systems.

4.1.3. LCIA results for the two product systems

Percentual LCIA results (reference is conventional detergent) for both product systems are summarized in Fig. 9 and indicate there is only little difference left (except for OD). Even in the case of increased numbers of bottle reusages (three and five times), the new “expectedly more sustainable” product system shows only insignificant environmental benefits. Consumer behavior related to cleaning water usage has much stronger environmental influence than bottle reuse.

To explain this phenomenon, Fig. 10 is incorporated to visualize contributions exemplarily for the conventional product system. The “bottle life cycle” encompasses impacts of all bottle life cycle phases, including EoL. Main contributors for all impacts (except for OD) are use phase and detergent production. These two phases are almost identical for both systems, which is why their contributions resemble.

Based on scenario analysis and contribution results, relevant impact categories emerge, showing strong sensitivity to factors such as number of refill cycles, wrong sorting, and RC. These are chosen for final consolidation of results in section 4.3.

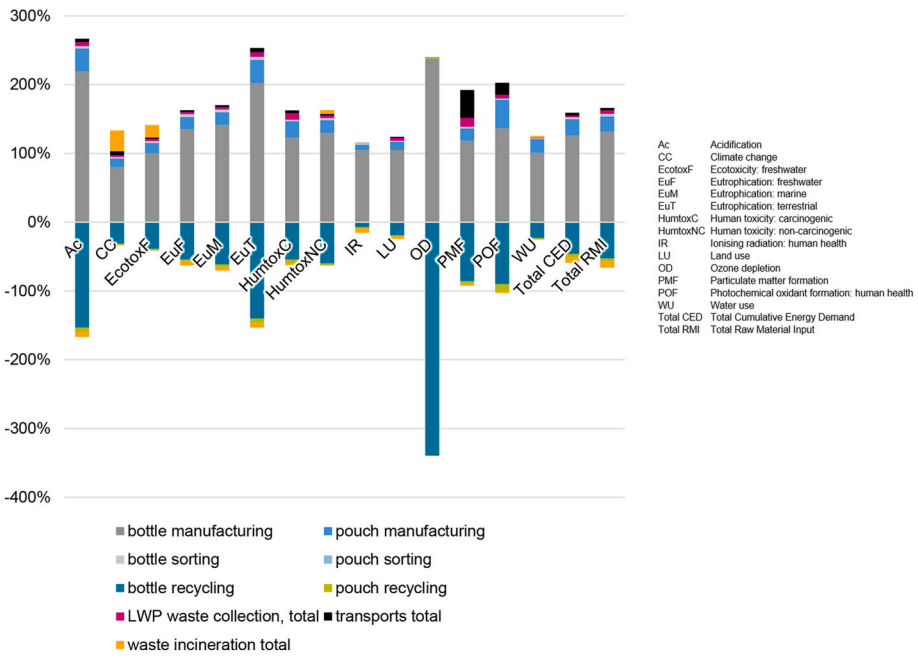


Fig. 7. Contributions to environmental impacts of refill packaging system (1 cycle).

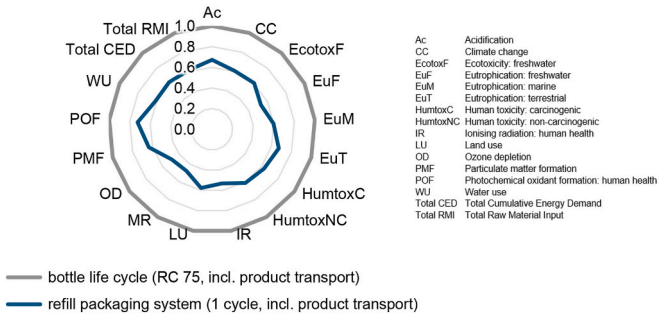


Fig. 8. Relativized LCIA results for conventional bottle and refill packaging system, including different product contents transported.

4.2. Circularity assessment of the two different packaging designs

As circularity assessment, MCI and PCI are applied for both packaging designs (for rationale, see section 3.6). The following input data is prepared to calculate MCI of both products exemplarily for one reuse cycle (see Table 4). Complementarily, results for core circularity indicators from ISO 59020 can be retrieved from Annex E as their informativeness does not differ from Table 4 but presents data in a more aggregated way.

To calculate utility, the packaging's service life is used (L), while intensity of use refers to product content (U). Each unit of (conv) and (conc) theoretically provides one FU "manual dishwashing of 180 place settings". Life span is unitless as it describes the lifetime of packaging in relation to the FU. If the number of reuse cycles varies, utility input data (L, U) and relevant mass shares (M) are adjusted. This way, it is considered how often the bottle is reused (L) and to which shares each product contributes to the FU (U, M). Table 4 shows the conventional detergent bottle's lifetime is doubled. Meanwhile, the pouch is only used for half of the FU and during half of the usage phase. These input parameters are used to compute results shown in Table 5.

A direct comparison between both packaging designs shows the conventional bottle performs better compared to the stand-up pouch. For variation of number of refills and RC, input parameters are provided in supplementary material, so that the following analysis focuses on results (see Fig. 11).

Evidently, the highest MCI score is achieved with one reuse cycle. While higher numbers of reusages reduce circularity following the original MCI, larger RC imply higher circularity. The lower level of PCI results compared to MCI results derives from its formulation, e.g., factor removal (see Annex D). However, since neither MCI nor PCI considers the reduced mass per FU, MCI_{red} as an MCI extension is employed, with results presented in Fig. 12.

The results differ significantly, as the ideal number of refills shifts to five reuses. Beyond this point, the effects of mass reduction and reuse are outweighed by the absolute mass of virgin materials exceeding recycled mass used.

4.3. Consolidation of LCA vs. CE indicators – linkages and trade-offs

MCI and PCI complement LCA by bridging emission-based and resource-efficiency perspectives, enabling a more robust and multidimensional assessment (Rigamonti and Mancini, 2021; Sazdovski et al., 2024). The most important results from previous sections are summarized in Fig. 13. Next to CC, only impact categories that emerged as most sensitive are presented to maximize clarity of analysis, sacrificing previous holisticness to a certain extent.

As circularity indicator (MCI), the relativized result is presented to avoid unequal interpretation (see 3.7). Regarding RC, there are no trade-offs between circularity and environmental impacts. The higher the RC, the stronger is circularity and environmental performance. Concerning number of refill cycles, a conflict is perceived between circularity and environmental impacts. While enhanced reuse cycles reduce environmental burden, circularity decreases. Although it must be noted that circularity decrease occurs at a much higher number of refill cycles, if the reduced mass per FU is considered. Yet Fig. 12 answers one research question: for most examined indicators (except OD), the refill packaging system achieves results superior to the conventional bottle (RC 75%).

5. Discussion & analysis

5.1. Analysis of LCA results

LCA results indicate the refill packaging system outperforms the

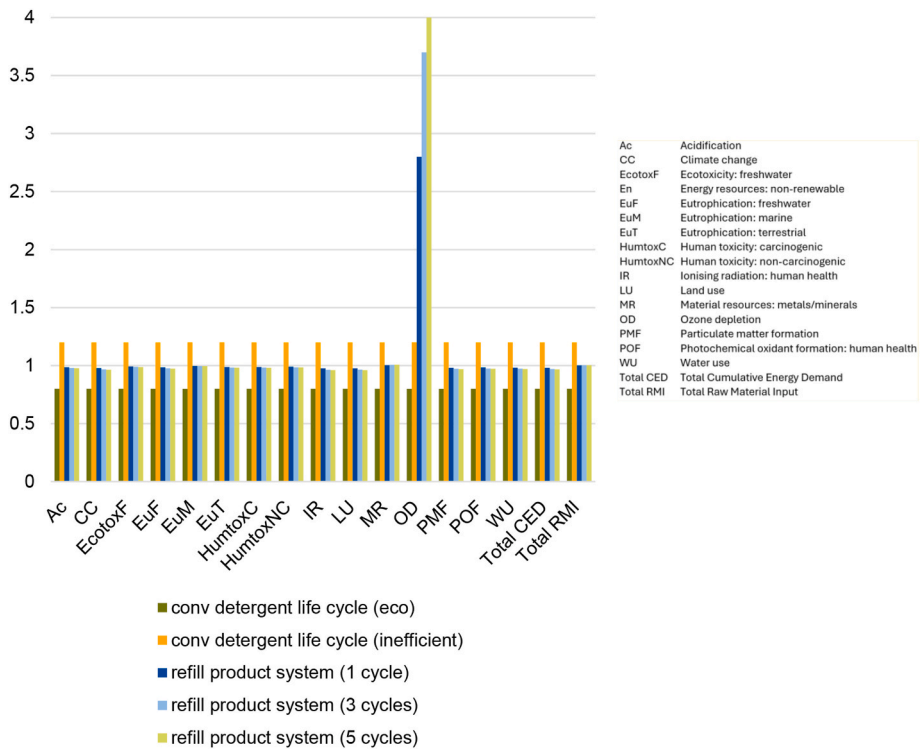


Fig. 9. Relativized LCIA results for conventional and refill product systems, including one, three, and five refill cycles.

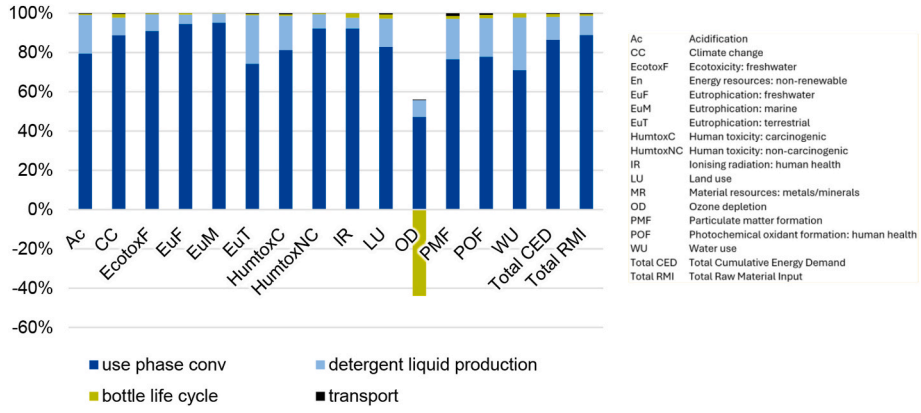


Fig. 10. Environmental impact contributions for conventional detergent product system.

conventional PET bottle (RC 75%) on almost all environmental impact categories (except OD). This exception is, in particular, due to substitution of virgin PET production depending on terephthalic acid, whose OD potential is disproportionately high. The largest variance due to changes in RC is found in OD again due to terephthalic acid. Notably, the PET bottle (RC 100%) outperforms the refill system (1 cycle) in multiple impact categories (Ac, EuT, OD, PMF, POF) due to larger masses of virgin materials substituted at EoL.

While occasional trade-offs are observed among environmental impact categories, they often shift in similar directions or exhibit dominant patterns of change. These patterns manifest the general impression of the refill pouch being more environmentally friendly, in line with Nessi et al. (2014) studying refill systems for liquid detergent. However, often, substantially higher environmental burdens for reusable packaging can be noted due to larger amounts of packaging material required to enhance durability (Cottafava et al., 2021; Thomassen et al., 2024). This issue is circumvented since reused packaging initially intended to be a single-use design is not altered, but sufficient for multiple usages. Furthermore, reuse systems often cause additional transportation trips for return, which is avoided as the bottle is refilled at home (Muranko et al., 2021).

This study captures circular burdens and benefits upstream and downstream. However, this study simplifies consideration of recyclate quality, despite its criticality. A more granular approach could significantly influence results and may be integrated into future research, assuming extensive data availability. In addition to recyclate quality, the

Table 4
Input data for MCI calculations for both packaging designs (values for one reuse cycle).

Input data definition (EMF and Granta Design, 2019)		unit	Conventional detergent			Detergent concentrate
			PET bottle	HDPE cap	PP label	LDPE pouch
M	Product mass	g	12.6	1.6	0.4	2.4
F_r	Share of product mass fed by recycled feedstock	%	75	-	-	-
C_r	Share of product mass collected for recycling	%	81	81	81	50
E_c	Efficiency of recycling process of product at EoL	%	63	67	74	70
E_f	Efficiency of recycling Process to produce recycled feedstock for product	%	63	-	-	-
L	Actual average life span of product	-	1	1	1	½
L_{av}	Industry average life span of product	-	½	½	½	½
U	Actual average number of achieved FUs during use phase	-	½	½	½	½
U_{av}	Industry average number of achieved FUs during use phase	-	½	½	½	½

Table 5
Indicators calculated using input data from Table 4 to compute MCI for both plastic packaging designs (values for one cycle).

Parameter definition (EMF and Granta Design, 2019)		unit	Conventional detergent			Detergent concentrate
			PET bottle	HDPE cap	PP label	LDPE pouch
V	Material that does not originate from recycling, reuse, or biological material from sustainable production	g	3.15	1.60	0.40	2.4
W	Total mass of unrecoverable waste associated with product	g	7.06	0.52	0.12	1.38
X	Utility	-	2.00	2.00	2.00	1.00
F(X)	Utility Factor	-	0.45	0.45	0.45	0.90
LFI	Linear Flow Index (components)	-	0.39	0.71	0.68	0.82
MCI_i	Material Circularity Index (components)	-	0.82	0.68	0.69	0.26
MCI_{tot}	Material Circularity Index (total) for **one refill scenario**	-	**0.73**			

quality of its actual application field could be considered more explicitly. After all, the usage of rPET is considered highly advantageous, although most rPET stems from beverage bottles useable for higher-quality food-grade applications (Conversio, 2024a; Eriksen et al., 2019). Furthermore, previous studies investigated how different EoL modelling approaches affect results, as significant deltas may occur (Ekvall et al., 2020; Sanabria Garcia et al., 2025). The European-wide harmonized Product Environmental Footprint (PEF) provides guidelines for allocating impacts and credits from recycling and energy

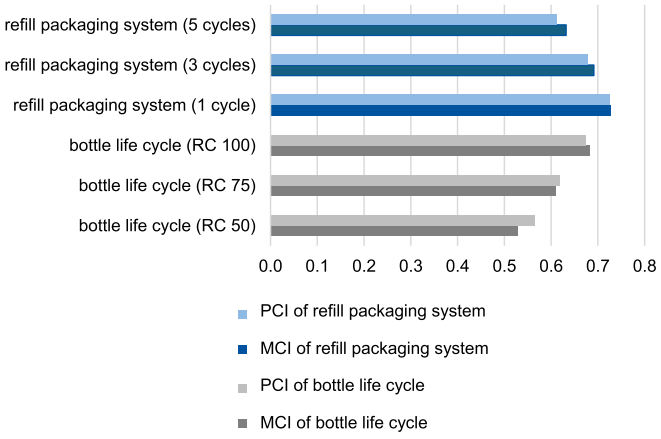


Fig. 11. PCI and MCI results for different scenarios of both packaging designs.

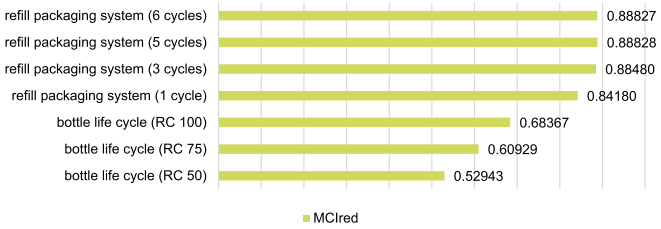


Fig. 12. MCI_{red} results for different scenarios of both packaging designs.

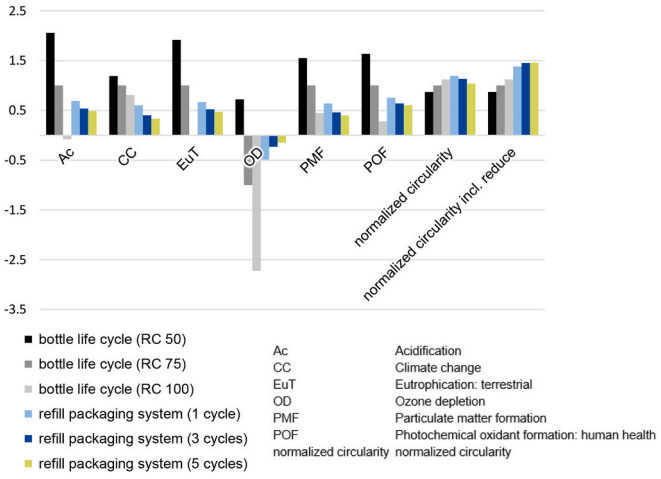


Fig. 13. Relativized LCIA and CE indicator results for both packaging designs with varied RC and reuse cycles.

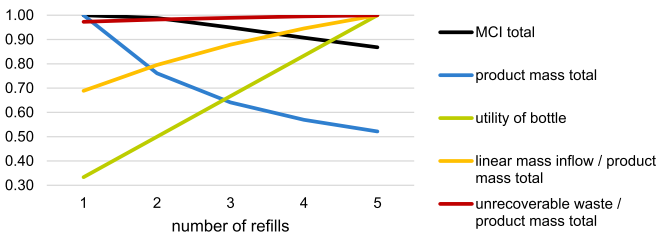


Fig. 14. Presentation of relative MCI results and input parameters with 100% as each maximum value.

recovery through the "Circular Footprint Formula" (CFF) (European Commission, 2025a; Kerps et al., 2024). This approach requires numerous parameters, including an allocation factor reflecting supply and demand of secondary materials (Zampori and Pant, 2019). Due to its methodological complexity (Obrecht et al., 2022; Zampori and Pant, 2019) the CFF application exceeds this study's sensitivity analysis scope while being an interesting outlook.

For LCA, three assessment scopes are investigated – the packaging-only, packaging + product transport, and the full-product scope to enable more targeted insights into environmental performances and to relativize significance of packaging-only assessments. For all impact categories, the use phase provides the highest contribution (except for OD), mainly due to significant amounts of electricity needed to heat up water. One major product system difference is the phase in which detergent dilution occurs, mainly affecting transport. Results for packaging-only and product scope show comparably low transport contribution (see Figs. 7 and 10) unlike for refill-and-reuse systems, where heavier packaging material and assistive infrastructure are required (Muranko et al., 2021; Tua et al., 2020).

A hybrid scope is added covering both packaging systems and transported volumes in direct comparison, excluding the in-balance identical detergent production and use phase. The packaging + product transport scope leverages environmental benefits of the refill system in comparison to the reference (see Fig. 10). Second major difference between both product systems is the packaging design (see section 3.1). Although full-product-scope demonstrates other parameters, such as water temperature, have much higher potential in reducing product systems' environmental impacts than packaging design. Practical implications may be that highest potential in reducing environmental impacts of both product systems lies in consumers' hands. Utilizing low water temperatures and economical proportions of detergent is highly beneficial. In essence, results of different assessment scopes showcase how conclusions or decisions on circular packaging designs can be distorted or misplaced if evaluations consider only packaging-level, in line with Bher and Auras (2024).

5.2. Analysis of circularity results

A direct circularity comparison between both packaging designs shows the conventional packaging performs better compared to the pouch. Root is the bottle RC of 75%, while the pouch consists of virgin material. Nevertheless, thanks to the bottle reusage, the highest circularity score is achieved by the refill packaging system (one cycle). Previous works investigating circularity of single-use vs. reusable packaging systems come to similar results (Rasines et al., 2024; Sazdovski et al., 2024; Thomassen et al., 2024).

Unexpectedly, higher numbers of reusages reduce circularity due to the growing impact of refill pouches' low circularity. This unintuitive effect requires a deeper analysis facilitated by Fig. 14 presenting relative results of MCI input parameters and results for refill and reuse scenarios from 1 to 5. It is evident the total mass of material required per FU decreases. However, the linear inflow and outflow in relation to the total product mass increase.

This effect depicts a real-life dilemma (reduced material vs. restorative material) and reveals a limitation of MCI formulation. Reduced mass per FU is not sufficiently considered (Sazdovski et al., 2024). Therefore, another supplementary input parameter is added to MCI formulation, the "reduced mass share" (see 3.6.1). With its inclusion, the circularity peaks at five reuse cycles. Beyond this point, the absolute virgin product mass exceeds the recycled mass (even with 100% RC for the bottle). It appears reasonable that product masses should not be further incrementally reduced if this requires larger absolute amounts of virgin material inputs.

The higher the RC, the higher the circularity and distance between MCI and PCI results. This effect is caused by the growing influence of additionally integrated sorting efficiency. Overall, circularity assessment supports the refill packaging design is favorable in comparison to the bottle despite the discussed adverse effect of reuse cycles. The PCI is employed for sensitivity analysis as it considers greater detail. By reproducing the same trends as MCI but with lower absolute scores due to formulation differences, PCI strengthens confidence in MCI results.

Another limitation of the applied circularity assessment is the lack of distinction between different material loss fates such as landfill, littering, and energetic recovery. For the latter, slightly higher (energetic) circularity could be argued. In addition, it must be noted that PCI does consider mass and quality loss occurring in plastic recycling in a simplified way via efficiencies and a quality factor (Bracquené et al., 2020). If more extensive data availability on recyclate's physical properties is given, more complex indicators could be tested, e.g., the circularity potential by Schulte et al. (2023b). Furthermore, PCI does not encompass frequency of multiple product life cycles nor lifetime restrictions (Schulte et al., 2023a). It is evident though, the maximum number of possible plastic recycling cycles depends, among other things, on plastic types (Eriksen et al., 2019; Pinter et al., 2021). Thus, it appears crucial to tap into insights of this complex ongoing research field and integrate them into circularity assessments of plastic packaging designs. Furthermore, future work could use the temporal storage effect of carbon as an indicator. Generally, including a larger number of circularity indicators could diversify and alter results.

5.3. Discussion of Circular-LCA consolidation

Comparison between LCIA results and circularity indicators (MCI and PCI) reveals alignment in general trends. Both environmental and circularity performance are mostly ameliorated with a shift from conventional packaging to the refill packaging system. However, exceptions emerge, e.g., in case of increased refill cycles. While environmental profiles improve steadily with more reuse cycles – in all impact categories, except OD – circularity indicators exhibit an opposite trend latest after five refill cycles. This divergence underscores a fundamental methodological difference between LCA and circularity metrics: while LCA captures up- and downstream impact and emissions-based performance, MCI is a mass- and utility-weighted measurement of material efficiency, making it sensitive to compositional system shifts not necessarily reflecting environmental gains (Rigamonti and Mancini, 2021; Samani, 2023). This trade-off between circularity and environmental performance could be overcome if LDPE pouches were produced with RC. For this study's purpose – to explore and analyze key parameters – generating an aggregated score via MCDM is not needed. Nevertheless, future works could reduce complexity and surpass identified trade-offs by applying MCDM (Tzeng and Huang, 2011).

5.4. General limitations

Including secondary and tertiary packaging could influence comparative results (Molina-Besch et al., 2019). Similarly, product loss rates could be different between pouches and bottles, again affecting comparative results. Neither impacts of littering nor chemical recycling are included in this study, although potentials for pouches and bottles might differ. After all, despite extensive data acquisition, viability of estimations could be verified in more depth as data quality and uncertainty assessment could be conducted.

5.5. Alignment with PPWR, ISO 59020, and DPP

This work examines several levers of PPWR, such as recycled content requirements, reusability, refill systems, design for recyclability, and packaging minimization. Their application is analyzed, and respective

environmental and circularity impacts are quantified. Furthermore, design for recycling criteria listed by PPWR are applied, such as separability of components, sorting, and recycling efficiencies. Provided results can be used by operators to demonstrate proposed reuse models offer mostly environmental benefits as required by PPWR. The methodological framework applied is oriented towards ISO 59020 by following its principles, e.g., ensuring meaningful outcomes and relevant system boundaries. Complementary methods applied are identified as very similar to core circularity indicators suggested by ISO 59020 (see Annex E). Collected and documented data on packaging masses and material compositions, recycled content, recyclability-relevant features, and EoL-routing can provide a quantitative basis for circularity-related metadata that PPWR and ISO 59000 anticipate. Moreover, identification of sensitive parameters in our analysis (e.g. effective recycling rates and EoL substitution of primary materials) highlights which attributes are most critical to record in DPPs to support verifiable circularity and environmental claims.

6. Summary & conclusion

This research contributes to methodological discussion and operationalization of CE assessments on nano (product) level. This is accomplished by first identifying and critically reflecting on the most suitable existing CE assessment approaches for a product use case covering multiple R-strategies in the plastic packaging sector. An existing framework with criteria matching this study's requirements, such as avoiding double counting and measuring multiple R-strategies, is adopted. Accordingly, an integrated Circular-LCA is executed with MCI and PCI as product-level circularity indicators to compare a single-use detergent bottle with a concentrate pouch and its refill-and-reuse packaging system. This approach is to a large extent in line with ISO 59020 requirements and reveals synergies and trade-offs between environmental outcomes and circularity while covering most differences between the two packaging and product systems. MCI is selected for its simplicity, wide application, and comparability. PCI is added to provide a complementary, more granular perspective on EoL parameters as a sensitivity analysis within Circular-LCA. Supplementally, an adjusted MCI is suggested, taking into account the R-strategy reduce.

Results support a shift from conventional single-use packaging towards refill-and-reuse configuration, with even a single refill cycle mostly reducing environmental impacts of packaging-only scope by on average 35% and substantially increasing circularity. Circularity peaks at one reuse cycle and then declines, due to outweighing influence of the refill pouches 100% virgin feedstock. Although this effect is relativized, if mass reduction as a key R-strategy is integrated, as the adjusted MCI peaks at five reuse cycles. Introducing RC for refill packaging potentially mitigates this trade-off. Findings of such indicator limitations and conflicts in CE objectives highlight the value of combining life cycle-based methods with circularity indicators to assess product systems more holistically to avoid burden shifting. However, the full-product scope reveals environmental savings from changed packaging designs are relatively unremarkable due to vastly outbalancing impacts during production and use phase. Therefore, it is crucial for circular sustainable packaging development to exceed packaging level when conducting informative assessments for decision-making.

Among the circular strategies evaluated, Reduce, Reuse, and Recycle emerge as the dominant levers in improving both environmental and circularity performance, offering clear net benefits without major trade-offs (enabled by a strategic Rethink of the product system). Conversely, Recover, as lower-order strategy, offers supplementary advantages but exhibits more constrained influence and a higher likelihood of category-specific trade-offs.

Results of this study showcase the complexity of plastic packaging assessment in the context of CE. Sensitive parameters are emphasized to guide future assessments for practitioners. Product-specific findings are especially relevant for liquid household product packaging in European market contexts with mature waste infrastructure, providing recommendations on more circular and sustainable packaging design required by PPWR. Moreover, this work provides guidance on how to systematically evaluate multiple R-strategies oriented towards ISO 59000 requirements. It exemplifies respective data needs, which is of great importance given the growing interest in CE practices and related DPP initiatives.

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

During the preparation of this work the authors used OpenAI GPT in order to rephrase and shorten text passages. After using this tool, the authors reviewed and edited the content as needed and takes full responsibility for the content of the published article.

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CRediT authorship contribution statement

Annalisa Karin Nolte: Conceptualization, Formal analysis, Methodology, Project administration, Writing – original draft. **Milad Goudarzi:** Data curation, Methodology, Writing – original draft. **Kathrin Greiff:** Conceptualization, Supervision, Writing – review & editing.

Declaration of competing interest

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

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.jclepro.2026.148822.

Annex A. Literature background

A.1 Methodological development of CE assessment (extension)

Measuring the progress of CE has become an important and rapidly evolving area of research (Kristensen and Mosgaard, 2020). Ferrante et al. (2025) conducted a systematic literature review to examine the evolution of CE assessment. The results show that LCA, followed by MCDM, is the

most widely used method for assessing the environmental impact of circular systems over their entire life cycle (Ferrante et al., 2025). The transformation towards sustainable CE requires not only increased circularity, but also consideration of environmental, social, and economic aspects across the entire value chain (BMUKN, 2024). A holistic assessment of all these aspects is challenging, partly due to the growing diversity of product systems and business models within the CE (Ahmed et al., 2022).

A.2 CE indicators and assessment frameworks on nano (product) level (extension)

CE indicator selection processes are complex since suitable indicators need to be identified among a wide variety of options, depending on specific use cases and assessment objectives (Ahmed et al., 2022; Pauliuk, 2018; Saidani et al., 2019). Ahmed et al. (2022) have therefore developed a framework that includes step-by-step instructions for the indicator selection process for different CE levels. Similarly, Saidani et al. (2019) developed a taxonomy that classifies indicators into 10 categories to assist with case-specific indicator selection. Luthin et al. (2023) apply six of Saidani's ten categories in order to identify suitable CE indicators for integration into a Circular Life Cycle Assessment (Circular-LCA) framework. The systematic literature review by Oliveira et al. (2021) underscores the need for such a consolidated methodology to be used in LCSA in parallel with circularity indicators.

One of the main advantages of MCI (recommended by Luthin et al. (2023)) is its flexible structure, which allows researchers to adapt or extend it according to the specific requirements of their case studies (Gonçalves et al., 2024; Tashkeel et al., 2021; Vadoudi et al., 2022). Collectively, studies, e.g., of Gonçalves et al. (2024) and Vadoudi et al. (2022) reflect a growing effort to adapt circularity metrics to packaging contexts, with recurring themes such as reusability, data granularity, and material degradation pointing to key methodological challenges in CE indicator design. In case of extensive data availability, circularity potential of recycled plastic can be, e.g., measured via a functional requirement matrix approach comprising mass and quality conservation based on physical properties (Schulte et al., 2023b). In addition, circularity is not just about a single reutilization of material but should ideally comprise resource usage over multiple product life cycles (Schulte et al., 2023a).

A.3 Previous CE assessments of plastic packaging (extension)

Besides reduce, the R-strategy reuse is often applied through reusable vs. single-use packaging comparisons, examining their environmental performances (Brock and Williams, 2020; Thomassen et al., 2024; Tua et al., 2019). Often, the main aim is to identify the conditions under which reusable alternatives may yield environmental benefits. Cottafava et al. (2021) evaluated four single-use cup systems against their reusable counterpart across various use-phase and EoL scenarios. While their approach offers clarity by isolating the reuse benefits, it limits the consideration of broader circularity strategies. Thomassen et al. (2024) conducted a similar comparative LCA of rice packaging systems, analyzing a conventional single-use plastic film against a newly designed reusable alternative. As with Cottafava et al. (2021), the reusable packaging exhibited a substantially higher impact per unit in its baseline configuration due to its increased material mass and more complex design.

Several recent contributions have shifted their focus towards more holistic CE assessments of packaging systems. Papo and Corona (2022) conduct a cradle-to-grave LCA of two packaging alternatives for non-beverage High-Density Polyethylene (HDPE) bottles made of: virgin versus RC. To foster holistic CE assessments, the *Buddiepack* research project developed a circularity indicator Excel tool, which aims to facilitate consideration of social, economic, environmental, and circularity aspects when comparing single-use with alternative reusable products (Buddie-Pack, 2025).

Annex B. Overview of CE indicator applications in the plastic packaging sector

Table 6
Overview of CE assessment studies in the plastic packaging sector and the applied or suggested CE indicators

Reference	Description	Applied CE-Indicators
Chairat and Gheewala (2023)	circularity assessment of open-/closed-loop recycling alternatives; PET bottle-to-bottle vs bottle-to-fiber using LCA	MCI
Gonçalves et al. (2024)	CE analysis and material flow analysis in Portugese LWP sector	a modified version MCI
Karayılan et al. (2021)	Application of CE approaches for minimization of environmental impacts in plastic packaging sector	MCI
Lonca et al. (2020)	Closed-loop recycling strategies in PET sector of the US	MCI
Matos et al. (2023)	classification of 30 MCI indicators at micro level regarding CE strategies and identification of relevant indicators for plastic packaging sector	MCI, PCI, VRE, CPI, SCI, TRP, CEI, GRI, PLCM
Matos et al. (2024)	Indicator review and taxonomy for use in CE and eco-design	MCI, CEIP, VRE
Niero and Kalbar (2019)	Framework for selecting CE metrics through a life-cycle lens; beer industry case study	MRS, MCI
Pauer et al. (2019)	Recommendation of one-dimensional categorized CE indicators for food packaging: input/output/energy	Recycled content, Reuse rate, etc.
Rasines et al. (2024)	Circular-LCA of fresh fruit packaging; reusable vs single use crates	MCI, MRS, CI
Roos Lindgreen et al. (2021)	CE indicators review for glass & PET bottles from grey literature	MCI, MRS, RESCOM, UL3600, CTI
Sazdovski et al. (2024)	Comparative study of secondary packaging of fresh food in Spain; single-use cardboard vs reusable crates	MCI, PCI
Syu et al. (2022)	CE assessment in food-grade plastics	MRS, CEV, PLCM, MRI, MCI
Tashkeel et al. (2021)	Economic-based CE assessment of LWP waste	a cost-normalized version of MCI
Thomassen et al. (2024)	Comparative Circular-LCA of reusable vs. single use rice packaging	Pauer et al. (2019) indicators, MCI
Vadoudi et al. (2022)	Circular LCA of multi-layer plastic packaging	a modified version MCI
Šuškevičė and Kruopienė (2021)	Festival beverage cup reuse models and CE impacts	MCI

Annex C. Further specification of LCI

C.1 Production of detergent liquid

The production of detergent liquid was modelled based on a typical industrial blending process that utilizes proprietary formulations. Careful attention was given to translating generic ingredient labels into specific chemical compounds identifiable in the ecoinvent database. A consolidated formulation was developed through cross-comparison of industry standards (European Union, 2015; Moura and Da Silva, 2024; Yeser Chemicals, 2023). For the *conventional* product variant, a standard diluted formulation was established. For the *concentrate* product variant, adjustments were made to reflect reduced water content according to dilution guidelines provided by Henkel (120 ml concentrate mixed with 330 ml of water). Energy consumption during production was estimated at 0.94 kWh per liter of detergent liquid using inventory data from a report of the European Union for hand dishwashing detergents (2015). Since the focus of this study is on CE assessment methodology rather than detergent formulations, a simplifying assumption is applied, setting the density of all chemical compounds and the mixed detergent (concentrate) at 1 g/cm^3. To determine appropriate electricity datasets in ecoinvent, voltage requirements were cross-checked with specifications from a commercial industrial mixer (Chasing Machine Ltd., 2025).

C.2 Packaging and manufacturing

The packaging and manufacturing phase involves inputs such as plastic granulates and electricity, while the output is the filled and packaged product. The content volumes can be retrieved from the products' labels. The masses of the packaging components were determined using a research lab balance with a precision of 0.1 mg, but were reported at a resolution of 0.1 g. It is assumed that Henkel does not manufacture packaging components in-house; instead, these components are outsourced to third-party suppliers who utilize local market plastic granulates to produce items like bottle preforms and lids via injection molding or label rolls and pouch film rolls, produced via film extrusion. The final packaging assembly is assumed to take place at Henkel's facility using the following machines. For conventional, the bottle is formed, filled, capped (Changzhou MIC Machinery Ltd, 2025b) and labeled (Changzhou MIC Machinery Ltd, 2025a). The pouch is formed, filled, and sealed using a stand-up pouch forming machine (SN Maschinenbau GmbH, 2025). Electricity consumption for this stage was estimated using technical configurations of machinery, while noting that real operational energy consumption tends to be lower than rated power capacity. We assume a general factor of 0.7 throughout this study to compute operational energy consumption from rated power specifications, following advice from expert interviews. Secondary and tertiary packaging – such as transport cartons and pallets – were excluded from this assessment due to their insignificant contribution to overall system impact (Kang et al., 2013; Molina-Besch et al., 2019).

Sensitivity analysis – RC of PET bottle

An empirical study concludes that 75% represents the optimal RC threshold that can be maintained indefinitely without significant loss of physical or chemical quality (Pinter et al., 2021). Although it is important to note, the study subjects a closed-loop recycling system over eleven cycles (including solid-state polycondensation). This conclusion aligns with current industry practices. Petcycle – a major actor in the German beverage packaging market, accounting for approximately 8% of PET bottle circulation – recommends a RC of 75% for its member companies (PETCYCLE, 2020). Such benchmarks reflect a more realistic and technically grounded standard for RC that balances quality, process limitations, and market-scale feasibility. Hence, sensitivity analysis includes a variation of RC for the PET bottle.

C.3 Use phase

The use phase inventory is based on literature and product-specific data relevant to dishwashing practices in Germany, with a focus on typical consumer behavior, water and energy use, and product dosing. For an average use (in between full-sink and direct application), 10 ml per 4 place settings (PS) is adopted. Upscaling the 450 ml content of the conventional product, the cleaning of 180 PS can be achieved, which is the FU of the study. The amount of water and energy used during use phase is adopted from Stamminger et al. (2007). The wastewater produced during manual washing of the 180 PS is assumed to be average wastewater treated in an average European wastewater treatment plant. As a conservative default scenario, it is assumed that consumers reuse and refill the detergent bottle only once.

C.4 Transportation processes

Transportation processes between life cycle stages were included in the LCI, considering the following routes: packaging supplier to manufacturer, manufacturer to distributor, distributor to local retailer, retailer to household, collection from household to sorting plant, sorting plant to recycling plant, sorting plant to energy recovery plant, and recycling plant to energy recovery plant. For all transport steps, a fixed distance of 100 km was assumed except for the transport to the household (5 km) and collection from household to sorting plant (120,5 km) (Bulach et al., 2022). The producing company is situated in Düsseldorf (Henkel, 2025), and the consumer is assumed to be living in Aachen (distance ~ 100 km by roads, see Fig. 14 (Google, 2026)). Within this region of Germany, numerous companies are situated processing LWP plastic waste and producing secondary plastics. Therefore, the simplification of assuming 100 km per transport distance appears likely and sufficiently conservative. For fuel efficiency, EURO4 is assumed following PEF methodology (European Commission, 2021). Due to the very low contribution of transport to the overall LCA results (see section 4.1) a sensitivity analysis is renounced.

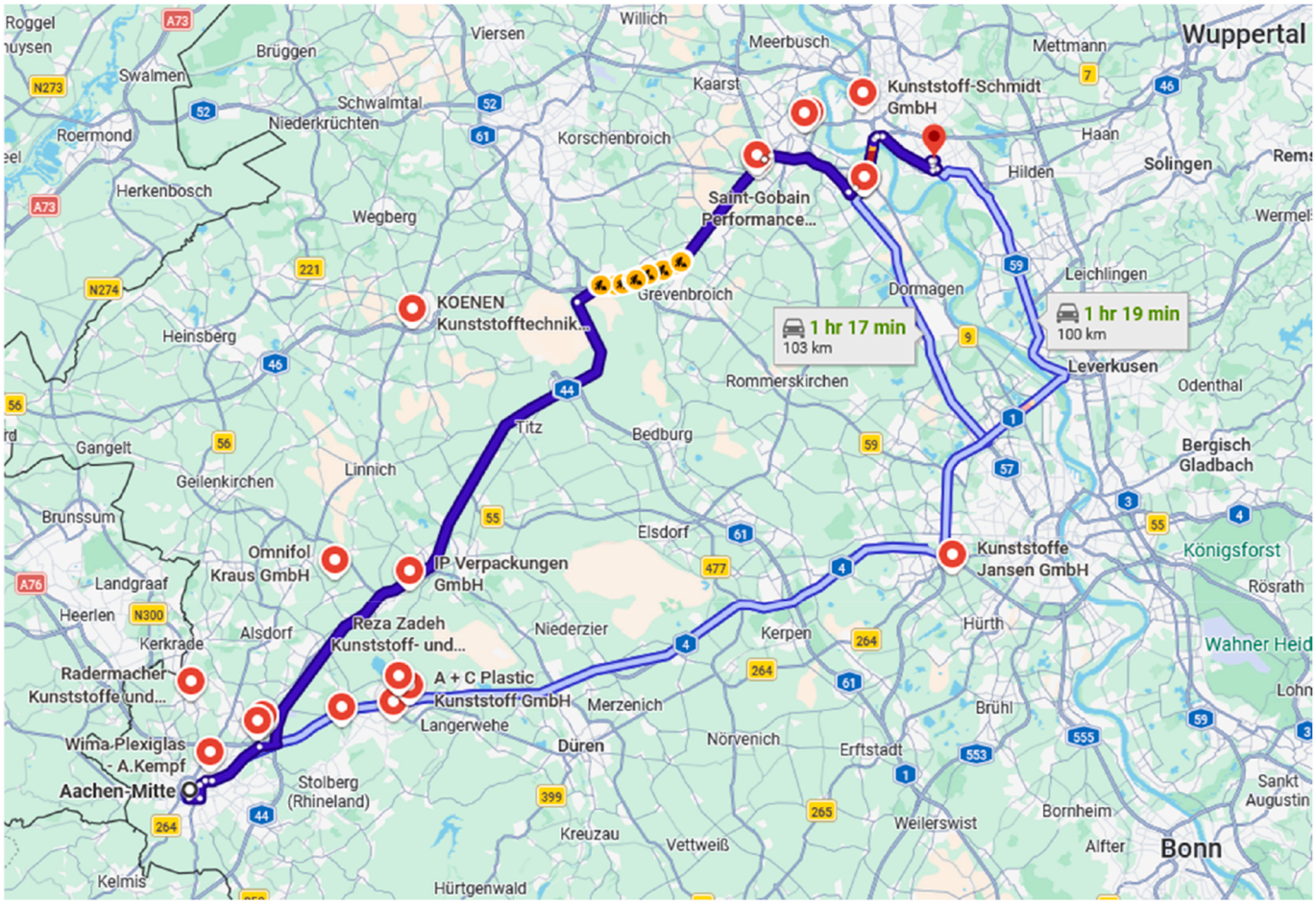


Fig. 15. Three possible routes and surrounding region between Henkel (Düsseldorf) and consumer (Aachen), including some companies working with plastics along the routes (Google, 2026)

C.5 End of life (EoL) collection

Once the detergent products are depleted, their empty packaging remains. In Germany, light-weight packaging waste (LWPW) is usually collected from households and transported to waste sorting facilities as a municipal service. Following Bulach et al. (2022), the logistics were categorized into two routes: the indirect and direct route. The indirect route accounts for 70% of the transported waste and involves an initial distance of 15 km to a mediary station, followed by an additional 100 km to reach the sorting plant. In contrast, the direct route represents 30% of the total transportation and consists of a straight journey of 100 km to the sorting plant. For both routes additional 10 km are considered for near-field transportation from households via trucks (Bulach et al., 2022).

C.6 Light-weight packaging (LWP) waste sorting (part II)

It must be noted that LWP waste sorting plants in Germany vary largely in the extent to which the state-of-the-art processing technology is used. There is an ongoing technical development. However, the implementation depends highly on price developments on the plastic market and legal requirements (e.g., recycling quotas) (Knappe et al., 2021). Defining an "average" sorting plant is therefore a challenging task, which was attempted via a literature basis and complemented by interviews with experienced experts. Following the LWP sorting flow chart of Knappe et al. (2021), the majority of LWPW sorting processes is a considerable cascade of energy-intensive NIR-sorters required to produce different plastic type concentrates. To allocate a reasonable share of energy expenses for the sorting of both packaging designs, a classification of single sorting processes is carried out. Any sorting process that is specifically required to produce a plastic output fraction is classified as *part p* (from **p**lastic). Any other processes are defined as *part np* (**n**ot **p**lastic). For *part p*, the energy requirement per ton LWPW input into the sorting facility is computed. For sorting *part np*, the energy requirement per ton input is determined as well, though the plant input is multiplied by the average plastic share of the sorting plant's output fractions to allocate these expenses exclusively to sorted plastics. To conclude, the input plastic packaging masses of the two different designs are multiplied by the sorting energy expenses per ton to obtain their allocated share.

C.7 Recycling modelling and avoidance of double counting

How the recycling modelling is implemented in this study is exemplified on PET, as our case study includes input and output flows of recyclate for this material. For the other materials, the recycling modelling follows the same approach as far as recyclate is produced. The approach is visualized in Fig. 16. The PET-RC of the bottle is modelled using the input flow *"PET, recycled"*, mostly created based on primary data from expert interviews. This

flow is provided by the created process *"PET, recycled input production"* which includes all expenses occurring to produce rPET (sorting, recycling, interim transports) .

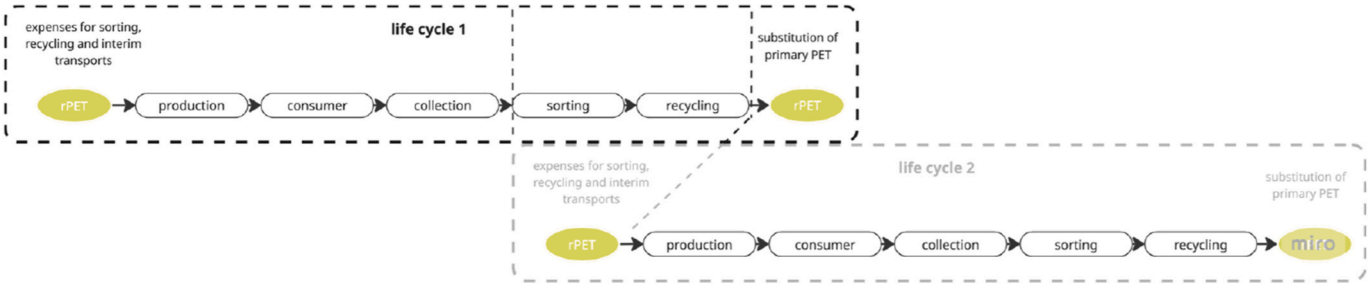


Fig. 16. Exemplified modelling of recycled PET (rPET) in openLCA

In this case study, the recyclate production at the EoL is modelled by applying the avoided burden approach and utilizing the ecoinvent output flow "polyethylene terephthalate, granulate, bottle grade". This means that primary PET is assumed to be substituted by the produced recyclate. This avoidance of primary materials to produce unspecified other products gives credit for the examined life cycle 1. The unspecified next material life cycle 2 is not examined, but could be theoretically modelled the same way, accounting for expenses of recycled input flows and giving credits in case recyclate is produced at the EoL.

Annex D. Detailed circularity indicator selection and formulation

The criteria set applied by Luthin et al. (2023) coincides with the requirements for this study and comprises the following criteria (Saidani et al., 2019):

- **Levels**: only micro level addressing indicators are relevant (in this case, micro level comprises nano level)
- **Performance**: only addressing technical (intrinsic) performance (since LCSA addresses sustainability of CE concepts and double counting should be avoided)
- **Loops**: at least three R-Strategies must be addressed by the CE indicators (to ensure inclusion of different CE strategies, emphasize CE goes beyond recycling, avoiding overweighting of single CE strategies)
- **Unit**: must be calculated with real measurements/numbers as input data and result in quantitative output to allow objective comparison
- **Dimension**: transparent, individual components and their weighting were traceable
- **Transversality**: only general indicators are selected

The VRE and CEIP, recommended for plastic packaging circularity assessment by Matos et al. (2024), are excluded based on the criteria "performance" and "unit" (Luthin et al., 2023).

D.1 Material Circularity Indicator (MCI)

The (product) material circularity indicator MCI_p ranges from 0.1 for a fully linear product to 1 for a fully circular product. It is a quantitative indicator and measures the degree of circularity of products by taking into account the use of virgin, recycled, and reused materials or components, EoL recycling, product life span, and product utility. The MCI_p is based on two sub-indicators: the Linear Flow Index *(LFI)* and the Utility Factor $F(X)$. The calculation is conducted using formulas D.1 and D.2 (EMF and Granta Design, 2019):

$$MCI_p^* = 1 - LFI \times F(X) \tag{D.1}$$

$$MCI_p = max\left(0, MCI_p^*\right) \tag{D.2}$$

The *LFI* quantifies the proportion of linear material flows considering virgin material used *(V)* in the product, the product mass *(M)* and direct waste produced *(W)*, see formula D.3. The direct waste produced is composed of unrecoverable waste as shares of the product's material not being recovered at the EoL (W_0), unrecoverable waste generated when producing recycled feedstock for the product (W_F) and unrecoverable waste produced during recycling of (shares) of the product (W_C), see formula D.4. (EMF and Granta Design, 2019)

$$LFI = \frac{V + W}{2M + \frac{W_F - W_C}{2}} \tag{D.3}$$

$$W = W_0 + \frac{W_F + W_C}{2} \tag{D.4}$$

$F(X)$ builds a function over the utility of a product (X), taking into account the lifetime of a product (L) compared to the industry average lifetime (L_{av}), and number of uses of the product (U) compared to the industry average number of uses (U_{av}) (EMF and Granta Design, 2019):

$$F(X) = \frac{0.9}{X} \tag{D.5}$$

$$X = \left(\frac{L}{L_{av}}\right) \times \left(\frac{U}{U_{av}}\right) \quad \text{(D.6)}$$

In comparison to the MCI, the PCI requires a more extensive knowledge of the examined product system and a greater amount of data input, introducing a multitude of new variables to compute. However, since the PCI is based on the MCI formulation but incorporates some general advancements, only the main distinctions between MCI and PCI calculation, which are relevant for this case study, are outlined in the following section. (Bracquené et al., 2020)

D.2 Product Circularity Indicator (PCI)

Similar to the MCI, the PCI takes into account the *LFI* and the utility factor *(X)*, though the factor of 0.9 is removed (see equation (D.7)). (Bracquené et al., 2020)

$$PCI = 1 - \frac{LFI}{X} \quad \text{(D.7)}$$

In contrast to the MCI, the recycling efficiency is not simplified as one term, but a differentiation between the efficiency of material separation (E_{ms}) and recycled feedstock production (E_{rfp}) is implemented. Since this case study focuses on the EoL phase and material recovery of different plastic packaging designs, it is highly relevant to acknowledge separate efficiencies for sorting and recycling plants to enable a more precise analysis of potential material losses. Hence, the unrecoverable waste consists of waste from feedstock production (W_{fp}), waste from component production (W_{cp}), post-use waste, or in other words, material sent to energy recovery or landfill at end-of-use (W_u), waste generated during material separation (W_{ms}), and waste generated during recycled feedstock production (W_{rfp}), see equation (D.8). (Bracquené et al., 2020)

$$W = W_{fp} + W_{cp} + W_u + W_{ms} + W_{rfp} \quad \text{(D.8)}$$

In addition, a net exchange of recovered recycled material is deduced by distinguishing recycled material recovered (R_{out}) and recycled material used for feedstock production within the same product system (R_{in}) (see equation (D.9)). In this context, a quality factor of the material entering and leaving the product system is introduced as *Qin* and *Qout* equation (D.10). (Bracquené et al., 2020)

$$R = |R_{in} - R_{out}| \quad \text{(D.9)}$$

$$LFI = \frac{V + W + Q_{in}R_{in} + Q_{out}R_{out}}{V_{linear} + W_{linear}} \quad \text{(D.10)}$$

For applying the PCI to products made of a certain number of materials or components *(i)*, a mass-based weighting *(M_i)* method, as in equation (D.11), can be used. With the help of this equation, the calculations of individual PCIs for each component *(PCI_i)* can be combined into a *PCI_{total}* (Bracquené et al., 2020).

$$PCI_{total} = \frac{\sum_i M_i * PCI_i}{\sum_i M_i} \quad \text{(D.11)}$$

The same weighing approach is applied when computing the MCI results. More detailed information can be found in the respective publication (Bracquené et al., 2020).

Annex E. Discussion of ISO 59020 indicators

The following table demonstrates the large extent of content-wise overlapping between the MCI (EMF and Granta Design, 2019) and the resource inflow and outflow circularity indicators from ISO 59020 (2024). There are only two indicators that are not fully equivalent – the actual reused content and actual recirculation outflows […] in the biological cycle (ISO, 2024) and the collected fraction of product mass for reuse and collected fraction for composting (EMF and Granta Design, 2019). However, both indicators or parameters are equal to zero and irrelevant for this case study, as no biological or reused materials appear.

Table 7
Circularity indicators from ISO 59020 (2024) and their equivalent or overlapping MCI input parameters (EMF and Granta Design, 2019)

Indicator category	Circularity indicator	Summary description	mandatory/ optional	MCI equivalent/overlap
Resource inflows	A.2.2 Average percent reused content of an inflow (X)	Fraction of input material resources that is reused content.	mandatory	Fraction of the product mass feedstock from reused material
Resource inflows	A.2.3 Average percent recycled content of an inflow (X)	Fraction of input material resources that is recycled content.	mandatory	Fraction of the product mass feedstock from recycled material
Resource inflows	A.2.4 Average percent renewable content of an inflow (X)	Fraction of material resources inflow (X) that is sustainably produced renewable content.	mandatory	Fraction of product mass feedstock from sustainable production (biological material except reuse/recycle)
Resource outflows	A.3.2 Average lifetime of product or material relative to industry average	Indicator of time that an output resource (e.g., product) will remain in use compared to an industry average for the resource.	optional	actual average lifetime of product/industry-average lifetime of a product of the same type
Resource outflows	A.3.3 Percent actual reused content derived from outflow (X)	Fraction of outflow that is reused.	mandatory	Fraction of product mass collected for reuse

(*continued on next page*)

Table 7 (*continued*)

Indicator category	Circularity indicator	Summary description	mandatory/ optional	MCI equivalent/overlap
Resource outflows	A.3.4 Actual % recycling rate of outflow (X)	Fraction of outflow that is recycled.	mandatory	Fraction of product mass being collected for recycling * Efficiency of recycling process
Resource outflows	A.3.5 Percent actual recirculation of outflow in the biological cycle	Fraction of outflow content that is recirculated at end of life for safe return to the biosphere and meets the qualifying conditions for recirculation.	mandatory	Fraction of product mass from biological material (sustained production) collected for energy recovery

Table 8
Mandatory circularity indicator (ISO, 2024) results for the conventional detergent bottle (conv) and the concentrate pouch (conc), and different numbers of refill and reuse cycles, and recycled content (RC) of the bottle

Circularity indicator	conv	conc	One refill cycle		Three refill cycles		Five refill cycles		conv (RC = 50 %)	conv (RC = 100 %)
	bottle incl. label & cap	pouch	bottle incl. label & cap	pouch	bottle incl. label & cap	pouch	bottle incl. label & cap	pouch	bottle incl. label & cap	pouch
A.2.2 Average percent reused content of an inflow (X)	0 %	0 %	0 %	0 %	0 %	0 %	0 %	0 %	0 %	0 %
A.2.3 Average percent recycled content of an inflow (X)	65 %	0 %	65 %	0 %	65 %	0 %	65 %	0 %	43 %	86 %
A.2.4 Average percent renewable content of an inflow (X)	0 %	0 %	0 %	0 %	0 %	0 %	0 %	0 %	0 %	0 %
A.3.2 Average lifetime of product or material relative to industry average	1	1	2	1	4	1	6	1	1	1
A.3.3 Percent actual reused content derived from outflow (X)	0 %	0 %	0 %	0 %	0 %	0 %	0 %	0 %	0 %	0 %
A.3.4 Actual % recycling rate of outflow (X)	51.63 %	35.00 %	51.63 %	35.00 %	51.63 %	35.00 %	51.63 %	35.00 %	51.63 %	51.63 %
A.3.5 Percent actual recirculation of outflow in the biological cycle	0 %	0 %	0 %	0 %	0 %	0 %	0 %	0 %	0 %	0 %

Annex F. Implications of impact category selection

In methodological terms, these substitutions imply a shift from impact-oriented resource depletion indicators as defined in EF 3.1 towards pressure-based indicators that quantify total energy and material flows associated with a product system. The resource use categories in EF 3.1 aim to approximate potential future scarcity or extraction constraints through characterization modelling, whereas CED and RMI remain closer to the inventory level and express the absolute amount of primary energy and raw materials mobilized per product system (Frischknecht et al., 2015; Mostert and Bringezu, 2019). Accordingly, this study's results for energy and resource use should be interpreted as indicators of resource intensity and potential contribution to absolute resource use reduction (e.g., in CE or planetary-boundary context), rather than as indicators of scarcity-related damage.

A further implication is that the resource-related results of this study are not directly comparable to EF 3.1 studies that strictly apply the default resource categories, especially when it comes to aggregated EF single scores. In line with the broader LCA literature, which acknowledges that no single resource indicator can fully capture all relevant aspects of resource sustainability (Hellweg and Milà i Canals, 2014; Sala et al., 2018), we treat CED and RMI as complementary indicators used alongside EF 3.1 impact categories rather than as a full replacement of the EF method. This choice may lead to some correlation between CED and other impact categories such as CC. However, potential double counting is avoided by not aggregating these indicators into a single score and by interpreting them explicitly as separate dimensions of environmental performance.

Annex G. Sensitivity analysis results for recycling yields and substitution factors

Fig. 17 and Fig. 18 present the sensitivity analysis results for best and worst-case substitution factors and for conservative recycling yields for both packaging systems.

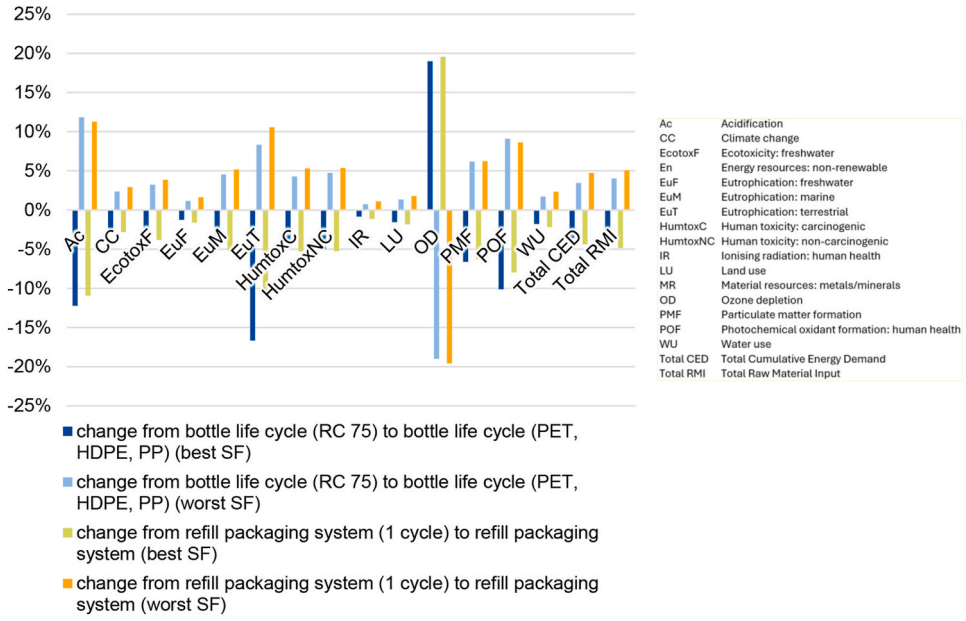


Fig. 17. Sensitivity analysis results for best and worst-case substitution factors for both packaging systems

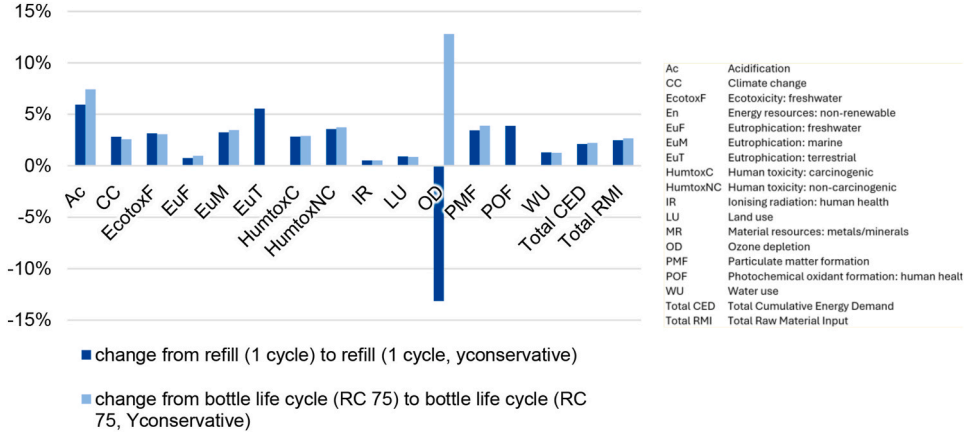


Fig. 18. Sensitivity analysis results for conservative recycling yields for both packaging systems

Data availability

Data will be made available on request.

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