



Comparative life cycle assessment of centralized versus distributed high-density polyethylene recycling chains

Davide Gotti, Daniele Landi^{*}, Christian Spreafico, Davide Russo

Department of Management, Information and Production Engineering, University of Bergamo, Via Salvecchio 19, Bergamo, Italy

ARTICLE INFO

Keywords:

Life Cycle Assessment (LCA)
Plastic waste management
Circular Economy (CE)
Plastic waste recycling
High-Density Polyethylene (HDPE)

ABSTRACT

Plastic waste management presents a growing global challenge, and recycling is an important strategy for mitigating plastic's environmental impact and promoting circularity. Although centralized recycling chains are well established, the potential of small- and medium-scale distributed recycling facilities to improve waste collection in urbanized areas and foster recycling in remote regions remains underexplored. This study uses life cycle assessment (LCA) to compare the environmental performance of three recycling chain scenarios for rigid high-density polyethylene plastic packaging waste: centralized, semi-distributed, and fully distributed. Using the ReCiPe Midpoint (H) method and data from a recycling facility in Northern Italy, this study applies the product perspective calculation method in which recycled plastic replaces virgin plastic, and environmental credits are assigned accordingly. The results show that the semi-distributed scenario offers the most effective compromise between logistical efficiency and operational scale and achieves the lowest environmental impact, with an average reduction of $48\% \pm 13.9\%$ across 18 impact categories and savings of 1.42 kg CO₂eq per kilogram of recycled granulate compared to the centralized system. The distributed scenario is second, while the centralized system demonstrates the highest environmental burden, primarily because of the impact of long-distance logistics.

1. Introduction

The environmental performance of high-density polyethylene (HDPE) mechanical recycling (MR) is governed by both process efficiency and the spatial organization of the recycling infrastructure. The interaction between facility scale and geographic distribution introduces critical trade-offs between operational efficiency and transport-related impacts, which substantially affect overall system sustainability. Although large-scale facilities may offer process-level advantages, their environmental benefits can be offset by increased logistical burdens in low-density regions (Bradshaw et al., 2025; Selamoglu, 2024). Existing LCA studies have predominantly focused on centralized configurations, with limited exploration of the logistical–environmental trade-offs across alternative recycling scenarios (Kreiger et al., 2014; Aguirre-Villegas et al., 2025). This study addresses this gap by conducting a high-resolution, system-level comparison of centralized, semi-distributed, and fully distributed recycling configurations, providing data-driven insights for infrastructure planning across diverse

territorial contexts. The most important aspect of the paper is the comparison of the recycling chains, and the need to implement efficient plastic waste recycling for environmental reasons is self-evident without discussion of the EU targets.

Plastic packaging constitutes approximately 61 % of the total post-consumer plastic waste stream (Plastics Europe, 2022). Despite its important role in resource recovery, the post-consumer plastic packaging waste (PPW) share collected separately for recycling in 2017 was only approximately 41 % within the EU nations (Antonopoulos et al., 2021). Transitioning to a circular economy (CE) requires advanced recycling strategies to reduce dependence on virgin polymers and mitigate environmental burdens linked to plastic waste. HDPE is a versatile and recyclable thermoplastic that represents approximately 17 % of global plastic production (Gandhi et al., 2021) and is predominantly used in rigid packaging and household applications. As such, it is a strategic target due to its high recyclability and potential for substantial life cycle impact reduction (Gandhi et al., 2021; Plastics Europe, 2022). Despite increasing awareness of environmental challenges and potential

^{*} Corresponding author. Via Pasubio 7b, 24044, Dalmine, BG, Italy.

E-mail addresses: davide.gotti@unibg.it (D. Gotti), daniele.landini@unibg.it (D. Landi), christian.spreafico@unibg.it (C. Spreafico), davide.russo@unibg.it (D. Russo).

<https://doi.org/10.1016/j.jclepro.2025.147062>

Received 30 May 2025; Received in revised form 13 October 2025; Accepted 8 November 2025

Available online 24 November 2025

0959-6526/© 2025 The Authors. Published by Elsevier Ltd. This is an open access article under the CC BY license (<http://creativecommons.org/licenses/by/4.0/>).

CO₂ emissions reductions of 60 %–70 % through recycling compared to virgin production (Muralikrishna and Manickam, 2017; Bataineh, 2020), recycling rates in Europe remain low, estimated between 10 % and 15 % (European Recycling Industries' Confederation, 2020), consistent with global rates hovering around 9 %–12 % (OECD, 2022)."

Producing virgin HDPE (v-HDPE) consumes 124 billion MJ annually (Kreiger et al., 2014), while recycled HDPE (r-HDPE) embodies 24 % less energy (Grant and James, 2005). Failure to recycle HDPE leads to substantial waste of energy and material resources (Grant and James, 2005; Hammond and Jones, 2008). Recycling HDPE can reduce nonrenewable energy use by 40 %–85 % and GHG emissions by 25 %–75 %, while also diverting waste from landfills and reducing reliance on virgin production, offering substantial environmental benefits over single-use v-HDPE (Bataineh, 2020). HDPE waste management follows a dual pathway: 63 % is collected and processed in recycling facilities; the remaining 37 % undergoes final disposal, with 68 % incinerated and 32 % landfilled (Tonini et al., 2021). Mixed plastic waste recycling can provide environmental benefits over incineration and landfill, particularly in terms of global warming potential, provided that high-quality recyclate is produced (Shonfield, 2008).

Although HDPE recycling offers clear environmental advantages, its industrial-scale recycling is often not economically competitive with virgin polymer production (Arena et al., 2023; Lase et al., 2023; Uzosike et al., 2023). Logistical complexities, particularly long-distance transportation of waste and HDPE bales in centralized systems, can reduce or negate these benefits by significantly increasing energy use and emissions (Selamoglu, 2024). Existing literature has largely focused on incremental improvements within conventional infrastructures, such as optimized collection routes or energy efficiency upgrades, while overlooking alternative decentralized configurations that may reduce transport burdens and improve system-level sustainability (Kreiger et al., 2014).

Historically, polymer recycling has favored large, centralized facilities to use economies of scale for the production of low-value commodities (Missouri Department of Natural Resources, 2010). However, increasing the scale of material recovery facilities (MRFs) does not always translate into linear environmental benefits (Aguirre-Villegas et al., 2025). Although larger MRFs reduce per-unit energy consumption and GHG emissions through economies of scale, they require extensive transportation networks that introduce additional carbon burdens, particularly in regions with low population densities.

Large-scale MRFs benefit from economies of scale, with lower per-unit energy consumption and improved market access, yet the scale of these facilities also influences their overall environmental performance. In contrast, small-scale MRFs have higher per-unit energy costs but may offer logistical advantages in forming decentralized recycling networks and reducing transportation emissions. The hub-and-spoke model, where large centralized MRFs process materials from multiple collection points, was proposed as an efficient alternative to dispersed small-scale facilities, although its comparative environmental performance remains underexplored (Aguirre-Villegas et al., 2025; Bradshaw et al., 2025). While the scale dynamics of waste sorting facilities have received increasing attention, the environmental implications of scaling recycling plants remain insufficiently characterized.

The analysis of energy efficiency and economies of scale in MRFs is gaining research attention, but the available data present limitations and contradictory results, especially regarding facility size. Aguirre-Villegas et al. (2025) and Bradshaw et al. (2025) examined the energy consumption and carbon footprints of MRFs in the United States and observed a significant scale effect, with small MRFs showing electricity consumption up to two orders of magnitude higher than medium and large facilities. Crucially, they observed a significant discrepancy between actual operational consumption and predictions based on idealized models, such as that of Pressley et al. (2015), suggesting that idealized models tend to significantly underestimate actual energy consumption by not fully accounting for operational inefficiencies.

These findings contradict with the European data. Although European studies reported lower average consumption, they often lacked essential information about the size of the analyzed MRFs, making direct comparisons difficult. This reinforces the notion that the use of energy consumption values based on general averages, aggregated data without scale distinctions, or purely theoretical models introduces significant errors into environmental and economic assessments, as may have occurred in the study by Kerdlap et al. (2021), which used underestimated consumption data for small-scale cases.

Kreiger et al. (2014) demonstrated that distributed HDPE recycling can completely eliminate the transportation burden, outperforming the most efficient centralized recycling scenarios in energy consumption (Grant and James, 2005). However, their study was limited by its domestic context, which assumed that all recycling stages were integrated and performed by nonspecialized personnel using non-industrial equipment.

Semi-distributed and fully distributed recycling scenarios lead to reductions in total waste collection distances when compared to centralized systems (Kerdlap et al., 2020). However, increased GHG emissions from smaller vehicles offset environmental gains from shorter transport routes, ultimately resulting in higher overall logistics-related emissions. An important strength of their study is its use of an agent-based model to capture complex system dynamics using real, detailed logistics data. Nonetheless, insufficient consideration of infrastructure scale within the supply chain is a major limitation; their study scales material flows inaccurately, leading to distorted, unreliable results, and makes arbitrary, unsubstantiated assumptions about the scaled system flows by considering energy consumption to be lower for small-scale facilities than for large-scale facilities, thereby contradicting the economies of scale highlighted in more recent studies. Moreover, the higher sorting rate reported for smaller facilities compared to larger ones conflicts with the residual rate data provided by Bradshaw et al. (2025).

This study provides a holistic evaluation using integrated scenarios with variations in facility location and scale, exploring the potential of small- and medium-scale facilities located near plastic collection centers to minimize the long-distance transportation required by conventional centralized systems. To address gaps in the literature, data from direct and recent third-party real-world case studies are used to scale facility sizes appropriately and quantify the flows within recycling chain scenarios.

This approach provides more robust results than recent studies on the assessment of distributed scenarios. To quantify the crucial roles of logistics and facility scale in the overall environmental sustainability of the HDPE recycling chain, this study compares the sustainability of three distinct CE options, a centralized system and two distributed systems using more integrated and localized approaches. (1) Centralized recycling: The traditional scenario, involving a large-scale, dedicated centralized recycling facility, a centralized large-scale sorting facility serving multiple collection centers, and transportation of waste bales and recycled granules. (2) Semi-distributed recycling: A scenario involving medium-sized sorting centers positioned strategically to reduce transportation distances, bridging the gap between fully centralized and fully distributed systems. (3) Collection center-integrated recycling: A modular, container-based recycling facility at the plastic collection center, eliminating the transport of waste bales to separate recycling plants and significantly reducing logistical inefficiencies.

This study delivers a high-resolution LCA comparison of centralized and distributed HDPE recycling chains, evaluating how strategic facility placement and scale influence transport-related impacts, energy efficiency, and overall environmental performance. The aim is to identify the most effective configuration for minimal environmental impact.

2. Materials and methods

2.1. Goals and scope

The core function analyzed is the generation of r-HDPE granules within the complete recycling system; the functional unit (FU) is defined as 1 kg of r-HDPE granules. This definition is applied consistently across all three scenarios. The product system includes the full recycling chain, covering all relevant stages and facilities involved in the processes required to produce the defined FU.

The system boundaries established for this assessment, which focuses on the production of 1 kg of r-HDPE granules, are defined based on the methodological guidelines outlined in the Joint Research Centre (JRC) report *Environmental effects of plastic waste recycling* (Tonini et al., 2021). This study uses a cradle-to-gate LCA framework to systematically evaluate the environmental impacts over the complete recycling pathway, from post-consumer plastic waste collection to recycled granulate leaving the factory gate. The system boundaries encompass the initial operations involved in the supply of waste feedstock; collection, transport, and sorting of the target HDPE fraction destined for treatment at the MRF; and all core MR processes (separation and reprocessing stages) required to transform sorted waste into granulate. Importantly, the assessment also accounts for the management of process material losses during the sorting and recycling phases, assuming that their final treatment is via established incineration and landfill pathways. Furthermore, the analysis incorporates environmental credits associated with the primary production processes of the v-HDPE to be substituted by the recycle, ranging from raw material extraction to polymer synthesis. The system boundary ends precisely when the final granulated r-HDPE output exits the recycling facility gate, ready for downstream manufacturing applications. Consequently, all subsequent life-cycle stages, including product manufacturing, the usage phase, and end-of-life management of the final product, are excluded from the scope of this assessment. This exclusion is valid because the assumed identical downstream effects do not alter the relative environmental comparison of the investigated recycling systems. Following the product category rule (PCR) guidelines for “plastics in primary forms”, the system boundaries are extended to include relevant upstream activities, such as consumable materials production, comprehensive logistics covering inter-facility transport and waste disposal haulage, and downstream final management of all internally generated waste materials, including sorting contaminants and sludge and pulp from water treatment.

Several elements are intentionally excluded from system boundaries. Uncollected waste from the collection stage falls outside the initial scope. Using an attributional framework focused on the primary product (r-HDPE) and the cut-off approach at the waste generation point, recyclable co-products generated throughout the recycling chain and environmental burdens from the original product’s primary production phase are excluded; the latter is entirely attributed to the product’s first life cycle. Additionally, in compliance with the PCR “plastic waste and scrap recovery (recycling)”, processes related to capital goods manufacturing (production equipment, buildings, machinery, and trucks), personnel travel, and research and development activities are excluded from the scope.

This approach ensures methodological consistency, enhances the reproducibility of the results, and aligns with the standardized system boundaries recognized by third-party entities. This carefully defined scope is justified by the study’s primary objective: to assess the environmental impacts directly linked to recycling chain operations. Fig. 1 shows the system boundaries for the entire recycling chain. Flowcharts for each phase are presented in the following sections, along with outlines of the system boundaries for each phase.

This work is confined to the environmental assessment of recycling chain configurations, with economic aspects such as economies of scale, feasibility, and market constraints excluded from the analysis.

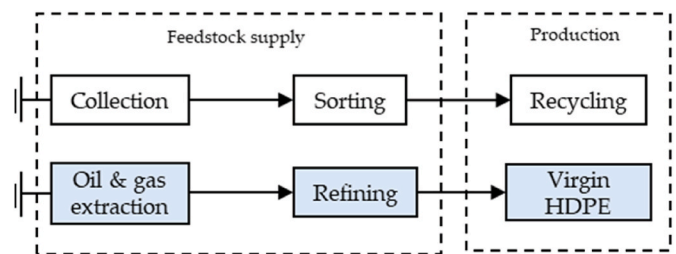


Fig. 1. System boundaries of the recycling chain. Black continuous-line boxes indicate induced processes, while grey boxes indicate processes that are avoided (via virgin material substitution). HDPE: High-density polyethylene.

2.2. Process description

The plastic recycling CE chain consists of five facilities interconnected through logistics and serves as the core focus of this study. The material flow analysis includes the following waste management processes: generation, collection, sorting, and recycling (Antonopoulos et al., 2021). Fig. 2 shows the flow diagram of a recycling chain for mixed PPW conversion into HDPE granules. The processes included in this system are identified by their respective rates, indicating the efficiency of each facility. The term ‘rate’ defines the waste management process in its entirety by considering the total material flow, whereas the ‘transfer coefficient’ applies to individual waste streams in each processing stage. This distinction is purely terminological because both terms represent the same mathematical quantity and are used interchangeably throughout this study. In this study, the material entering the recycling chain is assumed to be mixed plastic packaging waste (MPW), which is defined as sorted waste exclusively comprising plastic with minimal contaminants. The composition of MPW is highly complex and variable because of the considerable heterogeneity of incompatible polymers, additives, and contaminants in the stream, which poses primary technical challenges. Additionally, there is a notable lack of detailed MPW composition data, which are typically linked to commodity market sectors with limited or no information on the specific polymers involved (Garcia-Gutierrez et al., 2023). Waste composition is assumed to vary geographically and depends on local collection policies. Reference flows will subsequently be introduced, based on which the sorting plants have been sized for this study (see Fig. 3).

During the recycling process (Fig. 2), source-segregated post-consumer plastic waste is collected through residential curbside collection, drop-off programs, deposit redemption systems, and commercial collection programs. Residential curbside collection accounts for over 60 % of HDPE recovery (Franklin Associates, 2018). This plastic waste is initially transported to a collection center, where it is compacted, and then sent to a sorting facility. Subsequently, the sorted HDPE is forwarded in bales to a recycling plant for processing.

Collection systems represent designated locations where plastic waste, such as MPW from domestic trash, is gathered and compacted. In some cases, initial manual sorting is also conducted. Broadly, the collection stage encompasses all waste retrieval activities, and its collection efficiency (η_c) is intrinsically linked to the upstream segregation capacity, which depends on domestic waste separation practices and the specific collection system used. Efficiency data are uncertain and variable; collection systems vary between countries and even among municipalities. This study investigated the recycling of source-separated plastic waste collected within a system that promotes the inclusion of all plastic fractions and thus may also contain impurities, contaminants, and nonrecyclable residues. Various plastic waste compositions are analyzed and used as a basis for sorting facility sizing in the different scenarios.

The environmental performance of the collection stage is predominantly driven by transportation logistics with fuel combustion contributing significantly to Global Warming Potential and Particulate Matter

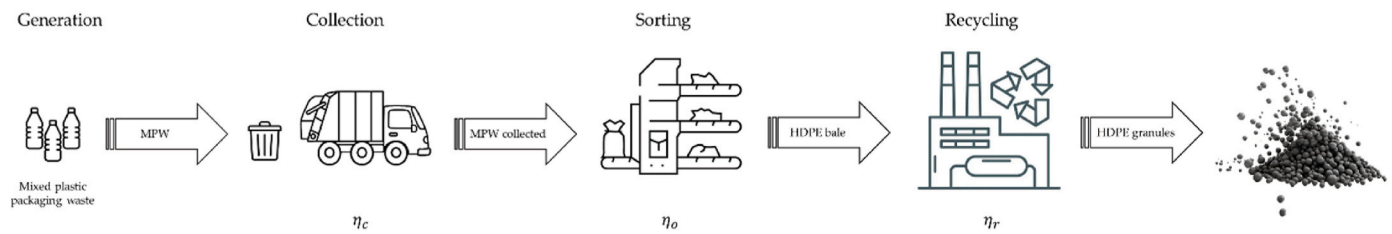


Fig. 2. HDPE recycling chain.

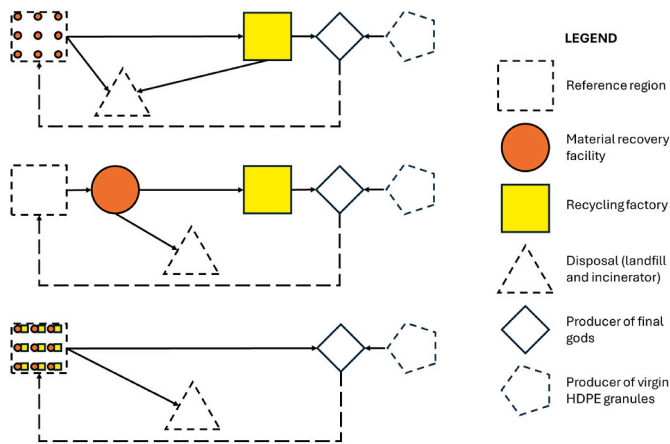


Fig. 3. Recycling chain scenarios.

Formation (Franklin Associates, APR, 2022). The impact of this phase is highly context-dependent, shaped by variables such as population density, collection scheme design, and fleet efficiency. However, Life Cycle Inventory (LCI) data for this stage remain limited and poorly harmonised across studies (Rikhter et al., 2022). Energy demand for collection and transport, excluding compaction, can reach 18.9 % of the total for recycled HDPE production, highlighting the importance of optimized logistics (Bataineh, 2020). In contrast, compaction operations contribute only marginally to the overall environmental burden of the collection phase (Tonini et al., 2021). As the gateway to material recovery, this phase represents a structural bottleneck through which inefficiencies may propagate across the entire recycling chain (Helmes et al., 2022).

MRFs, or sorting plants, process separately collected post-consumer plastic packaging waste (PPW) by segregating it into polymer-specific fractions suitable for downstream recycling operations (Antonopoulos et al., 2021). The sorting efficiency (η_o) is typically expressed through transfer coefficient, which reflect either the sorting rate (Antonopoulos et al., 2021) or recovery rate (Kleinhans et al., 2021) achieved for different PPW streams (Lase et al., 2023). As highlighted by Pressley et al. (2015), the design and configuration of separation equipment within MRFs are strongly influenced by the composition of the incoming waste stream, which in turn affects both material recovery performance and energy consumption.

Despite variations in process layout, state-of-the-art MRFs typically comprise five core technical stages (Cimpan et al., 2016; Dri et al., 2018). The process begins at the tipping floor, where feeding and pre-conditioning operations ensure a consistent material flow by opening bags and loosening compacted recyclables. The subsequent pre-sorting stage involves the removal of oversized items, contaminants, and materials unsuitable for sorting, often achieved by separating the flexible film fraction from rigid plastics using equipment such as film grabbers and ballistic separators, supplemented by manual quality control. The main sorting stage employs a sequence of physical separation technologies, integrating mechanical, pneumatic, and sensor-based systems,

including disc screens, magnets, and optical scanners, to segregate rigid items according to polymer type and color. Refining operations further enhance material purity through additional sorting and quality control, followed by the final product handling stage, where recovered recyclable fractions are typically compacted and baled to optimize storage and transport efficiency (Bradshaw et al., 2025).

From a life cycle perspective, the sequence of automated operations described above gives rise to two principal environmental hotspots. The first is electricity consumption, which exerts a significant influence on impact categories such as Global Warming Potential, Fossil Resource Scarcity, and Ionizing Radiation Potential. These burdens are closely tied to the composition and carbon intensity of the regional electricity grid, with Ionizing Radiation Potential being particularly sensitive to the proportion of nuclear energy within the mix (Pressley et al., 2015; Treyer et al., 2014; Arfelli et al., 2024).

The second environmental hotspot arises from the management of the non-recyclable residual fraction. Due to the intrinsic heterogeneity of the input stream, an initial pre-sorting stage is required to remove non-plastic contaminants, inevitably resulting in a residual output typically classified as non-recyclable mixed plastics (Rigamonti et al., 2014). The handling of this stream necessitates a dedicated downstream pathway involving additional transport, treatment, and final disposal operations. The management of the residual stream requires a dedicated downstream pathway involving transport and final disposal, typically landfilling or incineration (Bradshaw et al., 2025), each associated with distinct and often underestimated environmental burdens in conventional life cycle assessments (Sabate and Kendall, 2024). Landfilling of plastic-rich residuals contributes significantly to long-term impacts, particularly in terms of permanent land occupation and resource depletion (Zhang et al., 2023).

Although passive in nature, sanitary landfills are prone to material losses and exhibit notable burdens related to freshwater ecotoxicity, primarily due to the long-term leaching of additives and contaminants (Sabate and Kendall, 2024). Initial assessments by the European Commission (2012) identified heavy metals such as chromium as key contributors to toxicity, an observation corroborated by more recent findings on the release of molybdenum and vanadium from landfilled plastic waste (Zhang et al., 2023). Incineration, by contrast, enables partial energy recovery but results in elevated impacts across several critical categories. The complete oxidation of fossil carbon contributes substantially to climate change, while the release of pollutants such as NO_x , SO_2 , and heavy metals, notably mercury (European Commission, 2012), generates significant burdens in Human Toxicity and Terrestrial Acidification (Dri et al., 2018; Tonini et al., 2021). The specific polymer composition of the waste stream is also critical: chlorine-containing plastics (e.g., PVC) can lead to the formation of highly toxic dioxins and furans, while nitrogen-rich polymers (e.g., PU) may increase NO_x emissions (Antelava et al., 2019). Incomplete combustion can result in plastic-containing ash, creating a secondary pathway for microplastic pollution (Sabate and Kendall, 2024).

Consequently, the sorting efficiency of the plant, its ability to recover clean polymer fractions while minimizing non-recyclable residues, emerges as a key determinant of downstream environmental burdens. Even modest reductions in the residual stream can yield disproportionately large benefits, underscoring the importance of optimizing

sorting performance not only for material recovery but also as a strategic lever to mitigate the environmental load of residual management. This observation aligns with recent findings by Helmes et al. (2022), who emphasize that sorting performance and energy demand are critical drivers of the environmental profile of recycling systems: low sorting efficiency or elevated electricity consumption, particularly in small-scale facilities, can significantly erode the net environmental benefits of recycling.

MRFs are classified by size based on the recyclable material quantities processed: small-scale facilities handle less than 18–45 tons per day (5000–12,000 tons/year), whereas large-scale facilities process over 180 tons per day (45,000 tons/year). This study examined plastic waste with low contaminant levels from a separate collection and considered three distinct compositions to represent the investigated scenarios. The sorting rates for each MRF scenario are calculated using residue rates from the literature. Medium-sized MRFs have the lowest average residue rate (8 %), followed by small-scale MRFs (13 %) and large-scale facilities (19 %) (Bradshaw et al., 2025). Although large-scale facilities have higher residue rates than small-scale MRFs, they often manage part of their residues internally or transfer them to nearby processing units, thereby reducing secondary transportation needs and improving overall recycling supply chain efficiency. One primary challenge for small-scale MRFs is the generation of a significant fraction of nonrecyclable residues, which account for 13 % of the processed materials on average. These residues must be landfilled or sent to larger treatment facilities, increasing costs and the environmental impact of transportation. Furthermore, small-scale MRFs have lower levels of automation and operational efficiency. However, they are more geographically dispersed, reducing their need for long-distance transportation of recyclables, although the smaller transport vehicles used in these systems can have higher environmental impacts. A detailed inventory of a representative medium-scale sorting facility, including the process layout and equipment-specific energy consumption, is provided in the Appendix (Fig. A1 and Table A10).

The environmental contribution of the sorting stage within the overall recycling chain is far from negligible. In a recent consequential LCA of HDPE bottle packaging in Europe, Helmes et al. (2022) found that the combined “collection and sorting” stage can account for up to 20 % of the total climate change impact in MR systems, exceeding the burden associated with downstream reprocessing. Comparable contributions were also reported for fossil and mineral resource scarcity. These findings align with energy-based results from Bataineh (2020), who estimated that sorting and separation processes represent 26.7 % of total energy demand, making them the second-largest energy contributor after reprocessing. Inefficient sorting increases the proportion of material sent to final disposal, thereby reducing overall resource recovery and undermining the environmental benefits of recycling.

Material Recovery Facilities and recycling plants represent two critical stages within the plastic MR value chain. Recovered films and residual rigid plastics are first subjected to size reduction through granulation, producing polymer flakes. These flakes are then purified via friction cleaning to remove surface contaminants. Subsequent classification and washing are performed based on polymer type and colour, typically using optical sorting or density separation techniques. The process concludes with the extrusion of clean flakes to produce the final recycled product in granulated form.

After HDPE bales are sorted, they are sent to MR facilities, where they are washed and reprocessed into secondary raw materials in the form of recycled granules used in consumer goods manufacturing (Antonopoulos et al., 2021). The MR facility's efficiency η_r is described by the recycling rate parameter, defined here as the ratio of the target product (HDPE granules) to the amount of incoming material in bales. This process efficiency refers to the recycling rates (Antonopoulos et al., 2021) or recycling yields (Lase et al., 2023) of sorted bales. Additives including plasticizers, fillers, flame retardants, antioxidants, and colorants, which constitute approximately 7 % of plastics produced, remain

unaffected during MR and can affect recycled material properties, including strength, weight, and durability (Arena et al., 2023). The use of chemical agents (NaOH, detergents) can improve purity but creates wastewater disposal challenges (Uzosike et al., 2023).

Specifically, the MR chain processes are as follows:

1. Transportation: HDPE waste bales arrive at the MR facility.
2. Sorting: HDPE bales contain contaminants, including other plastics (low-density polyethylene [LDPE], polyethylene terephthalate [PET]). To enable recycling, the materials must be purified by separating HDPE from other plastics.
3. Grinding: The plastic is ground or cut into smaller pieces with increased surface areas for washing and drying. However, these pieces are still too large for reprocessing at this stage.
4. Washing: Plastics are often contaminated with food or other waste; washing removes these impurities.
5. Drying: The plastic must be dried completely before proceeding to additional processing steps.
6. Granulation: The plastic is ground into flakes to be remolded. This process turns the small pieces made during the grinding process into micro-pellets for compounding (Hou et al., 2018).
7. Compounding: The polymer is melted in an extruder; the extrudate is cooled to a solid phase upon exiting the die and then pelletized for reprocessing (Gandhi et al., 2021).

From a life cycle perspective, the MR stage involves several key environmental hotspots. Foremost among these is the electricity demand associated with core unit operations. Grinding, drying, and particularly extrusion and granulation (pelletising) are consistently identified as major contributors to Global Warming Potential and Fossil Resource Scarcity, especially within carbon-intensive energy systems (Uzosike et al., 2023; Hu et al., 2025). The second environmental hotspot concerns water consumption and effluent management. The washing phase requires considerable freshwater input and produces wastewater streams containing detergents, polymer fines, and organic contaminants. The treatment of these effluents can substantially influence impact categories such as Freshwater Ecotoxicity and Eutrophication, positioning this stage as a primary contributor to aquatic environmental burdens (Arena et al., 2023; Martín-Lara et al., 2022). In certain contexts, the water demand associated with recycling operations may be sufficiently high to offset part of the climate change benefits when compared to landfilling (DEFRA, 2022). A third critical environmental burden arises from non-recoverable material losses. Real-world facility assessments have shown that the final disposal of these process residues can dominate the overall life cycle, potentially tripling the environmental impact of the recycling system when managed through conventional landfilling without gas capture (Martín-Lara et al., 2022). Moreover, the persistence of legacy additives, such as plasticizers and flame retardants, may significantly affect Human and Ecosystem Toxicity, depending on their behaviour and fate during the extrusion phase.

The MR stage contributes approximately 15 % of the total climate change impact in HDPE systems, primarily due to the energy demand associated with washing, extrusion, and pelletising operations (Helmes et al., 2022). Although its climate burden is lower than that of collection and sorting, it remains a significant contributor to fossil and mineral resource depletion. The environmental performance of this stage is highly sensitive to both the regional energy mix and the quality of the incoming feedstock. Energy-based findings by Bataineh (2020) reinforce its relevance, indicating that reprocessing operations alone account for 54.4 % of the total energy demand within the HDPE recycling system.

Finally, the transport stage represents a cross-cutting component of the recycling chain and contributes non-negligibly to its overall environmental footprint. Emissions from fuel combustion in road freight are a major driver of impacts across multiple environmental categories, particularly Global Warming Potential. Road transport accounts for

approximately 27 % of total EU greenhouse gas emissions, reflecting an enduring dependence on fossil fuels, which supply 93 % of its final energy consumption (European Commission, 2020). Finally, the transport stage represents a cross-cutting component of the recycling chain and contributes non-negligibly to its overall environmental footprint. Emissions from fuel combustion in road freight are a major driver of impacts across multiple environmental categories, particularly Global Warming Potential. Road transport accounts for approximately 27 % of total EU greenhouse gas emissions, reflecting an enduring dependence on fossil fuels, which supply 93 % of its final energy consumption (European Commission, 2020). In addition to climate-related burdens, road transport is a significant source of air pollutants, responsible for 36 % of total NO_x emissions, 11 % of PM_{2.5}, and 8 % of non-methane volatile organic compounds (NMVOCs) within the EU (European Commission, 2020). The environmental relevance of NO_x is particularly notable, as it acts as a precursor to several impact categories, including terrestrial acidification, photochemical ozone formation, and secondary particulate matter formation (Fries and Hellweg, 2014).

Despite its importance, the logistical component is often underestimated in life cycle assessments. Depending on system configuration, transport distances, and vehicle efficiency, it can account for between 5 % and 55 % of total GHG emissions in integrated waste management systems (Liao et al., 2024). Mitigation strategies should therefore encompass not only route optimization but also informed decisions regarding vehicle capacity and fleet design, both of which can substantially reduce system-wide environmental burdens.

Our analysis compares three distinct configurations for HDPE recycling, each employing a specific structure and scale of facilities to reflect different territorial contexts and logistical approaches:

- **Centralized Recycling:** This scenario is modeled with a single, large-scale collection center feeding a single, large-scale sorting facility (MRF). This sorting facility, in turn, processes materials from this single collection center and forwards them to a single, large-scale recycling plant. This configuration is designed to maximize economies of scale at each processing stage, relying on a centralized logistics network.
- **Semi-distributed Recycling:** This configuration involves 10 collection centers that serve 3 medium-scale sorting facilities (MRFs). These medium-scale MRFs then consolidate their processed materials and forward them to a single, large-scale recycling plant. This model aims to balance economies of scale at the final recycling stage with reduced transport distances by decentralizing the sorting operations, with 3 sorting facilities strategically positioned to serve multiple collection points.
- **Distributed Recycling:** This scenario features a highly decentralized approach, comprising 50 collection centers. These centers supply 10 small-scale sorting facilities (MRFs), which are often co-located with or directly linked to 10 small-scale recycling plants. This configuration emphasizes proximity and localized processing to minimize logistical impacts, acknowledging that individual facility-level economies of scale may be limited.

These assumptions regarding the number and scale of facilities for each scenario are critical for our analysis, as they directly influence the modeled logistical requirements, energy consumption, and overall environmental performance. Further details on the capacity assumptions for each facility type can be found in the Supplementary Information (Table S1).

2.3. Life cycle inventory

The life cycle inventory was developed using a combination of primary and secondary data sources. Primary data were obtained from a case study conducted throughout the calendar year 2024 in Lombardy (Northern Italy), focusing on large-scale mechanical reprocessing of

post-consumer HDPE. Secondary data were sourced from the Ecoinvent database (version 3) to represent virgin HDPE production, consumable inputs, and end-of-life processes, including landfilling, incineration, transport, and electricity generation. A detailed breakdown of inventory flows is provided in Table A12 of the Appendix. Additionally, tertiary data from peer-reviewed literature were used to model the sorting facilities, small-scale mechanical reprocessing units, and the recycling system's logistical framework, encompassing the allocation of material flows across the interconnected processing stages and the scalability of the collection centers and sorting plants of various sizes.

This LCA evaluated the environmental performance, which depends on the recycling infrastructure scale and configuration, by modeling the processing of separately collected MPW from municipal solid waste (MSW) using two reference input flow scenarios: (i) for the centralized chain, an MSW plastic fraction stream, originating primarily from household waste streams and co-collected nonplastic contaminants (Rigamonti et al., 2014; Arena et al., 2023); and (ii) for the distributed chain, a stream characterizing average EU capture rates applied to generated PPW (Antonopoulos et al., 2021). Detailed compositions are provided in Table A3–A7 in the Appendix. This study focused on plastics sourced from separate collection schemes, thus excluding the potential environmental implications of managing mixed plastics recovered from residual MSW streams, which lie beyond the defined scope (Plastics Europe, 2022; Tonini et al., 2021). The JRC report on *Best Environmental Management Practice for the Waste Management Sector* (Dri et al., 2018) defines the capture rate (separate collection rate) as the ratio of the mass of a waste material segregated into a separate stream at the generation point to the total material mass generated, expressed as a percentage (Directive, 2008/98/EC; Antonopoulos et al., 2021). The remaining fraction that is not captured by the collection system will likely be directed to alternative waste management pathways (landfill, incineration, or retention within the unsorted mixed waste stream). The overall collection efficiencies were calculated as 0.58 for the centralized scenario (Rigamonti et al., 2014) and 0.41 for the distributed scenarios (Antonopoulos et al., 2021). Different collection efficiencies were modeled explicitly and propagated through the system while maintaining a consistent FU, ensuring methodological consistency and comparability. The energy consumption associated with the compaction station, which plays a role in plastic waste management, was relatively low. The average energy consumption across the three stations was 0.09 MJ per kilogram of compacted plastic waste (Arena et al., 2003). The principal environmental burdens of waste collection systems are predominantly attributed to their energy-intensive transportation, which causes significant air emissions; the energy consumption of compaction stations in plastic waste management is comparatively limited (Tonini et al., 2021).

The sorting rate is defined as the ratio of the mass of the target recyclable HDPE bale output stream from an MRF to the mass of the recyclable input stream entering the facility (Antonopoulos et al., 2021). The indicator (sorting rate) should be calculated at the plant level and expressed as the percentage of recovered material sent for recycling from the packaging plastic input stream; it can also be calculated for a single stream. The sorting rate achieved at the plant level by the MRF is an effective indicator when assessing the environmental performance of the sorting process (Dri et al., 2018). HDPE exhibits the highest sorting rate when considered a single stream flow among polymeric materials (Antonopoulos et al., 2021). Currently, sorting and recycling rates vary considerably among the EU member states and are influenced by the quantity, type, purity, and complexity of materials processed and the variety, type, and arrangement of technologies used in recycling facilities (Villanueva and Eder, 2014; Cimpan et al., 2016; Ragaert et al., 2017).

For large-scale MRFs, the reference waste stream was MPW, specifically post-consumer plastic packaging. This composition was based on estimates from Lase et al. (2023), who conducted a material flow analysis at the European level. The waste stream contains various fractions,

including polymers from multiple sectors, metals, textiles, paper, and inert materials. The quantities and compositions were established through direct measurements performed by the Italian National Consortium for the Collection and Recycling of Plastic Packaging (Corepla, 2023). The specific reference MSW stream composition was outlined by Arena et al. (2003), who showed that it contains approximately 70 % nonrecyclable plastic packaging and 30 % other materials. After the sorting phase, the HDPE content of this stream was determined to be 9 % (Arena et al., 2003). Rigamonti et al. (2014) also examined the plastic composition of MSW. Using data from Corepla (2023) to characterize the plastic waste composition, they found that, after the collection phase, HDPE represents approximately 9 % of the total; these findings are similar to those of Arena et al. (2003). For the medium-scale MRF facility, the waste stream was MPW sourced from residential plastic packaging; specifically, the waste comprised used liquid containers made from PET or polyethylene (PE) and gathered via a mono-material collection system. The collected plastic waste composition was based on data from Corepla (2023) for Italy and supplemented by data from sorting stations and reconditioning units provided by the companies considered in the study (Arena et al., 2003). The study reported a sorting rate of approximately 19 %. The sorting rate for small-scale MRFs was derived from the data presented by Antonopoulos et al. (2021), which considered a PPW stream generated in the EU in 2017, with its composition based on a report by Plastics Europe (2022). Subsequently, the waste quantities and compositions that actually entered the sorting facility were calculated using capture rates based on figures reported by Deloitte Sustainability, 2017. The quantity of HDPE relative to the total incoming stream was approximately 16 %, which corresponded to the plant's sorting rate for the target HDPE bale. The sorting rates and standard deviations of the target materials at the MRFs reported by Antonopoulos et al. (2021) were used to estimate the sorting and recycling system efficiencies while accounting for data variability. The scale factor plays an essential role in terms of energy consumption and environmental effects because of the different technologies implemented. The operational processes of an MRF were defined using secondary data from the literature (Rigamonti et al., 2014; Pressley et al., 2015) while addressing and overcoming the limitations highlighted in previous studies (Kerdlap et al., 2020, 2021).

Detailed methodological assumptions and data are provided in the Appendix, with inventory data for equipment used in sorting facilities summarized in Table A10. This study used the average consumption data from MRFs reported by Bradshaw et al. (2025) and Aguirre-Villegas et al. (2025), which indicate that large MRFs, with greater material sources and advanced sorting technologies, have electricity consumption per ton of recyclable material processed that ranges up to two orders of magnitude lower than that of small MRFs (5–90 kWh/t versus 300–550 kWh/t, respectively). Specifically, small-scale MRFs consume over 60 times more electricity than large-scale MRFs, with an average consumption of 373.9 kWh per ton of waste processed compared to only 6.1 kWh for large-scale facilities. Medium-scale MRFs use approximately six times more electricity per ton than their larger counterparts, with an average consumption of 36 kWh per ton of waste processed (Perugini et al., 2005). In line with the findings of Bradshaw et al. (2025), Pressley et al. (2015) reported an energy consumption range of 4.7–7.8 kWh per ton of input waste for advanced sorting facilities. These observations are further corroborated by North American data from a life cycle inventory commissioned by the Association of Plastic Recyclers (APR), which reports a weighted average electricity consumption of approximately 16.4 kWh per metric ton of post-consumer material processed in MRFs (APR, 2022).

The recycling process was analyzed in this study using the case of a large-scale recycling facility in Northern Italy. The data for this reprocessing facility were based on direct real figures. Energy consumption data for the small-scale version were calculated and scaled from those of the larger facility using insights from Uzosike et al. (2023), who investigated and detailed the efficiencies of extruders of various

sizes. Electricity consumption data measured directly from the large-scale recycling facility indicated an average of approximately 935.85 kWh per ton of processed waste compared to 1149.87 kWh for the scaled-down small-scale facility. Water consumption (3.02 m³/ton) and the various consumable materials are assumed to be the same for both facility sizes, based on the assumptions of Kerdlap et al. (2021). The reprocessing rate of 70.27 % calculated from the real case was assumed to be the same for both facilities, in line with Kerdlap et al. (2021), who reported a slightly higher value of 76 %. Perugini et al. (2005) and Arena et al. (2003) suggested an even better recycling rate of 88 %. A detailed breakdown of the input and output flows is provided in Table A11 in the Appendix, while a comprehensive description of the actual MR facility, including process diagrams and life cycle inventory (LCI) data, is available in Appendix A2 (see Fig. A25–A27 and Table A18–A19). The detailed workflow is illustrated in Figs. A25 and A26 in the Appendix.

Transport logistics during the collection stage were modeled using tertiary data from the JRC report by Tonini et al. (2021), which included details of collection schemes, distances, and transport types. This study accounted for the varying distances and transport modes between facilities within the recycling chain. Transport types were partially based on the assumptions of Kerdlap et al. (2021), while representative distances for average national-scale logistics configurations were selected to reflect realistic operating conditions. For the three recycling chain configurations centralized, semi-distributed, and distributed, the following transport distances and vehicle types were considered at each stage:

- Collection center to MRF: Distances of 90 km, 30 km, and 10 km were assumed for the centralized, semi-distributed, and distributed configurations, respectively. Light commercial vehicles were used in all cases.
- MRF to recycling facility: Distances of 90 km, 180 km, and 0 km were considered. Trucks over 32 tonnes were employed for the centralized and semi-distributed scenarios, while vehicles between 7.5 and 16 tonnes were used in the distributed configuration.
- MRF to incinerator/landfill: Distances of 10 km, 45 km, and 45 km were assumed. Transport was carried out using trucks between 7.5 and 16 tonnes, with light commercial vehicles additionally used in the distributed scenario.
- Recycling facility to goods producer: Distances of 90 km, 90 km, and 216 km were considered. Freight vehicles and trucks between 7.5 and 16 tonnes were used in the centralized and semi-distributed configurations, while light commercial vehicles were employed in the distributed case.

The residuals, defined as materials received by MRFs that cannot be recovered as recyclable materials, include contaminants, nonrecyclable materials, and other waste. Some of these residuals may technically be recyclable but are not captured during the separation and recovery processes because of various factors (Bradshaw et al., 2025). According to Bradshaw et al. (2025), residuals typically account for 13 % (range: 1 %–20 %) of the total for small-scale facilities, 8 % (range: 1 %–10 %) for medium-scale facilities, and 19 % (range: 3 %–39 %) for large-scale facilities. These findings indicate that larger facilities may experience higher incoming contamination levels or may show reduced material recovery efficiency compared to smaller facilities. Larger MRFs tend to report higher residual rates, primarily because of increased contamination at the input stage, the predominant use of automated sorting technologies (which may not achieve optimal efficiency), and reduced reliance on manual sorting compared to smaller facilities (Bradshaw et al., 2025). The residuals are assumed to be allocated in percentage terms as follows: 68 % for incineration and 32 % for landfills (Tonini et al., 2021).

Within the LCA context, the product environmental footprint (PEF) calculation methodology was used, integrating the circular footprint formula (CFF) to quantify the environmental impacts of recycled ma-

terial production and allocate the environmental burdens of recycling and virgin material production across subsequent product value chains. The potential recycled material production impacts are calculated using the CFF in Eq. (1). The net impact of replacing virgin material with recycled material was calculated as the difference between the environmental impact of introducing recycled material and that of producing displaced virgin material, per Eq. (2). This impact is adjusted for quality differences using the substitution factor $\zeta = Q_s/Q_p$, representing the ratio of secondary material quality to primary material quality, to provide a more accurate assessment of virgin material replacement. The delta (Δ) in Eq. (2) represents the resulting burden or savings relative to the amount of virgin material that could potentially be substituted in an unspecified application for the given polymer. To account for the substitution of virgin polymers with recycled alternatives, the default substitution factors recommended by the PEF methodology were applied. These factors, derived from technical and economic assessments of material performance, reflect the quality differences between recycled and virgin materials. For HDPE, a substitution factor $\zeta = 0.9$ (Rigamonti et al., 2014; Tonini et al., 2021) was used to quantify virgin polymer displacement. A positive Δ indicates environmental savings, whereas a negative Δ signifies a burden (Bataineh, 2020).

$$I_{mp} = (I_c + I_o + I_r) \cdot A + I_v \cdot (1 - A) \cdot \zeta \left[\frac{\text{impact}}{t} \right] \quad (1)$$

$$\Delta I = I_{mp} - I_v \cdot \zeta \quad (2)$$

where I_{mp} is the impact of recycled material production, I_c is the impact of plastic collection (impact/t), I_o is the impact of sorting activities (impact/t), I_r is the impact of recycling operations (impact/t), and I_v is the impact of virgin material production (impact/t). A positive ΔI indicates environmental savings, signifying that the use of recycled HDPE results in a lower environmental burden compared to virgin HDPE production (considering the quality substitution factor). Conversely, a negative ΔI signifies an environmental burden, meaning that the use of recycled HDPE results in a higher environmental burden. A is the default allocation factor for the allocation of market impacts; the PEF recommends a value of 0.5 for recycled plastics. ζ is the substitution factor that accounts for the quality differences between recycled and virgin materials. In this study, only one-time recycling of HDPE was considered, with a maximum of two life cycles. However, HDPE resin can typically be recycled at least 10 times (Gandhi et al., 2021). According to the PEF guidelines, $A = 0.5$ represents market equilibrium, where recycled content and end-of-life recycling are given equal importance. Furthermore, although the PEF methodology prescribes a default range of 0.2–0.8 for A , depending on market conditions, it also acknowledges that material- or application-specific values should be used when available (Zampori and Pant, 2019). The use of standardized allocation factors, as proposed by Zampori and Pant (2019) and Tonini et al. (2021), improves comparability across studies and avoids the over-crediting of recycled systems. However, this approach may reduce sensitivity to process improvements or to variations in recycled quality, especially in high-performance closed-loop systems. Although a default substitution factor of $\zeta = 0.9$ is recommended for HDPE, this value may still overestimate actual displacement in cases where recycled polymers show inferior mechanical properties or require blending to meet technical specifications.

2.4. Life cycle impact assessment (LCIA)

In accordance with ISO 14044, the LCIA phase involves classifying process emissions into impact categories in which they may contribute to effects on human health or the environment. This study used a midpoint approach, applying characterization factors to express the relative impact of each substance compared to a reference substance, thus ensuring consistency and transparency in the interpretation of the

results. For recycled resins, the impacts are separated by life cycle stages to highlight contributions from collection, sorting, reprocessing, and logistics. The following sections present the LCIA results for the selected impact categories.

Midpoint indicators (Huijbregts et al., 2017):

- Climate change: The global warming potential (GWP) quantifies the contributions of different GHGs to global warming, expressed in kg CO₂-equivalents to air.
- Stratospheric ozone depletion: The ozone depletion potential (ODP) measures the stratospheric ozone layer reduction caused by ozone-depleting substances, expressed in kg CFC-11-equivalents to air.
- Ionizing radiation: The ionizing radiation potential (IRP) evaluates the increase in ionizing radiation exposure from radionuclides, quantified in kBq Co-60-equivalents to air.
- Fine particulate matter formation: The particulate matter formation potential (PMFP) reflects the increased fine particulate matter (PM_{2.5}) formation caused by precursor pollutants (e.g., NH₃, NO_x, and SO₂), expressed in kg PM_{2.5}-equivalents to air.
- Photochemical ozone formation (human health): The human ozone formation potential (HOFPP) quantifies human exposure to ozone from NO_x and nonmethane volatile organic compound emissions, expressed in kg NO_x-equivalents to air.
- Photochemical ozone formation (ecosystems): The ecosystem ozone formation potential (EOFP) assesses ozone formation's impact on ecosystems with a focus on plant health, expressed in kg NO_x-equivalents to air.
- Terrestrial acidification: Terrestrial acidification potential (TAP) evaluates the effects of acidifying emissions (e.g., SO₂, NO_x, and NH₃) on terrestrial ecosystems, expressed in kg SO₂-equivalents to air.
- Freshwater eutrophication: The freshwater eutrophication potential (FEP) measures the impact of nutrient pollution, particularly phosphorus, on freshwater ecosystems, expressed in kg P-equivalents to freshwater.
- Toxicity (human health): The human toxicity potential (HTP) quantifies the potential health impacts of toxic chemicals, including carcinogenic and noncarcinogenic effects, expressed in kg 1,4-DCB-equivalents to urban air.
- Toxicity (ecosystems): The ecotoxicity potential (ETP) measures the impact of toxic emissions on ecosystems, including aquatic and terrestrial environments, expressed in kg 1,4-DCB-equivalents to industrial soil.
- Water use: The water consumption potential (WCP) evaluates the water volume consumed relative to the water extracted, expressed in m³ water-equivalents consumed.
- Land use: The land occupation potential (LOP) quantifies biodiversity loss caused by land use changes, particularly in agriculture, expressed in m²-year annual cropland-equivalents.
- Mineral resource scarcity: The surplus ore potential (SOP) measures the potential depletion of mineral resources, particularly metals such as copper, expressed in kg Cu-equivalents.
- Fossil resource scarcity: The fossil fuel potential (FFP) quantifies the scarcity of fossil fuels, expressed in kg oil-equivalents.

Fig. 4 compares the environmental impact results I_{mp} for the various recycling chain scenarios. The centralized recycling chain scenario demonstrated the highest impact across all indicators, with the exception of ionizing radiation and water consumption, where the distributed recycling scenario showed higher impacts. The semi-distributed recycling scenario demonstrated the most favorable performance, with consistently lower environmental impacts than the centralized scenario by $-48\% \pm 13.87\%$ (standard deviation, SD) across all environmental indicators. A numerical comparison of the three recycling scenarios is provided in Table A13 in the Appendix. These results highlight the clear environmental advantages of the semi-distributed configuration, which

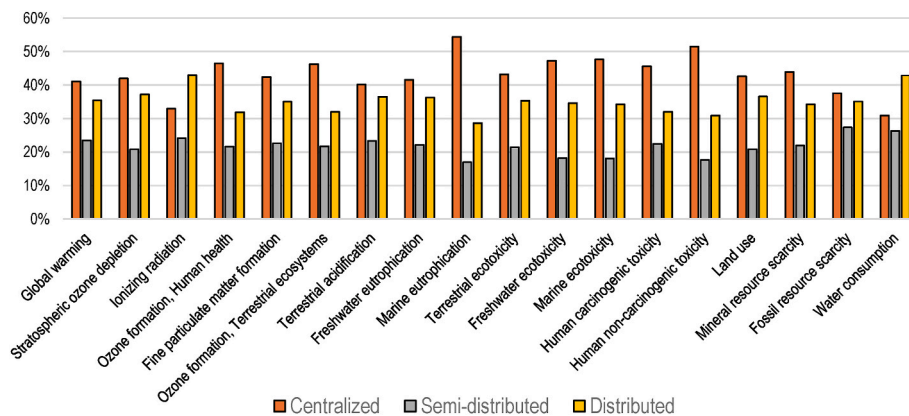


Fig. 4. Percentage comparisons of environmental impacts I_{mp} of the three recycling scenarios evaluated using midpoint (H) method indicators (in accordance with Joint Research Centre (JRC) guidelines). These percentages represent the impact of each scenario relative to the impact of producing virgin HDPE. The 100 % bar implicitly represents the environmental impact of producing 1 kg of virgin HDPE.

accomplished the most favorable trade-off between plant size and logistical efficiency. Specifically, significant impact reductions were observed for categories related to toxicity and ecosystem health, including marine eutrophication (−69 %), human noncarcinogenic toxicity (−66 %), and freshwater ecotoxicity (−62 %) (Wickerts et al., 2024). These reductions can plausibly be attributed to shorter transportation distances and the reduced reliance on long-haul fuel-intensive logistics, which contribute to emissions of pollutants such as NO_x and SO_2 . Furthermore, the semi-distributed scenario also demonstrated notable improvements in global warming (−43 %) and terrestrial acidification (−42 %), indicating a more efficient energy and resource profile at the plant level. Although the distributed configuration showed improved environmental performance over the centralized system, the semi-distributed model provided an optimal balance, achieving substantial impact reductions while avoiding the increased operational and infrastructural burdens that may affect smaller-scale distributed systems. A breakdown of category-specific impact reductions is available in Table A15 of the Appendix. This highlights the importance of using geographically optimized medium-scale facilities as a systemic strategy to reduce environmental impacts across multiple midpoint categories.

Fig. 5 presents the percentage change in environmental impact relative to virgin HDPE production, calculated as $(\Delta I / I_v) \times 100\%$. In this calculation, ΔI represents the net environmental impact difference between the recycling system and virgin HDPE production. As defined previously, a negative ΔI signifies an environmental saving (lower

impact), while a positive ΔI signifies an environmental burden (higher impact). These values were calculated following the CFF and substitution approach described by Tonini et al. (2021). When accounting for environmental credits related to the displacement of v-HDPE, the net environmental impact (ΔI) values consistently show greater environmental savings (negative values) across all scenarios compared to the initial midpoint results (I_{mp}). For instance, the semi-distributed configuration achieved an average environmental saving of 66.7 % in the global warming category, as indicated by the negative percentage in our calculation. These additional reductions are mainly attributed to the high environmental burden of v-HDPE production, particularly in categories such as climate change, fossil resource scarcity, and stratospheric ozone depletion (Tonini et al., 2021). In the fossil resource scarcity category, the environmental credits gained by replacing v-HDPE exceeded the total recycling process impacts in all three scenarios, with net reductions reaching −105 %, −144 %, and −112 % in the centralized, semi-distributed, and distributed configurations, respectively. A similar trend emerged in the global warming category, where the semi-distributed and distributed scenarios achieved net benefits of −96 % and −63 %, respectively. These results indicate that, in some categories, recycling can not only offset its own environmental footprint but also provide a net positive effect on the overall system balance. Other impact categories showed notable improvements, particularly when they are sensitive to transport emissions and fossil-based energy use. In the semi-distributed scenario, marked reductions were observed for

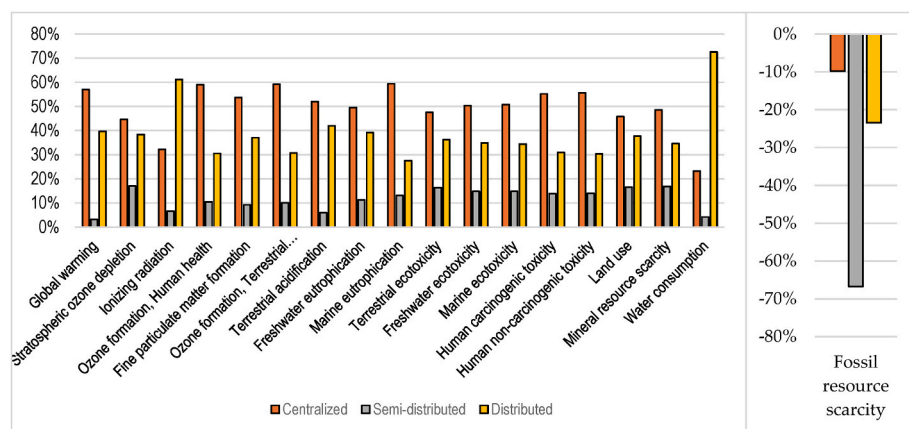


Fig. 5. Percentage comparisons of the environmental net impacts (ΔI) of the three recycling scenarios evaluated using midpoint (H) method indicators. The percentages represent the relative environmental burden or saving of each scenario compared to virgin HDPE production, calculated as $(\Delta I / I_v) \times 100\%$. A negative percentage indicates a saving (lower impact than virgin HDPE), while a positive percentage indicates a burden (higher impact than virgin HDPE). The impact of producing virgin HDPE (I_v) is implicitly represented as 0 % in this relative comparison.

photochemical ozone formation–human health (–75 %), terrestrial acidification (–77 %), and marine eutrophication (–91 %), confirming the benefits of the reduced transport distances and optimized process configurations (Tonini et al., 2021; Wickerts et al., 2024).

3. Results and discussion

The following analysis seeks to identify system-level environmental hotspots by examining the underlying drivers of the observed impacts. The subsequent figures illustrate the contribution of each life cycle stage to the overall environmental performance across all indicators, for each of the three recycling chain configurations. Impacts are disaggregated by process stage as follows: IT_c refers to the collection phase, including compaction but excluding transport; IT_o denotes sorting operations; IT_r corresponds to MR; IT_j_l represents the final treatment of non-recyclable residues; IT_v accounts for the production of virgin HDPE allocated to the recycled output; and IT_t encompasses all transport activities throughout the recycling chain.

Fig. 6 (1) shows that in the centralized recycling chain scenario, the highest impacts were related to the transportation phase, which included 12 of the 18 indicators. On average, transportation contributed 48.4 % ± 17 % (SD) to the total environmental impact. For marine eutrophication, freshwater ecotoxicity, marine ecotoxicity, and human noncarcinogenic toxicity, the most significant contribution came from the final disposal stage of nonrecyclable waste and various contaminants, with shares of 65.6 %, 46 %, 45.4 %, and 45.1 % of the total impact, respectively. Conversely, for fossil resource scarcity and water consumption, the most impactful stage was the allocated virgin production share, contributing 52 % and 40.7 % of the total impact, respectively. Fig. 6(2) presents the environmental impacts associated with the semi-distributed scenario. In this configuration, v-HDPE production emerged as the most impactful phase across eight of the total

impact indicators, contributing on average 32.4 % ± 15 % (SD) to the overall environmental burden. This was followed by the transportation phase, which dominated six impact indicators, with an average contribution of 28.7 % ± 12 % (SD). The final disposal stage of nonrecyclable waste and various contaminants was identified as the most significant contributor to the three impact indicators. Finally, the sorting phase showed the highest contribution in the case of mineral resource scarcity. Fig. 6(3) illustrates the distribution of environmental impacts for the distributed scenario. In this case, the sorting phase dominated the impact profile as the most burdensome stage across eight impact indicators, contributing on average 27.7 % ± 8 % (SD) to the overall environmental burden. This was followed by the transportation phase, which made the highest contribution to five indicators, with an average contribution of 25.4 % ± 10 % (SD). The production of v-HDPE was the leading contributor to three indicators, while the final disposal stage of nonrecyclable waste and various contaminants showed the greatest influence on marine eutrophication and human noncarcinogenic toxicity.

Fig. 7 illustrates the phase-wise breakdown of the net environmental impacts when considering the substitution of virgin material with recycled material for each of the three recycling scenarios. To complement the scenario-specific contributions, a comparative analysis was conducted to evaluate the environmental burden of each life cycle phase across the three system configurations. This phase-oriented perspective, presented in Fig. 8, identifies the dominant contributing scenario for each phase and highlights the inter-scenario variability in environmental impacts across the eighteen ReCiPe Midpoint (H) indicators, as reflected by the error bars.

When limited to compaction activities, the collection phase contributes minimally to the overall environmental burden, averaging 3.8 % ± 2.1 %, 5.1 % ± 1.8 %, and 4.1 % ± 1.2 % in the centralized, semi-distributed, and distributed scenarios, respectively. However, when transport associated with residential curbside collection is included, the

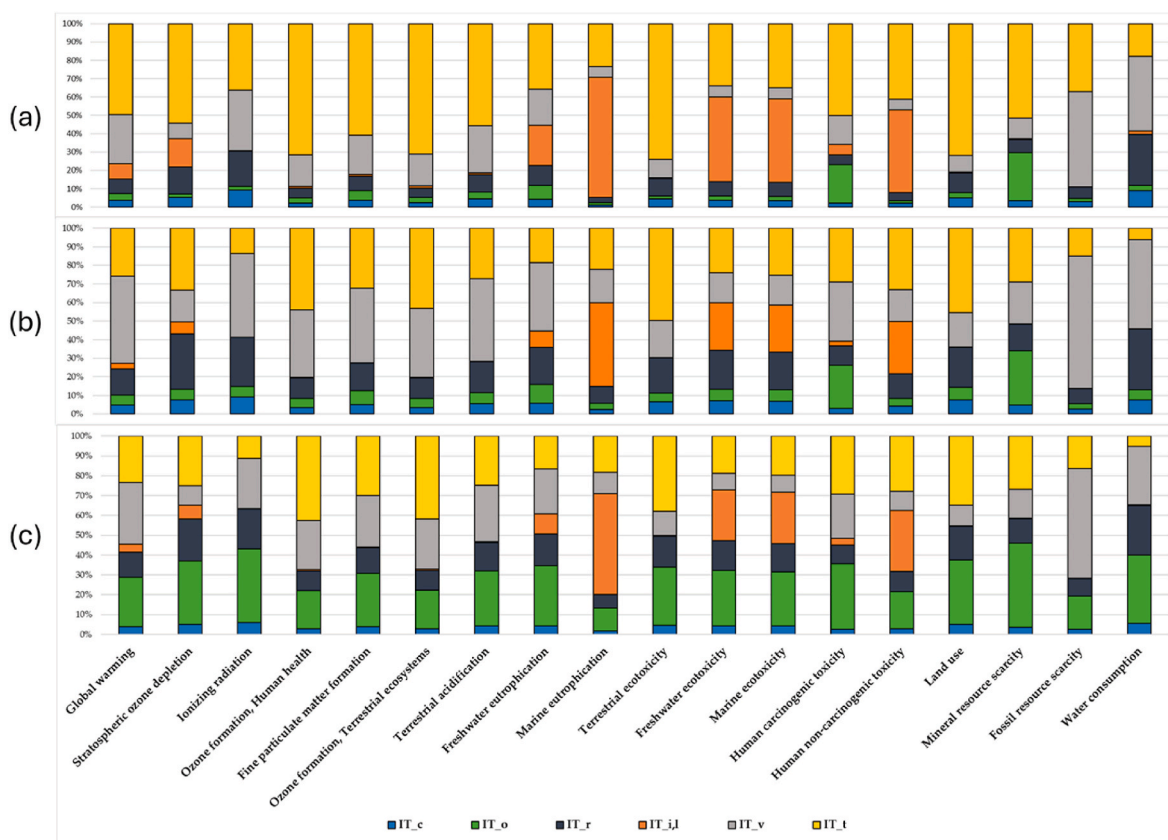


Fig. 6. Percentage characterizations of environmental impacts for the stages evaluated using midpoint (H) method indicators for (a) centralized recycling chain, (b) semi-distributed recycling chain, and (c) distributed recycling chain.

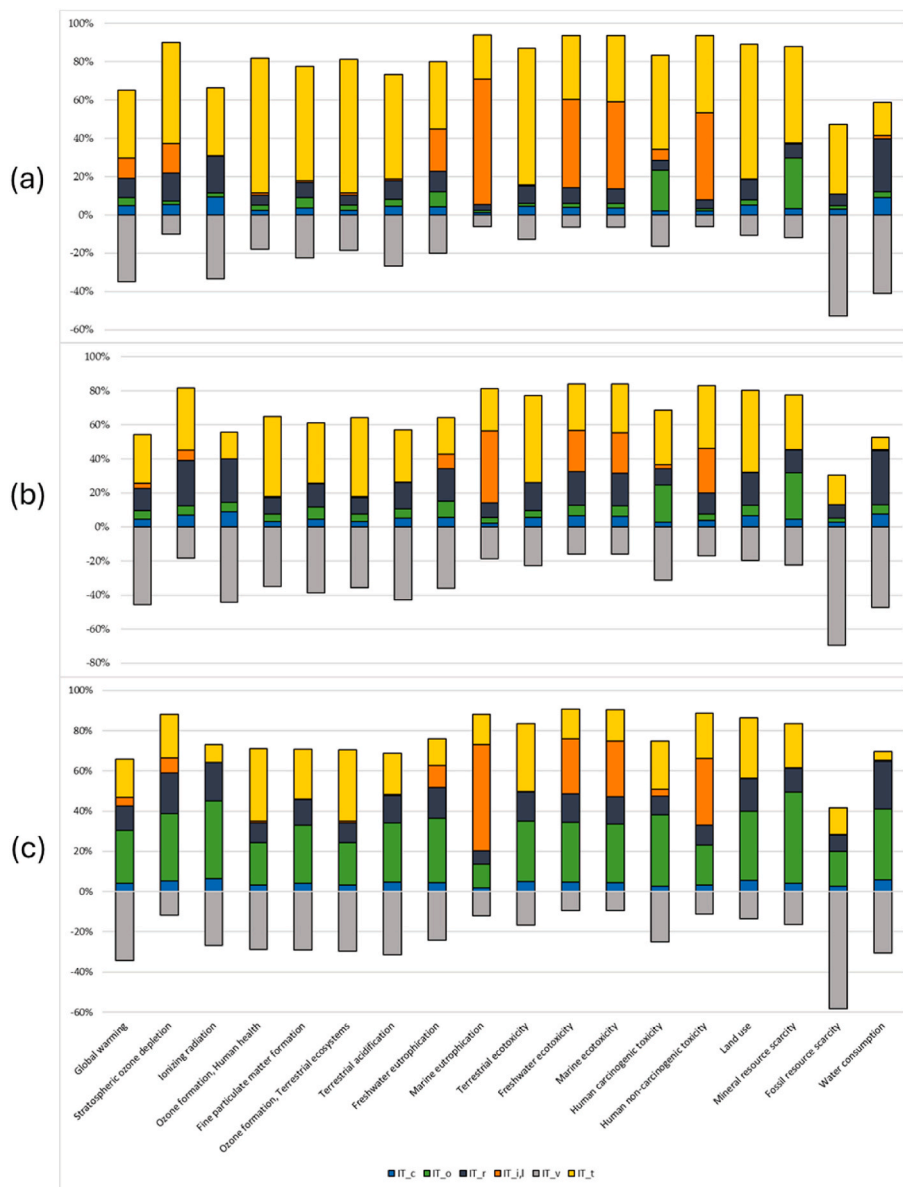


Fig. 7. Percentage characterizations of net environmental impacts (ΔI) for the stages evaluated using midpoint (H) method indicators for (a) centralized recycling chain, (b) semi-distributed recycling chain, and (c) distributed recycling chain. Each stacked bar represents the total net impact (ΔI) for each scenario. The components of the stacked bar indicate the contribution of each life cycle stage to this net impact: IT_c (Collection, excluding transport), IT_o (Sorting), IT_r (Material Recovery/Recycling), IT_i,l (Final Disposal of Non-recyclable Residues), IT_v (Virgin Material Avoidance, i.e., credit for not producing virgin HDPE), and IT_t (Transport). A negative contribution of IT_v signifies a saving achieved by avoiding virgin material production. The total height of the bar represents the sum of these contributions, showing the overall net environmental impact (ΔI) relative to virgin HDPE production.

average contribution rises to $10.8 \% \pm 2.85 \%$, $14.4 \% \pm 3.37 \%$, and $12.0 \% \pm 3.16 \%$, respectively. Emissions from transport in this phase alone account for 12 %, 28 %, and 42 % of total transport-related impacts in the centralized, semi-distributed, and distributed configurations. This limited contribution is primarily attributable to the low energy intensity of compaction operations, the short distances involved in local waste collection, and the marginal influence of this phase on downstream processing. Aggregated across all configurations, the centralized system accounts for the largest share of impacts from the collection phase (38.8 %), followed by the distributed (33.1 %) and semi-distributed (28.1 %) layouts, as illustrated in Fig. 8.

The sorting phase emerges as the most configuration-sensitive stage within the recycling chain, displaying marked disparities in its contribution to the overall environmental burden. On average, it accounts for $5.2 \pm 6.7 \%$ and $7.4 \pm 6.3 \%$ of total impacts in the centralized and semi-distributed systems, respectively, with considerable intra-system

variability (median values: 3 % and 5 %). By contrast, the distributed layout contributes an average of $27.7 \pm 8.0 \%$, with notably lower dispersion across impact indicators. When impacts are aggregated across all configurations, the distributed system alone accounts for 74.6 % of the total burden associated with sorting, compared to 13.8 % and 11.6 % for the centralized and semi-distributed approaches. This pronounced imbalance reflects the significant energy inefficiencies inherent to small-scale MRFs, typical of distributed systems. As shown in Fig. A2 (Appendix), electricity consumption is the dominant contributor to the environmental burden of the sorting phase in the distributed scenario, accounting for $61.3 \pm 20.0 \%$ of total impacts. In contrast, the centralized system is primarily influenced by packaging consumables, particularly steel strapping, LDPE film, and wooden pallets, which together account for $96.4 \pm 2.0 \%$. These materials disproportionately affect impact categories such as Human Carcinogenic Toxicity and Mineral Resource Scarcity. Fig. A3 further disaggregates these contributions by

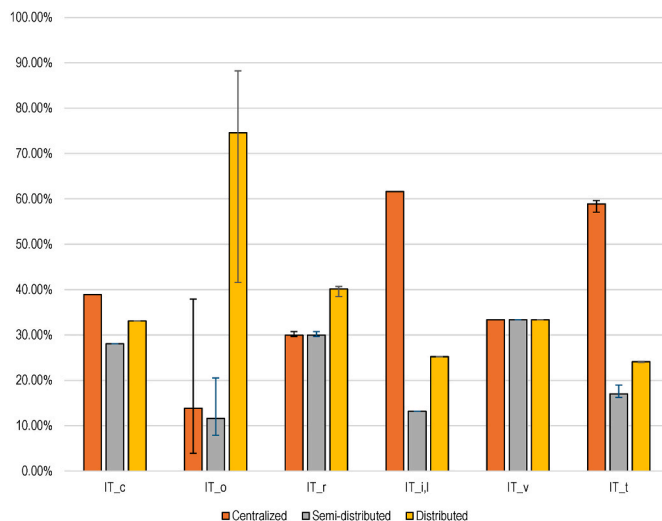


Fig. 8. Average contributions per configuration and life cycle phase, normalised across scenarios. Error bars show standard deviation over ReCiPe Midpoint (H) indicators, reflecting intra-phase variability.

material type, underscoring their role in upstream resource intensity and toxicity potential. In distributed systems, elevated electricity demand drives most impact categories, thereby reducing inter-indicator variability. Conversely, in centralized and semi-distributed layouts, lower electricity use allows the environmental burden of packaging materials to emerge more prominently in specific categories. Although the centralized facility benefits from greater energy efficiency, its lower sorting performance necessitates a higher input volume, resulting in a greater environmental burden than that observed in the semi-distributed system, which achieves a more favorable balance between energy demand and sorting effectiveness. This divergence in dominant inventory flows, electricity in distributed systems versus packaging materials in centralized and semi-distributed configurations, accounts for both the magnitude and variability of sorting impacts across scenarios, as clearly illustrated by the extended error bars in Fig. 8.

The recycling stage contributes, on average, $9.3 \pm 5.8\%$, $16.1 \pm 6.6\%$, and $13.3 \pm 4.3\%$ of the total environmental impact in the centralized, semi-distributed, and distributed scenarios, respectively. When impacts are aggregated, the burdens are almost equally shared between the centralized and semi-distributed configurations (31.17% each), reflecting their common reliance on a large-scale reprocessing facility. In contrast, the distributed system accounts for a slightly higher share (37.67%) due to the use of multiple decentralized modular units. The variability across scenarios remains relatively limited, confirming electricity consumption as the predominant driver of this phase. As discussed in Section 2.2, extrusion and granulation are particularly energy-intensive operations. In addition to energy use, this stage imposes a notable burden on water resources, owing to the freshwater requirements of the washing phase and the associated wastewater treatment. The Stratospheric Ozone Depletion indicator also reveals a significant impact, driven by both the regional electricity mix and the treatment of impurities such as paper pulp and residual sludge. These processes contribute to Freshwater and Marine Eutrophication as well, although their relative importance is moderated by the more substantial impacts arising from the downstream treatment of non-recyclable residues. Overall, while this stage represents the technological core of the recycling chain, it does not emerge as the dominant contributor to the total environmental burden.

The treatment of non-recyclable residues has a disproportionately high impact on specific environmental categories, particularly Marine Eutrophication, Freshwater Ecotoxicity, and Human Non-Carcinogenic Toxicity. In the centralized scenario, reduced sorting efficiency

significantly increases the volume of residual waste requiring disposal, thereby amplifying the associated environmental burdens. On average, this phase contributes 14.5% , 7.8% , and 9.5% of total system impacts in the centralized, semi-distributed, and distributed configurations, respectively. Although these contributions are substantial, indicator sensitivity remains elevated, with median values ranging from 0.8% to 1.5% across scenarios and a distinctly skewed distribution. When impacts are aggregated, the centralized configuration accounts for 61.62% of the cumulative burden associated with this phase (Fig. 8), followed by the distributed (25.22%) and semi-distributed (13.16%) systems. Inter-scenario variability is minimal due to the near-identical distribution of impacts across categories, highlighting the critical role of sorting efficiency in shaping downstream environmental pressures and reinforcing the significance of end-of-pipe processes within centralized systems.

Transport consistently emerges as one of the most impactful and configuration-sensitive stages within the system. On average, it contributes $48.7 \pm 16.7\%$, $28.7 \pm 12.4\%$, and $25.4 \pm 9.8\%$ of total environmental impacts in the centralized, semi-distributed, and distributed configurations, respectively. As shown in Fig. 8, the centralized system alone accounts for 59.24% of the cumulative transport-related burden, followed by the semi-distributed (21.20%) and distributed (19.56%) layouts. This pronounced disparity reflects the distinct logistical structures inherent to each configuration. Fig. A4 (Appendix) provides a detailed breakdown of transport-related impacts, disaggregated by individual route segments. Among these, the leg connecting the collection center to the sorting facility consistently ranks as the most burdensome in both centralized and semi-distributed systems, accounting for over 80% of logistics-related burdens in the centralized configuration, over 40% in the semi-distributed, and over 36% in the distributed scenario, primarily due to longer distances covered by smaller, less efficient vehicles. In centralized configurations, characterized by fewer but more spatially dispersed nodes, vehicle type emerges as a critical determinant of transport-related environmental performance.

To enhance the interpretation of the system's environmental performance, an upstream aggregation of impacts was carried out, encompassing all pre-recycling operations, including collection, sorting, final disposal of non-recyclable residues, and associated transport activities, as illustrated in Fig. A5 (Appendix). The aggregated analysis reveals that the upstream phase constitutes the principal environmental hotspot across all configurations, with average total contributions of $70.47 \pm 15.98\%$ for the centralized system, $43.26 \pm 13.60\%$ for the semi-distributed layout, and $54.95 \pm 11.48\%$ for the distributed configuration. These findings clearly indicate that the upstream phase is the dominant contributor to overall environmental impacts in all scenarios, primarily due to the burdens associated with logistics and pre-treatment operations.

The base case assumes a constant, linear waste flow through the chain, with one facility per stage operating under steady-state conditions. The analysis was subsequently extended to full system scale by explicitly modelling logistics and capacity constraints across the chain. Nominal capacities for collection centers, material recovery facilities, recycling plants, and thermal treatment units show considerable variation in the literature.

Crístiu et al. (2024) classify sorting centers as small ($\leq 50 \text{ kt y}^{-1}$), medium ($50\text{--}100 \text{ kt y}^{-1}$), and large ($>100 \text{ kt y}^{-1}$), with similar thresholds reported for incinerators. For collection centers, the corresponding ranges are $\leq 25 \text{ kt y}^{-1}$, $25\text{--}50 \text{ kt y}^{-1}$, and $>50 \text{ kt y}^{-1}$. Bradshaw et al. (2025) define small MRFs as processing $<1 \text{ kt y}^{-1}$, medium as $1\text{--}10 \text{ kt y}^{-1}$, and large as $>10 \text{ kt y}^{-1}$. Alternative classifications from Colville and McFeron (1994) and Tetra Tech Canada (2019) distinguish small plants ($5\text{--}12 \text{ kt y}^{-1}$) from large plants ($\approx 45 \text{ kt y}^{-1}$). Empirical data from Bradshaw et al. report operational ranges of $0.16\text{--}0.61 \text{ kt y}^{-1}$ for small sorting facilities, $3.73\text{--}7.58 \text{ kt y}^{-1}$ for medium-scale plants, and $390\text{--}560 \text{ kt y}^{-1}$ for large installations. Similarly, Kerdlap et al. (2021) estimate large sorting facilities at $8.93\text{--}30.91 \text{ kt y}^{-1}$ and small ones at

1.768–3.071 kt y⁻¹; on a daily throughput basis, these correspond to 25–30 t d⁻¹ for large plants and 5–6 t d⁻¹ for small ones. These daily ranges were also applied to recycling facilities, where large plants typically receive 25–30 t d⁻¹ and small ones 5–6 t d⁻¹. Scaling these figures to the industrial case study, where the reference recycling plant processes 13.7 kt y⁻¹, yielded an estimated nominal capacity of approximately 2.75 kt y⁻¹ for a small-scale recycling facility.

In the model, collection centers were parameterized at 25 kt y⁻¹ (±20 %), 75 kt y⁻¹ (±20 %), and 125 kt y⁻¹ (±20 %) for small, medium, and large facilities, respectively. Sorting centers were set at 8.5 kt y⁻¹ (±29 %), 35 kt y⁻¹ (±29 %), and 175 kt y⁻¹ (±14 %). Recycling plants were configured at 2.75 kt y⁻¹ (±9 %) for small-scale operations and 13.7 kt y⁻¹ (±5 %) for large-scale facilities. These values were dynamically adjusted by the genetic optimization algorithm within the specified bounds to maximize system performance. Scaling relationships across varying waste volumes are illustrated in Fig. A6, highlighting how infrastructure requirements shift with regional feedstock availability. Capacity variability was assumed to decrease with increasing process complexity, reflecting tighter operational tolerances in downstream MR compared to upstream collection and sorting.

The optimization targeted facility utilization rates of 85–95 % of nominal capacity (as shown by the shaded band in Fig. 9), minimized mass-balance deviations, and constrained the total number of facilities

to reduce infrastructure investment and operational complexity. Table A14–A15 (Appendix) present the optimized facility capacities derived from the algorithm for six representative geographical scenarios. Fig. 9 summarizes the outcomes, showing that in low-population-density areas, centralized systems tend to be oversized, making distributed chains more suitable. In more densely populated regions, semi-distributed configurations can be appropriately dimensioned, while centralized systems are generally viable only in predominantly urban contexts, where feedstock volumes consistently support large-scale operations. Logistical mismatches between facility capacity and feedstock availability may lead to underutilization or congestion, both of which undermine the economic performance and profitability of the recycling chain.

To assess environmental impacts across diverse geographic contexts, location-specific distance multipliers were applied (Fig. A7, Appendix) to systematically adjust transport distances from a baseline scenario, thereby reflecting regional variability. The urban scenario was selected as the reference case, based on the premise that densely populated areas typically benefit from proximity to established infrastructure networks, resulting in shorter inter-facility distances throughout the supply chain.

A comprehensive environmental assessment was conducted for all three processing configurations, centralized, semi-distributed, and distributed, across six distinct geographic zones, using the optimized

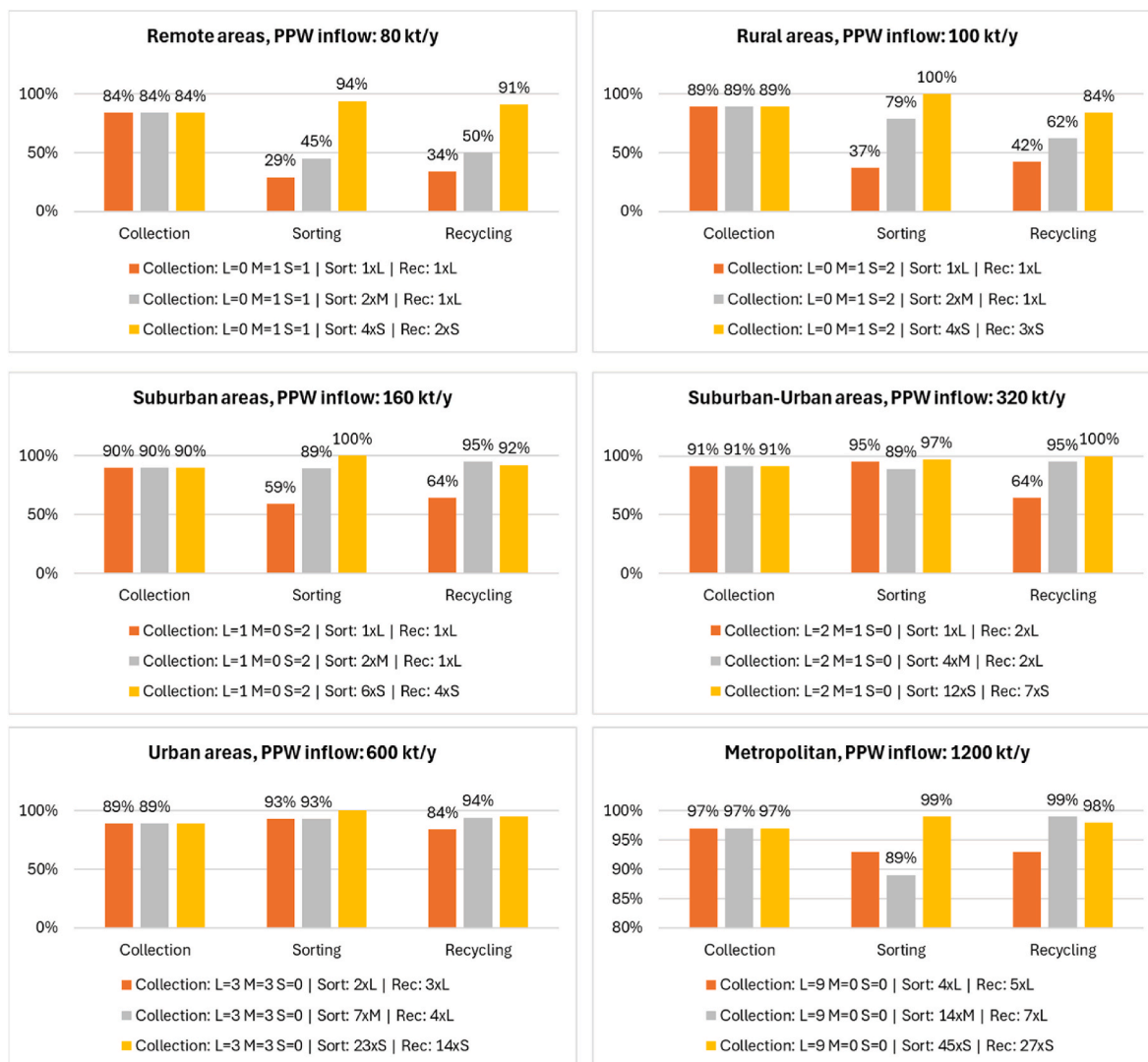


Fig. 9. Plant utilization rates in six different strategic regional configurations (where L = Large-scale facilities, M = Medium-scale facilities, S = Small-scale facilities).

facility layouts derived from the genetic algorithm. Fig. 10 presents the distribution of environmental impacts for the Global Warming Potential indicator, offering key insights into the geographic sensitivity of HDPE recycling systems.

The results show that transport-related impacts represent a significantly larger share of the total environmental burden in low-population-density areas, with this effect most pronounced in centralized configurations. Across all geographic contexts, the semi-distributed scenario consistently emerges as the most balanced option, optimizing processing efficiency while minimizing transport requirements. Notably, a clear convergence trend is observed in the performance of centralized systems as population density increases, with urban and metropolitan areas exhibiting markedly lower transport penalties compared to rural and remote regions.

4. Sensitivity analysis

To assess the robustness of the comparative evaluation and to explore how key logistical and operational parameters shape the environmental performance of HDPE recycling systems, a comprehensive sensitivity analysis was conducted. This included both scenario-based and probabilistic approaches, focusing on critical hotspots identified in the baseline model: transportation logistics, facility efficiency, and the management of non-recyclable residues. Scenario-based analyses investigated the effects of varying travel distances, vehicle types, geographic contexts, and energy demand assumptions, while a Monte Carlo simulation was used to quantify uncertainty propagation across the collection and sorting stages. The results provide quantitative insights into system resilience and highlight the primary drivers of environmental impact across the different configurations.

The first key parameter, travel distance, was varied by $\pm 30\%$ relative to the base case. Results indicate that the semi-distributed configuration consistently delivers the lowest environmental impacts, while the centralized system remains the most sensitive to changes in transport length. Nonetheless, targeted optimization of routing could substantially reduce its footprint, especially in categories driven by fuel combustion, such as ozone formation, fine particulate matter, terrestrial ecotoxicity, land use, and acidification. Summary results are provided in Fig. A8 and Table A16 (Appendix).

The second key parameter, vehicle category, was assessed through a targeted sensitivity analysis, with summary results shown in Fig. 11 and full breakdowns quantified in Table A17 (Appendix). The analysis compared light commercial vehicles with medium-duty lorries (3.5–7.5

t) within the centralized configuration, reflecting divergent practices reported in the literature. Switching to medium-duty lorries led to substantial reductions in transport-related impacts: up to 48.9 % for ozone formation and 41.6 % for fine particulate matter, with an average decrease of 29 % across all impact categories. Despite these improvements, the semi-distributed system remains the most balanced option overall. These findings suggest that logistical optimization, particularly through vehicle selection and payload densification, could render the centralized system competitive or even superior to distributed alternatives. A further analysis extended the vehicle variation across all three configurations, with comparative results presented in Fig. A9–A10 (Appendix).

In the distributed scenario, the sorting phase emerges as a critical hotspot, primarily due to the high energy intensity of small-scale facilities. To explore its influence, a sensitivity analysis was conducted assuming a 50 % reduction in electricity demand. Results show an average impact decrease of $12.54\% \pm 3.57$, with the most affected categories being those sensitive to the electricity mix: ionizing radiation (-19.2%), water consumption (-17.3%), and land use (-16.5%). These findings indicate that energy efficiency improvements in decentralized sorting could yield meaningful environmental benefits. Fig. A11 in Appendix illustrates the comparative performance of the three configurations under this improved-efficiency scenario, confirming that the semi-distributed system remains the most favorable option overall.

The final disposal stage emerges as a hotspot for several impact categories, particularly in the centralised scenario, where low sorting efficiency and high residue rates in large-scale facilities amplify environmental burdens. A sensitivity analysis assuming a 50 % reduction in residue rates is presented in Appendix Figure A12, showing notable improvements in key indicators. Marine eutrophication decreased by 33 %, while freshwater ecotoxicity, marine ecotoxicity, and human non-carcinogenic toxicity each declined by approximately 23 %. Nonetheless, the centralised configuration remains less favorable compared to the other system options.

Increasing the sorting efficiency of large-scale facilities by 67 % results in a 40.19 % reduction in impacts from collection, sorting, and disposal, and a $38.78\% \pm 0.76\%$ decrease in transport-related impacts. As shown in Appendix Figure A13, this shift makes the distributed configuration the least favorable across most categories. A further sensitivity analysis on medium-scale efficiency ($\pm 30\%$) confirms the robustness of the base-case results (Appendix Figure A14).

A combined sensitivity analysis addressing both transport mode and sorting efficiency demonstrates that targeted optimization within the

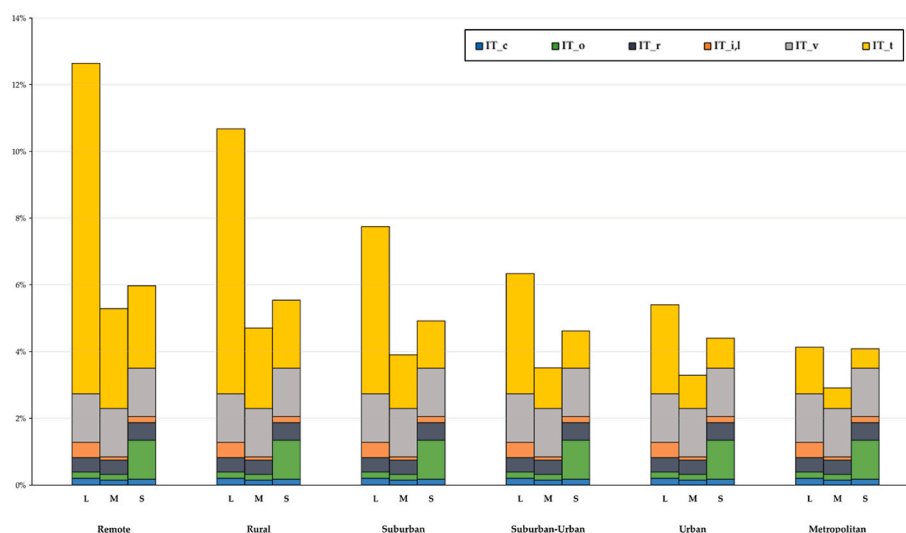


Fig. 10. Relative environmental impact distribution across geographic scenarios and system configurations for global warming potential, with phase-specific breakdown (where L = Large-scale facilities, M = Medium-scale facilities, S = Small-scale facilities).

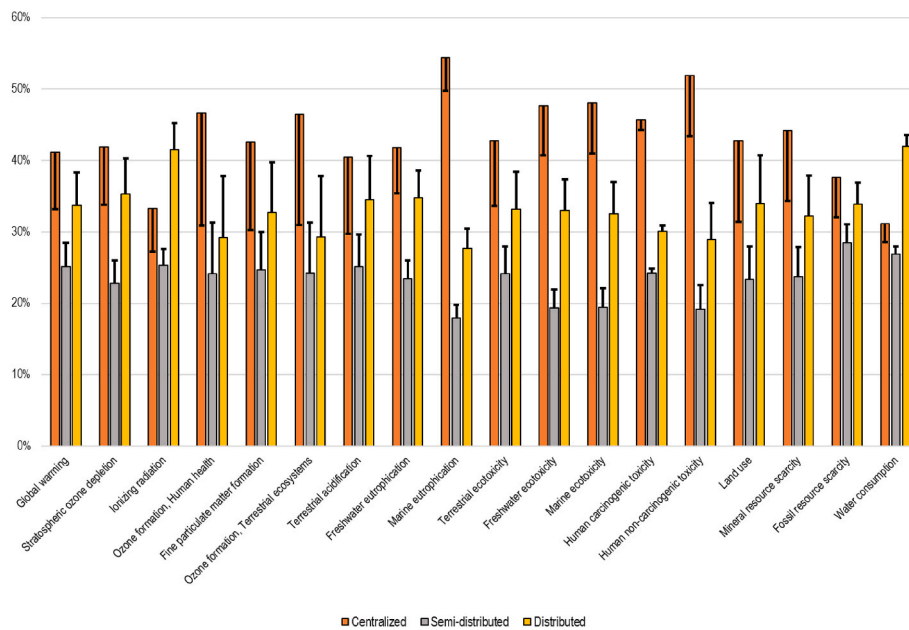


Fig. 11. Relative environmental impacts (I_{imp}) for the three recycling chain configurations under the alternative centralized scenario employing medium-duty lorries for the collection-centre-to-sorting-facility (co) segment.

centralised scenario can substantially enhance its environmental performance. As illustrated in Fig. A15 (Appendix), under these improved conditions, the centralised and semi-distributed configurations exhibit comparable or superior results across the majority of impact categories. Nonetheless, elevated burdens persist for Marine Eutrophication and Human Carcinogenic Toxicity, indicating residual limitations in the centralised approach.

Finally, a Monte Carlo simulation with 10,000 iterations was performed to evaluate uncertainty propagation and the robustness of scenario rankings within the HDPE recycling chain. The results confirmed the semi-distributed configuration as the best overall performer across all impact categories, with statistically significant differences in mean global warming potential values: 1.97 kg CO₂eq for the semi-distributed model, compared to 2.44 kg for the distributed and 2.94 kg for the centralized configuration. A variance decomposition analysis identified sorting efficiency as the most influential parameter, accounting for 43.9 % of the explained variance in global warming impacts. This was followed by HDPE collection efficiency (32.7 %), while the actual recycling process efficiency contributed only 6.0 %. Positively correlated sampling further validated the stability of the comparative outcomes. Full methodological details and statistical outputs are provided in Appendix Figures A16–A25 and Table A3–A9.

5. Conclusion

This study explored the inherent trade-off between the environmental benefits of recycling and the associated effects of transportation by examining alternative logistical configurations for HDPE recycling. Decentralized recycling scenarios, characterized by smaller facility scales and reduced transportation distances, are proposed as alternatives to conventional centralized systems. By comparing the integrated models directly with conventional centralized recycling and non-recycling systems, this study quantifies the impact of reduced transportation distances on the overall environmental sustainability of HDPE recycling. The limitations of previous studies that have either overlooked or inadequately accounted for the critical role of logistics in the CE of plastics are also addressed.

This study clearly demonstrates that the semi-distributed scenario yields the lowest environmental impact because of a favorable trade-off

between optimized logistics and a facility scale that is not reduced excessively, allowing for both operational efficiency and environmental sustainability. This synergy provides lower direct burdens and greater environmental credits, particularly in fossil- and transport-sensitive impact categories.

Sensitivity and geographic analyses reinforced the optimality of the semi-distributed configuration, highlighting reduced transport burdens and balanced process efficiencies as key drivers of its superior environmental performance. Monte Carlo simulation subsequently confirmed the statistical robustness of these findings, with the semi-distributed scenario consistently outperforming alternatives across all uncertainty scenarios.

One limitation of this study lies in the assumption of source-separated plastic waste streams, which may not accurately represent conditions in regions lacking dedicated collection systems. The nature of upstream waste collection is a critical determinant of system performance, and substantial geographical variability introduces uncertainty that can significantly affect both efficiency and environmental outcomes. A further limitation is the exclusive focus on environmental indicators, with economic aspects addressed only qualitatively and without a full cost-effectiveness comparison of the recycling configurations. Moreover, although the framework accounts for geographic variation and facility scaling, the logistics modelling is based on literature-derived distance assumptions, which may not fully capture site-specific routing efficiencies in all regional contexts.

CRediT authorship contribution statement

Davide Gotti: Investigation, Formal analysis, Data curation, Conceptualization. **Daniele Landi:** Writing - conceptualization - review and editing, Writing – original draft, Supervision, Methodology, Funding acquisition. **Christian Spreafico:** Writing – Conceptualization - Review and editing, Supervision. **Davide Russo:** Writing – review & editing.

Declaration of competing interest

The authors have declared no conflict of interest.

Appendix A. Supplementary data

Supplementary data to this article can be found online at <https://doi.org/10.1016/j.jclepro.2025.147062>.

Data availability

No data was used for the research described in the article.

References

- Aguirre-Villegas, H.A., Bradshaw, S.L., Benson, C., 2025. Effect of scale on the energy and carbon footprint of material recovery facilities (MRFs) in the United States. Preprint. <https://doi.org/10.2139/ssrn.5131964>.
- Antelava, A., Damilos, S., Hafeez, S., et al., 2019. Plastic Solid Waste (PSW) in the context of Life Cycle Assessment (LCA) and sustainable management. *Environ. Manag.* 64, 230–244. <https://doi.org/10.1007/s00267-019-01178-3>.
- Antonopoulos, I., Faraca, G., Tonini, D., 2021. Recycling of post-consumer plastic packaging waste in the EU: recovery rates, material flows, and barriers. *Waste Manag.* 126, 694–705. <https://doi.org/10.1016/j.wasman.2021.04.002>.
- Arena, U., Mastellone, M.L., Perugini, F., 2003. Life cycle assessment of a plastic packaging recycling system. *Int. J. Life Cycle Assess.* 8, 92–98. <https://doi.org/10.1007/BF02978432>.
- Arena, U., Parrillo, F., Ardolino, F., 2023. An LCA answer to the mixed plastics waste dilemma: energy recovery or chemical recycling? *Waste Manag.* 171, 662–675. <https://doi.org/10.1016/j.wasman.2023.10.011>.
- Arfelli, F., Roguszewska, M., Torta, G., Iurlo, M., Cespi, D., Ciacci, L., Passarini, F., 2024. Environmental impacts of food packaging: is it all a matter of raw materials? *Sustain. Prod. Consum.* 49, 318–328. <https://doi.org/10.1016/j.spc.2024.06.032>.
- Association of Plastic Recyclers (APR), 2022. Life cycle inventory of 100% postconsumer HDPE and PET recycled resin from postconsumer containers and packaging. Report prepared by Franklin Associates, A Division of ERG. <https://plasticsrecycling.org/images/library/APR-Life-Cycle-Inventory-of-Postconsumer-HDPE-and-PET-Recycled-Resin-from-Postconsumer-Containers-and-Packaging.pdf>.
- Bataineh, 2020. Life-cycle assessment of recycling postconsumer high-density polyethylene and polyethylene terephthalate. *Adv. Civ. Eng.* 1. <https://doi.org/10.1155/2020/8905431>.
- Bradshaw, S.L., Aguirre-Villegas, H.A., Boxman, S.E., Benson, C.H., 2025. Material recovery facilities (MRFs) in the United States: operations, revenue, and the impact of scale. *Waste Manag.* 193, 317–327. <https://doi.org/10.1016/j.wasman.2024.12.008>.
- Cimpan, C., Høiby, A.W., Pretz, T., Astrup, T.F., 2016. Techno-economic assessment of central sorting at material recovery facilities. *J. Clean. Prod.* 112 (5), 3920–3932. <https://doi.org/10.1016/j.jclepro.2015.07.086>. *Pagine*.
- Corepla, 2023. Rapporto di Sostenibilità 2023. Consorzio Nazionale per la Raccolta, il Riciclo e il Recupero degli Imballaggi in Plastica. https://www.corepla.it/wp-content/uploads/2024/11/COREPLA_Rapporto-di-sostenibilita-2023.pdf.
- Cristiu, D., d'Amore, F., Bezzo, F., 2024. Economic and environmental optimisation of mixed plastic waste supply chains in Northern Italy comparing incineration and pyrolysis technologies. *Comput. Chem. Eng.* 180, 108503. <https://doi.org/10.1016/j.compchemeng.2023.108503>.
- DEFRA, 2022. Life cycle assessment of the plastic packaging waste management hierarchy. WRAP (Waste Resour. Action Program.) UK Depart. Environ. Food Rural Affairs. https://cdn.ca.emap.com/wp-content/uploads/sites/6/2022/12/DEFRA_PLASTIC_WASTE_HIERARCHY_LCA_REPORT-v4.0-002_0.pdf.
- Deloitte Sustainability, 2017. Blueprint for plastics packaging waste: quality sorting & recycling. Final Rep. <https://www2.deloitte.com/content/dam/Deloitte/my/Documents/risk/my-risk-blueprint-plastics-packaging-waste-2017.pdf>.
- Dri, M., Canfora, P., Antonopoulos, I., Gaudillat, P., 2018. Best Environmental Management Practice for the Waste Management Sector. Joint Research Centre. JRC Research Reports JRC111059.
- European Commission, 2012. Life Cycle Indicators for Resources, Products and Waste: Waste Management. Joint Research Centre. <https://doi.org/10.2788/50351>. JRC Technical Reports EUR 25520 EN.
- European Commission, 2020. Determining the Environmental Impacts of Conventional and Alternatively Fuelled Vehicles Through LCA. Publications Office of the European Union. <https://data.europa.eu/doi/10.2834/91418>.
- European Recycling Industries' Confederation (EuRIC), 2020. Plastic recycling factsheet. <https://euric.org/resource-hub/position-papers/euric-plastic-recycling-fact-sheet>. (Accessed 20 May 2025).
- Franklin Associates, 2018. Life cycle impacts for postconsumer recycled resins: PET, HDPE, AND PP. The association of plastic recyclers. <https://plasticsrecycling.org/resources/2018-life-cycle-impacts-for-postconsumer-recycled-resins-pet-hdpe-pp/>. (Accessed 20 May 2025).
- Fries, N., Hellweg, S., 2014. LCA of land-based freight transportation: facilitating practical application and including accidents in LCIA. *Int. J. Life Cycle Assess.* 19, 546–557. <https://doi.org/10.1007/s11367-013-0657-2>.
- Gandhi, N., Farfaras, N., Wang, N.-H.L., Chen, W.-T., 2021. Life cycle assessment of recycling high-density polyethylene plastic waste. *J. Renew. Mater.* 9, 1463–1483. <https://doi.org/10.32604/jrm.2021.015529>.
- Grant, T., James, K., 2005. Life Cycle Impact Data for Resource Recovery from C&I and C&D Waste in Victoria. RMIT University, Centre for Design.
- Hammond, G., Jones, C., 2008. Inventory of Carbon & Energy (ICE), v1.6a. University of Bath.
- Helmes, R.J.K., Goglo, P., Salomoni, S., van Es, D.S., Vural Gursel, I., Aramyan, L., 2022. Environmental impacts of end-of-life options of biobased and fossil-based polyethylene terephthalate and high-density polyethylene packaging. *Sustainability* 14, 11550. <https://doi.org/10.3390/su141811550>.
- Hou, P., Xu, Y., Taiebat, M., Lastoskie, C., Miller, S.A., Xu, M., 2018. Life cycle assessment of end-of-life treatments for plastic film waste. *J. Clean. Prod.* 201, 1052–1060. <https://doi.org/10.1016/j.jclepro.2018.07.278>.
- Hu, A.H., Ting, C.-Y., Ouattara, A., Chen, W.-T., Kuo, C.-H., 2025. Life cycle assessment of recycling polyethylene terephthalate (PET): a comparative case study in Taiwan. *Recycling* 10, 98. <https://doi.org/10.3390/recycling10030098>.
- Huijbregts, M.A.J., Steinmann, Z.J.N., Elshout, P.M.F., Stam, G., Verones, F., Vieira, M., Zijp, M., Hollander, A., van Zelm, R., 2017. ReCiPe2016: a harmonised life cycle impact assessment method at midpoint and endpoint level. *Int. J. Life Cycle Assess.* 22, 138–147. <https://doi.org/10.1007/s11367-016-1246-y>.
- Kerdlap, P., Purnama, A.R., Low, J.S.C., Tan, D.Z.L., Barlow, C.Y., Ramakrishna, S., 2020. Environmental evaluation of distributed versus centralized plastic waste recycling: integrating life cycle assessment and agent-based modeling. *Proced. CIRP* 90, 689–694. <https://doi.org/10.1016/j.procir.2020.01.083>.
- Kerdlap, P., Purnama, A.R., Low, J.S.C., Tan, D.Z.L., Barlow, C.Y., Ramakrishna, S., 2021. Comparing the environmental performance of distributed versus centralized plastic recycling systems: applying hybrid simulation modeling to life cycle assessment. *J. Ind. Ecol.* 26, 252–271. <https://doi.org/10.1111/jiec.13151>.
- Kleinhans, K., et al., 2021. Development and application of a predictive modelling approach for household packaging waste flows in sorting facilities. *Waste Manag.* 120, 290–302. <https://doi.org/10.1016/j.wasman.2020.11.056>, 1 February 2021.
- Kreiger, M.A., Mulder, M.L., Glover, A.G., Pearce, J.M., 2014. Life cycle analysis of distributed recycling of post-consumer high density polyethylene for 3-D printing filament. *J. Clean. Prod.* 70, 90–96. <https://doi.org/10.1016/j.jclepro.2014.02.009>.
- Lase, I.S., Tonini, D., Caro, D., Albizzati, P.F., Cristóbal, J., Roosen, M., Kusenberg, M., Ragaert, K., Van Geem, K.M., Dewulf, J., De Meester, S., 2023. How much can chemical recycling contribute to plastic waste recycling in Europe? An assessment using material flow analysis modeling. *Resour. Conserv. Recycl.* 192, 106916. <https://doi.org/10.1016/j.resconrec.2023.106916>.
- Liao, N., Lü, F., Zhang, H., He, P., 2024. Optimizing the greenhouse gas emissions of waste transfer and transport: an integration of life cycle assessment and vehicle routing problem. *Waste Manag.* 189, 314–324. <https://doi.org/10.1016/j.wasman.2024.08.034>.
- Martin-Lara, M.A., Moreno, J.A., Garcia-Garcia, G., Arjandas, S., Calero, M., 2022. Life cycle assessment of mechanical recycling of post-consumer polyethylene flexible films based on a real case in Spain. *J. Clean. Prod.* 365, 132625. <https://doi.org/10.1016/j.jclepro.2022.132625>.
- Missouri Department of Natural Resources, 2010. Recycling economics: higher costs are an illusion. <https://eiera.mo.gov>. (Accessed 20 May 2025).
- Muralikrishna, I.V., Manickam, V., 2017. Life cycle assessment. In: *Environmental Management: Science and Engineering for Industry*, first ed. Butterworth-Heinemann, Oxford, pp. 57–76.
- Organisation for Economic Co-operation and Development, 2022. Global plastics outlook: economic drivers, environmental impacts and policy options. https://www.oecd.org/en/publications/global-plastics-outlook_de747aef-en.html. (Accessed 12 April 2025).
- 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>.
- Plastics Europe, 2022. Plastics – the facts 2022. <https://plasticseurope.org/knowledge-hub/plastics-the-facts-2022/>. (Accessed 12 April 2025).
- Pressley, P.N., Levis, J.W., Damgaard, A., Barlaz, M.A., DeCarolis, J.F., 2015. Analysis of material recovery facilities for use in life-cycle assessment. *Waste Manag.* 35, 307–317. <https://doi.org/10.1016/j.wasman.2014.09.012>.
- Ragaert, K., Delva, L., Geem, K.G., 2017. Mechanical and chemical recycling of solid plastic waste. *Waste Manag.* 69, 24–58. <https://doi.org/10.1016/j.wasman.2017.07.044>.
- Rigamonti, L., Grosso, M., Møller, J., Martinez Sanchez, V., Magnani, S., Christensen, T. H., 2014. Environmental evaluation of plastic waste management scenarios. *Resour. Conserv. Recycl.* 85, 42–53. <https://doi.org/10.1016/j.resconrec.2013.12.012>.
- Rikhter, P., Dinc, I., Zhang, Y., et al., 2022. Life Cycle Environmental Impacts of Plastics: a Review. NIST GCR 22-032. National Institute of Standards and Technology. <https://doi.org/10.6028/NIST.GCR.22-032>.
- Sabate, K., Kendall, A., 2024. A review evaluating the gaps in plastic impacts in life cycle assessment. *Cleaner Environ. Sys.* 14, 100205. <https://doi.org/10.1016/j.cesys.2024.100205>.
- Selamoglu, M., 2024. Waste management and logistics from a sustainability perspective. In: *Sustainable Development in 21st Century Through Clean Environment*. Kripa Drihti Publications, Pune, pp. 66–69.
- Shonfield, P., 2008. LCA of Management Options for Mixed Waste Plastics. *Waste and Resources Action Programme (WRAP)*.
- Tonini, D., Garcia-Gutierrez, P., Nessi, S., 2021. Environmental Effects of Plastic Waste Recycling. Joint Research Centre. JRC Research Reports JRC122455.
- Treyer, K., Bauer, C., Simons, A., 2014. Human health impacts in the life cycle of future European electricity generation. *Energy Policy* 74, S31–S44. <https://doi.org/10.1016/j.enpol.2014.03.034>.
- Uzosike, C.C., Yee, L.H., Padilla, R.V., 2023. Small-scale mechanical recycling of solid thermoplastic wastes: a review of PET, PE, and PP. *Energies* 16, 1406. <https://doi.org/10.3390/en16031406>.

- Villanueva, A., Eder, P., 2014. End-Of-Waste Criteria for Waste Plastic for Conversion. Joint Research Centre. <https://doi.org/10.2791/13033>.
- Wickerts, S., Arvidsson, R., Nordelöf, A., Svanström, M., Johansson, P., 2024. Prospective life cycle assessment of sodium-ion batteries made from abundant elements. *J. Ind. Ecol.* 28, 116–129. <https://doi.org/10.1111/jiec.13452>.
- Zampori, L., Pant, R., 2019. Suggestions for Updating the Product Environmental Footprint (PEF) Method. Joint Research Centre. JRC Research Reports JRC115959.
- Zhang, O., Takaffoli, M., Ertz, M., Addar, W., 2023. Environmental impact assessment of plastic waste management scenarios in the Canadian context. *Environments* 10, 213. <https://doi.org/10.3390/environments10120213>.