



Chemical Recycling of Plastic Waste: Comparative Evaluation of Environmental and Economic Performances of Gasification- and Incineration-based Treatment for Lightweight Packaging Waste

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Abstract

Chemical recycling (CR) – in enabling the use of plastic waste back as secondary carbon feedstock for production – could play a complementary role to mechanical recycling in supporting the transformation from a linear to a circular carbon economy. To date, research has predominantly focused on assessing technological aspects associated with CR of pure plastic waste streams. Little is known about its potential for treating low-quality and mixed plastic waste fractions which are unsuitable for conventional recycling and are currently incinerated. To address this gap, this investigation utilizes an integrated approach comprising of life cycle assessment and techno-economic analysis to evaluate the environmental and economic performance of CR of lightweight packaging waste via waste gasification compared to direct and indirect incineration in Germany. Results show that CR can contribute significantly – irrespective of the energy mix – to reducing climate change, terrestrial acidification, and fossil resource scarcity. In terms of economic performance, findings suggest that while CR requires higher capital investment, a multi-pronged approach which encompasses upscaling, waiver of carbon dioxide certificate costs, and price premium for CR products could increase profitability of CR to incineration. This study provides empirical support for the potential contribution of CR to complement existing strategies to combat the plastic waste challenge, and insights into market conditions which could promote its economic attractiveness. Additionally, it provides comprehensive inventory data for conventional and alternative waste treatment plants for lightweight packaging waste to inform future research on systemic assessment of CR technologies and their contributions to a circular economy.

Keywords Circular carbon economy · Life cycle assessment (LCA) · Techno-economic analysis (TEA) · Waste incineration · Plastic waste

Abbreviations

AP Acidification potential
CEPCI Chemical Engineering Plant Costs Index

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<i>CN</i>	Climate neutral
<i>CR</i>	Chemical recycling
<i>DPP</i>	Dynamic payback period
<i>EPS</i>	Energy provision scenario
<i>EREG</i>	Environmental regulation
<i>ES</i>	Economic scenario
<i>FCI</i>	Fixed capital investment
<i>FFP</i>	Fossil fuel potential
<i>GHG</i>	Greenhouse gas
<i>GWP</i>	Global warming potential
<i>LCCA</i>	Levelized cost of carbon abatement
<i>LWP</i>	Lightweight packaging
<i>MC</i>	Market conditions
<i>MRF</i>	Materials recovery facility
<i>NPV</i>	Net present value
<i>PRED</i>	Predominantly renewable energy supply
<i>PS</i>	Plant scaling
<i>RDF</i>	Refuse-derived fuel
<i>SI</i>	Supplementary Information
<i>TCI</i>	Total capital investment
<i>UBA</i>	German Environment Agency
<i>WC</i>	Working capital

Introduction

In recent years, the global plastic crisis arising from increasing plastic consumption in combination with improper plastic disposal has captured the global attention and is putting immense pressure on policymakers, science, and industry to react [1]. As a highly versatile and affordable material with a wide range of applications, plastic is integral to applications in diverse sectors ranging from consumables, household & industrial appliances, medical equipment, communications, to mobility [2]. However, the stability of plastic materials also translates into a serious disposal problem. Improper disposal of plastic waste has led to widely publicized environmental pollution on land and sea, and is associated with adverse effects not only for flora and fauna but also for humans [3–5]. To combat the plastic challenge, policymakers are expected to develop stricter regulations ranging from the reduction of plastic consumption especially for single-use plastics, enhanced eco-design, to higher recycling quota [6, 7]. At the same time, science and industry are increasingly in the spotlight to develop sustainable solutions to increase the circularity of plastics [8–12].

In developing solutions and measures to address the plastic crisis, the focus is been increasingly placed on lightweight packaging (LWP) materials. LWP waste represents a mixture of consumable packaging made from polymers, ferrous & non-ferrous metals, paper & cardboard, aseptic containers (i.e. beverage cartons), and various composite materials [13]. At 40%, it accounts for the largest share of the plastic production in Europe [14]. As such, it is not surprising that the environmentally friendly disposal/treatment of LWP waste is a key focus of socio-political debates about the plastic waste challenge, and correspondingly a priority for policymakers [14–16]. According to EUROSTAT [17], a significant share of emerging LWP waste in Europe is treated either via material recycling or

direct incineration, with the latter being a treatment technique still widely applied in European countries. Incineration reduces waste volume, weight, and pollution potential while generating electricity and/or heat as useful products [18]. Additionally, the extraction of minerals as construction materials and “pure” metals make it an attractive alternative to landfilling [18–22]. However, the downside of incineration is that via the incineration process potentially recyclable carbon-containing LWP fractions such as plastics and cardboard will be converted into greenhouse gas (GHG) emissions. This represents not only a loss of secondary carbon resources which could be channeled back as secondary feedstock for production, it furthermore contributes to global warming [23–25].

Hence, as part of its strategy to support the transition towards a sustainable and efficient economy, the European Union is targeting higher recycling rate of LWP within its New Circular Economy Action Plan [6]. One objective is to further increase the proportion of LWP waste processed in material recovery facilities (MRFs) as a preparatory step for mechanical (material) recycling [7]. Mechanical recycling is defined as methods to convert waste materials into secondary raw materials without changing the basic structure of the material [19, 26]. Through a thorough separation of individual packaging types, MRFs produce homogeneous and high-quality material streams for recyclate production via mechanical recycling. The use of such recyclates for chemical production instead of the direct incineration of LWP waste promotes the transition from a linear economy (i.e. where plastic waste is incinerated following consumption/utilization) to a circular economy (i.e. circularity) where it is channeled back into the production cycle as a secondary feedstock [1, 27]. Additionally, an increase in the use of such high-quality domestic plastic waste streams to replace imported fossil feedstock for packaging production also contributes to primary resource conservation and a reduction of CO₂ leakages along international supply chains [28]. In binding carbon atoms of carbon-containing LWP waste in products rather than in the form of CO₂ emissions after incineration, it furthermore contributes to reducing the carbon footprint of packaging materials and to the achievement of a net-zero GHG emissions society [25, 29].

However, circularity of plastics is easier said than done. Although the EU Directive 94/62/EC states that EU countries must recycle 50% of all LWP made of plastic by 2025 (and 55% until 2030) [7], the example of Germany – a country which despite a modern and well-established plastic waste recycling system in the form of the dual system (“*Duales System*”) [30, 31] only achieves recycling rates of less than 30% for post-consumer plastics [32] – points to the limits of mechanical recycling in meeting the ambitious EU goal for packaging recycling. Thus, to increase the circularity of packaging waste and to meet recycling targets, new recycling concepts and technologies are urgently required to complement mechanical recycling [17]. In this context, chemical recycling as an alternative option to incineration of LWP waste which is not mechanically recyclable has been eliciting increasing attention and interest not only from industry and academics, but also from policymakers and the civil society [8–10, 12, 33–35].

The term chemical recycling (CR) is generally used to describe technologies where carbon-containing waste is broken down into its basic chemical building blocks, which can then be used as raw materials for the production of new consumables such as plastics and chemicals [36, 37]. CR could complement mechanical recycling in increasing the circularity of plastics and supporting the transition from a linear to a circular carbon economy [1, 23, 24, 29, 38, 39]. However, there is a lack of quantitative data and scientific/technical literature to support a comparative evaluation of the strengths and weaknesses of CR against conventional recycling and energy recovery pathways. The predominant attention today – academic and otherwise – is on technical issues related to CR, with existing literature focusing on new technical

developments [10, 40–43], life cycle assessments [23, 44–46], as well as energy and material balances [43, 47, 48]. Furthermore, the predominant focus on pure plastic waste streams as potential CR feedstock has led to significant controversy in the socio-political realm as it presents a direct competition with mechanical recycling techniques [9, 49]. Moreover, despite its potential relevance and contribution to plastic circularity, studies that utilize a multi-dimensional and holistic approach (i.e. considering integration in existing production infrastructures and processes) to comparatively evaluate CR of LWP waste against conventional incineration and mechanical recycling options are, to the best of the authors' knowledge, missing. Such insights would be invaluable in contributing to socio-political discussions about the role of CR in the waste hierarchy, especially for the sustainable management of plastic waste [50].

To address these gaps in the extant literature, this research utilizes a case study approach to investigate quantitative impacts associated with CR of LWP waste against conventional LWP waste treatment pathways. It contributes to discussions about LWP waste treatment through extending the focus from an isolated technical evaluation to integrated environmental and economic assessments of CR in Germany. Specifically, the environmental assessment focuses on a life cycle assessment of indicators namely climate change, terrestrial acidification, and fossil resource scarcity. Economic performance is assessed via techno-economic analysis of total capital investment (TCI), net present value (NPV), dynamic payback period (DPP), and levelized cost of carbon abatement (LCCA). The environmental and economic assessments support a comparative evaluation of three LWP waste treatment pathways namely (1) direct incineration, (2) partial incineration, and (3) CR.

Note that in the investigation, it is assumed that both partial incineration and CR are integrated into current best practices whereby homogeneous and high-quality LWP waste streams are first extracted in MRFs for subsequent mechanical recycling. The heterogeneous and low-quality residues from the MRF sorting process are then either incinerated in the partial incineration pathway, or used as raw material for a CR plant. In this way, CR does not compete with mechanical recycling for pure plastic waste streams. Rather, it complements mechanical recycling to raise the recycling quota by presenting a recycling alternative to incineration for residue LWP waste fractions which are not suitable for mechanical recycling [51].

The paper is structured as follows: the “[Study Context](#)” section provides detailed information on the study context in Germany. This supports an assessment of the transferability of findings, conclusions, and recommendations of the current study to other research contexts, e.g. to other countries with different waste management systems. The “[Methodology](#)” section presents the inventory dataset for this study. In addition to an overview of the characteristics of LWP waste and treatment technology, methodologies utilized for the environmental and economic calculations are also presented. The “[Results and Discussion](#)” section presents the results and discusses the comparative evaluation of the three different LWP waste treatment pathways in terms of their environmental and economic performance. Finally, the paper concludes with a summary of study implications and a discussion about its limitations and transferability to other contexts or subsequent studies.

Study Context

With its dual system, Germany has a structured source separation system in place where LWP waste is collected separately from other waste (i.e. residual municipal solid waste, paper/cardboard waste, glass bottles, etc.) in yellow containers and sacks (“Gelbe Tonne” and “Gelber Sack”) [51]. The systems implementation was motivated by the enactment of

the “Packaging Ordinance” in the early 1990s which anchored the “polluter pays principle” in Germany’s waste management legislation [52]. Since then, companies that introduce LWPs to the market are obliged to contribute financially to their collection and recovery. Today, more than 2 million tonnes of LWP waste are collected annually by the dual system and are processed in MRFs [53].

The thorough separation of the individual packaging types in MRFs facilitates the production of homogeneous and high-quality material streams for recycling. However, the degree of separation depends on the efficiency of the sorting process. Inevitably, MRFs produce a residual fraction of materials (i.e. sorting residues) that are either too small for sorting or consist of multiple partially bonded materials that do not meet the quality standards for subsequent mechanical recycling processes [13]. According to the German Nature and Biodiversity Conservation Union (NABU), more than 50% of LWP waste ends up as sorting residues which are not suitable for mechanical recycling [54]. Currently, these residues are typically processed as refuse-derived fuel (RDF) for energy and/or heat recovery in incineration facilities ranging from RDF power plants, cement works, to coal-fired power plants [51]. However, as sorting residues from LWP waste are carbon-rich, a potential alternative treatment pathway to incineration is the utilization of LWP sorting residues as secondary carbon feedstock for chemical production via CR.

Since the 1970s, a keen interest in CR development is evident from the deployment of a variety of experimental-, pilot-, and industrial-scale facilities in different countries [10, 55]. The term CR generally refers to the decomposition of carbonaceous waste either thermally or by solvents into synthesis gas monomers, oligomers, or higher hydrocarbons as products [10]. These products represent feedstocks for the production of plastics or other value products such as fertilizers, paints, or pharmaceuticals [8, 37, 56, 57]. According to Mamani et al. [10], CR technologies can be generally categorized in solvolysis, pyrolysis, and gasification [10]. However, not all CR technologies are suitable for processing LWP sorting residues with mixed compositions and low qualities. To date, only gasification technologies have been operated on an industrial-scale for such purpose. For example, in Berrenrath from 1993 to 1997 and in Schwarze Pumpe from 1995 to 2007 [40, 55, 58, 59], LWP sorting residues from the dual system served as co-feedstock with coal for syngas generation and subsequent methanol production in Germany. Another prominent example is Showa Denko K.K. in Kawasaki/Japan, where a plastic waste gasification plant has been in operation since 2003 to generate hydrogen for ammonia production [60].

In view of its previous and current practical application for the CR of plastic waste, this research evaluates gasification as the alternative CR treatment pathway to incineration pathways for LWP waste. Unlike incineration pathways where electricity and heat are the desired products, during the gasification process waste is broken down into a CO- and H₂-rich syngas under high pressure and high temperature conditions with oxygen and/or steam as gasification agents. In subsequent downstream processes, syngas can be converted to platform chemicals such as methanol, which can then be further synthesized via a methanol-to-olefins process to olefins, an important intermediate raw material for the production of a broad spectrum of chemical products including virgin plastics [25, 61, 62].

Figure 1 illustrates the three LWP waste treatment pathways considered in this study. In Pathway I (i.e. reference scenario), LWP waste is channeled directly for incineration in RDF power plants without pre-sorting (i.e. no material recovery in MRF). Note that this does not reflect the case in Germany as LWP waste is pre-sorted in MRFs for mechanical recycling before sorting residues are sent as RDF for incineration (i.e. partial incineration). Nevertheless, in view that incineration is widely practiced and/or considered for deployment in numerous EU countries (and worldwide), the direct

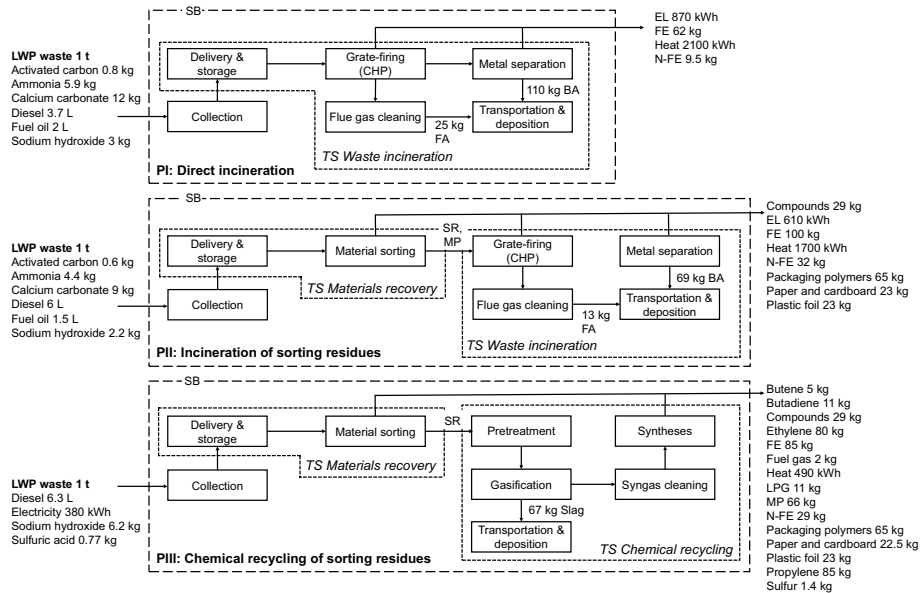


Fig. 1 System environment and results of inventory calculations. BA: Bottom ash. CHP: Combined heat and power generation. EL: Electricity. FA: Fly ash. FE: Ferrous scrap. LPG: Liquefied petroleum gas. LWP: Lightweight packaging. MP: Mixed plastics. N-FE: Non-ferrous scrap. SB: System boundaries. SR: Sorting residues. TS: Treatment step. Underlying values rounded to two significant digits. Energy figures are reported net of energy self-use

incineration pathway (PI) provides a theoretical basis of comparison to illustrate the environmental and economic impacts for recycling efforts of LWP waste via partial incineration in Pathway II (PII), as well as via CR in Pathway III (PIII).

Methodology

The assessment of the three LWP waste treatment pathways, i.e. incineration (PI), partial incineration (PII), and CR (PIII), builds on Voss et al. [46] and is conducted via three individual steps: First, a comprehensive inventory dataset for the treatment of LWP waste at three representative plant sites, reflecting PI to PIII, is generated (see the “Inventory Dataset” section). Second, based on the developed inventory dataset, an life cycle assessment of impacts along midpoint indicators climate change, terrestrial acidification, and fossil resource scarcity for the treatment of one tonne of LWP waste is conducted for each plant site (see the “Environmental Assessment” section). Third, techno-economic analyses of TCI, NPV, DPP, and LCCA are performed to evaluate economic plant site performances (see the “Techno-economic Analysis” section).

Inventory Dataset

Definition and Characterization of LWP Waste in Germany

For the investigation, LWP waste is defined as waste material that is assigned to the waste code number EAV-15010601-U for "light weight packaging" in the disposal statistics by the German Federal Statistical Office [53]. Note that German authorities utilize EAV-15010601-U as a subgroup for the European waste code number EAV-150106 for "mixed packaging" [53, 63]. The assumed LWP waste composition as presented in Table S1 in the Supplementary Information (SI) relies on data from a study conducted by the German Environment Agency (UBA) [64]. Chemical composition of each sub-fraction is based on Riber et al. [65], resulting in chemical characteristics for LWP waste – e.g. water content, lower calorific value, and carbon content – as presented in Table S2 in the SI.

Technological System

Scientific publications, governmental technical reports, LCI databases, and process modeling results serve as data sources for the investigated treatment technologies. Specifically, data on waste collection and waste transportation is drawn from Larsen [66] and Olesen [67]. Data for waste incineration is obtained from technical reports by UBA [51, 68] and corresponding BAT-documents by the European Commission [69, 70]. Data on materials recovery is drawn from research by Cimpan et al. [13]. Finally, data on chemical recycling is obtained from chemical process modeling via ASPEN Plus V10 based on the approach by Keller et al. [23]. Detailed material flow assumptions are presented in the following paragraphs. Aggregated inputs and outputs for each treatment pathway are illustrated in Fig. 1.

Collection and transportation LWP waste collection is modeled assuming the combustion of 3.1 L diesel tonne^{-1} in a waste collection truck [66]. Transportation of waste treatment outputs such as bottom ash, fly ash, slag, and sorting products is modeled under the assumption of a Euro-5 semitrailer truck with a cargo volume of 10 tonnes travelling fully loaded, consuming 0.021 L diesel $\text{tonne}^{-1} \text{ km}^{-1}$ [67].

Waste incineration Incineration of LWP waste and LWP waste sorting residues is based on RDF incineration in German RDF power plants as described by UBA [51, 68]. The process consists of four steps, namely 1) delivery and storage, 2) grate-firing, 3) flue gas cleaning, and 4) metal separation:

- Step 1: Delivery and storage of LWP waste after collection are modeled via the combustion of 0.5 L diesel tonne^{-1} in a wheel loader that operates at 33% average load factor [71]. Note that waste incineration does not include delivery and storage in PII, as it describes the second treatment step after materials recovery (cf. Fig. 1).
- Step 2: Grate-firing consumes 2 L fuel oil tonne^{-1} input as auxiliary fuel [70] and cogenerates heat and electricity. Cogeneration efficiencies are set to 37% for heat and 15% for electricity relative to the lower heating value of the input [68]. Please note that assumed values correspond to net values i.e. after deduction of internal energy consumption. Excess heat and electricity are fed into a local heat grid and the

German electricity grid, respectively, to substitute conventional energy. Bottom ash from grate-firing is processed in Step 4.

- Step 3: Gas treatment for airborne emissions resulting from grate-firing is assumed with a wet flue gas cleaning based on data provided by UBA [51] and EC [70]. Specifically, the system consists of a dry electrostatic precipitator, a two-stage water scrubber unit, a selective catalytic reforming unit, and a fly-flow adsorption. Supply consumption is set to 0.8 kg activated carbon, 5.9 kg ammonia, 12 kg calcium carbonate, and 3 kg sodium hydroxide tonne^{-1} input [51, 70].
- Step 4: A metal separation unit – comprising of two overbelt magnets, a bar sizer, an impact crusher, a flip-flop sieve, and an eddy current separator – with recovery efficiencies of 80% for ferrous and 19.5% for non-ferrous metals is assumed [72].

Process-specific air emission of treatment include 750 g NO_x (4.3 g N₂O), 260 g CO, 150 g SO₂, and 2.5 g CH₄ tonne^{-1} input [73, 74]. Fossil CO₂ emission are calculated based on the fossil carbon content of input in EASETECH (cf. “Life Cycle Assessment Model”). Solid emissions include fly ash and bottom ash which are treated as follows: Fly ash is assumed to be transported 700 km to a salt mine (cf. process “collection and transportation”) where it is utilized as backfilling material consuming 1.8 L diesel and 28 kWh electricity tonne^{-1} fly ash [75]. Remaining bottom ash after metal separation is transported 2 km to a landfill, modeled with the combustion of 1 L diesel tonne^{-1} in a 89 KW drivable machine charged to 36% on average [76]. Note that transportation via truck is generally considered one-way due to the assumption of optimized logistics.

Materials recovery Materials recovery is modeled based on data provided by Cimpan et al. [13] for MRFs in Germany. Treatment consists of two treatment steps, namely 1) delivery and storage and 2) material sorting:

- Step 1: Delivery and storage include reception of LWP waste and a pre-conditioning in form of bag opening and size reduction consuming 0.84 L diesel and 4.1 kWh electricity tonne^{-1} LWP waste input [13].
- Step 2: Material sorting consists of i) a conditioning step with sieving steps or ballistic separation, ii) a primary sorting step for first separation into material groups, iii) a refining step to increase material output quality i.a. via manual sorting, and iv) a product handling step to prepare products for downstream processors via e.g. balling [13]. Assumed sorting efficiencies are based on data provided by Cimpan et al. [13] and UBA [51] for materials recovery processes with extensive plastic sorting (cf. SI, Table S3). Supply consumption includes 1.4 L diesel and 87 kWh electricity tonne^{-1} LWP waste input [13].

Material recovery products include i) recyclable materials, ii) sorting residues, and iii) mixed plastics. For i), recyclable materials such as packaging polymers, plastic foil, and metals are transported 20 km to a recycling facility [13]. For ii), sorting residues are transferred to the next treatment step i.e. waste incineration or CR assuming the combustion of 0.5 L diesel tonne^{-1} in a wheel loader that operates at 33% average load factor [71]. For iii), mixed plastics are assumed to be treated via incineration based on Cimpan et al. [13]. As PII includes a corresponding treatment step, mixed plastics are assumed to be handled together with sorting residues (cf. Fig. 1). As PIII does not include a corresponding treatment step, mixed plastics are transported 20 km to an external RDF power plant assuming

environmental impacts based on process “waste incineration” (see above) and costs based on Cimpan et al. [13] as displayed in Table 1.

CR Based on Keller et al. [23] and Poganietz et al. [86], CR is modeled via four individual process steps: 1) pretreatment, 2) gasification, 3) gas treatment, and 4) syntheses:

- Step 1: Pretreatment includes shredding and compacting of sorting residues consuming 3.7 kWh electricity tonne⁻¹ input [86].
- Step 2: Gasification is assumed to be realized via fixed-bed gasification with liquid slag extraction (BGL type) [23].
- Step 3: Gas treatment of raw product syngas includes i) a water scrubbing unit, ii) a two-stage, adiabatic CO shift unit, iii) a selective acid gas removal unit, and iv) a three-stage Claus plant [23].
- Step 4: Syntheses of purified and conditioned syngas to olefins is realized via i) a syn-gas-to-methanol synthesis, ii) a methanol-to-olefins synthesis, and iii) a olefins recov-

Table 1 Assumed values for global warming potential (GWP100), acidification potential (AP), fossil fuel potential (FFP), and prices for supplies and products. *: CHP selling price. EM: EASETEACH modeling. Underlying values rounded to two significant digits. All prices assumed for year 2021

	Per	GWP100 [kg CO ₂ eq.]	AP [kg SO ₂ eq.]	FFP [kg oil eq.]	Price [€]	Sources
<i>Supplies</i>						
Activated carbon	kg	5.1	0.0078	2.4	2.4	[77, 78]
Ammonia	kg	2.3	0.00084	0.94	0.3	[77, 78]
Calcium carbonate	kg	0.064	0.00026	0.025	0.10	[77, 78]
Diesel	kg	0.5	0.0012	1.1	0.52	[77, 78]
Fuel oil	L	0.43	0.00097	1.2	0.58	[77, 78]
Sodium hydroxide	kg	0.95	0.0014	0.36	1.1	[77, 78]
<i>Products</i>						
Butadiene	kg	1.5	0.0019	1.5	0.7	[77, 78]
Butene	kg	1	0.0023	1.4	0.7	[77, 78]
Compounds	kg	1.1	0.0026	0.6	0	[13, 77, 79]
Ethylene	kg	1.4	0.0027	1.6	1.02	[77, 80]
Ferrous scrap	kg	1.9	0.0032	0.38	0.17	[13, 77]
Fuel gas	kg	0.61	0.00048	1.3	0.43	[77, 78]
Heat	kWh	0.26	0.00021	0.084	0.024*	[77, 81, 82]
Liquefied petroleum gas	kg	0.79	0.0018	1.3	0.47	[77, 78]
Mixed plastics incineration	kg	1.1	-0.00032	-0.38	0.017	EM, [13]
Non-ferrous scrap	kg	5.3	0.014	1.8	0.41	[13, 77]
Packaging polymers	kg	1.5	0.0027	1.5	0.12	[13, 77]
Paper and cardboard	kg	0.69	0.002	0.28	0.05	[13, 77]
Plastic foil	kg	1.5	0.0027	1.5	0.11	[13, 77]
Propylene	kg	1.4	0.0027	1.6	0.95	[77, 80]
Sulfur	kg	0.66	0.0014	0.79	0.11	[77, 78]
<i>Supply and product</i>						
Electricity	kWh	0.42	0.00035	0.11	0.081*	[77, 83–85]

ery stage [23]. For i) syngas-to-methanol synthesis, a single stage quasi-isothermal fixed bed reactor of the Lurgi type is assumed. For ii) methanol-to-olefins synthesis, assumed process configuration refers to the UOP HYDRO methanol-to-olefins process [23]. In iii) the olefins recovery stage, product gas obtained from methanol-to-olefins process is introduced to an olefin recovery process that includes a sodium hydroxide scrubber, a cryogenic separation unit, a three-stage refringent plant, and an adsorption & extractive distillation unit to recover butene and butadiene.

Note that data for treatment steps 2) gasification, 3) gas treatment, and 4) synthesis is produced with the chemical process modeling software ASPEN Plus V10 as presented by Keller et al. [23]. Aggregated supply consumption includes 280 kWh electricity and 9.1 kg sodium hydroxide tonne^{-1} sorting residue input. Treatment products of the CR process are assumed with 4.2 kg butadiene, 1.9 kg butene, 0.76 kg fuel gas, 710 kWh heat as process steam that can be used for versatile applications at the plant site and/or for export, 16 kg liquified petroleum gas, and 240 kg olefins (120 kg ethylene, 130 kg propylene) tonne^{-1} sorting residue input. Airborne emissions are accounted for with 710 kg fossil CO_2 , 0.042 kg CO, 0.13 kg SO_2 , and 0.1 kg NOx tonne^{-1} sorting residue input. 99 kg slag tonne^{-1} sorting residue input are transported 2 km to an adjacent landfill where it is deposited, modeled with the combustion of 1 L diesel tonne^{-1} in a 89 KW drivable machine charged to 36% on average [76].

Environmental Assessment

ISO standard 14,040:2006 which defines life cycle assessment requirements is utilized to assess the environmental life cycle impacts of all treatment pathways [87].

Goal and Scope

Goal is to facilitate a comparative assessment of environmental impacts associated with the three pathways for LWP waste treatment, namely direct incineration in a RDF power plant without pre-sorting, incineration of sorting residues from MRF, and CR of sorting residues from MRF. The functional unit is defined as the treatment of one metric tonne of LWP waste, with fractional composition and chemical characteristics further described in Tables S1 and S2 in the SI. The geographical scope is Germany and the temporal scope is set to 2021 until 2028 as assumptions regarding waste composition or technology efficiency are based on current knowledge [79].

System Boundaries

As illustrated in Fig. 1, system boundaries extend from the point of waste collection in households to treatment product distribution. Waste generation upstream impacts are excluded as a zero-burden approach is applied [79]. Nevertheless, upstream impacts associated with supply utilization and downstream impacts associated with treatment product distribution are included via system expansion with a background system (cf. Table 1). Corresponding environmental footprints for supplies are drawn directly from GaBi [77]. In contrast, footprints for treatment products are calculated based on data provided by Montejo et al. [79] for market product substitution ratios that reflect quality/material losses

during recycling processes and reduced market acceptance. Specifically, ferrous and non-ferrous scraps substitute primary steel and aluminum at substitution ratios of 87% and 79%, respectively. Paper & cardboard waste substitutes primary paper at a ratio of 82%. Both packaging polymers and plastic foil substitute primary low-density polyethylene at a substitution ratio of 81%. Compounds are assumed to contain 74% paper, 22% plastic foil, and 4% aluminum based on the material composition of beverage cartons [79]. Material loss during decomposition is set to 22%. After decomposition, individual components substitute primary market products at ratios defined above. Eventually, chemical recycling products, electricity, and heat substitute market products at a ratio of 100% as conventional quality can be expected [46]. Footprints for supplies and market products are drawn from GaBi [77]. Note that all applied footprints refer to specific or market average values, indicating the attributional mode of this assessment. Moreover, environmental impacts associated with plant construction are beyond the scope of the current investigation.

Life Cycle Assessment Model

The software EASETEACH V3.1.7. [88] – well established for waste management research [89–91] – is applied for the environmental assessment. EASETECH is developed at the Department of Environmental Engineering, Technical University of Denmark (DTU), and focuses i.a. on a detailed consideration of fractional composition and chemical characteristics of waste. In EASETECH, LWP waste can be modeled by combining predefined sub-fractions such as standard packaging polymers, plastic foils, and paper waste. Based on the assumed composition, the software then determines key characteristics of LWP waste including energy content, carbon content, and water content (cf. SI, Table S2 for chemical characteristics). At the same time, it allows a realistic representation of sorting processes as individual sorting efficiencies can be assumed for different sub-fractions.

Impact Assessment

Keller et al. [23] identified midpoint indicators climate change, terrestrial acidification, and fossil resource scarcity as relevant impact categories for the examination of incineration and CR via gasification for alternative carbon carriers in Germany. Note that further life cycle impact categories used in holistic analyses such as marine eutrophication or land use are not analyzed in the current investigation. Regarding impact assessment methods, impact on climate change is determined per mechanism global warming potential for a period of 100 year (GWP100) in line with the Fifth Assessment Report (AR5) of IPCC [92]. Biogenic CO₂ emissions are considered climate-neutral in the investigation. Impact on terrestrial acidification is determined per mechanism acidification potential (AP) and impact on fossil resource scarcity per mechanism fossil fuel potential (FFP) based on the ReCiPe2016 framework by Huijbregts et al. [93].

Environmental Sensitivity Analysis

As the German energy system is currently undergoing substantial transformations [94], different compositions for energy supply, i.e. energy provision scenarios (EPS), are examined in more detail. Via sensitivity analysis, using data provided by the European Commission [95] for electricity provision and by BDI [81] for heat provision, the influence of changes

in energy mix on environmental impacts is determined. Specifically, three EPS scenarios are investigated:

- Scenario EPS-BASIC with current emission factors for heat and electricity supply (0.26 kg CO₂eq., 0.00021 kg SO₂eq., 0.084 kg oil eq. kWh⁻¹ heat; 0.42 kg CO₂eq., 0.00035 kg SO₂eq., 0.11 kg oil eq. kWh⁻¹ electricity),
- Scenario EPS-PRED for predominantly renewable energy supply (0.17 kg CO₂eq., 0.00015 kg SO₂eq., 0.064 kg oil eq. kWh⁻¹ heat; 0.18 kg CO₂eq., 0.00019 kg SO₂eq., 0.074 kg oil eq. kWh⁻¹ electricity),
- Scenario EPS-CN for nearly climate neutral energy supply via wind power (0.012 kg CO₂eq., 0.000024 kg SO₂eq., 0.0033 kg oil eq. kWh⁻¹ heat; 0.011 kg CO₂eq., 0.000023 kg SO₂eq., 0.00312 kg oil eq. kWh⁻¹ electricity).

Please note that indirect effects of energy provision such as impacts on metal or chemical production processes are beyond the scope of the current investigation.

Techno-economic Analysis

For the techno-economic analysis, each LWP waste treatment pathway is represented with a corresponding treatment plant with an assumed capacity of 300,000 tonnes LWP waste year⁻¹ based on capacity data for modern waste treatment plants in Germany [68]. Widely utilized techno-economic analyses including TCI, NPV, DPP, and LCCA for these plants are performed based on approaches by Peters et al. [96], Sinnott and Towler [97], and Friedman et al. [98]. Note that 2021 is assumed as the investment base year.

Total Capital Investment (TCI)

TCI is defined as fixed capital investment (FCI: manufacturing and plant facilities capital) plus working capital (WC: capital required for plant operation) (cf. Eq. 1) as suggested by Peters et al. [96]. FCI is calculated based on a power factor applied to capacity ratio approach [96]. Specifically, FCI data for individual plant components (cf. SI, Table S4) is drawn from the literature. This is multiplied by the ratio of targeted capacity to reference capacity raised to the power of 0.7 to reflect economies-of-scale [96], and then multiplied by a temporal price index ratio to account for temporal price developments [96]. For temporal price indexing, the Chemical Engineering Plant Costs Index (CEPCI) is utilized and extrapolated to 2021 based on first order linear regression (cf. SI, Table S5). Investment data for individual plant components for waste incineration is drawn from UBA [51]. For materials recovery, investment data from Cimpan et al. [13] is utilized. For CR, data provided by Poganietz et al. [86] is referred to. Note that WC is estimated at 15% of the FCI as suggested by Peters et al. [96].

$$TCI = FCI + WC = \left(\sum_{pc} FCI_{pc} \left(\frac{cap}{cap_{rp}} \right)^{\kappa^{inv}} \left(\frac{pindex}{pindex_{ry}} \right) \right) (1 + wcr) \quad (1)$$

with

cap Capacity [t]
cap_{rp} Capacity of reference plant [t]

<i>FCI</i>	Fixed capital investment [€]
<i>FCI_{pc}</i>	Fixed capital investment for plant component [€]
<i>pc</i>	Plant component index [-]
<i>pindex</i>	Price index in year of construction [-]
<i>pindex_{ry}</i>	Price index in reference year [-]
κ^{inv}	Scaling power factor [-]
<i>WC</i>	Working capital [€]
<i>wcr</i>	Working capital rate [%]

Net Present Value (NPV)

As presented in Eq. 2, NPV represents the sum of all discounted cash flows over the entire life of an investment reduced by the FCI in investment base year [96]. Referring to Bator et al. [99], a construction period of 3 years for all treatment plants is assumed. Plant operation thus starts in year 2024 and is assumed at 30 years with 90% plant availability for all treatment plants [100–102]. Considered cash flows during plant operation are based on Peters et al. [96] as presented with Eqs. 3 to 13. Prices for supplies and treatment products are based on data provided by UN Comtrade [78] and Cimpan et. al. [13] as presented in Table 1. Other relevant parameters for calculations such as discount rate, tax rates, and inflation rates are presented in Table 2. Labor requirements are assumed at 91,000 employee hours year⁻¹ for PI, 200,000 for PII, and 180,000 for PIII based on personnel requirements in German plants [13, 86, 103, 104]. Note that a share of 15% of the employee hours is assumed to be skilled labor instead of unskilled labor [96, 105].

Table 2 General economic assumptions for treatment cost calculation. LWP: Lightweight packaging. All prices assumed for year 2021. Underlying values rounded to two significant digits

	Unit	Value	Source
Administration percentage	%	25	[96]
CO ₂ certificate price	€/t	25	[106]
CO ₂ certificate price inflation	%	5	[107]
Consumables percentage	%	20	[96]
Corporate income tax rate	%	15	[108]
Depreciation period (linear)	a	10	[109]
Discount rate	%	8	[100]
Insurance percentage	%	1	[96]
Labor cost inflation rate	%	2.6	[110]
Labor power factor	%	0.25	[96]
Local tax percentage	%	2	[96]
Maintenance & repair percentage	%	4	[96]
Overhead percentage	%	50	[96]
Price inflation	%	1.4	[111]
LWP waste gate fee	€/t	140	[13]
Skilled employee hour	€/h	80	[112]
Supervision percentage	%	15	[96]
Transportation cost rate	€/km	0.07	[51]
Unskilled employee hour	€/h	63	[112]

$$NPV = \sum_{n=1}^t \frac{\sum_{k=1}^{10} CF_{n,k}}{(1+i)^n} - FCI \quad (2)$$

with

$CF_{n,k}$	Annual cashflow [€]
FCI	Fixed capital investment [€]
i	Discount rate [%]
k	Cashflow index [-]
NPV	Net present value [€]
t	Investment period [a]

$$CF_1 = FCI (insur + lTax) \quad (3)$$

with

CF_1	Local taxes and insurances [€]
FCI	Fixed capital investment [€]
$insur$	Insurance percentage [%]
$lTax$	Local taxes percentage [%]

$$CF_2 = FCI maint (1 + consum) \quad (4)$$

with

CF_2	Maintenance & repair [€]
$consum$	Consumables percentage [%]
FCI	Fixed capital investment [€]
$maint$	Maintenance & repair percentage [%]

$$CF_3 = \sum_s requi_s price_s \quad (5)$$

with

CF_3	Supplies [€]
$price_s$	Price for s [€/t]
$requis$	Requirements s [t]
s	Supply [-]

$$CF_4 = (sEmploy sRate + usEmploy usRate) (1 + supervis) \quad (6)$$

with

CF_4	Labor [€]
$sEmploy$	Skilled employee hours [h]
$sRate$	Hourly rate for skilled labor [€/h]
$supervis$	Supervision percentage [%]
$usEmploy$	Unskilled employee hours [h]
$usRate$	Hourly rate for unskilled labor [€/h]

$$CF_5 = CF_4 \text{ overhd} \quad (7)$$

with

CF_4 Labor costs [€].
 CF_5 Overhead [€].
 $overhd$ Overhead percentage [%]

$$CF_6 = \frac{CF_4 \text{ admin}}{1 + \text{supervis}} \quad (8)$$

with

$admin$ Administration percentage [%]
 CF_4 Labor costs [€]
 CF_6 Administration costs [€]
 $supervis$ Supervision percentage [%]

$$CF_7 = \sum_{em} \text{outp}_{em} \text{costs}_{em} \quad (9)$$

with

CF_7 Environmental expenses [€]
 costs_{em} Price for handling/treatment of em [€/t]
 em Emission/contaminant [-]
 outp_{em} Output of em [t]

$$CF_8 = \sum_{sto} \text{outp}_{sto} \text{dist}_{sto} \text{tRate} \quad (10)$$

with

CF_8 Transportation costs [€]
 dist_{sto} Transportation distance for sto [km]
 outp_{sto} Output of sto [t]
 sto Solid treatment output [-]
 tRate Transportation cost rate [€/km]

$$CF_9 = \sum_{prod} \text{outp}_{prod} \text{price}_{prod} \quad (11)$$

with

CF_9 Product revenues from sold products [€]
 outp_{prod} Output of $prod$ [t]
 price_{prod} Selling price for $prod$ [€/t]
 $prod$ Product [-]

$$tinc = \left(\sum_{k=1}^9 CF_k - depr \right) \quad (12)$$

$$CF_{10} = \begin{cases} tinc \text{ itax} & tinc > 0 \\ 0 & tinc \leq 0 \end{cases} \quad (13)$$

with

CF_k	Annual cashflow [€]
CF_{10}	Income tax rate [€]
$depr$	Plant depreciation [€]
$itax$	Corporate income tax rate [%]
k	Cashflow index [-]
$tinc$	Taxable income [€]

Dynamic Payback Period (DPP)

As presented in Eq. 14, DPP represents the required time period to recover the initial FCI [113]. In other words, DPP refers to the first year of investment amortization. Note that DPP has integer values as cash flows are generally assumed to occur at the end of each year.

$$\sum_{n=1}^{DPP} \frac{\sum_{k=1}^{10} CF_{n,k}}{(1+i)^n} - FCI = 0 \quad (14)$$

with

$CF_{n,k}$	Annual cashflow [€]
DPP	Dynamic payback period [a]
FCI	Fixed capital investment [€]
i	Discount rate [%]
k	Cashflow index [-]

Levelized Costs of Carbon Abatement (LCCA)

As presented in Eq. 15, LCCA describes an investment on the basis of euros per tonne of reduced CO₂eq. [98]. For the calculation, the annual cost surpluses compared to the reference technology are divided by the annual CO₂eq. savings over the entire life of the investment. Note that PI is defined as the reference process for LCCA calculations for PII and PIII as it provides the theoretical basis to illustrate the environmental and economic impacts for recycling efforts of LWP waste in this study (cf. “Study Context” section).

$$LCCA = \sum_{n=1}^t \frac{\sum_{k=1}^{10} CF_{n,k} - CFR_{n,k}}{CA_n} \quad (15)$$

with

$CA_{n,k}$	Annual carbon emission abatement [t]
$CF_{n,k}$	Annual cashflow [€]
$CFR_{n,k}$	Annual reference cashflow [€]
k	Cashflow index [-]
$LCAA$	Levelized cost of carbon abatement [€/t]
t	Period of plant operation [a]

Economic Sensitivity Analysis

Plant scaling, environmental regulation, and recycling product market conditions are analyzed via economic sensitivity analysis as they have been identified in previous studies to have the potential to affect the economic performance of the investigated treatment pathways [46]. Specifically, four economic scenarios (ES) are investigated:

- Scenario ES-BASIC for a plant capacity of 300,000 tonnes⁻¹ LWP waste per year working at current market conditions and environmental regulations in Germany,
- Scenario ES-PS for plant scaling to 600,000 tonnes⁻¹ LWP waste per year (+ 100%),
- Scenario ES-EREG for environmental regulation adjustment to exclude CO₂eq. emissions of CR from emission trading, i.e. no certificate costs for PIII, which is currently not the case [114],
- Scenario ES-MC for market condition adjustment to double attainable prices for CR products on the market compared to conventional products (+ 100%).

Results & Discussion

Figure 1 presents inventory results for the material inputs and outputs of all investigated treatment pathways. Results show that PI produces heat and electricity as central treatment outputs from LWP waste, while PII and PIII produce a variety of products for material recycling and/or versatile applications at chemical plant sites including plastic production. Results from environmental impact assessment are presented and discussed in the next section while results from the techno-economic analysis are presented in the “[Economic Impacts](#)” section. Note that additional results in the form of a fossil carbon balance and an energy balance are provided in Tables S6 & S7 in the SI.

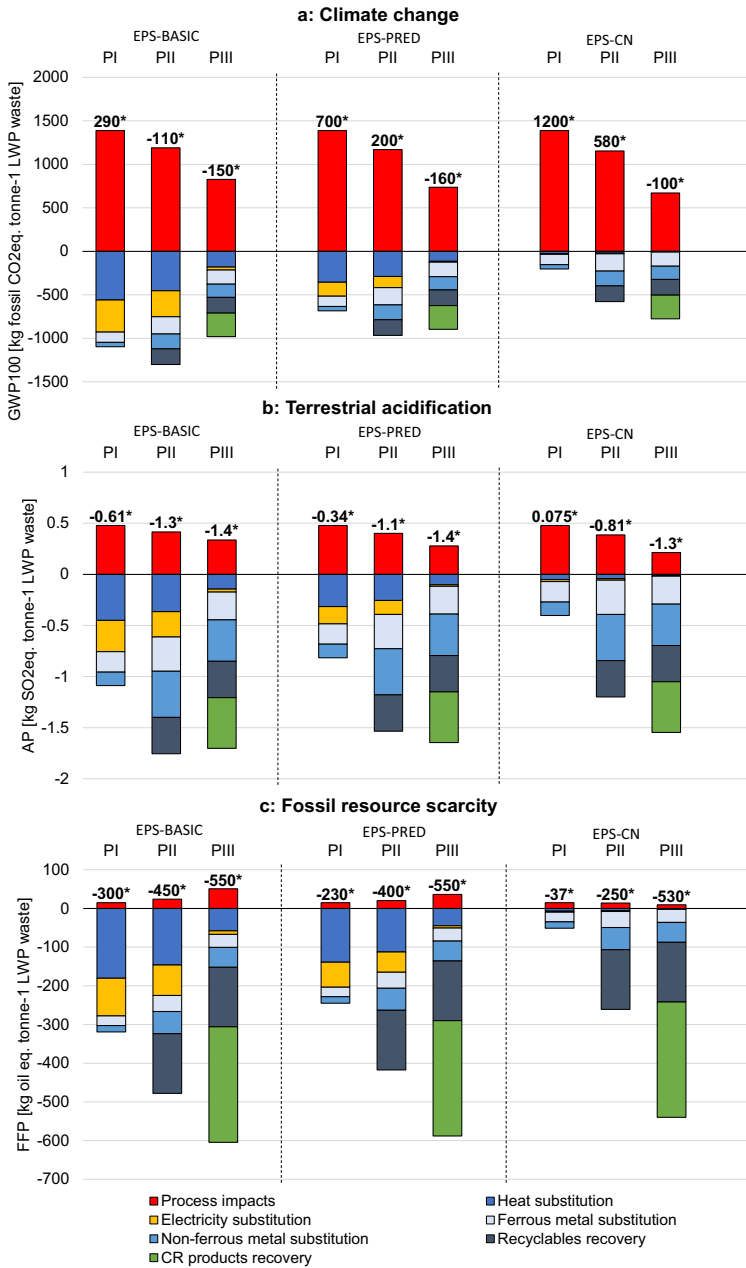


Fig. 2 Global warming potential, terrestrial acidification, and fossil resource scarcity results. AP: Acidification potential. CN: Climate neutrality. EPS: Energy provision scenario. FFP: Fossil fuel potential. GWP: Global warming potential. LWP: Lightweight packaging. PRED: Predominantly renewables. *: Resulting sum. Underlying values rounded to two significant digits

Environmental Impacts

Figure 2a–c present results from climate change, terrestrial acidification, and fossil resource scarcity assessments, respectively. For the analysis, impacts are divided into 1) process impacts, 2) heat substitution, 3) electricity substitution, 4) ferrous metal substitution, 5) non-ferrous metal substitution, 6) recyclables recovery, and 7) CR products recovery. Results of the contribution analysis indicate that heat and electricity substitution have a major effect on the environmental viability of all impact categories. In the following paragraphs, results for the investigated scenarios will be discussed for each individual impact category.

Climate Change

In scenario EPS-BASIC, PI exhibits the highest GWP100 at 290 kg CO₂eq tonne⁻¹ LWP waste, whereas PII and PIII are associated with a GWP100 reduction at -110 and -150 kg CO₂eq tonne⁻¹ LWP waste, respectively. The high GWP100 reduction of PIII is due to low process emissions and chemical product substitution. In scenario EPS-PRED, assuming a predominant renewable energy supply, both PI and PII produce net positive CO₂eq. emissions at 700 and 200 kg CO₂eq tonne⁻¹ LWP waste, respectively, as emission credits for the substitution of electricity and heat are significantly reduced. In contrast, GWP100 reduction associated with PIII remains similar to that in scenario EPS-BASIC. In scenario EPS-CN with a climate neutral energy supply, while the GWP100 reduction effect of PIII is reduced by 50% to -100 kg CO₂eq tonne⁻¹ LWP waste, the GWP100 of PI and PII will further increase to 1200 and 580 kg CO₂eq tonne⁻¹ LWP waste, respectively. Results thus strongly point to the GWP reduction potential of CR-based pathway PIII – irrespective of energy systems – in reducing carbon emissions associated with LWP waste treatment in comparison to incineration-based pathways PI and PII.

Terrestrial Acidification

Across all scenarios, PIII maintains a consistent contribution to reducing AP as a measure for the impact on terrestrial acidification at -1.4 to -1.3 kg SO₂eq. tonne⁻¹ LWP waste. This is because the lower credits for heat substitution in scenarios with increasing renewable energy supply are balanced with lower indirect SO₂eq. emissions that are induced by electricity consumption. In contrast, while incineration-based pathways also contribute to reducing AP, their contribution, in particularly for PI, is dependent on the energy supply in Germany. Hence, as process emissions and emission credits from electricity production become increasingly reduced corresponding to an increasing share of renewable energy supply, AP reduction also decreases considerably. In the case of PI, it reduces from -0.61 in Scenario BASIC to -0.34, and 0.075 kg SO₂eq. tonne⁻¹ LWP waste. Taken together, while all treatment pathways contribute to a AP reduction, the contribution of CR pathway is largely independent of energy supply and remains consistent even assuming renewable energy supply. In contrast, the contribution of incineration-based pathways to AP reduction is dependent on the energy mix and exhibits a tendency to decrease as the share of renewables in the energy mix increases.

Fossil Resource Scarcity

Generally, all pathways contribute to a reduction in FFP as measure for the impact on fossil resource scarcity. For the incineration-based pathways PI and PII, this is primarily due to

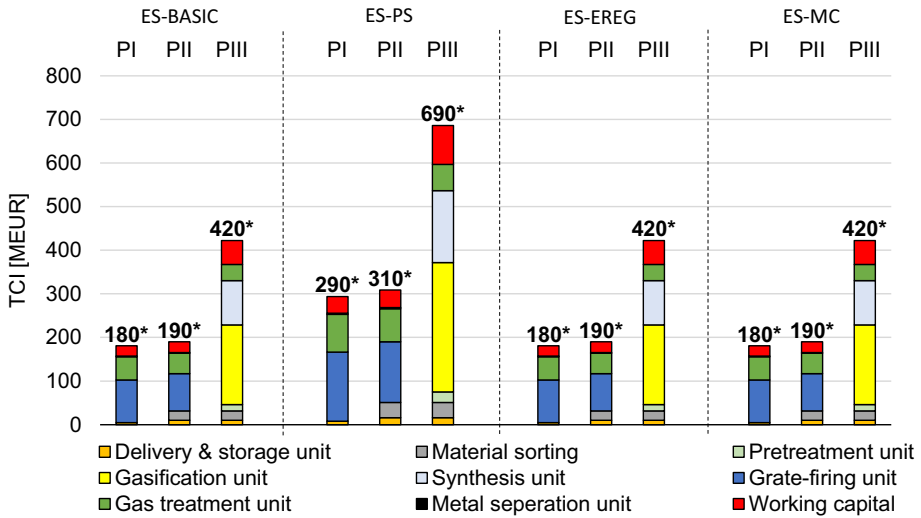


Fig. 3 Total capital investment (TCI). CN: Climate neutrality. EREG: Environmental regulation. ES: Economic scenario. MC: Market conditions. MEUR: Million euros. *: Resulting sum. Underlying values rounded to two significant digits

the substitution of conventional energy. In contrast, for the CR pathway PIII, this is due to the substitution of crude-oil-based olefins and by-products. Similar to climate change and terrestrial acidification, the contribution to decreasing fossil resource scarcity for PIII is observed to be relatively independent of the energy supply (i.e. remains consistent between -530 to -550 kg oil eq. tonne⁻¹ LWP waste). However, as with climate change and terrestrial acidification, the contributions of incineration-based pathways PI and PII are observed to be dependent on the energy mix and reduce significantly as the German energy supply becomes increasingly renewable.

Economic Impacts

Total Capital Investment (TCI)

Generally, PIII requires higher TCI than incineration-based pathways (see Fig. 3). Main drivers for TCI in PIII are investments in the gasification unit and the synthesis unit while grate-firing unit and the gas treatment unit represent the key TCI components of PI and PII. In scenario ES-BASIC, TCI for PIII is observed to be more than double that of PI and PII (i.e. 420 compared to 180 MEUR and 190 MEUR, respectively). Plant scaling is observed to have a significant impact on investments for all pathways. Specifically, for scenario ES-PS, an increase of about 60% TCI (i.e. to 290, 310, and 690 MEUR for PI, PII, and PIII, respectively) is observed for a capacity increase of 100% for all pathways. The results are in line with reported investments for waste incineration plants [69], materials recovery facilities [13], and small scale gasification plants [115], and indicate that upscaling leads to significant economies-of-scale regardless of treatment pathways. No impact on TCI is observed for scenarios ES-REG and ES-MC as assumptions for TCI are analog to scenario ES-BASIC.

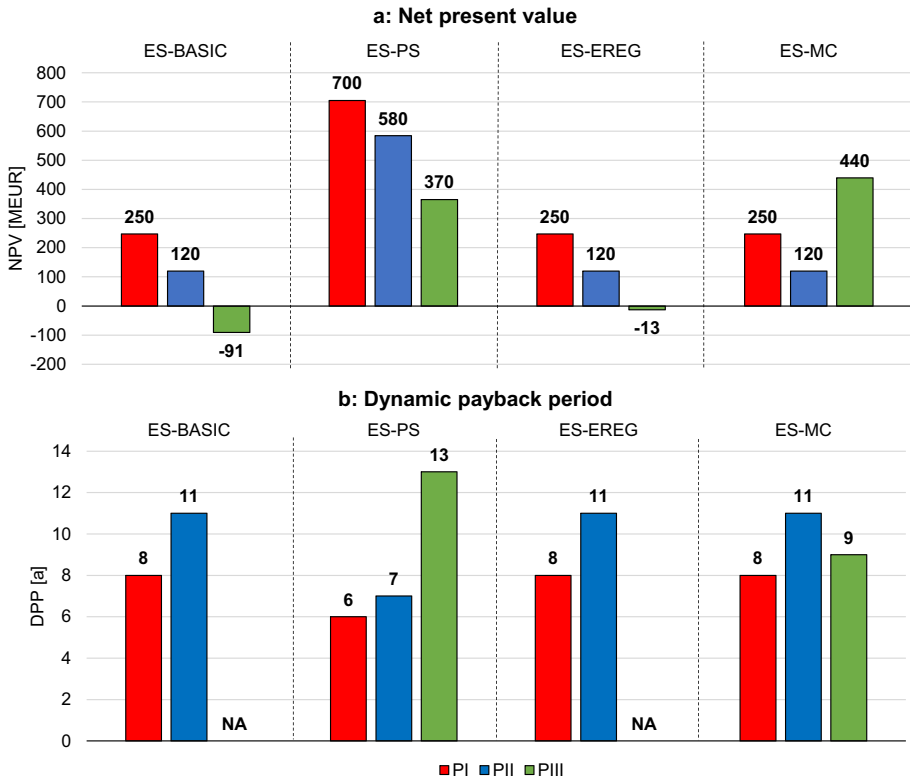


Fig. 4 Net present value (NPV) and dynamic payback period (DPP). CN: Climate neutrality. EREG: Environmental regulation. ES: Economic scenario. MC: Market conditions. MEUR: Million euros. NA: Not applicable. PS: Plant scaling. Underlying values rounded to two significant digits

Net Present Value (NPV)

In scenario ES-BASIC, NPV is highest for PI at 250 MEUR, followed by PII at 120 MEUR and PIII at -91 MEUR (see Fig. 4a). This indicates that PI and PII are profitable while PIII incurs financial losses. Upscaling is observed to have a significant positive impact on profitability for all pathways. When plant capacity is doubled to 600,000 tonnes LWP waste year⁻¹, PIII will exhibit a positive economic return (i.e. 370 MEUR). However, incineration-based pathways will also experience increased profitability at almost double that of the CR pathway (i.e. 700 and 580 MEUR for PI and PII, respectively). While environmental regulation (i.e. CO₂ certificate costs waived for CR) reduces the losses associated with PIII (i.e. from -91 MEUR in scenario ES-BASIC to -13 MEUR), it does not reach higher economic attractiveness than incineration-based pathways. The largest impact on the economic viability of PIII compared to incineration-based pathways is the realization of a price premium for CR products. In scenario ES-MC where market prices for CR products are assumed to be double that of conventional chemicals, CR pathway PIII – at 440 MEUR – will be much more profitable than incineration-based pathways (i.e. 250 and 120 for PI and PII, respectively).

Dynamic Payback Period (DPP)

As illustrated in Fig. 4b, DPP is only relevant for pathways with positive NPV (i.e. investment will be repaid within the investment timeline). Under scenario ES-BASIC, direct incineration with a DPP of 8 years exhibits a better project feasibility than indirect incineration with a DPP of 11 years. With upscaling of plant capacity, the project attractiveness of incineration-based pathways improves with a reduced DPP of 6 years. In contrast, the CR pathway will not be as economically attractive as incineration-based pathways in terms of DPP as it requires twice as long for the investment to be repaid. Nevertheless, even with the longer DPP, the investment will be paid back in less than half of the investment lifespan of the CR plant (i.e. at 33 years). Not surprisingly, environmental regulation relating to CO₂ certificate costs for CR is observed to have no impact on DPP for incineration-based pathways compared to scenario ES-BASIC. Assuming a price premium for CR products, a similar DPP for PIII (i.e. 9 years) and PII (i.e. 8 years) is observed.

Levelized Costs of Carbon Abatement (LCCA)

LCCA is calculated for PII and PIII using PI as the reference pathway. The energy mix is observed to exert a significant impact on carbon emissions for LWP waste treatment pathways (see the “Climate Change” section). This will have a corresponding impact on LCCA. The difference in energy mixes is thus accounted for in the LCCA via the energy scenarios EPS-BASIC, EPS-PRED, and EPS-CN.

Not surprisingly, for all economic scenarios (i.e. ES-BASIC, ES-PS, ES-EREG, and ES-MC), LCCA for both indirect incineration (i.e. PII) and CR (i.e. PIII) – compared to PI as reference pathway – will decrease as the proportion of renewables in the energy mix increases (see Fig. 5). Interestingly, under existing market conditions i.e. ES-BASIC, while PII will incur lower LCCA compared to PIII under EPS-BASIC condition (i.e. 130 compared to 180 € tonne⁻¹ CO₂ eq.), this tendency is reversed as the proportion of renewables in the energy mix increases. Specifically, PIII incurs less LCCA compared to PII in EPS-PRED and EPS-CN at 93 and 62 € tonne⁻¹ CO₂ eq. compared to 110 and 89 € tonne⁻¹ CO₂ eq., respectively. This indicates that with increasing renewables in the energy mix, the attractiveness of CR compared to indirect incineration (i.e. in terms of reduced LCCA) will correspondingly increase.

For PII, the only economic scenario which has an impact in reducing its LCCA is ES-PS. For PIII, not only do plant scaling (ES-PS) and a waiver of CO₂ certificate costs for CR (ES-EREG) lead to a significant reduction of LCCA; in the case of a price premium for CR products (ES-MC), negative LCCA values indicate that carbon saving and cost reduction can be attained at the same time.

Taken together, assuming an increasing proportion of renewable energy supply in the future as economies strive for carbon neutrality and net-zero, CR will present an economically attractive alternative to indirect incineration to abate carbon emissions across diverse economic scenarios.

Conclusion

This study quantitatively evaluates the environmental and economic impacts associated with gasification as an alternative chemical recycling treatment pathway to direct and indirect incineration pathways for lightweight packaging waste. To support the comparative

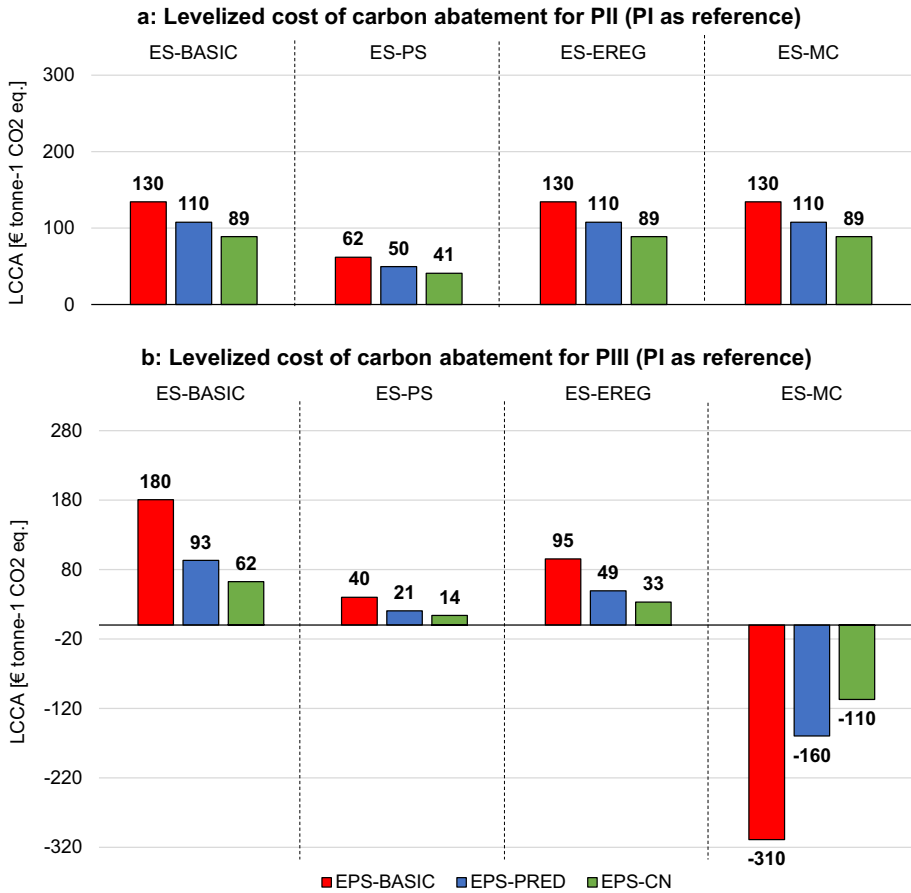


Fig. 5 Levelized cost of carbon abatement (LCCA). CN: Climate neutrality. EPS: Energy provision scenario. ES: Economic scenario. EREG: Environmental regulation. MC: Market conditions. PRED: Predominantly renewable energy supply. PS: Plant scaling. Underlying values rounded to two significant digits

life cycle assessment and techno-economic analyses, a comprehensive inventory dataset is developed to enable in-depth assessments of material flows, environmental impacts, and economic performance of conventional and innovative treatment pathways for lightweight packaging waste. This study provides novel insights into environmental impacts which are associated with the investigated treatment pathways in terms of climate change, terrestrial acidification, and fossil resource scarcity, as well as their sensitivity to changes in the energy system. Additionally, it also enables an empirical overview of associated economic impacts in terms of total capital investment, net present value, dynamic payback period, and levelized cost of carbon abatement, as well as their sensitivity to changes in economic conditions in terms of plant scaling, changes in environmental regulations (i.e. waiver of CO₂ prices for chemical recycling), as well as market conditions (i.e. price premium for CR products).

Study results indicate that chemical recycling of sorting residues of lightweight packaging waste in the form of gasification has a significant positive environmental potential

which is currently unexploited. Unlike incineration-based pathways whose environmental impacts are dependent on emission credits they receive for substitution of electricity and heat, the contribution of chemical recycling to reducing impacts on climate change, terrestrial acidification, and fossil resource scarcity are shown to be relatively independent of the energy mix. Results thus suggest that as economies transform their energy systems towards renewable energy, the environmental contribution of incineration-based treatment of lightweight packaging waste will decrease considerably, whereas that of chemical recycling will remain relatively stable.

From an economic standpoint, the high total capital investment of chemical recycling compared to incineration-based pathways may limit its economic attractiveness for investors. However, results also point to how a multi-pronged approach which integrates generous plant scaling with favorable environmental regulation for chemical recycling and a price premium for chemical recycling products could significantly increase its economic competitiveness such that chemical recycling is comparable, and even more profitable under certain conditions to incineration-based pathways.

To conclude, this research extends the isolated focus of extant chemical recycling studies on technical issues to assess environmental and economic impacts which are associated with the implementation of chemical recycling in the form of waste gasification in comparison to incineration-based treatment pathways. The integrated approach utilized is conducted in line with proven practices of life cycle assessment and techno-economic analysis, and accounts for both upstream and downstream impacts of plastic waste treatment. Key advantages of the applied methods include a detailed and reliable inventory dataset as well as inferences for the realization of chemical recycling in the form of gasification at plant level.

Despite the novel insights which are obtainable through this approach, a central limitation is the limited transferability of attained results to a large-scale, systemic implementation of chemical recycling. Moreover, the focus on Germany also limits the direct transferability of study findings to other nations and contexts. Future research could thus benefit from an integration of data and results from this study with sophisticated systemic modeling approaches, and a consideration of the role of chemical recycling in diverse and wider economies (e.g. in the European Union). Such a systemic approach could lead to a deeper understanding of how chemical recycling could complement other strategies in the complex and interconnected systems of waste treatment, chemical production, and energy provision – including the associated infrastructural, regulatory, and financial dynamics – to address the global plastic waste challenge.

Notwithstanding the identified limitations of the present study and the potential avenues for future research, this study provides a comprehensive assessment of environmental and economic impacts associated with chemical recycling compared to established incineration-based treatment pathways for lightweight packaging waste. It thus contributes to research and development in this field by providing empirical evidence of environmental benefits of chemical recycling as well as economic data on required investments and economic performance. Study results strongly suggest that chemical recycling could play a role as a complementary building block to mechanical recycling to close material cycles in circular economies. It furthermore provides decision-makers with insights into the framework conditions which would be necessary to enable the realization of chemical recycling's potential in addressing the plastic waste challenge.

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Data Availability Not applicable.

Code Availability Not applicable.

Declarations

Conflict of Interest The authors declare no competing interests.

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