Resources, Conservation & Recycling 226 (2026) 108674

Contents lists available at ScienceDirect

Resources, Conservation & Recycling

journal homepage: www.sciencedirect.com/journal/resources-conservation-and-recycling

Life cycle assessment and life cycle costing of emerging circular flexible plastic food and non-food packaging

Anna-Sophie Haslinger *, Trang T. Nhu, Erasmo Cadena, Gwenny Thomassen, Jo Dewulf, Sophie Huysveld

Department of Green Chemistry and Technology, Research Group STEN, Ghent University, Coupure Links 653, 9000 Ghent, Belgium

ARTICLE INFO

Keywords:
Life cycle assessment (LCA)
Life cycle costing (LCC)
Emerging technologies
Flexible plastic packaging
Circular economy

ABSTRACT

Closed-loop recycling of food (F) and non-food (NF) multi-material multilayer (MuMu) flexible plastic packaging remains challenging for state-of-the-art technologies hence it is currently incinerated and landfilled (baseline). Two innovative recycling scenarios are compared, using a basket-of-products approach for LCA and LCC: one producing NF packaging and another producing both F and NF packaging enabled by tracer technology. This involves advanced mechanical and physical recycling, followed by the production of novel polyethylene (PE)-based mono-material multilayer packaging with PE post-consumer recyclates (PCR), compared against the baseline. The innovative scenarios outperform the baseline across climate change, fossil and minerals and metals resource use, freshwater ecotoxicity, particulate matter, and acidification achieving 16 % to 52 % impact reductions, with LCC results showing a similar trend (41–49 %). The loops are not fully closed in mass terms, due to losses but also laminate design with PCR is challenging and does not enable full uptake.

1. Introduction

The proposed Packaging and Packaging Waste Regulation (PPWR) of the European Parliament and the Council aims to reduce packaging waste, while promoting the incorporation of post-consumer recycled (PCR) materials. A key provision mandates that all packaging is recyclable by 2030 (European Commission, 2022a; European Parliament, 2024), driving innovation in flexible plastic packaging (FPP), particularly in design for and from recycling (CEFLEX, 2020; Martínez Leal et al., 2020). However, producing recyclable flexible laminates with PCR poses both technical and regulatory challenges, particularly for food packaging. The inclusion of PCR in food-contact applications is strictly regulated under EC Regulation 2022/1616, as amended by EC Regulation 2025/351, which restricts its use to materials derived exclusively from waste plastics previously used in food packaging or from recycling processes approved for safety by the European Food Safety Authority (European Commission, 2025; 2022b). The latter is the case for polyethylene terephthalate (PET) (EFSA Panel on Food Contact Materials, Enzymes and Processing Aids (CEP) et al., 2024), and for recycling of HDPE and PP in controlled closed-loop product systems for producing new crates for fruits and vegetables (EFSA Panel on Food Contact Materials, Enzymes, Flavourings and Processing Aids (CEF), 2013). Furthermore, EFSA has approved the recycling of polypropylene (PP), styrene acrylonitrile resin, and acrylonitrile butadiene styrene tableware from airline on-board services (EFSA Panel on Food Contact Materials, Enzymes and Processing Aids (CEP) et al., 2021).

The development of high-performance packaging has resulted in the adoption of multi-material multilayer (MuMu) structures (Ragaert et al., 2017). Traditionally, MuMu plastic packaging is incinerated (Koinig et al., 2022) or landfilled (Eunomia and Plastic Recyclers Europe, 2020; Plastics Recyclers Europe, 2023) due to inadequate collection, sorting, and recycling infrastructure. This is further driven by the separation challenges (Tamizhdurai et al., 2024) linked to its complex composition of polymers such as polyethylene (PE), PET, polyamide (PA), and PP, along with aluminum layers (Freitag et al., 2024; Ragaert et al., 2017). Moreover, sorting technologies such as Near Infrared (NIR) spectroscopy cannot identify MuMu properly (Schmidt et al., 2022) nor reliably distinguish between food (F) and non-food (NF) packaging waste, hindering closed-loop F packaging recycling (Geueke, 2018; Kusch et al., 2021). Additionally, polymers or materials are frequently non-miscible (Schmidt et al., 2022), contaminations and non-intentionally added substances (Horodytska et al., 2020) result in inconsistent regranulation behaviors and quality issues (Auer et al., 2023; Soares et al., 2022), along with odor issues (Horodytska et al., 2020). Additionally low

* Corresponding author.
E-mail address: annasophie.haslinger@ugent.be (A.-S. Haslinger).

https://doi.org/10.1016/j.resconrec.2025.108674
Received 2 April 2025; Received in revised form 15 October 2025; Accepted 27 October 2025
Available online 7 November 2025
0921-3449/© 2025 The Authors. Published by Elsevier B.V. This is an open access article under the CC BY-NC license (http://creativecommons.org/licenses/by-nc/4.0/).

economic viability and limited market demand further exacerbate the issue (Tamizhdurai et al., 2024). In conclusion, current recycling systems are primarily designed for mono-materials (Schmidt et al., 2022) which has driven recent advancements in mechanical, chemical, thermochemical and biochemical recycling techniques (Tamizhdurai et al., 2024).

In the context of circular FPP, overcoming these challenges necessitates enhanced collection infrastructure (CEFLEX, 2020), advanced sorting such as tracer-based sorting (TBS) technology using fluorescent chemicals (Kusch et al., 2021), and innovative recycling technologies, including advanced mechanical recycling with delamination and deinking (Ügdüler et al., 2023, 2022), physical recycling via selective dissolution (Kaiser et al. 2017; Kol et al. 2021) or chemical recycling (Ragaert et al., 2017). Furthermore, advanced decontamination techniques, including deodorization, are essential to enhance the suitability of PCR for high-performance applications (Horodytska et al., 2020). Moreover, implementing circular design strategies, such as mono-material multilayer (MoMu) laminates, is crucial to enhance recyclability (CEFLEX, 2020).

In this context, Life Cycle Sustainability Assessment (LCSA) plays a crucial role in guiding informed decision- and policy-making for emerging technologies (Haase et al., 2022; Soares et al., 2022). LCSA aims to assess the three pillars of sustainability namely environment, economy, and society using methodologies such as Life Cycle Assessment (LCA), Life Cycle Costing (LCC), and Social Life Cycle Assessment (S-LCA) (UNEP/SETAC et al., 2011; Valdivia, 2024.). According to Pihkola et al. (2022), the assessment of criticality and raw material circularity should be considered alongside the traditional pillars, presented as complementary topics in the Orienting LCSA framework (Isasa et al., 2024). Furthermore, material flow analysis (MFA) is a widely used method for quantifying circular economy performance (Barkhausen et al., 2023; Lase et al., 2022), with a clear overlap to LCA due to materials flows as base of the inventory (Barkhausen et al., 2023). Examples of other economic assessment approaches for guiding decision-making include Techno-Economic Assessment (TEA) and Cost-Benefit Analysis (CBA) (Bianchi et al., 2021). While these economic assessment methods are simpler and more straightforward, LCC is deemed to be the most consistent framework in combination with LCA (Hunkeler et al., 2008) due to its focus on cost distribution over a product's life cycle. However, compared to LCA and despite e.g., the SETAC guidebook for environmental LCC (Hunkeler et al., 2008), the LCC methodology is less mature (Valdivia et al., 2021) and standardized, and applications vary from one research to another (Taelman et al., 2020).

Despite increasing research efforts, a LCSA or even a combined LCA and LCC of circular FPP remains absent in the scientific literature. The research gap exists because previous studies have either not focused on circular FPP, have applied only one type of assessment (e.g., LCA) or other types of economic assessments (i.e., not LCC), or have not addressed closed-loop systems that return plastic waste to plastic packaging. In the field of combined sustainability assessments (LCA, LCC, and potentially S-LCA) of food plastic packaging, Wohner et al. (2020) did not focus on FPP but conducted an environmental and economic assessment comparing ketchup packaging in PP bottles versus glass. Regarding environmental assessments of plastic recycling, studies such as Garcia-Gutierrez et al. (2023); Kusch et al. (2021) and Tonini et al. (2021), did not address closed-loop systems but focused solely on PCR materials. Furthermore, the study by Tunçok-Çeşme et al. (2024) on FPP only focused on the environmental assessment. In the context of economic approaches, Larrain et al. (2021) and Walker et al. (2020) and Bashirgonbadi et al. (2022) did not apply LCC or address a closed-loop system. Larrain et al. (2021) and Walker et al. (2020) used TEA to either evaluate different post-consumer plastic waste treatments or multilayer packaging recycling via solvent-targeted recovery and precipitation, respectively. Similarly, Bashirgonbadi et al. (2022) assessed an improved mechanical recycling process through CBA.

Based on these research gaps, this work investigates the environmental and economic performance of two innovative scenarios with the aim of achieving circular flexible F and NF packaging through LCA and LCC, respectively. More specifically, the case study includes the EoL treatment of flexible F and NF packaging waste and the production of innovative, recyclable MoMu laminates with PE PCR (applicable for either F or NF packaging). The study compares two innovative scenarios: one where F and NF packaging waste remain combined within the sorted PE-rich household plastic packaging waste stream -enabling only NF packaging laminate production with PE PCR- against, another one where TBS separates F packaging (with tracers) and NF packaging waste (without tracers), enabling the production of both F and NF packaging laminates with PE PCR. The tracer application in F packaging guarantees that the resulting PCR originates solely from food-contact materials in compliance with EC Regulation No. 2022/1616, as amended by EC Regulation 2025/351 (European Commission, 2025, 2022b). To this end, the study examines two innovative advanced recycling technologies, i.e., physical recycling through selective dissolution for mixed F and NF waste or only the NF fraction (separated from the F fraction via TBS), and advanced mechanical recycling with delamination and deinking for the F fraction sorted through TBS. The innovative scenarios are compared to the current situation of F and NF FPP and their linear EoL treatment (incineration with energy recovery and landfilling). An MFA is applied to identify material losses across the scenarios and therefore the contribution to closed-loop recycling or circularity performance in general. Moreover, a combination of LCA and LCC of circular FPP especially for F application and closed-loop recycling is unique. This novelty is further strengthened by its foundation on primary data collected in close collaboration with industry experts and based on technology validated and demonstrated in an industrially relevant environment. This study further adds value to LCC by (i.) using the basket of product approach and (ii.) incorporating Green Premium cost aspects that represents the additional value associated with environmentally preferable alternatives (Köveker et al., 2023).

2. Materials and methods

This section explains the LCA and LCC methodology applied in this case study. There are four phases in an LCA study based on ISO 14040/44: (i.) goal and scope definition; (ii.) inventory analysis; (iii.) impact assessment, and; (iv.) interpretation (ISO 14040, 2006; ISO 14044, 2006). These phases also serve as the basis for the LCC (Bachmann et al., 2024; Ciroth et al., 2011). The methodological guidelines used in the LCA study are based on the Product Environmental Footprint (PEF) approach (European Commission, 2018; European Commission. Joint Research Centre., 2022), and the draft Product Environmental Footprint Category Rules (PEFCR) for flexible packaging (Quantis and Amcor, 2022). Given its highest scientific robustness and transparency among LCC (Bianchi et al., 2021), the study follows the cLCC method, which primarily assesses direct internal costs throughout a product's life cycle (Bianchi et al., 2021; Ciroth et al., 2011; Hunkeler et al., 2008). These costs are borne by one or more actors in the life cycle but are evaluated here from the perspective of a single actor (Bianchi et al., 2021). Examples of cost components include investment costs, operational costs, and sales revenues, with the latter considered a negative cost (Carlsson Reich, 2005). Furthermore, an MFA was developed and visualized using Sankey diagrams. MFA is a systematic approach to quantify the flows and stocks of materials within a defined system, providing valuable insights into resource utilization and waste management strategies (Allesch and Brunner, 2015; Barkhausen et al., 2023) supporting both the LCA and LCC.

2.1. Step 1: goal and scope definition

This study evaluates the environmental and economic performance, and presents material flows of waste management strategies within the European circular economy. It compares innovative approaches with

current practices for recycling PE-rich flexible packaging waste from households, ultimately producing F and NF MoMu flexible packaging with PE PCR, as well as energy (electricity and heat recovered from incinerating residues). The PE-rich flexible packaging waste comprises both MoMu and MuMu structures, with a blend of F and NF packaging. MuMu structures are predominantly present in the F packaging fraction and sensitive NF packaging applications requiring chemical barriers, such as cleaning products (Freitag et al., 2024). Therefore, MuMu flexible packaging waste is a key focus of this study.

2.1.1. *System boundary*

The evaluation spans the EU Circular FoodPack project's timeframe

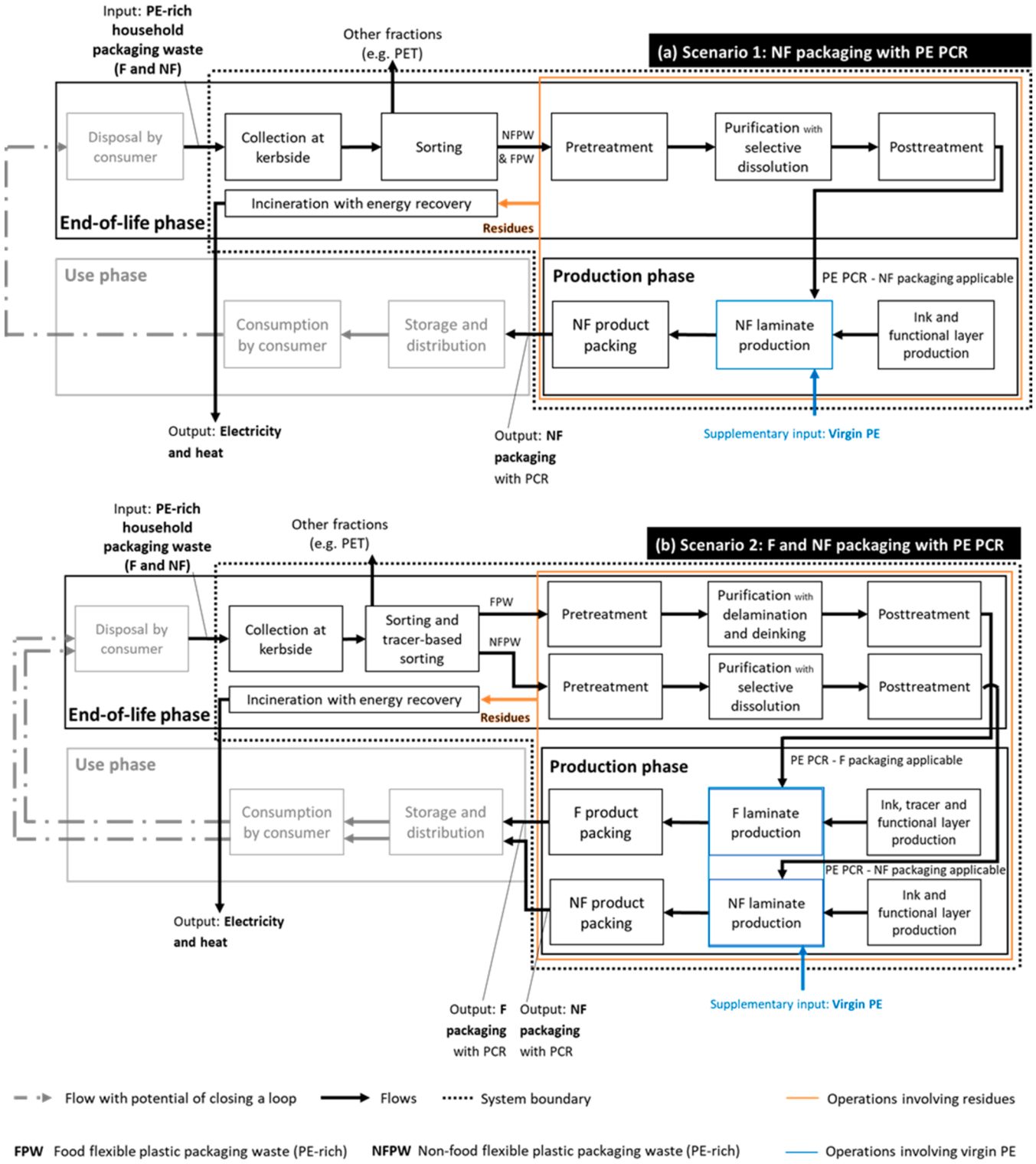

Fig. 1. Overview of the system boundaries for the innovative scenarios (a) S1: NF packaging with PE PCR and (b) S2: F and NF packaging with PE PCR. All recycling processes are (prospectively) assumed to take place at a facility in France, with transportation occurring between sorting, recycling, laminate production in Belgium, and final packaging production in Portugal.

(2021–2024) and includes prospective elements, particularly technologies currently undergoing pilot-scale and semi-industrial testing rather than full industrial operation. To bridge the gap to industrial-scale operations, industry expert estimates were incorporated (cfr. subsection 2.2). Moreover, this study is geographically focused on Europe, with the composition of PE-rich flexible packaging waste analyzed at three sorting plants located in France, Belgium, and Germany. The PE content is 90 wt% in the French sample (C-FRA), 85 wt% in the Belgian sample (C-BEL), and 89 wt% in the German sample (C-GER-310), comprising various PE grades (low density-PE, linear low density-PE, high density-PE). PP is present at 4 %, 6 %, and 2 % in the French, Belgian, and German samples, respectively. PET accounts for 2 % in the French and 1 % in the Belgian, and German samples, while PA constitutes 2 % in the French and 2 % and 3 % in the German and Belgium samples, respectively. The remaining fraction, labeled as residue, makes up 3 % in C-FRA, 6 % in C-BEL, and 5 % in C-GER-310. Please note that the numbers are rounded. NF packaging exhibits notably higher PE content, reaching up to 94 wt%, while F packaging displays greater variability in polymer composition (Freitag et al., 2024). Moreover, the laminate structures of the innovative MoMu packaging comprise multiple layers, including 50 %-54 % PE PCR, virgin PE, functional layers for processability, and functional barriers for F packaging applications. The proportion of PE PCR incorporation is guided by specific project objectives. For the F packaging laminate with 50 % PE PCR, the structure consists of: 1 % overprint varnish (OPV), 15 % machine-direction oriented PE (MDOPE) including functional barrier, 1 % ink with tracer, 1 % delamination primer, 2 % solvent-free adhesive, 50 % PE PCR, 2 % solvent-free adhesive, 10 % virgin PE, 1 % PE tie, 2 % EVOH, 1 % PE tie, and 15 % PE sealant. For the NF packaging laminate with 54 % PE PCR, the structure consists of: 1 % OPV, 22 % MDOPE (50 % virgin and 50 % PE PCR), 1 % ink, 2 % solvent-free adhesive, 43 % PE PCR, and 30 % PE sealant. The order of the list corresponds to the order of the layers, starting with the outer layer, while the percentage is calculated relative to the laminate's surface weight.

The system boundary of the circular FPP value chain covers the "waste to gate" phase, encompassing: (i.) collection, state-of-the-art (SoA) sorting, and TBS; (ii.) pretreatment (oversorting, shredding, washing, grinding, and float-sink separation); (iii.) purification via selective dissolution and/or delamination and deinking; (iv.) post-treatment (recompounding and deodorization); (v.) laminate production, including ink, tracers, and functional layers; and (vi.) packaging production, all within Europe (Fig. 1). These process steps, along with detailed inputs and outputs, are provided in the Supplementary Information- Annex 1 as process flow diagrams. The study excludes the use phase (storage, distribution, consumer consumption, and disposal) and other waste fractions collected together with the waste fraction under study, like PET and aluminum, which follow separate recycling pathways.

The assessment encompasses three main scenarios, with Fig. 1 illustrating the system boundaries for the innovative scenarios S1 (NF packaging with PE PCR) and S2 (F and NF packaging with PE PCR) and Fig. 2 depicting the S3 baseline scenario. In the S1 scenario (Fig. 1a), one ton of PE-rich household flexible packaging waste is processed through selective dissolution, producing a NF packaging laminate with 54 % PE PCR in a duplex MoMu structure for detergent tab pouches. Additionally, electricity and heat are generated as outputs from the incineration of residues with energy recovery. In S2 (Fig. 1b), one ton of PE-rich household flexible packaging waste, including tracers in F flexibles, is processed through TBS into F (25 %) and NF (75 %) flexible fractions. These shares were derived from analyses of the Belgian, French, and German PE-rich flexible packaging waste samples conducted by the waste manager within the project. The F fraction undergoes delamination and deinking, while the NF fraction is processed via selective dissolution. Outputs include F packaging (triplex MoMu laminate with 50 % PE PCR) for coffee pouches, NF packaging (duplex MoMu laminate with 54 % PE PCR) for detergent tab pouches, and electricity and heat generated from residue incineration with energy recovery. Selective dissolution was established for NF-applicable PE PCR, while delamination and deinking were designated for F-applicable PE PCR based on internal project discussions on the initial allocation of resources. Delamination refers to the separation of the different layers of the multilayer structure (Ügdüler et al., 2022) and involves the removal of adhesives between layers (Soares et al., 2022). In this case, a water-based delamination–deinking process using NaCl-based detergents is applied. Besides PE PCR, virgin PE, functional layers (functional barriers e.g., EVOH, solvent-free adhesive, delamination primer, coatings), tracers (in the case of F packaging) and ink are incorporated for the performance of the laminate structure. The design-for-recycling measures include next to the shift from MuMu to MoMu, as well the transition from solvent-based to solvent-free adhesives, which enhances separation efficiency during water-based delamination and complements the use of delamination primers in the laminate structure. The tracers applied in this study are inorganic fluorescent markers incorporated into the printing ink at ppm-level concentrations, embedded between virgin PE layers.

In the S3 baseline scenario (Fig. 2), one ton of PE-rich household flexible packaging waste is collected in two ways: 25 % is collected through selective collection, while 75 % is collected through mixed collection as municipal waste (Eunomia and Plastic Recyclers Europe, 2020). A portion of these flexible films (13 %) undergoes sorting after

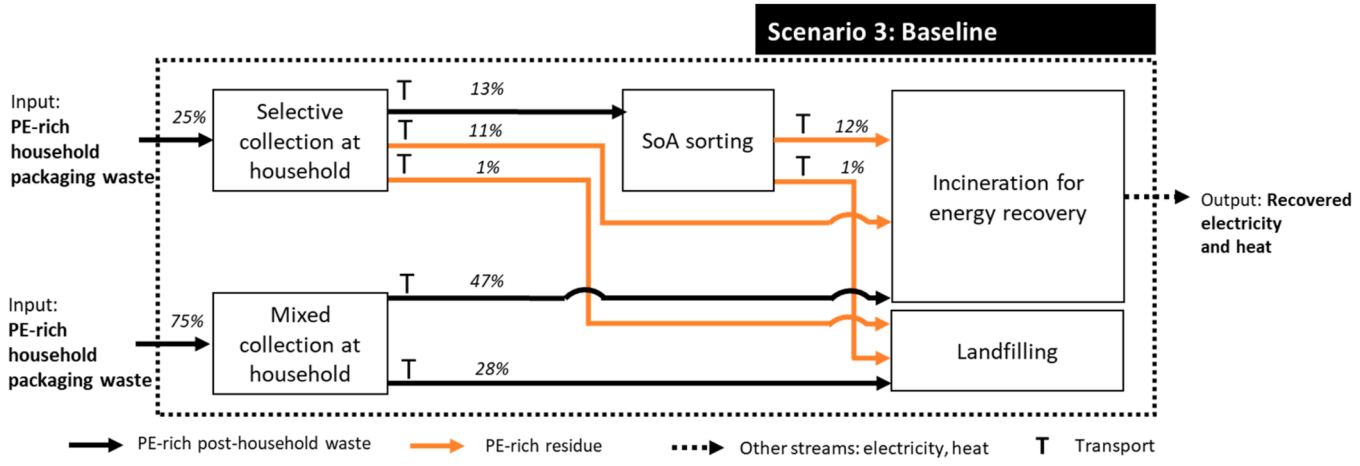

Fig. 2. Overview of the system boundaries for the S3: Baseline scenario. The Baseline scenario is based on European data (Eunomia and Plastic Recyclers Europe, 2020; Plastic Europe, 2022).

selective collection, where they are, however, sorted out as residues due to their MuMu structure (Eunomia and Plastic Recyclers Europe, 2020), of which 12 % are incinerated with energy recovery and 1 % goes to landfill (Plastic Europe, 2022). Additionally, 12 % of the selectively collected flexible films does not enter sorting (Eunomia and Plastic Recyclers Europe, 2020), with 11 % incinerated for energy recovery and 1 % sent to landfill (Plastic Europe, 2022). Regarding the 75 % mixed collection as municipal waste, a fraction of 47 % is incinerated with energy recovery and 28 % is sent to landfill (Plastic Europe, 2022). Overall, a fraction of 70 % is directed to incineration with energy recovery, while the remaining fraction of 30 % is sent to landfills (Eunomia and Plastic Recyclers Europe, 2020; Plastic Europe, 2022). The resulting outputs comprise the generation of electricity and heat from incineration of waste. The baseline scenario (S3) assumes no recycling for household flexible packaging focusing on MuMu, which is primarily used for energy recovery or fuel according to Eunomia and Plastic Recyclers Europe (2020).

2.1.2. Functional unit (FU)

This assessment adopts a waste/system perspective approach, as outlined by Jeswani et al. (2021). According to Tonini et al. (2021), the system perspective is suitable for conducting impact assessments and estimating potential savings resulting from changes in waste management systems. For studies involving multiple systems lacking direct comparability, a basket of products is useful as FU (Hauschild et al., 2018). This approach is used in waste management studies such as for plastic packaging waste, where different treatment scenarios produce varying amounts of different outputs, such as electricity from incineration and recycled material (Arena et al., 2003). In the basket of products approach, functional equivalence between the innovative and conventional/baseline systems is ensured by expanding each system by generating its respective alternative products or energy to achieve an equivalent overall output (Finnveden, 1999; Nakatani, 2014). In the context of this case study, the FU encompasses the treatment of one ton PE-rich household flexible packaging waste resulting in a basket of products, including (i.) F packaging, (ii.) NF packaging, and (iii.) energy (electricity and heat). System expansion accounts for additional NF or F packaging, and/or additional electricity and heat. For example, in the Baseline scenario, where only energy is recovered, system expansion includes additional F and NF packaging, produced entirely from virgin materials (0 % PE PCR). Table 1 represents the basket of products (S1-S3) in quantitative values. The energy values are attributed to the energy recovered from the incineration of residues generated throughout the value chain (S1 and S2) or the incineration of PE-rich FPP waste (S3).

The application for F packaging is coffee pouches, and for NF packaging, detergent pouches; for structure clarification, F/NF packaging refers to MoMu with PE PCR, while additional F/NF packaging refers to MuMu with only virgin materials.

Table 1
Functional unit – Basket of products (S1-S3), Note: values are rounded. When summing the products in the basket, (i.e., m^2, MWh, and GJ) the values are equal across the scenarios. For packaging the kg/m^2 vary depending on the structure.

Basket of products	(S1) NF	(S2) F and NF	(S3) Baseline
NF Packaging	12 476 m^2 (1305 kg)	9 299 m^2 (973 kg)	–
F Packaging	–	2 463 m^2 (307 kg)	–
Recovered Electricity	0.4 MWh	0.5 MWh	1.0 MWh
Recovered Heat	3.0 GJ	3.4 GJ	7.0 GJ
System Expansion: Additional NF Packaging	–	3 177 m^2 (442 kg)	12 476 m^2 (1735 kg)
System Expansion: Additional F Packaging	2 463 m^2 (238 kg)	–	2 463 m^2 (238 kg)
System Expansion: Additional Electricity	0.6 MWh	0.5 MWh	–
System Expansion: Additional Heat	4.0 GJ	3.6 GJ	–

2.2. Step 2: data inventory

The inventory combines data from industrial partners, semi-industrial and pilot-scale data from academic partners and research institutions, which have been validated (TRL 5) and/or demonstrated (TRL 6) in a industrially relevant environment (European Commission, 2015). Ink, tracer, and functional layer production are at TRL 5; TBS, delamination–deinking, selective dissolution, and deodorisation are at TRL 5–6; while pretreatment, recompounding, laminate conversion, and packing have reached TRL 6.

However, mass and energy data for the LCA, were obtained under varying conditions: full-scale industrial operation (e.g., pretreatment, laminate conversion, recompounding), semi-industrial scale for delamination and deinking, and pilot-scale for selective dissolution. The data quality was validated through multiple bilateral meetings. This resulted in data quality close to the maximum score, indicating strong representativeness and reliability. A detailed description of the data quality is provided in the Supplementary Information – Annex 3. Regarding LCA, primary data was collected per specified reference flow, e.g., per year, per ton of waste processed, or per m^2 of laminate produced, depending on the process. Based on the mass balance, the MFA represents the treatment of one ton of PE-rich household flexible packaging waste, resulting in the production of either NF packaging (S1) or a combination of F and NF packaging (S2).

Secondary sources, i.e., databases and literature, were primarily used to model the background processes, including the production of raw materials and chemicals used throughout the process, virgin plastics (PE and functional barriers for F laminate production), electricity, heat, fuel, and treatment of wastes. The main database utilized in this study is *ecoinvent v3.9*, using datasets which incorporate the impact of infrastructure. For packing, secondary data from *ecoinvent* were applied. For the details on the used datasets see Supplementary Information – Annex 1. Furthermore, the study assumed waste collection within the French market, with an average transport distance of 60 km based on data provided by the waste management company. All recycling processes were (prospectively) considered to take place at a French facility, with transport occurring between sorting, recycling, laminate production (Belgium), and final packaging production (Portugal). To align the data, particularly for delamination and deinking and selective dissolution, with industrial-scale operations, the data inventory was refined through consultations with industry experts. For instance, residue management was represented through incineration with energy recovery, while wastewater treatment was depicted through ultrafiltration for the delamination and deinking processes, combined with a closed-loop water reuse system. As the study focuses on European sites, site-specific datasets for electricity were incorporated (i.e., FR for waste management, BE for laminate production, GER for ink and tracer production and PT for product packing). Sensitivity analysis was performed to assess the effects of prospective 2050 EU electricity mixes. Furthermore, data on waste flows and treatment in the Baseline scenario (S3) was derived from Eunomia and Plastic Recyclers Europe (2020) and Plastic Europe (2022), reflecting mass balances for 2018 and 2020, respectively. The mass-balance and energy data for the system expansion of additional NF and F packaging laminate are derived from primary data, while the system expansion of heat and electricity are calculated from primary data.

Furthermore, cost data were made available for the LCC analysis by the project partners. In this study, cLCC is applied, considering only internal costs within the system boundary, while excluding external costs like health impacts from emissions (Bianchi et al., 2021). Regarding cLCC and following the cost-break down structure (CBS) outlined by Martinez-Sanchez et al. (2015), internal costs are

categorized into budget and transfer costs (e.g., taxes). Following Bachmann et al. (2024), the CBS is extended to include positive cash flows. Budget costs include Capital Expenditures (CAPEX), as well as Operating Expenditures (OPEX) (Bianchi et al., 2021) (cfr. Supplementary Information – Annex 1). CAPEX covers investments in land, buildings, equipment, installation, and project management, with cost data obtained from project partners, literature such as Bashirgonbadi et al. (2022) and Towler & Sinnott (2013), and online sources such as Focus Technology Co., Ltd. (2024). Land costs are not accounted for due the assumption of comparable land costs across the three scenarios as the required industrial land has already been purchased. OPEX consists of fixed costs (e.g., personnel, maintenance, insurance) and variable costs (e.g., utilities, raw materials, residual treatment), incorporating data from industrial partners and the Horizon 2020 MultiCycle project for cost data on laminate production. Transport costs are determined using EU-level collection cost data (Hestin et al., 2015), driver wages, and fuel prices (Economic Research Institute, 2025; Transport and Environment, 2015). Utility prices are based on EU average 2024 Eurostat data (Eurostat, 2024a, 2024b), reflecting recent fluctuations in energy costs across different regions. Transfer costs, such as corporate taxes calculated at 25 % of net profit (PWC, 2024) are included.

The cost structure is in line with the basket of products approach, ensuring that the costs of system expansion products, such as heat, electricity, and virgin packaging, are accounted for, in each of the three scenarios. For system expansion, production costs are estimated using literature data on heat and electricity costs from waste-to-energy processes, based on data from France (ADEME et al., 2023). The Chemical Engineering Plant Cost Index (CEPCI) was applied with a factor of 1.34 to adjust costs to 2024 values (Maxwell, 2025). It is assumed that the heat and electricity cost data excludes transfer costs, given that it is derived from the literature (ADEME et al., 2023), with limited cost structure discloser. Additionally, the production costs of virgin non-recyclable MuMu packaging are incorporated based on project information (industrial partners, literature and MultiCycle). Due to the sensitivity of profit margins and pricing information, positive cash flows are considered but are further processed during the impact assessment to ensure confidentiality (cfr. subsection 2.3), while accurately reflecting revenue differences across the scenarios. Hence, in addition to conventional cost considerations, this study introduces the Green Premium Adjustment Cost (GPAC), following the concept of Green Premium (Köveker et al., 2023). GPAC captures the incremental value of environmentally friendly products (e.g., packaging including PCR) compared to their virgin counterparts (e.g., 100 % virgin packaging). In this study, industry indicates that the newly introduced MoMu F and NF packaging including PCR commands a higher market price than conventional virgin MuMu F and NF packaging. Introducing GPAC is justified by the iterative approach, suggesting that, in the unavailability/ confidentiality of certain data, it may be necessary to redefine the goal and scope of the study (Bachmann et al., 2024), which leads in this case to the further processing of sensitive data. The LCI for both LCA and LCC can be found in the Supplementary information (Annex 1).

2.3. Step 3: impact assessment

2.3.1. LCA

The impact assessment was performed in Microsoft Excel, using the *ecoinvent* v3.9 life cycle inventory (LCI) database and analyzed by the Environmental Footprint (EF) 3.1 impact assessment method (European Commission. Joint Research Centre., 2019). For the assessment, sixteen impact categories in accordance with the EF method are available. Based on the draft PEFCR for flexible packaging (Quantis and Amcor, 2022) and literature, this study focuses on the analysis of the five most relevant impact categories for flexible packaging, including fossil resource use, mineral and metal resource use, climate change, freshwater ecotoxicity, and particulate matter, which typically account for 80 % of the total impact of the flexible packaging manufacturing (Quantis and Amcor, 2022). This relative contribution estimation involves a normalization and weighting process (Supplementary Information – Annex 2). In addition to the five aforementioned impact categories, acidification is also discussed in this study due to its relevance to the EoL treatment of plastic packaging waste (Civancik-Uslu et al., 2021; Huysveld et al., 2022; Lazarevic et al., 2010).

2.3.2. cLCC

It should be noted that, although conventional LCC does not formally include an impact assessment phase, this study follows the approach proposed by Bachmann et al. (2024) to ensure methodological consistency with the LCSA framework and the structure of LCA based on ISO 14040 (2006); ISO 14044 (2006). The cLCC formula presented in the study of Martinez-Sanchez et al. (2015) is adapted and applied in this study (Eq. 1), aligning with the cost structure outlined (cfr. Supplementary Information – Annex 1). It calculates the total costs as the sum of budget costs (UBC_i) and transfer cost (UT_i) per unit for all activities in the foreground system of the scenarios, scaled based on input quantities (W_i) (Bianchi et al., 2021; Martinez-Sanchez et al., 2015). To account for the basket of products, the sum of UBC_S and UT_S is applied for the system expansion (s), incorporating additional NF packaging, F packaging, electricity, and heat per scenario to maintain equivalent functionality.

To ensure equitable revenue consideration across all scenarios, GPAC is introduced as an incremental cost, calculated as the price difference between the innovative and conventional product (€/m^2) (Köveker et al., 2023). This cost is integrated into each scenario as an additional system expansion cost, complementing the system expansion costs associated with virgin NF or F packaging production (UBC_S and UT_S). Each system expansion component is then scaled according to its respective input quantity (I_s) (Table 1). This approach ensures that economic value is fairly accounted for in all scenarios, particularly by reflecting the higher market value of the innovative laminate including PCR. However, this adjustment is applied exclusively to packaging costs, as electricity and heat prices remain consistent across all three scenarios, eliminating the need for further adjustments.

$$cLLC = \sum_{i=1}^{n} [W_i \times (UBC_i + UT_i)] + [I_s \times (UBC_s + UT_s + GPAC_s)] \quad (1)$$

Cost formula applied in the cLCC for circular FPP costs

i: Foreground unit process involved in this study (cfr. Figs. 1 and 2)
s: System expansion process involved in this study
W_i: Input quantity to unit process *i*, can be waste or other intermediate products.
UBC_i: Unit budget cost of unit process *i* per waste unit (e.g., ton).
UT_i: Unit transfer cost of unit process *i* per waste unit (e.g., ton).
UBC_s: Unit budget cost of unit process *s* per unit (e.g., m^2).
UT_s: Unit transfer cost of system expansion unit process *s* per unit (e.g., m^2).
I_s: Input quantity of system expansion unit process *s*.
$GPAC_s$: Green premium adjustment cost of virgin packaging input *s* per product unit (e.g., m^2).

Moreover, following the recommendation of Bachmann et al. (2024), no discounting is applied. Furthermore, CAPEX is depreciated to achieve a cost per FU. For buildings, depreciation time is set to 20 years, while equipment investment is depreciated over a 7-year period, based on information provided by project partners. The results in subsection 3.3 are presented both with and without UT and GPAC to enhance the visualization of process contributions.

2.4. Step 4: interpretation

In this study, the overall results of the three scenarios (S1-S3) are compared, with a focus on the primary contributions to provide insights

into potential areas for process improvements. The interpretation also considers the quality of the data and assumptions made throughout the assessment. Therefore, scenario analyses (what-if analysis) are performed to assess the effects of different choices (Thomassen et al., 2019). Furthermore, a sensitivity analysis is recommended to evaluate the impact of associated uncertainties (Horn et al., 2021; ISO 14040, 2006), and data quality is evaluated using a pedigree matrix (Bachmann et al., 2024; Ciroth, 2009).

The scenario analyses evaluate the environmental impact of electricity in the foreground system and system expansion of S1-S3 based on the anticipated 2050 electricity mix for the EU (European Commission. Directorate General for Energy, 2021), to address the influence of the prospective energy transition. Additional information is provided in the Supplementary Information (Annex 3). The sensitivity analyses examine the economic impact of variations in energy and virgin PE costs. Regional variations in energy costs (Eurostat, 2024a, 2024b) necessitate a detailed regional feasibility assessment before implementation (Bashirgonbadi et al., 2022). Furthermore, periodic market fluctuations in virgin PE prices (European Environmental Agency, 2025) introduce a certain element of uncertainty. To account for these uncertainties, a sensitivity analysis was performed by adjusting the prices of virgin PE and utility costs (electricity and heat) by ±25 %, following the approach of Bashirgonbadi et al. (2022). These sensitivity analyses in subsection 3.4 focus on the UBC (multiplied by the input quantities $Wi/\ I_S$), due to the primary dependence of cost changes on the UBC, and for better visualization. Additionally, the assessment of data quality is provided in the Supplementary Information (Annex 4).

3. Results

The following subsections 3.1 to 3.4 present and discuss the LCI outcome (MFA), LCA and cLCC (undiscounted) results, followed by the scenario and sensitivity analyses.

3.1. LCI outcome (MFA)

Fig. 3 illustrates the MFA for S1 and S2, respectively, both starting with 1000 kg of PE-rich household flexible packaging waste, comprising both F and NF applications. This 1000 kg represents a fraction of 57.8 tons of selectively collected waste; however, for clarity and alignment with the FU, the analysis is based on this set quantity. Residual fractions from the sorting process, such as PET, aluminum, and paper, are excluded from the MFA, as they are directed toward separate recycling streams and are therefore considered beyond the scope of this study. In S1, the production of 1305 kg of NF packaging requires the addition of 631 kg of virgin PE. The residue rates are 8 % (79 kg) for pretreatment and 10 % (91 kg) for purification and post-treatment, with recompounding included in the purification stage. These rates are based on mass balances provided by industry and research institutes. Additionally, residue rates of 10 % (153 kg) for laminate production and 5 % (69

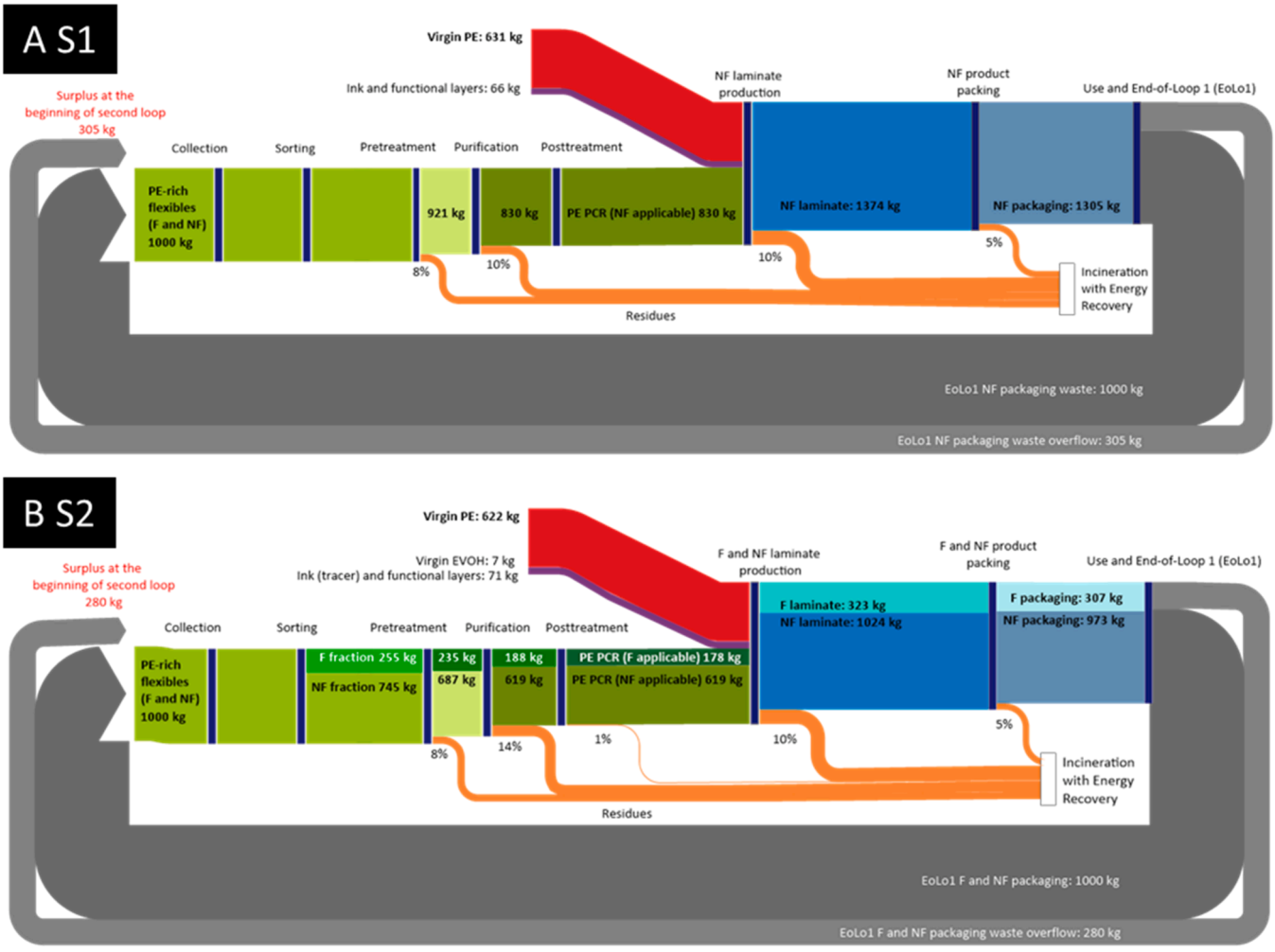

Fig. 3. MFA of S1 (A) and S2 (B), illustrating the treatment of one ton of PE-rich household flexible packaging waste, leading to the production of either NF packaging (S1) or a combination of F and NF packaging (S2). For S1, residue losses are distributed as follows: 8 % (79 kg), 10 % (91 kg), 10 % (153 kg), and 5 % (69 kg). For S2, the corresponding losses are: 8 % (78 kg), 14 % (115 kg), 1 % (10 kg), 10 % (150 kg), and 5 % (67 kg).

kg) for final packaging are derived from expert estimates within the industry. At the End-of-Loop 1 (EoLo1), S1 generates a surplus of 305 kg of PE-based NF flexible plastic packaging waste. S2 introduces the separation of F and NF fractions using TBS, with tracers embedded in the F fraction. This yields 178 kg of PE PCR applicable for F packaging and 619 kg of PE PCR applicable for NF packaging. Due to the higher incorporation of PCR—54 % in F packaging compared to 50 % in NF packaging—the need for virgin PE is reduced to 622 kg (compared to S1). This results in the production of 307 kg of F packaging and 973 kg of NF packaging. Residue rates are broadly comparable with S1, with 8 % (78 kg) for pretreatment, 10 % (150 kg) for laminate production and 5 % (67 kg) for final packaging. The exception is 14 % (115 kg) for purification, which is higher than in S1 due to the additional delamination and deinking steps required for the F fraction, which achieve an 80 % yield compared to the 90 % yield for the NF fraction. Posttreatment residues are 1 % (10 kg), regarding recompounding included in the purification stage of the F fraction. At the EoLo1, S2 generates a surplus of 280 kg of PE-based F and NF flexible plastic packaging waste. A key loss in both systems occurs during laminate production, specifically in the blown film coextrusion of PCR/PE film (see figure S7 and S8 in Supplementary Information – Annex 1), amounting to 113 kg in S1 and 102 kg in S2. Further details on the LCI outcomes for energy demand (electricity and heat) per process step are provided in the Supplementary Information (Annex 4).

Addressing the demand for comprehensive, evidence-based quantitative information, the combination of MFA and LCA offers potential for evaluating the environmental impacts of circular economy strategies (Barkhausen et al., 2023).

3.2. LCA outcome

The LCA results (Fig. 4) are first discussed in detail with respect to the climate change impacts. In addition, the impact categories fossil resource use, minerals and metals resource use, freshwater ecotoxicity, particulate matter and acidification are presented and discussed.

The LCA results indicate that the *climate change* impact can be reduced by 33 % (Fig. 4) when one ton of PE-rich flexible packaging waste from households is recycled via selective dissolution to produce NF packaging (with 54 % PE PCR in a MoMu duplex structure, S1), instead of being incinerated with energy recovery and landfilled (S3). A higher benefit (41 %) can be seen when shifting from the baseline S3 to S2, in which F and NF flexibles are separated and then purified by delamination/deinking and selective dissolution, respectively. The outputs of S2 include F packaging laminate (with 50 % PE PCR in a MoMu triplex structure) and NF packaging laminate (with 54 % PE PCR in a MoMu duplex structure), respectively. The reduction in impact when comparing S2 to S1 can mainly be explained by two reasons. First, energy use of delamination and deinking processes is around 4 times

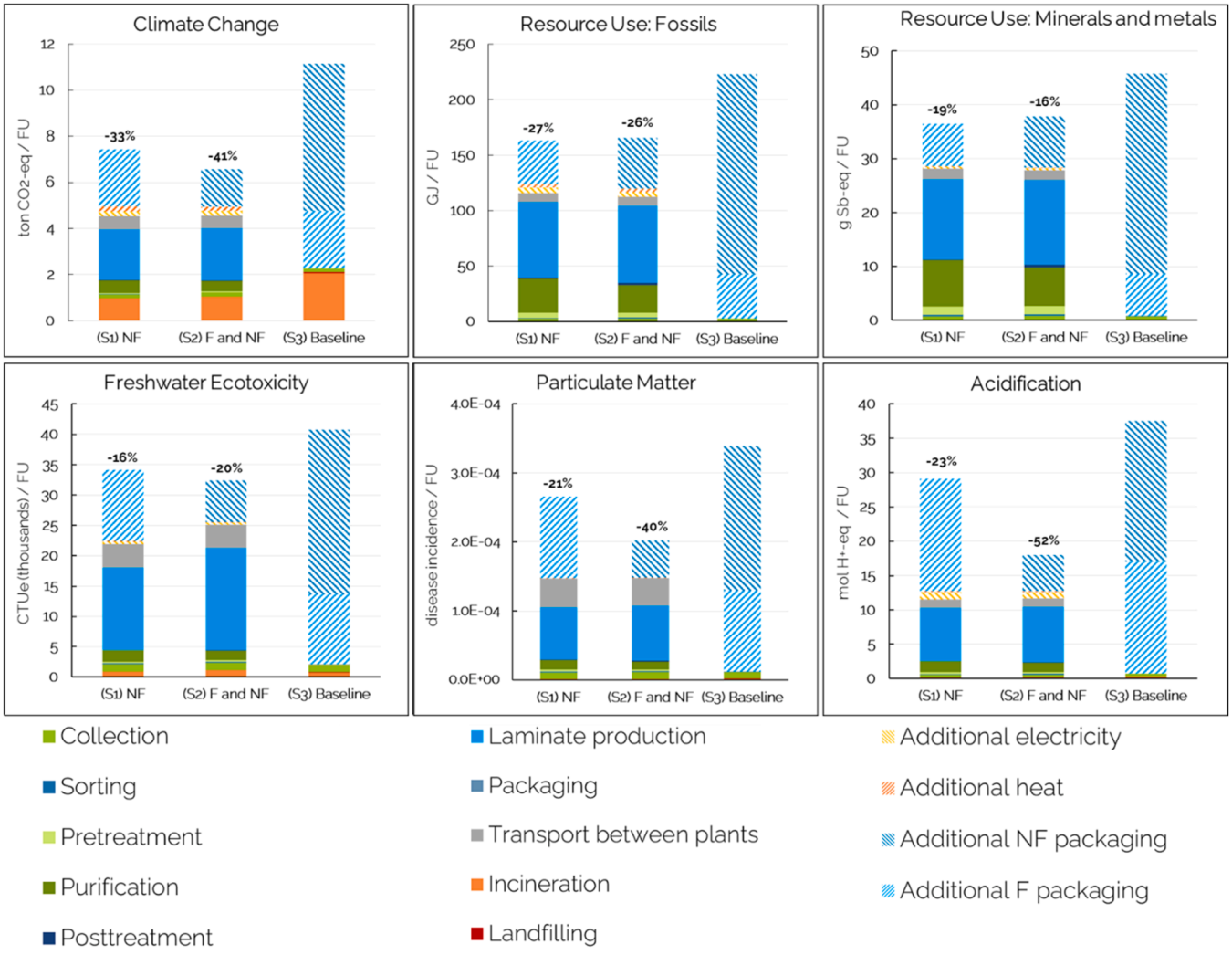

Fig. 4. Environmental impacts (S1-S3) across the six key impact categories climate change, fossil resource use, minerals and metals resource use, freshwater ecotoxicity, particulate matter and acidification; FU: treatment of one ton PE-rich household flexible packaging waste resulting in a basket of product including (i.) F packaging, (ii.) NF packaging, and (iii.) energy (electricity and heat).

lower compared to the selective dissolution process (cfr. Supplementary Information -Annex 4). Second, the additional NF packaging in S2 has a lower carbon footprint than the additional F packaging in S1 due to differences in their MuMu structure primarily driven by the presence of an aluminum layer. The MuMu NF packaging has a duplex structure without aluminum, whereas the MuMu F packaging consists of a triplex structure incorporating an aluminum layer.

Moreover, the primary contributions to the climate change impact of the innovative S1 and S2 are (i.) the laminate production (ranging from 30–35 % for MoMu and 25–33 % for additional MuMu NF and F packaging), particularly due to the remaining requirement of virgin material, which implies that the loop cannot be fully closed; (ii.) the incineration of residues (S1 13 % and S2 16 %), which indicates material losses (1–14 % per process step; cfr. subsection 3.1); (iii.) the purification with selective dissolution or delamination/deinking (S1 8 % and S2 7 %), influenced by energy demands; and (iv.) transport between plants (S1 7 % and S2 8 %) driven by transnational freight. An important consideration in interpreting the results is that the potential further upscaling of the innovative production processes could impact results, particularly concerning energy consumption (Thonemann et al., 2020), as larger-scale operations may enhance efficiency and reduce environmental impacts (Wang et al., 2024). Further research on the key factors driving the environmental impact of these processes is hence recommended to improve the process performance and energy efficiency. Moreover, it should be noted that experiments on selective dissolution, executed after fixing the scenarios in this research, have demonstrated that the purification efficiency of selective dissolution is sufficient to produce PE PCR also suitable for F packaging applications.

Across the other impact categories, S1 demonstrates a reduction of 16–27 % compared to S3, while S2 achieves a reduction of 16–52 %. S1 performs best in the categories of *fossil resource use* and *mineral and metal resource use*. The impact of the additional (MuMu) F packaging in S1 is lower in both categories compared to the additional (MuMu) NF packaging in S2, primarily due to differences in input mass (cfr. Table 1). Electricity consumption contributes considerably (e.g., in S1 33 % and 56 %, respectively) in the two categories, driven by the use of non-renewable energy sources (e.g., uranium) and copper in the distribution network, respectively. In contrast, S2 shows superior performance in *freshwater ecotoxicity, particulate matter* and *acidification*. The latter three impact categories exhibit a similar trend to climate change, driven by the presence of an aluminum layer in the additional (MuMu) F packaging. In comparison, the additional (MuMu) NF packaging demonstrates lower impacts per m^2 of laminate, with *freshwater ecotoxicity* reduced by a factor of two, *particulate matter* by a factor of three, and *acidification* by a factor of four. In general, the contribution of sorting and, for example, TBS in S2 is minimal, with TBS accounting for <1 % of the total across all impact categories.

In conclusion, the reduction in environmental impact across the six key categories is mainly driven by the transition in packaging design, moving from non-recyclable MuMu (composed of virgin PE, PET, and/or aluminum in this case study) to recyclable MoMu (PE-based, with a triplex structure containing 50 % PE PCR for F packaging and a duplex structure with 54 % PE PCR for NF packaging). As a result, a considerable portion of the studied environmental burdens associated with the linear system is effectively mitigated. However, it is relevant to point out that, though the PET-PE and/or aluminum based structure is the main driver for non-recyclable MuMu laminates, numerous other laminate structures remain non-recyclable due to their complex multi-material composition or other design constraints. Further research comparing the innovative MoMu food packaging including PCR with alternative laminate structures, such as virgin MoMu packaging, could be essential as virgin MuMu is gradually phased out, driven by design-for-recycling incentivized by legislation, such as PPWR (European Commission, 2022a; European Parliament, 2024). Moreover, it would be interesting to assess how these design-for-recycling measures perform in terms of environmental impacts across multiple recycling cycles. Furthermore, comparing innovative MoMu packaging with varying PE PCR content could provide valuable insights. Specifically, reducing the PE PCR content in F packaging could help decrease laminate thickness, addressing processability challenges (e.g. sealing) associated with thicker structures, based on project learnings. Moreover, the reliance on virgin material highlights that a fully closed-loop system cannot yet be achieved due to two primary factors: (i.) material losses occurring throughout the recycling process and (ii.) the restricted incorporation of PCR in laminates, driven by technical limitations and legislative constraints.

3.3. cLCC (undiscounted) outcome

The cost interpretation is conducted in two ways: first, by considering only UBC (excluding UT and GPAC), and second, by incorporating the UT and GPAC, represented in Fig. 5 a and b, respectively.

Excluding the UT and GPAC (Fig. 5a), the cLCC (undiscounted) results indicate a 26 % cost reduction for S2 compared to S3. In comparison, transitioning from S3 to S2 results in a lower cost reduction of 7 %. The total costs per FU, excluding UT and GPAC, amount to €4407 for S1, €5540 for S2, and €5975 for scenario S3. A detailed analysis considering UBC (Supplementary Information Annex 4) indicates that laminate production is the main cost driver in S1 (€1809) and S2 (€2491), whereas the additional NF packaging (MuMu structure) is the dominant contributor in S3 (€4892), next to additional F packaging (€ 745). Incineration and landfill expenses remain minimal at €84 and €48, respectively. The laminate cost is driven by OPEX costs i.e., virgin material costs such as virgin PE (S1 and S2) and innovative ink (S2). The cost of the ink is anticipated to decrease as batch sizes increase.

Including the UT and GPAC (Fig. 5b) increases cost reductions, showing a 44 % cost decrease in S1 compared to S3, while transitioning to S2 leads to a 49 % reduction. The increased cost advantage of S2 over S1 is due to the higher GPAC of the additional F packaging (MuMu structure) in S1 compared to the GPAC of the additional NF packaging (MuMu structure) in S2. The total costs per FU are €20,197 for scenario S1, €18,287 for scenario S2, and €35,756 for scenario S3. The GPAC is the primary contributing factor overall.

It is important to emphasize that while the products within the basket share the same functionality, they differ economically due to varying market prices and potential revenues. By incorporating the green premium concept into the basket-of-products approach, the study ensures a more comprehensive economic assessment and is consistent with the LCA approach. The innovative MoMu product generates higher economic value, which is reflected in the GPAC. Hence, failing to recycle PE-rich MuMu flexible packaging waste would lead to a loss of potential revenue from innovative MoMu packaging including PE PCR. When the GPAC of these advanced laminates is considered, overall system costs rise, further diminishing the economic viability of the baseline compared to the innovative recycling alternatives.

Furthermore, given the low costs associated with incineration and landfilling, it is recommended to assess and potentially expand landfill bans and/or increase incineration and landfill taxes e.g., specifically on plastic waste, on the European level. For example, in Belgium, a tax of €15 (indexed) per ton of general waste is applied to incineration. In France, an increase in the Taxe Générale sur les Activités Polluantes (TGAP) for incineration with energy recovery facilities is scheduled for 2025, with rates between €7.5 and €25 per ton of waste. This adjustment aims to enhance the competitiveness of reuse and material recovery (European Environmental Agency, 2023). For instance, De Weerdt et al. (2020) indicated that implementing a tax on plastic waste incineration can reduce plastic waste generation, in industrial settings. Furthermore, access to subsidies and innovation funding constitutes a critical enabler for small and medium-sized enterprises in developing and scaling such technologies.

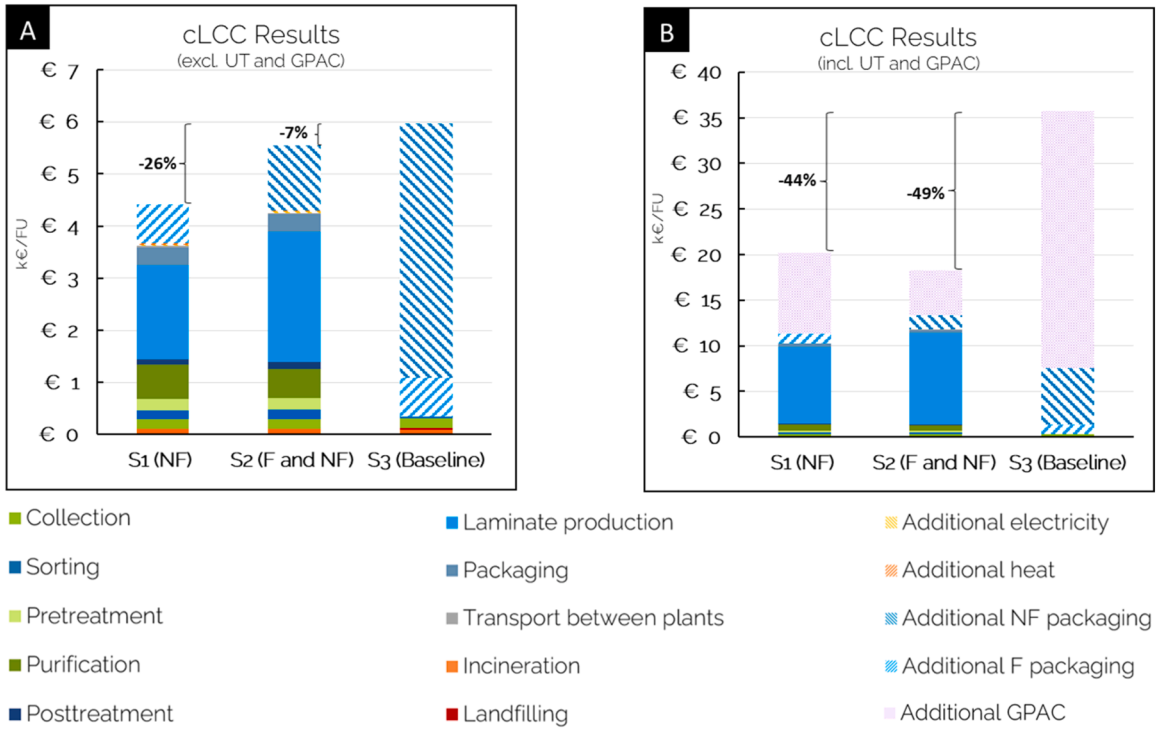

Fig. 5. cLCC (undiscounted) results (S1-S3), Figure A excludes the Green Premium Adjustment Costs (GPAC) and unit transfer costs (UT), while Figure B includes them. FU: treatment of one ton PE-rich household flexible packaging waste resulting in a basket of products including (i.) F packaging, (ii.) NF packaging, and (iii.) energy (electricity and heat).

3.4. Scenario and sensitivity analyses

The scenario analysis results are first discussed in relation to the LCA, followed by the sensitivity analysis of the cLCC (undiscounted) results. Detailed figures illustrating the analyses are available in the Supplementary Information (Annex 4). The scenario analysis in terms of LCA results is based on the impact of changes to the foreground electricity modeling, such as prospective European electricity in 2050 (country weighted-averages). This approach is motivated by the notable contribution of electricity to energy-related impact categories (see subsection 3.1). The variation in impact across the different categories ranges from -18 to +3 %. Overall, the most considerable improvement due to shifts in the electricity mix is observed in the impact category *fossil resource use*, with reductions of -18 % (S1) and -16 % (S2) compared to the present scenarios using site-specific electricity inputs. The category *fossil resource use* accounts for nuclear energy sources, in addition to fossil fuels. The majority of energy consumption occurs in France and Belgium. France currently has a high dependency on non-renewable energy, primarily due to its reliance on nuclear power (around 70 %) based on the *ecoinvent* database. However, with the projected decrease to 11 % nuclear in the EU 2050 electricity mix, the *fossil resource use* impact decreases. A slightly increase in impact is observed for *mineral and metal resource use* (+3 % in S1, +2 % in S2) and *freshwater ecotoxicity* (+1 % in both S1 and S2). The increase in both indicators is mainly driven by copper demand at the low-voltage level, largely due to its use in the distribution network. By 2050, copper demand is expected to increase due to the growing share of photovoltaic (PV) systems, which require copper for inverters and installation. In the EU country average for 2050, PV contribute 18 % to the energy mix, compared to current contributions of e.g., 8 % in Belgium and 4 % in France based on the *ecoinvent* database.

For *climate change*, the overall impact slightly decreases (-0.4 % in S1, -2 % in S2). The French electricity mix (0.09 kg CO_2-eq per kWh) used in the initial results performs better than the projected EU 2050 mix (0.17 kg CO_2-eq per kWh) due to a higher share of nuclear power and a lower reliance on natural gas (only 6 % in the French mix, based on the *ecoinvent* database). In contrast, the EU 2050 mix has a lower impact than Germany's (0.44 kg CO_2-eq per kWh) and Belgium's electricity mix (0.21 kg CO_2-eq per kWh) used in the initial results. This is driven by coal use for Germany and blast furnace gas treatment for Belgium. In contrast, the EU mix is primarily influenced by natural gas and petroleum, with only 1 % coal and no blast furnace gas treatment assumed (cfr. Supplement Information Annex 3). These compensatory effects are particularly visible in S1, where no considerable change is observed (-0.4 %). The same trend can be observed regarding *particulate matter* (-0.05 % in S1, 0.04 % in S2) and *acidification* (-1 % in S1 and S2). The overall conclusion remains the same, indicating that the innovative scenarios are more environmentally friendly. However, trade-offs between impact categories need to be carefully assessed.

The sensitivity analysis of the cLCC (undiscounted) results, focusing on the UBC (cfr. subsection 2.4), is conducted by adjusting the prices of virgin PE and utility costs (electricity and heat) by ±25 %, as outlined by Bashirgonbadi et al. (2022). The UBC change ranges from €910 for the S3 Baseline in the virgin PE sensitivity scenario to €23 in the Baseline for the heat cost sensitivity scenario. In particular, a 25 % decrease in the virgin PE price leads to the Baseline scenario having lower UBC than S2, though the difference is negligible (0.2 %). The electricity price change has a more pronounced effect on S1 (± €163) and S2 (± €159) than on the Baseline (± €77), likely due to the electricity consumption in the recycling value chain, while the heat price change has the least impact (± €23–54).

4. Discussion

The following subsections 4.1–4.3 provide further discussion, including the study's strengths and limitations, methodological considerations, and a comparison of the results with other case studies.

4.1. Strengths and limitations of the study

This study demonstrates novelty by utilizing primary data from innovative processes. Furthermore, it contributes to methodological advancement by combining LCA and LCC, applying the basket of products approach in both assessments, and incorporating the Green Premium concept within the LCC framework. The primary limitation of this study lies in data availability, which affects both data accuracy and the associated uncertainty. For instance, in the LCA of product packaging, the data inventory is based on an *ecoinvent* dataset that, according to expert estimates, are considered representative of the system. In particular, the sensitivity of revenue data, which cannot be disclosed publicly due to confidentiality concerns, limits the replicability of the LCC study when including the GPAC and UT, as these are based on this data point. Accordingly, the LCC analysis is carried out in two steps: first, by considering only the UBC (i.e., CAPEX and OPEX), and second, by incorporating UT (e.g., taxes) and GPAC. Additionally, data on F product packing production costs was unavailable. To address this gap, proxy data from NF product packing equipment costs was used. Other product packing cost components were derived from literature or based on assumptions, such as personnel costs being equivalent to those in laminate production, as detailed in the Supplementary Information (Annex 1). Moreover, UT is excluded from additional electricity and heat calculations, as the data is derived from the literature (ADEME et al., 2023) and net profit information is unavailable. However, as this represents a limited cost component, its overall influence on the results is assumed to be minimal. Additionally, as discussed by von Krogh et al. (2001), this study does not consider the potential for recycled plastic packaging waste to replace virgin material across multiple life cycles. Therefore, the potential impact of polymer degradation across multiple cycles remains open, due to data unavailability. Nevertheless, the continuous addition of virgin PE (around 50 %) (cfr. subsection 3.1) might mitigate degradation. According to default values for quality by Garcia-Gutierrez et al. (2023) recycled LDPE replaces 50 % of virgin LDPE functionality. It remains to be examined how the applied design-for-recycling measures (shift from MuMu to MoMu, solvent-free adhesive, delamination primer), together with the application of advanced sorting to reduce input waste heterogeneity, affect this value across multiple recycling cycles. Furthermore, tracers are not recycled, as they are incorporated into the ink, present at ppm-level concentrations hence being removed during the deinking process. Nevertheless potential carryover effects over multiple cycles require further investigation. Nonetheless, the tracers comply with European food-contact regulations and are Food and Drug Administration (FDA) approved (Polysecure, 2025).

4.2. Methodological discussions

From a methodological perspective, it is important to note that some impact categories in LCA are more advanced than others. The characterization reference models for *climate change* and *particulate matter* are well established, achieving a Level I rating, which indicates that they are recommended and satisfactory. In contrast, the model for *acidification* is rated Level II, suggesting it is recommended but requires some improvements. Meanwhile, the models for *freshwater ecotoxicity, fossil resource use*, and *mineral and metal* resource use are classified as Level III, indicating that they are recommended but should be applied with caution (Fazio et al., 2018). This should be taken into account when interpreting the overall LCA results.

Another key discussion point is that the applied methodology influences the analysis, addressing fundamental context-specific challenges in LCC. These challenges include: (i.) determining which types of costs should be assessed (e.g., private vs. social costs), (ii.) identifying the stakeholders for whom these costs are relevant (e.g., facility operators, households, public entities, or entire systems), and (iii.) defining the appropriate CBS (Martinez-Sanchez et al., 2015). In line with the guidance of Bachmann et al. (2024), the type of LCC applied, the cost boundary, and the stakeholder perspective is clarified in the following. This study applied cLCC. Both costs and positive cash flows (i.e., product revenue) are considered, with the latter represented through the GPAC. GPAC ensures equitable revenue consideration across all scenarios, as it is incorporated as an incremental cost within each scenario (represented as additive system expansion). In this way, the higher revenues of the innovative products compared to the virgin ones are appropriately accounted for, while being consistent with the basket of product approach using additive system expansion. The differences between this approach and the opposite approach, substitution, i.e., subtractive system expansion, and their effects on LCA modelling are described in more detail by Sanabria Garcia et al. (2025)

Furthermore, the assessment focuses solely on private economic effects, excluding environmental and social externalities, and leverages confidential data (i.e., product revenues) in a processed form to derive the GPAC (see subsection 2.3), ensuring fair comparison while respecting data sensitivity. However, environmental LCC (eLCC) or social LCC (sLCC) (Hunkeler et al., 2008) could be applied, incorporating externalities such as environmental costs (e.g., CO_2 emissions) and social costs (Bianchi et al., 2021). Moreover, the economic assessment is conducted from a waste management perspective. In this context, municipal waste management is treated as a single economic actor/-stakeholder, with the LCC accounting for all costs across the extended system (Carlsson Reich, 2005). Additionally, integrating a basket-of-products approach in LCC may introduce challenges typically associated with LCA, due to differences between the two methodologies. However, to ensure comparability across scenarios, maintain functional equivalence, and enable future integration within LCSA, the basket-of-products approach was retained. Moreover, no integration, aggregation, or weighting of LCA and LCC results was performed (Isasa et al., 2024), which could be explored in future research.

Furthermore, integrating the three pillars of sustainability would be valuable. The LCA and LCC results presented in this article complement Haslinger et al. (under revision) about S-LCA on the same case of flexible circular packaging. Insights from the complementary S-LCA by Haslinger et al., (under revision) reveal no instances of low or very low social performance across the innovative value chain and an equal performance of the innovative scenarios. An opportunity for improvement lies in enhancing data availability on the supply chain among value chain actors, supporting compliance with future legislation such as the Corporate Sustainability Due Diligence Directive (European Parliament and Council of EU, 2024). Combining these three assessments would provide a more comprehensive LCSA perspective.

Finally, it has to be highlighted that the input of PE-rich flexible packaging waste is smaller (in terms of surface area (m^2) or the number of packaging items) than the output of packaging, because the produced innovative packaging is not composed entirely of PE PCR; virgin materials are added and hence the PE PCR is distributed over more packaging items than the previous cycle. This modelling approach is suitable as long as the collection and sorting efficiency of this type of packaging remains low. In 2020, approximately 40 % of the films placed on the market were estimated to be collected for recycling. However, nearly all multilayer films were excluded from recycling due to design limitations, ultimately becoming part of the reject fractions that are typically incinerated (Plastics Recyclers Europe, 2023). This means that not all of the FPP that is produced ends up in recycling, as it can currently be seen that the multilayer flexible packaging waste does not enter the recycling value chain because major losses already occur during collection and sorting (detailed in subsection 2.1). However, if collection and sorting rates increase, the amount of packaging produced could exceed the amount of packaging waste generated in the previous cycle (if we assume packaging design with similar PE PCR shares), although losses occur during recycling. As the packaging markets cannot expand indefinitely, the surplus recyclates must be redirected to other markets besides FPP, including rigid food and non-food packaging and other

markets, which will affect the substitution of virgin materials and, hence, the sustainability effects. Recycling makes sense if there is sufficient market demand for recycled materials.

4.3. Comparison of this study's results with other case studies

Despite increasing research efforts, a detailed LCA and LCC of circular FPP particularly for F application and closed-loop recycling remains unique. To emphasize the novelty of this study, the following subsection presents a comparison with relevant studies in the field of (flexible) plastic packaging waste treatment and FPP production. In general, looking from a waste-management perspective, studies in this field do not extend beyond waste management and/or PCR such as Tonini et al. (2021), excluding the conversion into laminate and the product packaging stages (production phase). Furthermore, Huysman et al. (2017) compared various post-industrial plastic waste treatment scenarios, including closed-loop, semi-closed-loop, open-loop recycling, and incineration with energy recovery, but not for post-consumer plastic. Moreover, other studies primarily focus on conventional mechanical recycling, without considering advanced mechanical or physical recycling techniques that aim for closed loops, as explored in the current study. Lazarevic et al. (2010) provided a review of LCA studies on plastic waste management within the European recycling framework, particularly for household plastic packaging waste (Carlsson, 2002; Frees, 2002; Perugini et al., 2005; Raadal et al., 2008; Shonfield, 2008; von Krogh et al., 2001). Their findings indicate that recycling is generally preferred over incineration with energy recovery when assuming a 1:1 substitution ratio in energy-related impact categories. However, this advantage diminishes with lower substitution ratios or high organic contamination. Using the same substitution ratio, Civancik-Uslu et al. (2021) compared the environmental impacts and benefits of incineration with energy recovery, thermochemical recycling, and conventional mechanical recycling. The study examined four household plastic waste fractions (rigids and flexibles) and demonstrated that recycling outperforms incineration with energy recovery for resource consumption, global warming and acidification. Huysveld et al. (2022) focused on the same waste fractions as the former study, mainly to address lower substitution ratios by integrating technical and market substitutability for conventional mechanical recycling. In contrast to Civancik-Uslu et al. (2021), the study showed that conventional mechanical recycling is not always environmentally favorable over thermochemical recycling.

Furthermore, Kusch et al. (2021) examined the treatment of lightweight packaging in Germany using TBS technology, including mechanical recycling for well-sorted plastic fractions, compared to primary production. The study concluded that integrating TBS reduces climate change impact while producing high-quality recyclates (Kusch et al., 2021). These findings align with the current study, which also identifies TBS sorting with comparatively low environmental impact in the analysis. Moreover, Horodytska et al. (2020) analyzed the deinking process for treating post-industrial printed film waste. The study evaluated the impact of climate change, resources, ecosystem quality and human health for upcycling (including deinking), downcycling, and incineration by considering varying substitution ratios of recycled products. Upcycling performed better regarding climate change and resources considering market shares, increased substitution rates (0.4–0.8) and incineration substituting renewable energy sources (i.e., biogas and photovoltaic). Furthermore, Garcia-Gutierrez et al. (2023) analyzed the environmental and economic impacts of waste-treatment options for various plastic input-waste fractions such as treating large-format PE film waste using physical recycling via selective dissolution. Their study, which also stops at the PCR level, found that physical recycling offers the highest net savings for climate change compared to conventional mechanical recycling and incineration. In economic terms, incineration showed net savings due to higher revenues than costs, whereas recycling incurred net costs, as material recovery revenues did not fully offset expenses (Garcia-Gutierrez et al., 2023). The finding that incineration entails low costs aligns with this study.

Looking from a product perspective, the studies by Nhu et al. (2025) and Tunçok-Çeşme et al. (2024) on FPP, demonstrates that the integration of recycled content enhances environmental performance compared to completely virgin multi-material alternatives, supporting the findings of this study. However, Tunçok-Çeşme et al. (2024) analysis focused on metallized structures with recycled materials in MuMu (PET/PE) and MoMu (PP/PP) laminates without specifying the applications, rather than recyclable PE-based MoMu laminates with PCR applicable for F and NF packaging. Moreover, the origin of waste input, whether from consumer or industrial, was not clearly specified. Nhu et al. (2025), on the other hand, focused solely on delamination and deinking as the applied recycling technology.

Similar to available LCAs, the publications around the economic assessment of waste management focused on the EoL treatment phase. Furthermore, it was observed that economic assessments are often conducted separately from Life Cycle Assessment (LCA), resulting in non-comparable outcomes due to different system boundaries, as highlighted by Martinez-Sanchez et al. (2015). Faraca et al. (2019) conducted an eLCC analysis on rigid plastic recycling, demonstrating that achieving high-quality recycled plastic can lead to both environmental benefits and financial revenues, aligning with the findings of this study. Additionally, Larrain et al. (2021) employed TEA to evaluate the economic performance of conventional mechanical recycling of post-consumer plastic packaging waste fractions, similar to the ones studied by Civancik-Uslu et al. (2021). The study concluded that conventional mechanical recycling would only become profitable if a policy framework was implemented to incentivize the process. Walker et al. (2020) also conducted a TEA focusing on the treatment of multilayer packaging materials using solvent-targeted recovery and precipitation. Their findings indicated that the process results in deconstructed multilayer plastic waste streams, comprising PE, EVOH, and PET resins, at costs comparable to virgin resins. However, specific details of the methodology and cost components remain undisclosed. Furthermore, Bashirgonbadi et al. (2022) evaluated the economic viability of an improved mechanical recycling process, incorporating additional sorting, hot washing, and enhanced extrusion with deodorization. Their CBA revealed that these process improvements increase the economic value of the PCR.

5. Conclusions and perspectives

This study focused on two innovative waste management scenarios, specifically recycling PE-rich flexible packaging waste from households into F and NF mono-material multilayer flexible packaging with recycled PE, alongside energy recovery (electricity and heat recovered from incinerated residues). In conclusion, these innovative scenarios were found to be both environmentally (across the six key impact categories assessed) and economically (under the given assumptions) more sustainable than linear disposal (incineration with energy recovery and landfilling) and subsequent the production of virgin flexible packaging. More specifically, the first innovative scenario, where F and NF packaging waste remain combined within the PE-rich plastic waste stream sorted from households, employing selective dissolution and producing NF MoMu packaging reduces the environmental impacts across the six key categories by 16–33 % compared to the baseline scenario. A similar impact reduction (17–52 %) is observed for the second innovative scenario, where F and NF flexible packaging waste is separated based on the presence of tracers in F packaging, followed by purification through either delamination/deinking (F fraction) or selective dissolution (NF fraction). From an economic sustainability perspective, a similar trend is observed. The cLCC results showed a cost reduction (S1 41 and S2 49 %) in both innovative scenarios. By integrating the Green Premium concept into the basket-of-products approach, the study provides a more complete economic assessment by including the potential revenue differences between MuMu laminates and MoMu laminates including PCR.

Nevertheless, the baseline scenario reflected a certain level of cost competitiveness when focusing on UBC with an increase of 7 and 27 % to the innovative scenarios. This is particularly evident when considering the flexibility in PE virgin cost driven by market dynamics. Moreover, it is essential to consider introducing or updating taxes specifically for plastic waste incineration to help offset the currently low incineration costs and promote reuse and material recovery.

Key takeaways from the study are that the innovative scenarios demonstrate that fully closing the material loops remains challenging due to several factors, including current collection and sorting rates, alongside material losses during the recycling process. Additionally, technical and consumer safety constraints in laminate design especially for F packaging and sensitive NF packaging limit the incorporation of PCR (currently around 50 %). As shown in the MFA, the incorporation of virgin PE, results in a surplus of PE-based F and NF flexible plastic packaging waste at the end of the first loop. These challenges underscore potential opportunities for further research on chemical recycling technologies and virgin bio-based materials in the long term. Moreover, the laminate design is a key focus. It is crucial to incorporate PCR in a functional way, without unnecessarily increasing thickness, as this could lead to higher demand for virgin materials and create processability issues (e.g., sealing).

Future research could expand system boundaries to include multiple reuse cycles and evaluate the impact of repeated recycling on PCR material properties, alongside a comparison with chemical and conventional mechanical recycling methods. Project trials show that selective dissolution achieves sufficient purification efficiency to produce PE PCR applicable for both F and NF packaging, highlighting the need for further research on its sustainability performance in F packaging waste treatment. Moreover, a key discussion point is the influence of the applied methodology on LCC analysis. In a further study, eLCC or sLCC could be adopted, accounting for externalities. Additionally, exploring the integration, aggregation, and weighting of LCA, LCC, and S-LCA results along with the development and calculation of a circularity indicator would be valuable. Consequently, the integrated approaches are expected to further enhance consistency, and provide a more robust foundation for decision-making. Finally, considering criticality, future market dynamics, resource availability, and regulatory developments will be key for improving long-term sustainability projections.

Declaration of generative AI and AI-assisted technologies in the writing process

During the preparation of this work the author(s) used ChatGPT-4 in order to improve the readability of the text. After using this tool/service, the author(s) reviewed and edited the content as needed and take(s) full responsibility for the content of the publication.

Funding

This research was funded by the Circular Foodpack (CFP) project (ID: 101003806) under H2020-EU.3.5. - Societal Challenges - climate action, environment, resource efficiency and raw materials and H2020-EU.3.5.4. - Enabling the transition towards a green economy and society through eco-innovation.

CRediT authorship contribution statement

Anna-Sophie Haslinger: Writing – review & editing, Writing – original draft, Visualization, Methodology, Data curation, Conceptualization. **Trang T. Nhu:** Writing – review & editing, Validation, Supervision, Conceptualization. **Erasmo Cadena:** Writing – review & editing, Validation, Conceptualization. **Gwenny Thomassen:** Writing – review & editing, Validation. **Jo Dewulf:** Writing – review & editing, Funding acquisition, Conceptualization. **Sophie Huysveld:** Writing – review & editing, Validation, Funding acquisition, Conceptualization.

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.

Acknowledgments

The authors would like to thank the members of the CFP consortium who participated in the data collection and helped in identifying the life cycle sustainability performance of circular FPP.

Supplementary materials

Supplementary material associated with this article can be found, in the online version, at doi:10.1016/j.resconrec.2025.108674.

Data availability

Supplementary data associated with this article can be found, in the online version.

References

Ademe, Ruaro, J., Awiplan, Gass, M., Lebon, F., Bellenoue, K., 2023. Référentiel des coûts du service public de gestion des déchets en France métropolitaine-Données 2020 [WWW Document]. URL https://librairie.ademe.fr/(accessed 8.20.25).

Allesch, A., Brunner, P.H., 2015. Material flow analysis as a decision support tool for waste management: a literature review. J. Ind. Ecol. 19, 753–764. https://doi.org/10.1111/jiec.12354.

Arena, U., Mastellone, M.L., Perugini, F., 2003. Life cycle assessment of a plastic packaging recycling system. Int. J. Life Cycle Assess. 8, 92. https://doi.org/10.1007/BF02978432.

Auer, M., Schmidt, J., Diemert, J., Gerhardt, G., Renz, M., Galler, V., Woidasky, J., 2023. Quality aspects in the compounding of plastic recyclate. Recycling 8, 18. https://doi.org/10.3390/recycling8010018.

Bachmann, T.M., Van Der Kamp, J., Bianchi, M., Pihkola, H., Saavedra Del Oso, M., 2024. Life cycle costing as part of a life cycle sustainability assessment of products: methodology and case studies. Int. J. Life Cycle Assess. 29, 1863–1879. https://doi.org/10.1007/s11367-024-02347-1.

Barkhausen, R., Rostek, L., Miao, Z.C., Zeller, V., 2023. Combinations of material flow analysis and life cycle assessment and their applicability to assess circular economy requirements in EU product regulations. A systematic literature review. J. Clean. Prod. 407, 137017. https://doi.org/10.1016/j.jclepro.2023.137017.

Bashirgonbadi, A., Lase, I.S., Delva, L., Van Geem, K.M., Meester, S.D., Ragaert, K., 2022. Quality evaluation and economic assessment of an improved mechanical recycling process for post-consumer flexible plastics. Waste Manag. 11. https://doi.org/10.1016/j.wasman.2022.08.018.

Bianchi, M., Bachmann, T.M., van der Kamp, J., Riemer, M., Riva, F., Préat, N., Taelman, S.E., 2021. ORIENTING D1.3 critical evaluation of economic approaches.

Carlsson, A.-S., 2002. Kartläggning Och Utvärdering Av Plaståtervinning i Ett Systemperspektiv [Identification and Evaluation of Plastics Recycling in a System Perspective. IVL Swedish Environmental Research Institute, Stockholm.].

Carlsson Reich, M., 2005. Economic assessment of municipal waste management systems—case studies using a combination of life cycle assessment (LCA) and life cycle costing (LCC). J. Clean. Prod. 13, 253–263. https://doi.org/10.1016/j.jclepro.2004.02.015.

CEFLEX, 2020. Technical Report. Designing for a circular economy. Recyclability of polyolefin-based flexible packaging [WWW Document]. URL https://swissrecycle.ch/fileadmin/user_upload/pdfs/Firmen/Drehscheibe_KLW/CEFLEX-D4ACE-Phase-1-Guidelines-Technical-report-SPREAD-VIEW-low-res-June-2020.pdf (accessed 1.19.23).

Ciroth, A., 2009. Cost data quality considerations for eco-efficiency measures. Ecol. Econ. 68, 1583–1590. https://doi.org/10.1016/j.ecolecon.2008.08.005.

Ciroth, A., Hunkeler, D., Klöpffer, W., Swarr, T., 2011. Life cycle costing – a code of practice. Key Messag. Crit. Eval. https://www.greendelta.com/wp-content/uploads/2017/03/LCAXI_LCC.pdf accessed 1.19.23.

Civancik-Uslu, D., Nhu, T.T., Van Gorp, B., Kresovic, U., Larrain, M., Billen, P., Ragaert, K., De Meester, S., Dewulf, J., Huysveld, S., 2021. Moving from linear to circular household plastic packaging in Belgium: prospective life cycle assessment of mechanical and thermochemical recycling. Resour. Conserv. Recycl. 171, 105633. https://doi.org/10.1016/j.resconrec.2021.105633.

De Weerdt, L., Sasao, T., Compernolle, T., Van Passel, S., De Jaeger, S., 2020. The effect of waste incineration taxation on industrial plastic waste generation: a panel analysis. Resour. Conserv. Recycl. 157, 104717. https://doi.org/10.1016/j.resconrec.2020.104717.

Economic Research Institute, 2025. Driver salary [WWW Document]. URL https://www.erieri.com/salary/job/driver/belgium#:~:text=The%20average%20pay%20for%

20a,%E2%82%AC22%2C491%20and%20%E2%82%AC33%2C897. (accessed 2.3.25).
EFSA Panel on Food Contact Materials, Enzymes, Flavourings and Processing Aids (CEF), 2013. Scientific opinion on the safety evaluation of the process "CO.N.I.P." used to recycle polypropylene and polyethylene crates for use as food contact material. EFSA J. 11. https://doi.org/10.2903/j.efsa.2013.3157.
Eunomia, Plastic Recyclers Europe, 2020. Flexible films market in europe: state of play. Product. Collect. Recycl. Data.
European Commission, 2015. HORIZON 2020 WORK PROGRAMME 2014 –2015 General Annexes: G. Technology readiness levels (TRL) [WWW Document]. URL https://www.h2020.md/sites/h2020/files/1595112-18._general_annexes_wp2014-2015_rev1_en.pdf.
European Commission, 2018. In: PEFCR Guidance Document: Guidance for the Development of Product Environmental Footprint Category Rules (PEFCRs). Version 6.3.
European Commission, 2022a. COM(2022) 677 final. Proposal for a regulation of the european parliament and of the council on packaging and packaging waste, amending Regulation (EU) 2019/1020 and Directive (EU) 2019/904, and repealing Directive 94/62/EC.
European Commission, 2022b. Commission regulation (EU) 2022/1616 of 15 September 2022 on recycled plastic materials and articles intended to come into contact with foods, and repealing Regulation (EC) No 282/2008 [WWW Document]. URL https://eur-lex.europa.eu/legal-content/EN/TXT/PDF/?uri=CELEX:32022R1616 (accessed 8.3.23).
European Commission, 2025. COMMISSION REGULATION (EU) 2025/351 of 21 February 2025 amending Regulation (EU) No 10/2011 on plastic materials and articles intended to come into contact with food, amending Regulation (EU) 2022/1616 on recycled plastic materials and articles intended to come into contact with foods, and repealing Regulation (EC) No 282/2008, and amending Regulation (EC) No 2023/2006 on good manufacturing practice for materials and articles intended to come into contact with food as regards recycled plastic and other matters related to quality control and manufacturing of plastic materials and articles intended to come into contact with food [WWW Document]. URL https://eur-lex.europa.eu/legal-content/EN/TXT/PDF/?uri=OJ:L_202500351 (accessed 4.14.25).
European Commission. Directorate General for Energy, 2021. EU Reference Scenario 2020: energy, Transport and GHG emissions : Trends to 2050. Publications Office, LU.
European Commission. Joint Research Centre, 2019. Suggestions For the Update of the Environmental Footprint Life Cycle Impact Assessment: Impacts Due to Resource use, Water use, Land use, and Particulate Matter. Publications Office, LU.
European Commission. Joint Research Centre, 2022. Understanding Product Environmental Footprint and Organisation Environmental Footprint methods. Publications Office, LU.
European Environmental Agency, 2023. Technical note accompanying the EEA briefing 'economic instruments and separate collection – key instruments to increase recycling [WWW Document]. URL https://www.google.com/url?sa=t&rct=j&q=&esrc=s&source=web&cd=&cad=rja&uact=8&ved=2ahUKEwjZiPW9lrmLAxWNVKQEHW05G3EQFnoECBMQAQ&url=https%3A%2F%2Fwww.eea.europa.eu%2Fpublications%2Feconomic-instruments-and-separate-collection%2Ftechnical-note-accompanying-the-eea&usg=AOvVaw19QSMXp36woch6yv1tlKVT&opi=89978449.
European Environmental Agency, 2025. Competitiveness of secondary materials [WWW Document]. URL https://www.eea.europa.eu/en/circularity/thematic-metrics/business/competitiveness-of-secondary-materials (accessed 2.13.25).
European Parliament, 2024. PROVISIONAL AGREEMENT RESULTING FROM INTERINSTITUTIONAL NEGOTIATIONS Subject: Proposal for a regulation of the European Parliament and of the Council on packaging and packaging waste, amending Regulation (EU) 2019/1020 and Directive (EU) 2019/904, and repealing Directive 94/62/EC (COM(2022)0677 – C9-0400/2022 –2022/0396(COD)) [WWW Document]. URL https://view.officeapps.live.com/op/view.aspx?src=https%3A%2F%2Fwww.europarl.europa.eu%2FRegData%2Fcommissions%2Fenvi%2Finag%2F2024%2F03-15%2FENVI_AG(2024)760975_EN.docx&wdOrigin=BROWSELINK (accessed 2.5.25).
European Parliament and Council of EU, 2024. Directive (EU) 2024/1760 of the European Parliament and of the Council of 13 June 2024 on corporate sustainability due diligence and amending Directive (EU) 2019/1937 and Regulation (EU) 2023/2859Text with EEA relevance. [WWW Document]. URL https://eur-lex.europa.eu/legal-content/EN/TXT/PDF/?uri=OJ:L_202401760 (accessed 1.9.25).
Eurostat, 2024a. Electricity prices for non-household consumers - bi-annual data (from 2007 onwards) [WWW Document]. URL https://ec.europa.eu/eurostat/databrowser/view/nrg_pc_205/default/table?lang=de (accessed 1.31.25).
Eurostat, 2024b Gas prices for non-household consumers - bi-annual data (from 2007 onwards) [WWW Document]. URL https://ec.europa.eu/eurostat/databrowser/view/nrg_pc_203/default/table?lang=enhttps:%2F%2Fec.europa.eu%2Feurostat%2Fdatabrowser%2Fview%2Fnrg_pc_203%2Fdefault%2Ftable%3Flang%3Den.
Faraca, G., Martinez-Sanchez, V., Astrup, T.F., 2019. Environmental life cycle cost assessment: recycling of hard plastic waste collected at Danish recycling centres. Resour. Conserv. Recycl. 143, 299–309. https://doi.org/10.1016/j.resconrec.2019.01.014.
Fazio, S., Castellani, V., Sala, S., Schau, EM., Secchi, M., Zampori, L., Diaconu, E., 2018. Supporting Information to the Characterisation Factors of Recommended EF Life Cycle Impact Assessment methods: New Methods and Differences With ILCD. Publications Office, LU.
Finnveden, G., 1999. Methodological aspects of life cycle assessment of integrated solid waste management systems. Resour. Conserv. Recycl. 26, 173–187. https://doi.org/10.1016/S0921-3449(99)00005-1.
Focus Technology Co., Ltd., 2024. Film Roll Cutting Machine Film Roll Cutting Machine Automatic BOPP PE Pet Film Cutter Plastic Roll Cutting Machine [WWW Document]. URL https://havesino.en.made-in-china.com/product/UwXAvYudferc/China-Film-Roll-Cutting-Machine-Film-Roll-Cutting-Machine-Automatic-BOPP-PE-Pet-Film-Cutter-Plastic-Roll-Cutting-Machine.html?utm_source=ggshop&utm_medium=feed&utm_campaign=423-sljx-UwXAvYudferc&srsltid=AfAwrE5LrSCnMTTW6SH_M5TLRrqSANavD0fHbVOa65iGo_iWwTxb6nu7MK0 (accessed 10.5.24).
Frees, N., 2002. Miljømæssige Fordele Og Ulemper Ved Genvinding Af Plast [Environmental Advantages and Disadvantages the Recycling of Plastics Examples Based On Specific Products]. Miljøstyrelsen, Copenhagen.
Freitag, N., Schneider, J., Decottignies, V., Fell, T., Kucukpinar, E., Schlummer, M., 2024. Waste study on flexible food and non-food packaging: detailed analysis of the plastic composition of European polyethylene-containing Waste streams. Materials 17, 3202. https://doi.org/10.3390/ma17133202.
Garcia-Gutierrez, P., Amadei, A.M., Klenert, D., Nessi, S., Tonini, D., Tosches, D., Ardente, F., Saveyn, H., 2023. Environmental and Economic Assessment of Plastic Waste recycling: a Comparison of mechanical, physical, Chemical Recycling and Energy Recovery of Plastic Waste. JRC technical report. Publications Office of the European Union, Luxembourg. https://doi.org/10.2760/0472.
Geueke, B., 2018. Food packaging in the circular economy: overview of chemical safety aspects for commonly used materials. J. Clean. Prod. https://doi.org/10.1016/j.jclepro.2018.05.005.
Haase, M., Wulf, C., Baumann, M., Ersoy, H., Koj, J.C., Harzendorf, F., Mesa Estrada, L.S., 2022. Multi-criteria decision analysis for prospective sustainability assessment of alternative technologies and fuels for individual motorized transport. Clean Technol. Environ. Policy 24, 3171–3197. https://doi.org/10.1007/s10098-022-02407-w.
Haslinger, A.-S., Huysveld, S., Cadena, E., Dewulf, J., under revision. Methodological strategies for social life cycle assessment of an emerging technology: a case study on circular flexible plastic packaging.
Hauschild, M.Z., Rosenbaum, R.K., Olsen, S.I., 2018. Life Cycle Assessment: Theory and Practice. Springer International Publishing, Cham. https://doi.org/10.1007/978-3-319-56475-3.
Hestin, M., Faninger, T., Milios, L., 2015. Increased EU plastics recycling targets: environmental, economic and social impact assessment. Final Report. https://www.plasticsrecyclers.eu/wp-content/uploads/2022/10/increased-eu-plastics-recycling-targets.pdf. accessed 1.19.23.
Horn, R., Hong, S.H., Knüpffer, E., Alvarenga, R., Boone, L., Preat, N., Behm, K., Pajula, T., Pihkola, H., 2021. ORIENTING (D1.1) Critical evaluation of environmental approaches. EU Horizon 2020 project ORIENTING (GA No 958231).
Horodytska, O., Kiritsis, D., Fullana, A., 2020. Upcycling of printed plastic films: LCA analysis and effects on the circular economy. J. Clean. Prod. 268, 122138. https://doi.org/10.1016/j.jclepro.2020.122138.
Hunkeler, D.J., Lichtenvort, K., Rebitzer, G., Ciroth, A., 2008. Environmental life cycle costing 211.
Huysman, S., De Schaepmeester, J., Ragaert, K., Dewulf, J., De Meester, S., 2017. Performance indicators for a circular economy: a case study on post-industrial plastic waste. Resour. Conserv. Recycl. 120, 46–54. https://doi.org/10.1016/j.resconrec.2017.01.013.
Huysveld, S., Ragaert, K., Demets, R., Nhu, T.T., Civancik-Uslu, D., Kusenberg, M., Van Geem, K.M., De Meester, S., Dewulf, J., 2022. Technical and market substitutability of recycled materials: calculating the environmental benefits of mechanical and chemical recycling of plastic packaging waste. Waste Manag. 152, 69–79. https://doi.org/10.1016/j.wasman.2022.08.006.
Isasa, M., Lizasoain, E., Ugarte, R., Pihkola, H., Zanchi, L., Bachmann, T.M., Taelman, S. E., Hackenhaar, I.C., 2024. ORIENTING 2024 (D2.8) Recommendations for integration and interpretation of the LCSA results. EU Horizon 2020 project ORIENTING (GA No 958231).
ISO 14040, 2006. In: Environmental management — Life cycle assessment — Principles and framework. Switzerland.
ISO 14044, 2006. In: Environmental management — Life cycle assessment — Requirements and guidelines. Switzerland.
Jeswani, H., Krüger, C., Russ, M., Horlacher, M., Antony, F., Hann, S., Azapagic, A., 2021. Life cycle environmental impacts of chemical recycling via pyrolysis of mixed plastic waste in comparison with mechanical recycling and energy recovery. Sci. Total Environ. 769, 144483. https://doi.org/10.1016/j.scitotenv.2020.144483.
Kaiser, K., Schmid, M., Schlummer, M., 2017. Recycling of polymer-based multilayer packaging: a review. Recycling 3, 1. https://doi.org/10.3390/recycling3010001.
Koinig, G., Kuhn, N., Barretta, C., Friedrich, K., Vollprecht, D., 2022. Evaluation of improvements in the separation of monolayer and multilayer films via measurements in transflection and application of machine learning approaches. Polymers 14, 3926. https://doi.org/10.3390/polym14193926.
K.D. Kol, R., Roosen, M., Ügdüler, S., Van Geem, M., Ragaert, K., Achilias, S., De Meester, S., 2021. Recent advances in pre-treatment of plastic packaging waste, D.. In: Achilias, S. (Ed.), Waste Material Recycling in the Circular Economy - Challenges and Developments. IntechOpen, London. https://doi.org/10.5772/intechopen.99385.
Köveker, T., Chiappinelli, O., Kröger, M., Lösch, O., Neuhoff, K., Richstein, J.C., Sun, X., 2023. Green premiums are a challenge and an opportunity for climate policy design. Nat. Clim. Change 13, 592–595. https://doi.org/10.1038/s41558-023-01689-2.
Kusch, A., Gasde, J., Deregowski, C., Woidasky, J., Lang-Koetz, C., Viere, T., 2021. Sorting and recycling of lightweight packaging in Germany — climate impacts and options for increasing circularity using tracer-based-sorting. Mater. Circ. Econ. 3, 10. https://doi.org/10.1007/s42824-021-00022-6.
EFSA Panel on Food Contact Materials, Enzymes and Processing Aids (CEP), Lambré, C., Barat Baviera, J.M., Bolognesi, C., Chesson, A., Cocconcelli, P.S., Crebelli, R., Gott, D.M., Grob, K., Mengelers, M., Mortensen, A., Rivière, G., Steffensen, I.,

Tlustos, C., Van Loveren, H., Vernis, L., Zorn, H., Dudler, V., Milana, M.R., Papaspyrides, C., Tavares Poças, M.de F., Colombo, G., Comandella, D., Lioupis, A., Marano, R., Munoz Guajardo, I.P., Savini, E., Sfika, V., Tsochatzis, E., Volk, K., Lampi, E., 2024. Scientific Guidance on the criteria for the evaluation and on the preparation of applications for the safety assessment of post-consumer mechanical PET recycling processes intended to be used for manufacture of materials and articles in contact with food. EFSA J. 22. https://doi.org/10.2903/j.efsa.2024.8879.

EFSA Panel on Food Contact Materials, Enzymes and Processing Aids (CEP), Lambré, C., Barat Baviera, J.M., Bolognesi, C., Chesson, A., Cocconcelli, P.S., Crebelli, R., Gott, D.M., Grob, K., Mengelers, M., Mortensen, A., Rivière, G., Steffensen, I., Tlustos, C., Van Loveren, H., Vernis, L., Zorn, H., Dudler, V., Milana, M.R., Papaspyrides, C., Tavares Poças, M.de F., Lioupis, A., Lampi, E., 2021. Safety assessment of the process deSter, used to recycle plastic catering tableware for use as food contact materials. EFSA J. 19. https://doi.org/10.2903/j.efsa.2021.6947.

Larrain, M., Van Passel, S., Thomassen, G., Van Gorp, B., Nhu, T.T., Huysveld, S., Van Geem, K.M., De Meester, S., Billen, P., 2021. Techno-economic assessment of mechanical recycling of challenging post-consumer plastic packaging waste. Resour. Conserv. Recycl. 170, 105607. https://doi.org/10.1016/j.resconrec.2021.105607.

Lase, I.S., Bashirgonbadi, A., van Rhijn, F., Dewulf, J., Ragaert, K., Delva, L., Roosen, M., Brandsma, M., Langen, M., De Meester, S., 2022. Material flow analysis and recycling performance of an improved mechanical recycling process for post-consumer flexible plastics. Waste Manag. 153, 249–263. https://doi.org/10.1016/j.wasman.2022.09.002.

Lazarevic, D., Aoustin, E., Buclet, N., Brandt, N., 2010. Plastic waste management in the context of a European recycling society: comparing results and uncertainties in a life cycle perspective. Resour. Conserv. Recycl. 55, 246–259. https://doi.org/10.1016/j.resconrec.2010.09.014.

Martínez Leal, J., Pompidou, S., Charbuillet, C., Perry, N., 2020. Design for and from recycling: a circular ecodesign approach to improve the circular economy. Sustainability 12, 9861. https://doi.org/10.3390/su12239861.

Martinez-Sanchez, V., Kromann, M.A., Astrup, T.F., 2015. Life cycle costing of waste management systems: overview, calculation principles and case studies. Waste Manag. 36, 343–355. https://doi.org/10.1016/j.wasman.2014.10.033.

Maxwell, C., 2025. Cost Indices [WWW Document]. URL https://toweringskills.com/financial-analysis/cost-indices/(accessed 8.20.25).

Nakatani, J., 2014. Life cycle inventory analysis of recycling: mathematical and graphical frameworks. Sustainability 6, 6158–6169. https://doi.org/10.3390/su6096158.

Nhu, T.T., Haslinger, A.-S., Huysveld, S., Dewulf, J., 2025. Incorporating recyclates derived from household waste into flexible food packaging applications: an environmental sustainability assessment. Recycling 10, 142. https://doi.org/10.3390/recycling10040142.

Perugini, F., Mastellone, M.L., Arena, U., 2005. A life cycle assessment of mechanical and feedstock recycling options for management of plastic packaging wastes. Environ. Prog. 24, 137–154. https://doi.org/10.1002/ep.10078.

Pihkola, H., Cordella, M., Behm, K., Horn, R., Zamagni, A., Zanchi, L., Harmens, R., Sonderegger, T., Martinez, E.C., Bachmann, T.M., Van Der Kamp, J., Bianchi, M., Riva, F., Fehrenbach, D., Moraga, G.L., Isadora Hackenhaar, Moreno, L.S., Isasa, M., Taelman, S.E., Kujanpää, L., 2022. ORIENTING D2.3 - LCSA methodology to be implemented in WP4 demonstrations. https://doi.org/10.13140/RG.2.2.24300.13445.

Plastic Europe, 2022. Plastics—the facts 2022. [WWW Document]. URL https://plasticseurope.org/wp-content/uploads/2022/10/PE-PLASTICS-THE-FACTS_V7-Tue_19-10-1.pdf (accessed 1.19.23).

Plastics Recyclers Europe, 2023. Flexible films market in Europe. State of play production, collection and recycling data [WWW Document]. URL https://circularplasticsnl.org/wp-content/uploads/2023/12/2023-Flexible-Films-Market-in-Europe_State-of-Play_June-2023.pdf (accessed 9.26.23).

Polysecure, 2025. FDA approval for markers in plastics with food contact [WWW Document]. URL https://www.polysecure.eu/en/company/news.

PWC, 2024. Corporate income tax (CIT) rates [WWW Document]. URL https://taxsummaries.pwc.com/quick-charts/corporate-income-tax-cit-rates (accessed 7.25.24).

Quantis, Amcor, 2022. Draft Product Environmental Footprint Category Rules (PEFCR): Flexible Packaging - Version 1.0.

Raadal, H.L., Brekke, A., Modahl, I.S., 2008. Miljøanalyse Av Ulike Behandlingsformer For Plastemballasje Fra Husholdninger [Environmental Analysis of the Different Treatment For Plastic from Households. Østfoldforskning AS, Fredrikstad.].

Ragaert, K., Delva, L., Van Geem, K., 2017. Mechanical and chemical recycling of solid plastic waste. Waste Manag. 69, 24–58. https://doi.org/10.1016/j.wasman.2017.07.044.

Sanabria Garcia, E., Huysveld, S., Nachtergaele, P., Nhu, T.T., Dewulf, J., 2025. How multifunctionality modelling in LCA affects decision-making: the case of chemical recycling of plastic waste. Resour. Conserv. Recycl. 218, 108262. https://doi.org/10.1016/j.resconrec.2025.108262.

Schmidt, J., Grau, L., Auer, M., Maletz, R., Woidasky, J., 2022. Multilayer packaging in a circular economy. Polymers 14, 1825. https://doi.org/10.3390/polym14091825.

Shonfield, P., 2008. LCA of Management Options For Mixed Waste Plastics. WRAP, London.

Soares, C.T., de, M., Ek, M., Östmark, E., Gällstedt, M., Karlsson, S., 2022. Recycling of multi-material multilayer plastic packaging: current trends and future scenarios. Resour. Conserv. Recycl. 176, 105905. https://doi.org/10.1016/j.resconrec.2021.105905.

Taelman, S., Sanjuan-Delmás, D., Tonini, D., Dewulf, J., 2020. An operational framework for sustainability assessment including local to global impacts: focus on waste management systems. Resour. Conserv. Recycl. 162, 104964. https://doi.org/10.1016/j.resconrec.2020.104964.

Tamizhdurai, P., Mangesh, V.L., Santhosh, S., Vedavalli, R., Kavitha, C., Bhutto, J.K., Alreshidi, M.A., Yadav, K.K., Kumaran, R., 2024. A state-of-the-art review of multilayer packaging recycling: challenges, alternatives, and outlook. J. Clean. Prod. 447, 141403. https://doi.org/10.1016/j.jclepro.2024.141403.

Thomassen, G., Van Dael, M., Van Passel, S., You, F., 2019. How to assess the potential of emerging green technologies? Towards a prospective environmental and techno-economic assessment framework. Green Chem. 21, 4868–4886. https://doi.org/10.1039/C9GC02223F.

Thonemann, N., Schulte, A., Maga, D., 2020. How to conduct prospective life cycle assessment for emerging technologies? A systematic review and methodological guidance. Sustainability 12, 1192. https://doi.org/10.3390/su12031192.

Tonini, D., Garcia-Gutierrez, P., Nessi, S., 2021. Environmental Effects of Plastic Waste Recycling. European Commission. https://doi.org/10.2760/6309. JRC Technical Report.

Transport and Environment, 2015. Comparing US and EU truck fuel economy [WWW Document]. URL https://www.transportenvironment.org/wp-content/uploads/2021/07/2015_06_Comparing_US_EU_truck_fuel_economy_explantory_note_Final.pdf (accessed 7.5.23).

Towler, G, Sinnott, R, 2013. Capital Cost Estimating. In: Chemical Engineering Design. Elsevier, pp. 307–354.

Tunçok-Çeşme, B., Yıldız-Geyhan, E., Çiftçioğlu, G.A., 2024. Environmental life cycle assessment of two types of flexible plastic packaging under a sustainable circular economy approach. Sustainability 16, 3149. https://doi.org/10.3390/su16083149.

Ügdüler, S., De Somer, T., Van Geem, K.M., De Wilde, J., Roosen, M., Deprez, B., De Meester, S., 2022. Analysis of the kinetics, energy balance and carbon footprint of the delamination of multilayer flexible packaging films via carboxylic acids. Resour. Conserv. Recycl. 181, 106256. https://doi.org/10.1016/j.resconrec.2022.106256.

Ügdüler, S., Van Laere, T., De Somer, T., Gusev, S., Van Geem, K.M., Kulawig, A., Leineweber, R., Defoin, M., Van Den Bergen, H., Bontinck, D., De Meester, S., 2023. Understanding the complexity of deinking plastic waste: an assessment of the efficiency of different treatments to remove ink resins from printed plastic film. J. Hazard. Mater. 452, 131239. https://doi.org/10.1016/j.jhazmat.2023.131239.

Valdivia, S., 2024. Handbook On Life Cycle Sustainability assessment, Elgar Handbooks in Development. Edward Elgar Publishing, Cheltenham, UK ; Northampton, Massachusetts.

Valdivia, S., Backes, J.G., Traverso, M., Sonnemann, G., Cucurachi, S., Guinée, J.B., Schaubroeck, T., Finkbeiner, M., Leroy-Parmentier, N., Ugaya, C., Peña, C., Zamagni, A., Inaba, A., Amaral, M., Berger, M., Dvarioniene, J., Vakhitova, T., Benoit-Norris, C., Prox, M., Foolmaun, R., Goedkoop, M., 2021. Principles for the application of life cycle sustainability assessment. Int. J. Life Cycle Assess. 26, 1900–1905. https://doi.org/10.1007/s11367-021-01958-2.

UNEP/SETAC, Valdivia, S., Ugaya, C.M., Ciroth, A., Finkbeiner, M., Hildebrand, J., Klöpffer, W., Mazijn, B., Prakash, S., Sonnemann, G., Traverso, M., Vickery-Niederman, G., 2011. Towards a Life Cycle Sustainability Assessment. Making informed Choices On Products. UNEP/SETAC Life Cycle Initiative, Paris, France.

von Krogh, L., Raadal, H.L., Hanssen, O.J., 2001. Life cycle assessment of different scenarios 610 for waste treatment of plastic bottle used for food packaging [WWW Document]. URL https://norsus.no/wp-content/uploads/3901.pdf?v=1 (accessed 2.5.25).

Walker, T.W., Frelka, N., Shen, Z., Chew, A.K., Banick, J., Grey, S., Kim, M.S., Dumesic, J. A., Van Lehn, R.C., Huber, G.W., 2020. Recycling of multilayer plastic packaging materials by solvent-targeted recovery and precipitation. Sci. Adv. 6, eaba7599. https://doi.org/10.1126/sciadv.aba7599.

Wang, Q., Sun, T., Li, R., 2024. Does larger scale enhance carbon efficiency? Assessing the impact of corporate size on manufacturing carbon emission efficiency. Humanit. Soc. Sci. Commun. 11, 993. https://doi.org/10.1057/s41599-024-03474-8.

Wohner, B., Gabriel, V.H., Krenn, B., Krauter, V., Tacker, M., 2020. Environmental and economic assessment of food-packaging systems with a focus on food waste. Case study on tomato ketchup. Sci. Total Environ. 738, 139846. https://doi.org/10.1016/j.scitotenv.2020.139846.