Journal of Sustainability (2025) 1:2, 1-20
https://doi.org/10.55845/jos-2025-1270

RESEARCH ARTICLE

Comparative Life Cycle Assessment of a Pre-Commercial Chemical Recycling Process for Post-Industrial Plastic Waste

Darius Steegborn [1], Manfred Renner[1], Philip Biessey[1]*

Received: 4. September 2025 / Accepted: 25. November 2025 / Published: 4. December 2025

Abstract

Chemical recycling technologies are gaining attention, yet their environmental benefits remain debated due to high energy demand, low yields, and limited data quality. This study conducts a comparative life cycle assessment of a 130 kg/h pyrolysis plant converting post-industrial packaging waste into high-value chemicals. Its environmental performance is benchmarked against waste incineration and virgin chemical production. The chemical recycling process emits 1.4 $kg_{CO2\text{-}eq}$/kg feedstock, 50 % lower than for incineration. Considering product substitutions, emissions of the recycling process are reduced up to -0.9 $kg_{CO2\text{-}eq}$. Compared to virgin production, the chemical recycling process provides a suitable alternative, generating up to 40 % less CO_2 emissions. It also results in lower impacts on acidification and fossil fuel depletion, although freshwater and marine eutrophication are higher than those of fossil-based production. The results show high dependency on data and methodological assumptions that can reduce the emissions by up to 64 % but can increase them by 114 %.

Keywords: Chemical Recycling · Life Cycle Assessment · Pyrolysis · Plastic Waste Utilisation

1. Introduction

The global production of plastics has been steadily increasing in recent years (Plastics Europe, 2022a), and both the quantities produced (Lau et al., 2020) and the associated waste (Geyer, 2020) are expected to double by 2040, while only 9 % of the waste generated worldwide is recycled (OECD, 2022). At the same time, the European Union's Green Deal (European Commission, 2019) has set the goal of becoming carbon neutral by 2050. In order to achieve this goal for the plastic sector, which has a significant share of the total emissions (Chaudhari et al., 2022), the development and implementation of a circular economy for the plastic sector is an important step (Meys et al., 2021). The preferred pathways for a circular economy are waste prevention strategies such as reduce, refuse and reuse, followed by short-loop strategies such as repair, remanufacture and repurpose (Awino & Apitz, 2024). Still, from a circular economy perspective, resulting waste needs to be recycled rather than incinerated for recovery or disposed of (Awino & Apitz, 2024).

State-of-the-art technologies to recycle plastics are mechanical recycling processes that struggle with complex waste compositions, unsortable compound materials and impurities, resulting in lower-grade quality

---

*Corresponding author: biessey@rpe.rub.de
[1] Ruhr University Bochum, Universitätsstraße 150, 44803 Bochum, Germany

products (Dai et al., 2022; Ragaert et al., 2017). Therefore, to additionally convert those complex waste streams, the implementation of chemical recycling processes is needed to increase the recycling rate (Lase et al., 2023). Especially, the development of pyrolysis and liquefaction processes are discussed, which are able to convert those plastic wastes into a variety of monomers by cracking the polymer chains (Lechleitner et al., 2020; Zeller et al., 2021). The resulting products are mostly oil fractions that are reintegrated into the chemical industry (Biessey et al., 2023).

To assess the environmental impacts and potential savings of such processes, an ecological assessment of these technologies must be performed. While there is a variety of chemical recycling life cycle assessments (LCA) available today, recent review papers advise future studies to improve data quality, methodological boundary conditions, and data transparency (Antelava et al., 2019; Davidson et al., 2021; Pires Costa et al., 2022). Previous LCAs have shown that mechanical recycling has a lower environmental impact than pyrolysis and liquefaction, mostly due to its lower energy consumption (Meys et al., 2020). Consequently, chemical recycling processes should not focus on feedstocks which are suitable for mechanical recycling, indicating that both processes should be viewed as complementary technologies (Biessey et al., 2023; Davidson et al., 2021). The spectrum of remaining feedstocks for chemical recycling differs in their composition and impurities, leading to different products and product compositions (Genuino et al., 2023), which in turn influences the results of an LCA (Antelava et al., 2019). To transparently link LCA results to the respective feedstocks and to enable a comparison between different LCA studies, a feedstock characterisation should be stated. The state-of-the-art treatment of non-mechanically recyclable plastic waste is incineration. Past studies have sufficiently demonstrated that the incineration of plastic waste streams has a greater environmental impact than the utilisation via chemical recycling (Civancik-Uslu et al., 2021; Hermanns et al., 2023; Wang et al., 2015). These statements were mostly based on laboratory-scale investigations and respective literature data, or data obtained from process simulation. The interpretation and comparability of these prospective LCAs are linked to uncertainties arising from data limitations (Moni et al., 2020) and variations in scale-up methods (Erakca et al., 2024). To reduce the uncertainties, processes with a higher technology readiness level need to be evaluated (Pires Costa et al., 2022) and compared to the state-of-the-art. Therefore, the comparison is nevertheless considered an objective in life cycle studies. Other LCA studies shift the focus away from the waste treatment comparison, either by calculating the environmental impacts based on the derived products of the chemical recycling processes (Da Monteiro et al., 2022; Iribarren et al., 2012; Kulas et al., 2023) or, with a treatment and a product comparison, defining two objectives in one study (Das et al., 2022; Jeswani et al., 2021). For such a comparison, the products must meet the specific requirements and specifications of the respective substituted products. Therefore, separating and upgrading the derived product is an indispensable step to reintegrating them into the chemical industry (Civancik-Uslu et al., 2021; Kusenberg et al., 2024). Mainly thermal separation processes, such as distillation, are used for the fractionation of pyrolysis oils (Faisal et al., 2023), and the product oils are processed by hydrogenation to lower the olefin share and for the removal of heteroatoms (Kusenberg et al., 2022). During the distillation, the product oils are typically separated into three fractions: A naphtha fraction with a boiling point < 180 °C, a diesel fraction with a boiling point between 180 and 360 °C and a heavy oil fraction with a boiling point > 360 °C (Faisal et al., 2023; Zeb et al., 2023). During hydrogenation, the double bonds of the olefins are saturated with hydrogen and heteroatoms are removed in the presence of a catalyst. The process conditions in literature vary between 275 and 420 °C and 5 and 6.5 MPa. (Almohamadi et al., 2021; Dębek & Walendziewski, 2015; Djandja et al., 2021; Hita et al., 2015) As these upgrading steps can have a significant environmental impact, they should be considered in the methodology of LCAs (Holtkamp et al., 2024).

Still, even in most recent studies (Chari et al., 2025; Garcia-Garcia et al., 2024; Klotz et al., 2024; Schwartz et al., 2025; Tsangas et al., 2024), the process data is mostly derived from literature, process simulations or scaled laboratory experiments and only partly accounts for the emissions of product upgrading and separation, limiting their representativeness for near-commercial systems. Furthermore, the different product-oriented functional units in each study make it difficult to compare the technologies ecologically. To address these limitations, this study contributes with an LCA, using primary operational data from a technology readiness level (TRL) 8 continuous liquefaction plant to quantify real-world environmental performance, supplementing inventory data for product post-process with process simulation data and introducing a product-oriented functional unit to enhance the comparability between different studies. All methodological boundaries, as well as the environmental impacts of each process step, are reported individually, ensuring a transparent and traceable database for comparison with future LCA studies.

2. Materials and Methods

The LCA study was conducted according to the standards of DIN EN ISO 14040/44 (ISO, 2006a, 2006b) and in accordance with the specific chemical recycling guidelines of Chemical Recycling Europe (Sphera, 2025) and general life cycle inventory guidelines for the plastics sector as of Plastics Europe (Plastics Europe, 2022b). Even though social assessment of waste management has become an objective in recent literature (Ardolino et al., 2023; Gutierrez-Lopez et al., 2024), it is excluded from this publication.

2.1. Definition of goal and scope

Two objectives are defined in this study: The first objective is to compare environmental impacts for the treatment of two non-mechanically recyclable feedstocks in the chemical recycling process with the state-of-the-art, which is incineration for both feedstocks. To enable this comparison, a functional unit of 1 kg of feedstock is defined. The second objective is to compare the environmental impact of the chemical recycling products in relation to the environmental impact of conventional production routes of the respective products. Therefore, a second functional unit is defined as 1 kg of total obtained products (TOP). TOP is the sum of all chemicals that are declared as a product in the LCA. It is chosen as a functional unit because chemical recycling products vary depending on the chemical recycling technology and feedstocks used, and thus enables a comparison to future LCA studies that are based on different technologies. As this functional unit sums up all products, there are no calculated avoided burdens by side stream substitutions, which therefore enables a product-independent comparison to other recycling processes that treat similar feedstocks but result in other products. Although TOP aggregates product streams, individual environmental intensities are normalised per kilogram of product in the supplementary information.

The Life Cycle Impact Assessment was calculated in the software OpenLCA, using the method “Recipe Midpoint (H)”, which is widely used in the chemical industry (Santos et al., 2019). Since chemical recycling processes aim at providing chemical products with reduced CO_2 emissions compared to fossil-based routes, and the overall reduction of fossil resource dependencies by utilising waste material (Ellen MacArthur Foundation, 2017), the impact categories global warming potential (GWP) and fossil fuel depletion potential (FFDP) are considered for the assessment and discussion. As chemical production processes also have an impact on terrestrial and aquatic ecosystems, the categories marine (MEP) and freshwater (FEP) eutrophication potential and terrestrial acidification potential (TAP) are also discussed. The system was modelled using substitutions to account for the avoided production of fossil equivalents, consistent with recent chemical recycling LCA literature. The use of mass allocation was avoided, as the economic value of the different products, e.g., naphtha to caprolactam, differs by a factor of up to four (BusinessAnalytiq, 2025a, 2025b) as well as economic allocation, as the price of oil derivatives is fluctuating (BusinessAnalytiq, 2025a) leading to comparability issues with future or past LCA studies. For result simplification, individually for each impact category, process parts that had an impact of less than 1 % in the category are cut off.

2.2. System boundaries

The processes, including pretreatment and product upgrading, and the resulting system boundaries of liquefaction and the comparison processes are described in the following sections. The foreground system represents the considered process steps. The background system shows the origin of the inputs, the treatment method of the wastes and the substituted reference materials of the derived products.

2.2.1. Chemical recycling process The system boundaries of the recycling process are shown in Figure 1. The process was developed by Carboliq GmbH and is operated by Carbowest GmbH. The process has been assigned a technology readiness level of 8 due to its ability to process feedstocks as refuse-derived fuel and multilayer waste with a feedstock-dependent capacity of between 100 and 200 kg/h (Biessey et al., 2023). In the first step, the feed material is sieved for particles smaller than 2 mm and then run through non-ferrous and ferrous separators. Before being forwarded into the reactor, the feed is preheated, a zeolite catalyst is added, and the residual moisture is removed. The reactor operates electrically. Evaporating products are quenched and, subsequently, separated with a phase separator into an aromatic-rich and an olefin-/paraffin-rich fraction and then transferred to a storage tank. The gaseous phase is treated in a regenerative thermal oxidation plant. Solid residues of the reactor are collected in a container and disposed of as refinery sludge in hazardous substance incineration. The water obtained by drying the feedstock is sent to a wastewater treatment plant.

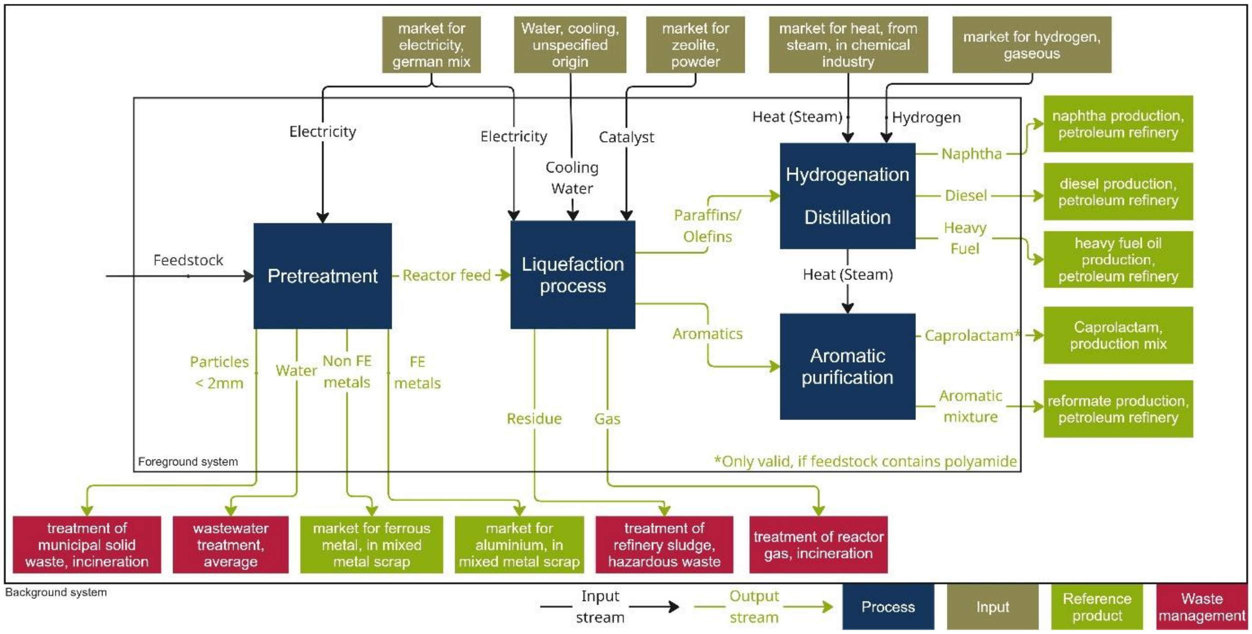

Figure 1. System boundaries of the life cycle inventory. The foreground system represents all material and energy flows. The background system shows the origin of the inputs (brown), the ways of treating the waste streams (red) and the substituted products (green)

The olefin- and paraffin-rich fraction is separated into three fractions in a distillation column: naphtha, which consists of carbon chains with a boiling point of less than 200 °C (GESTIS-Stoffdatenbank, 2024b), a diesel fraction (200 - 360 °C) (GESTIS-Stoffdatenbank, 2024a) and a heavy oil fraction (>360 °C). The resulting product streams were hydrogenated at 350 °C and 50 bar. As both feedstocks contain PA, the aromatic-rich fraction consists mainly of the PA monomer caprolactam and a BTEX fraction. As caprolactam is energy-intensive to produce (Hong & Xu, 2012) and considered a high-value chemical, it is isolated and purified. This is typically done in a thermal separation sequence (Hugo Fuchs, 1994; Losier, 1995; Tinge et al., 2000). The industrial purity is between 99.5 and 99.9 wt.-% (Domo Chemicals GmbH, 2022; Fibrant, 2024) and is set to 99.9 wt.-%. Unlike the other process steps, caprolactam purification is only considered if the feedstock contains PA. Energy for the upgrading of all oil products is provided by steam.

2.2.2. Incineration process The system boundaries and the mass balance of the incineration process are determined using Doka's municipal waste incineration model, which calculates the emissions based on the specific composition of the waste. The model uses the elemental composition of the feedstocks as an input, which are derived from the combustion analyses for both feedstocks shown in Table 1.

Table 1. Combustion analyses of the examined feedstocks A and B

Component	Feedstock A	Feedstock B	Unit
C	74.0	80.8	wt.%
H	12.0	11.5	wt.%
N	4.0	1.4	wt.%
S	-	< 0.1	wt.%
O	5.0	5.2	wt.%
Cl	-	0.2	wt.%
F	-	200	ppm
P	107	-	ppm
Ti	180	-	ppm
Si	960	-	ppm
Fe	260	-	ppm
Ca	3060	-	ppm
Al	390	-	ppm

With the transfer coefficients of the model, the emissions are categorised into direct air and water emissions and electrostatic precipitators (ESP) ash, boiler ash, bottom ash and scrubber sludge. (Doka, 2013) The resulting system boundaries are shown in Figure 2.

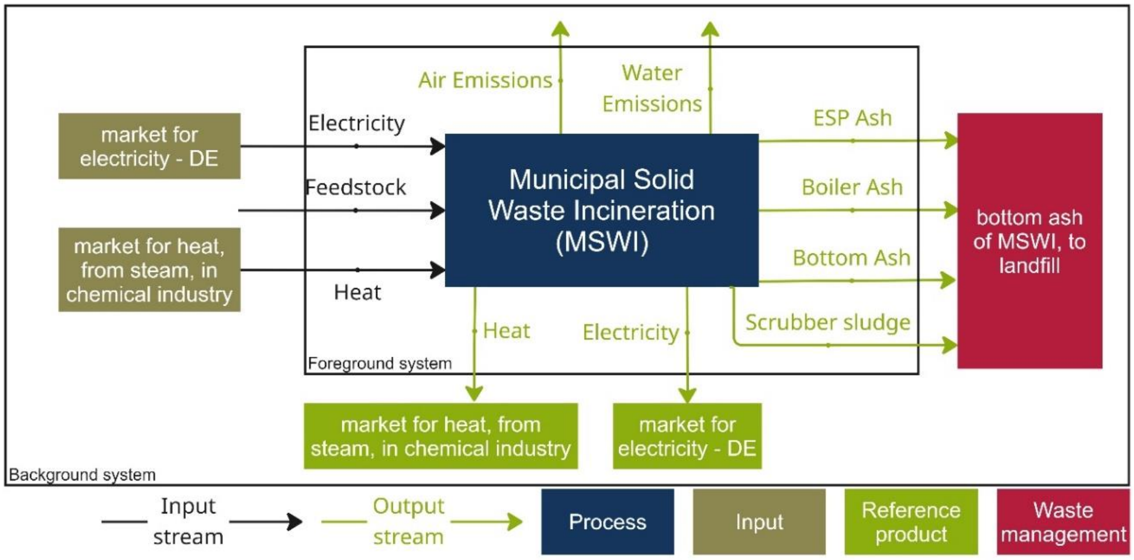

Figure 2. System boundaries of the incineration process based on the inventory model of Doka from 2013 (Doka, 2013)

The incineration process requires electricity, especially for flue gas treatment, and thermal energy to preheat the combustion air. Direct emissions generated are released into the atmosphere and groundwater. The ash residues are deposited in landfills. (Egli, 2005) In addition, the process produces both electrical and thermal energy. The quantities are calculated using the efficiencies taken from Doka's model with $\eta_{electrical}$ of 15.84 %

and $\eta_{thermal}$ of 28.51 % from the lower heating value (LHV) of the feedstock according to Equation 1 (Doka, 2013). The lower calorific value can be estimated specifically for the feedstock composition using Werner Boie's formula, Equation 2 (Boie, 1957). Where m is the mass fraction of the indicated elements carbon (*C*), hydrogen (*H*), oxygen (*O*), sulfur (*S*) and the water fraction (*W*).

$$q_{el} = \eta_{el} \times LHV\ (MJ/kg)\ and\ q_{heat} = \eta_{heat} \times LHV\ (MJ/kg) \qquad Eq.1$$

$$LHV = 33.9 \times m_C + 93.9 \times m_H - 10.8 \times m_O + 33.9 \times m_S - 33.9 \times m_W\ (MJ/kg) \qquad Eq.\ 2$$

2.3. Data acquisition

To obtain the mass balance for the foreground system of the recycling process, an experimental run was performed for each feedstock. The period for the runs depended on the amount of available feedstock. In a period of 51 hours for Feedstock A and 74 hours for Feedstock B, a total of 6747 kg and 8927 kg of feedstock were treated, which equals a mean capacity of 132 kg and 121 kg per hour. The feedstocks, wastes and products were weighed before and after the conducted period. The waste process gas was measured with a flowmeter, and the electricity supply for the entire process was measured with an electricity meter. The composition of the product oils was analysed using two-dimensional gas chromatography. The results were clustered by carbon chain length into paraffins, olefins, alkylbenzenes and single compounds > 0.5 wt.-%. The resulting component list was implemented in ASPEN Plus. The electrical and thermal energy requirement for the product purification and separation was calculated based on the ASPEN Plus simulation. Details on the separation sequence are provided in the supplementary information.

The background data for the provision of the inputs, the treatment of wastes and the substituted products are taken from the database ecoinvent 3.9.1 (ecoinvent, 2022). Therefore, the electrical energy mix is calculated using the grid from Germany in 2023 (Icha & Lauf, 2024). The obtained oil fractions substitute the respective refinery products naphtha, diesel and heavy oil. The caprolactam obtained from the aromatic-rich fraction substitutes for conventionally produced virgin caprolactam. The aromatic-rich BTEX fraction substitutes reformate. The inventory data for the caprolactam is not available in ecoinvent and is taken from the Carbon Minds database (Carbon Minds, 2024). The inventory data of the incineration case was calculated using the presented model by Doka.

3. Results

3.1. Life cycle inventory of the recycling process

Table 2. Life cycle inventory of the chemical recycling process based on 1 kg of feedstock

Input and Output Flows		Feedstock A	Feedstock B	Unit
Inputs	Feedstock	1.00	1.00	kg
	Catalyst	19.70	17.80	g
	Electricity Recycling Process	2.05	1.41	kWh
	Steam product oil upgrading	0.26	0.18	kWh
	Electricity product oil upgrading	0.16	0.08	kWh
	Hydrogen	45.50	23.90	g

Table 2 (Cont.). Life cycle inventory of the chemical recycling process based on 1 kg of feedstock

Input and Output Flows		Feedstock A	Feedstock B	Unit
Output	Paraffines/Olefins	0.75	0.40	kg
	of which Naphtha	0.15	0.06	kg
	of which Diesel	0.39	0.23	kg
	of which Heavy Oil	0.21	0.11	kg
	Wastewater	0.04	0.03	kg
	Aromatics	0.12	0.28	kg
	of which caprolactam	0.11	0.22	kg
	of which BTEX	0.01	0.06	kg
	Particles < 2 mm	0.00	0.01	g
	Ferrous metals	1.10	0.80	g
	Nonferrous metals	0.00	0.20	g
	Residue	0.00	0.15	kg
	Process gas	0.11	0.14	kg

The life cycle inventory of the recycling process, based on 1 kg of processed feedstock, is shown in Table 2. The results based on the functional unit of total obtained products are available in the supplementary information. The energy consumption for the treatment of 1 kg of feedstock is higher for Feedstock A with 2.47 kWh compared to 1.67 kWh for Feedstock B. Considering the paraffin/olefin and aromatic fraction as products, the conversion of Feedstock A is 87 % and for Feedstock B 68 %. The higher energy demand can therefore be correlated with the higher conversion rate of Feedstock A. The higher hydrogen consumption of Feedstock A is due to the higher share of olefinic oil. The amount of non-condensable gases is similar, with 11 % for Feedstock A and 14 % for Feedstock B. However, due to the lower conversion of Feedstock B, the residue is significantly higher at 15 % compared to almost 0 % for Feedstock A. Both feedstocks show a relatively low content of impurities as metals, moisture, small particles and sand, which is due to the post-industrial origin. Post-consumer plastics might lead to higher shares of impurities, which could lead to lower oil yields.

3.2. Life Cycle Impact Assessment

The Life Cycle Impact Assessment was calculated for both functional units, and the results are discussed for both objectives of the study in a separate chapter.

3.2.1. Comparison of the chemical recycling process to incineration The environmental impacts caused by the chemical recycling process and the incineration process are shown in Figure 3 for the categories GWP, MEP, FEP, FFDP, and TAP.

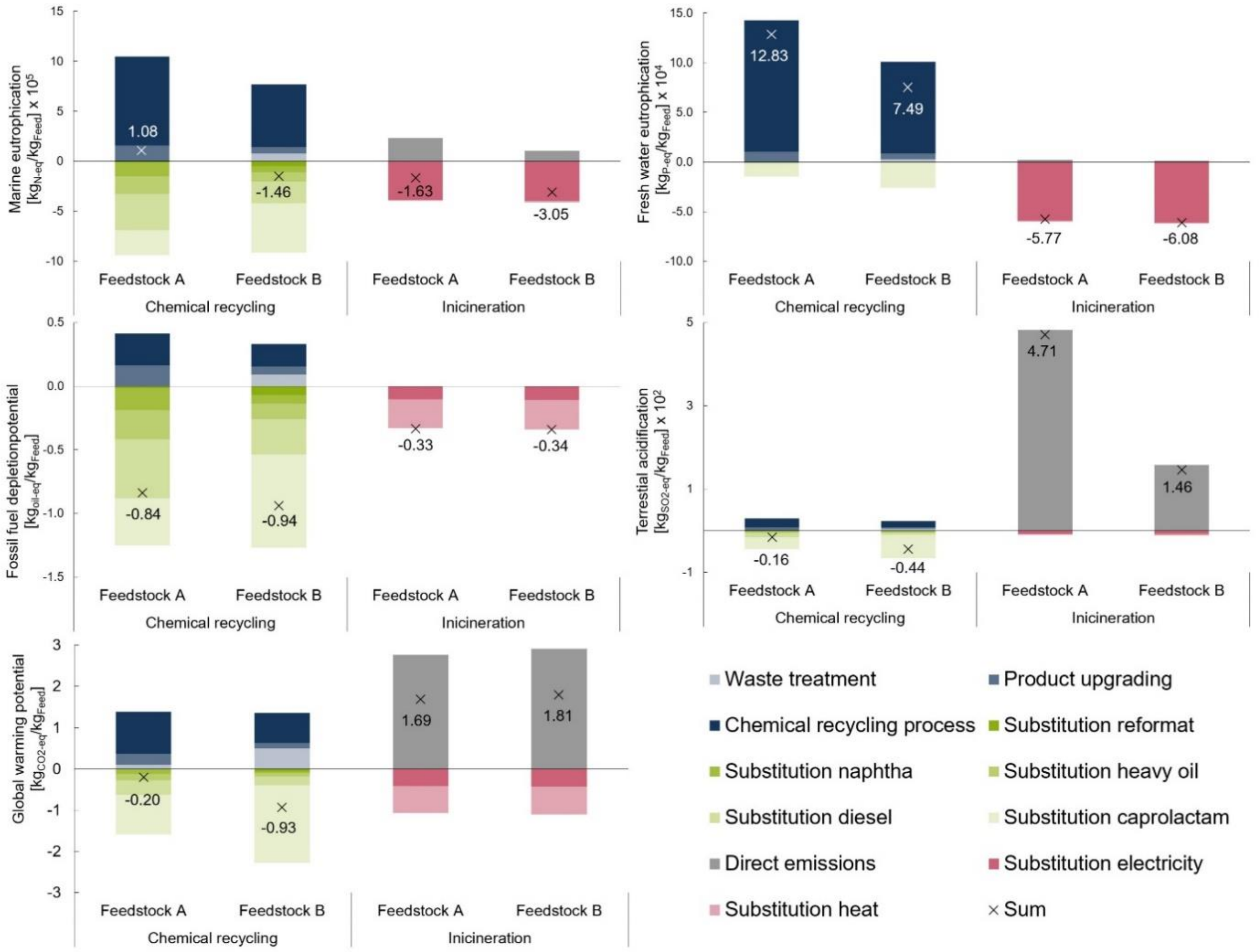

Figure 3. Calculated environmental impacts for the chemical recycling and incineration case based on 1 kg feedstock for the impact categories MEP (marine eutrophication potential, top left) and FEP (freshwater eutrophication potential, top right), FFDP (fossil fuel depletion potential, middle left), TAP (terrestrial acidification potential, middle right), and GWP (global warming potential, bottom left).

The amount of CO_2 equivalents generated by the chemical recycling process is 1.39 kg and 1.35 kg per kg feedstock, roughly the same for both feedstocks, although the origin varies: For Feedstock A, 73 % of the total emissions are caused by the chemical recycling process itself and 7 % by the waste treatment of the solid residue and gas phase. For Feedstock B, a proportion of only 53 % of the emissions is caused by the chemical recycling process and with 36 % a larger proportion is attributed to the treatment of the waste streams, which is due to the larger quantities of residue and gas phase. The impact of product stream upgrading also takes a significant share of both feedstocks, accounting for between 10 and 19 % of the CO_2 equivalents. The avoided emissions granted for the substitution of the produced chemicals depend not only on the overall conversion. They also depend on the yields of the individual products, which are influenced by the respective feedstocks. Caprolactam, which is obtained from the PA fraction in the feedstock, has a share of 60 - 82 % of the total avoided CO_2 equivalents. Despite it is only representing 13 - 32 wt.% of the total products obtained. This is due to the high energy demand and emissions associated with the fossil-based production of caprolactam. Accounting for the generated and the avoided emissions due to substitution, both feedstocks result in a net negative GWP. Feedstock A with -0.20 kg CO_2 per kg input and -0.93 kg for 1 kg of Feedstock B. The calculated emissions from the incineration process are also almost the same for both feedstocks A and B at 2.8 and 2.9 kg CO_2 per kg feedstock. Therefore, they are 99 % and 116 % higher compared to the chemical recycling scenario. Considering the substitutions, the ecological preference of the chemical recycling process even increases.

The MEP for feedstocks A and B, at 10.5 and 7.7 x 10^{-5} kg nitrogen equivalents per kg feedstock, are 5 and 7 times higher than for the incineration process. This is mainly due to the electrical energy used in the process, which contributes more than 80 % of the emissions. Nitrogen oxide emissions from the burning of fossil fuels such as coal, oil and gas and the further formation of nitrates, lead to damaging aquatic ecosystems. An increasing share of renewables, with only a mere fraction, can reduce emissions. Taking into account the avoided emissions for the substitutions, the MEP of the recycling process for Feedstock A is reduced to 1.1 and for B to a net negative -1.5 x 10^{-5} kg. This lowers the gap to the incineration but still has a higher environmental impact of 1.3 and 2.5 x 10^{-5} kg nitrogen equivalents for both feedstocks. The FEP of the recycling process is 14.3 and 10.1 x 10^{-4} kg phosphorus equivalent, which is significantly higher than the incineration process with 0.2 and 0.1 x 10^{-4} kg, favouring the incineration for this impact category. As for the MEP, about 90 % of the emission contribution is due to the electrical energy supply for the process. Product substitution has a comparatively smaller impact on the emissions of the recycling process compared to the other impact categories. It reduces FEP between 10 and 26 % for Feedstock A and B, while the calculated emissions for the incineration reduce to a net negative -5.8 and -6.1 x 10^{-4} kg phosphor equivalent.

The FFDP for Feedstock A is 0.42 kg oil equivalent per kg feedstock, which is 25 % higher than the FFDP for Feedstock B of 0.33 kg. This is mainly due to the higher energy demand for the process and the preparation of the products. The disposal of the waste streams has no impact on the FFDP for Feedstock A. However, it accounts for 27 % of the FFDP for Feedstock B due to the higher proportion of the gas and residual phase. Considering the substituted products, the FFDP for both feedstocks is reduced to a net negative of -0.84 and -0.94 kg oil equivalent. Despite the higher overall conversion of Feedstock A, the substituted amount of oil equivalent is the same for both feedstocks. This is because Feedstock B produces more aromatics and caprolactam, which require more energy to produce and therefore result in a higher FFDP. The incineration process has no direct fossil consumption and is therefore lower than the FFDP of the recycling process. Taking into account the substitutions, the FFDP calculates to a net negative of -0.33 and -0.34 kg oil equivalent per kg feedstock due to the substitution of electricity and heat. Both processes lead to a calculated saving of fossil resources, but the chemical recycling process, with twice to three times higher saving potential, is preferred.

The TAP of the recycling process of Feedstock A is 0.29 x 10^{-2} kg SO_2 equivalent per kg feedstock, which is 20 % higher than that of Feedstock B with 0.23 x 10^{-2} kg. This is due to the increased energy requirement of the reaction. By calculating the substitution, the TAP is reduced to net negative -0.16 and -0.44 x 10^{-2} kg SO_2 equivalent. Although the total product yield is higher for Feedstock A, the substituted value of the TAP is 35 % lower than for Feedstock B. This is due to the higher yield of caprolactam for Feedstock B, which is more emission-intensive to produce than the paraffin products. The TAP of the incineration process is 4.7 and 1.5 x 10^{-2} kg SO_2 equivalent caused by the direct emissions generated during the incineration. The difference between the feedstocks is due to the higher nitrogen content of Feedstock A. Comparing the processes, the environmental impact of the incineration of Feedstock A and B is 17 and 7 times higher than that of the recycling process, even increasing when taking into account substitution effects.

Both the direct emissions and the calculated substituted emissions vary between the two feedstocks. This means that the choice of feedstock with the linked overall conversion and yield of different products has a major impact on the environmental performance of the chemical recycling process. The evaluation of the comparison between the recycling process and incineration favours the chemical recycling process with lower emissions in the GWP, FFDP and TAP categories. When comparing the impact categories for the aquatic ecosystems, incineration has significantly lower environmental impacts in FEP and MEP.

3.2.2. Comparison of the chemical recycling process to virgin production By comparing the products of the recycling process with conventional production, the results are calculated based on the TOP. The TOP includes all upgraded olefin/paraffin and purified aromatic oil fractions defined in the system boundaries. Therefore, substitutions are no longer considered when calculating the emissions. The different total conversion of the two feedstocks changes the amount of waste entering the system for 1 kg of product. At a conversion rate of 87 %, 1.15 kg of waste is required for Feedstock A and 1.46 kg of waste is required for Feedstock B at a conversion rate of 68 %. The results of the environmental impact calculation of the recycling process and the conventional production of products are shown in Figure 4.

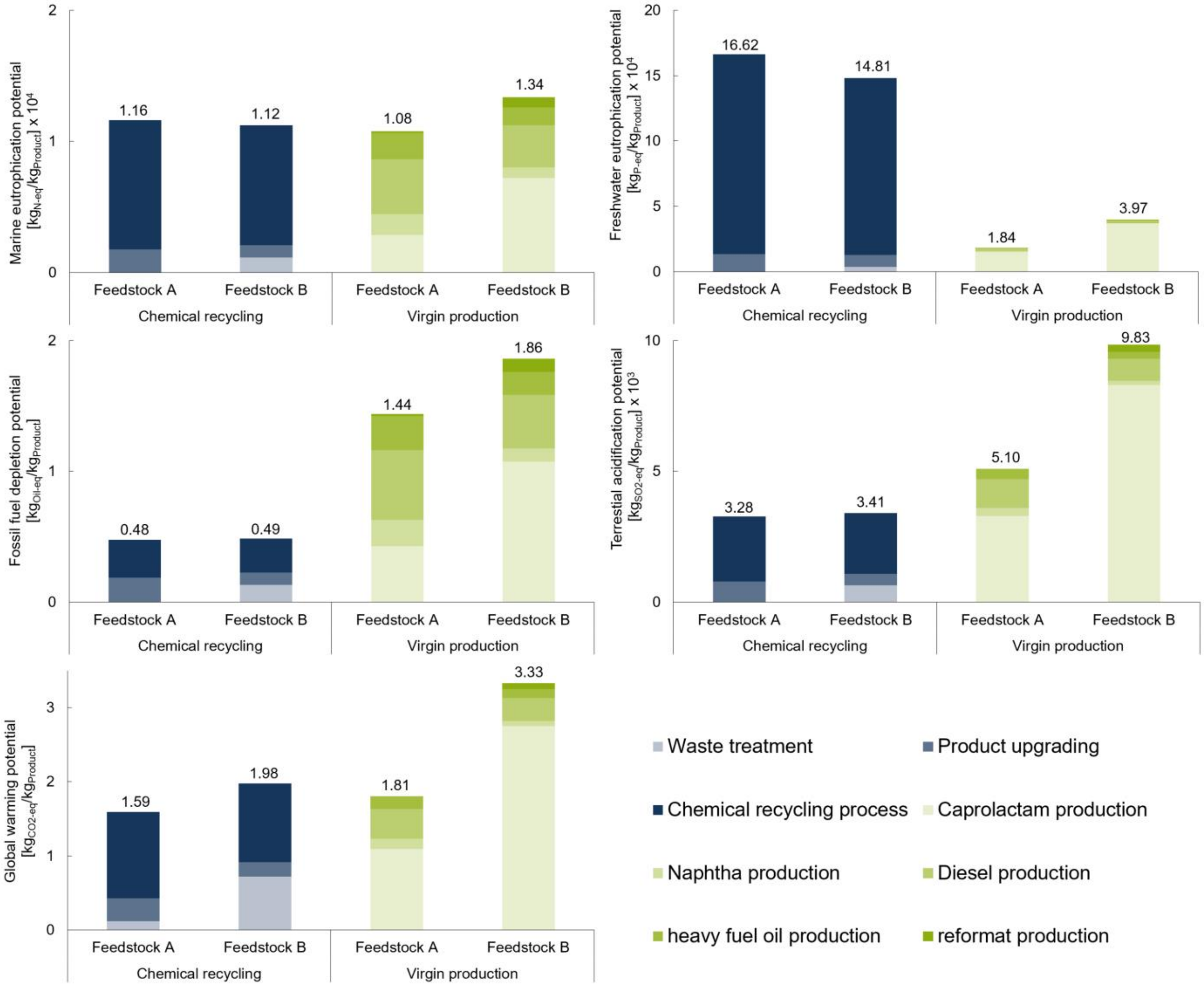

Figure 4. Calculated environmental impacts for 1 kg of obtained products by the chemical recycling process and the conventional production for the impact categories MEP (marine eutrophication potential, top left) and FEP (freshwater eutrophication potential, top right), FFDP (fossil fuel depletion potential, middle left), TAP (terrestrial acidification potential, middle right), and GWP (global warming potential, bottom left).

Feedstock B has a GWP of 2.0 kg CO_2 per kg of product, which is 25 % higher than Feedstock A's 1.6 kg, mainly due to higher emissions from the waste treatment of the generated solid residue and gas fraction. For both feedstocks, the recycling process generates fewer emissions than the fossil production, with 1.81 kg and 3.33 kg of CO_2 per kg of product, respectively. While the emissions are 12 % lower for Feedstock A, they are 40 % lower for Feedstock B due to the high caprolactam yield, showing that the product composition has an impact on the LCA results. The FFDP of the recycling process is similar for both feedstocks at 0.48 and 0.49 kg oil equivalent per kg product, while the fossil production contributes three and four times as much to the

FFDP at 1.4 and 1.9 kg. This shows that production using the chemical recycling process places less strain on fossil resources and can therefore contribute to a circular economy.

The TAP of the recycling process is also in the same order of magnitude for both feedstocks, at 3.3 and 3.4 x 10^{-3} kg SO_2 equivalent per kg products. The comparison process for fossil production with 5.1 and 9.8 x 10^{-3} kg shows significantly higher emissions of 55 and 188 %, indicating a higher pollution of terrestrial ecosystems. The MEP of the recycling process for Feedstock A and B is 1.16 and 1.12 x 10^{-4} kg nitrogen equivalent per kg product. The environmental impact of conventional production is in a similar range in both cases, at 1.08 and 1.34 x 10^{-4} kg nitrogen equivalent. The emissions from the fossil route are 7 % lower for Feedstock A, but 19 % higher for B, so there is no clear trend towards either scenario. The FEP of the recycling process is 16.6 and 14.81 x 10^{-4} kg phosphorus equivalent per kg of products. The emissions of the conventional route are 1.84 and 3.97 x 10^{-4} kg, which are 89 and 73 % less than the recycling process. This shows that the recycling process has a greater impact on freshwater ecosystems than the conventional route.

For the impact categories GWP, FFDP and TAP, the assessment shows that the recycling process has a lower environmental impact than the fossil route. For MEP, no clear statement can be made as the emission values are in the same order of magnitude. For FEP, conventional production has significantly lower emissions. The differences in the calculated environmental impacts between the different feedstocks are between 2 and 11 % based on Feedstock A, showing only small variations. The only exception is the GWP, where Feedstock B has 25 % higher emissions than Feedstock A, which is caused by the larger amount of treated waste streams. In contrast, the influence of the type of substituted products is much stronger, with differences ranging from 24 to 116 % between the two feedstocks. This means that the main influence on the environmental impact of this recycling process does not depend on the type of input material and the overall conversion, but rather on the type and quality of the products obtained.

The comparison of the results to other studies should be interpreted carefully, because every study uses a different database, assumptions, and methods. Nevertheless, it can give a categorisation of the results. The GWP is calculated in almost every study and can therefore be used for comparison, as shown in Table 3. From the waste perspective, the chemical recycling process with -0.20 and -0.93 $\text{kg}_{\text{CO2-eq}}$ per $\text{kg}_{\text{Feedstock}}$ shows lower emissions than most studies. This is mainly due to caprolactam products derived from PA in the waste, which account for up to 82 % of the total substitution credits. Without the isolation of the monomer of caprolactam, the GWP of ~ 0.75 to 0.95 $\text{kg}_{\text{CO2-eq}}$ would align with the studies used for comparison. This shows that the presence of PA in the feed leads to a promising scenario for chemical recycling from an environmental perspective. Comparing the results of this study, 1.6 to 2.0 $\text{kg}_{\text{CO2-eq}}$ per kg TOP, with other product-oriented LCA studies is challenging because the functional units used in the literature vary according to the type of product, whether oil derivatives or specific plastic granulates/films, as shown in Table 3. These differences lead to reported GWPs ranging from hardly to interpret, -1.2 to 5.34 kg CO_2-eq. To address the difficulty of comparing recycling processes that yield different products, the authors propose using the functional unit "TOP", which allows for an independent assessment of emission gaps relative to the average fossil-based pathway in future LCAs.

Table 3. Comparison of Global Warming Potential (GWP) results from this study with literature data. Abbreviations: TOP = Total Obtained Products; MPW = Municipal Plastic Waste; PE = Polyethylene; PP = Polypropylene; PS = Polystyrene; PA = Polyamide, PO = Polyolefin

Publication	**Feedstock**	**Data origin**	***Functional unit***		***GWP [kg CO_2-eq/functional unit]***	
			Waste perspective	**Product perspective**	**Waste perspective**	**Product perspective**
This study	Post-industrial PE/PA waste	TRL 8 process + simulation	1 kg post-industrial waste	1 kg TOP	-0.2 to -0.9	1.6 to 1.98
(Garcia-Garcia et al., 2024)	Contaminated MWP	Laboratory experiments	1 kg contaminated MPW	-	2,24	-

Table 3 (Cont.). Comparison of Global Warming Potential (GWP) results from this study with literature data. Abbreviations: TOP = Total Obtained Products; MPW = Municipal Plastic Waste; PE = Polyethylene; PP = Polypropylene; PS = Polystyrene; PA = Polyamide, PO = Polyolefin

Publication	Feedstock	Data origin	*Functional unit*		*GWP [kg CO_2-eq/functional unit]*	
			Waste perspective	**Product perspective**	**Waste perspective**	**Product perspective**
(Civancik-Uslu et al., 2021)	PP rigid/PS rigid /PO rigid/PE film	Simulation	1 ton of respective household waste	-	100/-1580/539/101	-
(Jeswani et al., 2021)	Light packaging waste	5 kta process plant	1 ton of light packaging waste	1t PE granulate	739	-447
(Chari et al., 2025)	MPW	Simulation + Literature	1 ton of MPW	1 t of PE/ PP film	778	-1379
(Yadav et al., 2023)	MPW	Simulation	-	1kg naphtha/mixed product/aromatic-rich product/olefin-rich product	-	2.74/4,64/2.2/5.34
(Hermanns et al., 2023)	sorted MPW	Literature	1 kg sorted MPW	-	0.41 to 0.46	-
(Klotz et al., 2024)	Plastic waste	Literature	1 kg plastic	-	-0.8 to 1.5	-

3.3. Sensitivity analysis to data uncertainties and methodological assumptions

The LCA results are calculated based on two experimental trials of the TRL 8 liquefaction plant. To assess the impact of deviations in the yield of the chemical recycling process on the GWP, two scenarios are modelled: an increase and a decrease of 5 % yield. The provision of process energy modelled with the German electricity grid, with a share of 51 % renewables, has the highest impact on the GWP, with up to 92 % of the total emissions. Since the composition of the electricity mix not only varies from region to region but also changes over time, three additional scenarios were investigated. In that the future development from 80 % renewables towards a fully renewable energy grid is considered, as well as a fully fossil grid. Furthermore, the GWP of the hydrogenation was calculated using hydrogen with average global emissions data from ecoinvent with 2.7 kg CO2-eq per kg H2. However, the emissions related to hydrogen production vary from 11.5 kg CO2 (grey hydrogen) (Kleijne et al., 2024) to 2.0 kg CO2 (green hydrogen) (Maniscalco et al., 2024). Two scenarios, assessing the influence of grey and green hydrogen supply on the GWP, are assessed. The GWP deviation of the scenarios compared to the assessed GWP is shown in Figure 5.

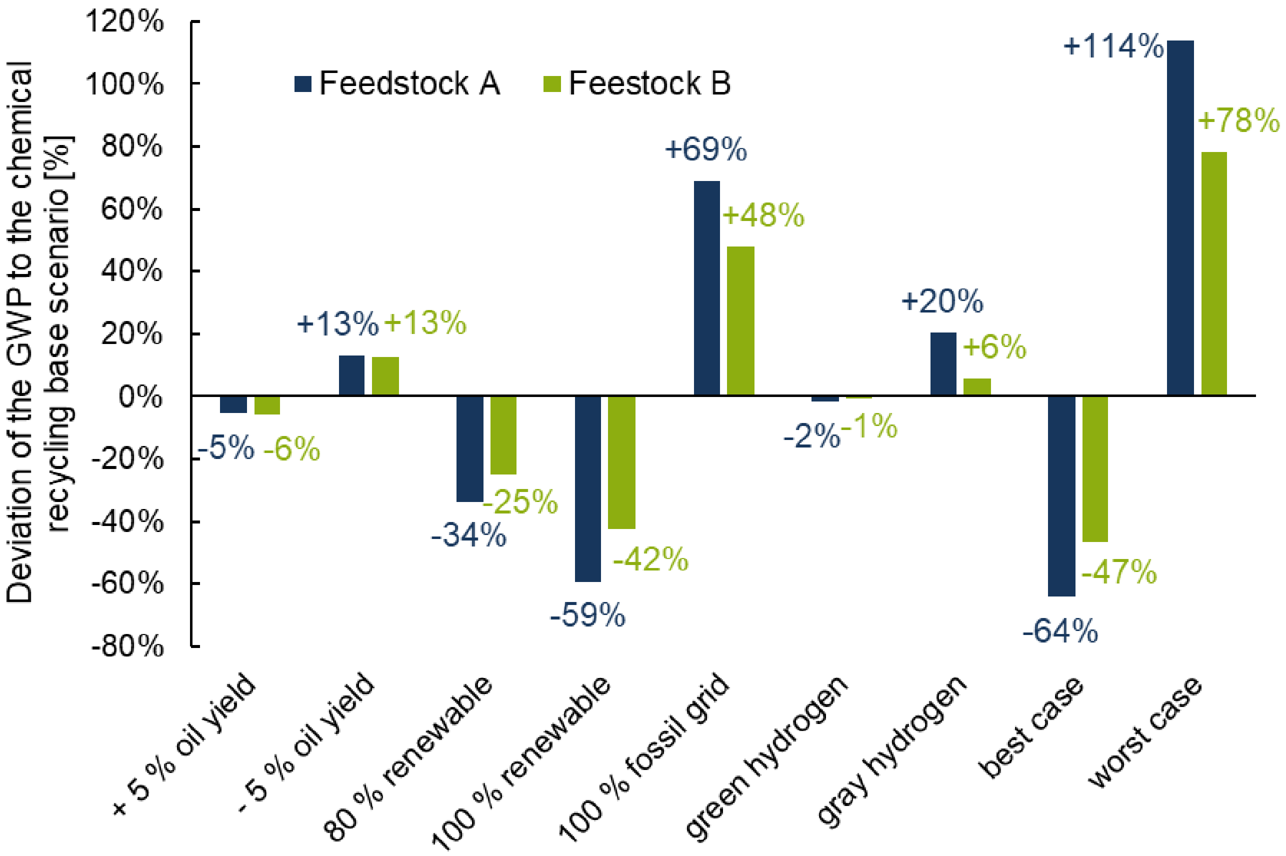

Figure 5. Changes to the GWP of the chemical recycling process for +/- 5 % product yield, chosen electricity grid power supply, the source of hydrogen supply, best case (+ 5 % yield, fully renewable, green hydrogen), and worst case (-5 % yield, fossil grid, grey hydrogen).

Increasing the oil yield can reduce the GWP of the recycling process by up to 6 %, while a reduction of the oil yield increases the GWP by up to 13 % for both feedstocks. Main contributors are the increased utility per kg total product, as well as the higher share of residue and gaseous side streams that need to be treated. The greenhouse gas emissions for the electricity supply reduce from 445 g CO_2 equivalent per kWh to 236 g CO_2-eq. for 80 % renewables 2030 and 75 g CO_2 with a fully renewable grid. For a fully fossil grid, it increases to 877 g CO_2-eq. The emissions from the recycling process to produce 1 kg of products can be reduced to 0.65 kg CO_2-eq with a fully renewable energy grid. This equals a 59 % emission reduction of the recycling process when achieving a fully renewable power grid. However, powering the recycling process with a fossil power grid, the emissions for the recycling process would drastically increase by up to 69 % to 2.92 kg CO_2-eq. While the use of green hydrogen only reduces the total emissions by 2 %, the use of grey hydrogen significantly increases the total process emissions by up to 20 %.

A best-case scenario for increased yield, fully renewable power supply and use of green hydrogen shows that the GWP can be reduced by up to 64 % but is still associated with unavoidable emissions. The unavoidable emissions divide in the waste treatment of the side streams with a share of 63 %, heat and hydrogen supply for product upgrading with 13 %, catalyst with 12 % and the electricity production with renewables with 12 %. To further reduce the environmental impact of the recycling process, the solid and gaseous waste streams generated need to be utilised. Also, for product upgrading with distillation and hydrogenation, the chemical industry needs to be decarbonised. Examples for such improvements are recent research in the utilisation of solid residue to improve bitumen characteristics in road construction (Gargiulo et al., 2023) or the electrification of the chemical industry (Gabrielli et al., 2023; Mallapragada et al., 2023). However, a worst-case scenario with a reduction of yield, fossil power grid and grey hydrogen supply increases the emissions by up to 114 %, leading to GWPs of up to 3.52 kg CO_2 for 1 kg TOP. In this case, the chemical recycling process would lead to a higher GWP than virgin production and therefore would not be the ecologically preferred production route.

4. Conclusion

In this study, the environmental impact of a TRL 8 pre-industrial chemical recycling plant located in Germany is determined and compared to the state-of-the-art incineration and the conventional, fossil-based production route of the recycled products for two different feedstocks. The assessed impact categories are GWP, FFDP, FEP, MEP and TAP. Exemplified for the GWP, the emissions caused by the chemical recycling process calculates to 1.4 $kg_{CO2\text{-}eq}$ for both feedstocks. Major parts of the emissions are attributable to the provision of energy for the process with at least 53 %, up to 36 % from waste treatment, and up to 19 % from product upgrading. This shows that the treatment of generated waste streams and the product upgrading have a significant influence on the environmental impact and should therefore not be neglected in LCA studies. Considering product substitution, the emissions of the recycling process can be calculated to net negative -0.2 kg_{CO2} per $kg_{Feedstock}$ for feedstock A and -0.9 kg_{CO2} for feedstock B. The chemical recycling process showed 50 % lower GWP compared to incineration. It also shows lower environmental impacts in the FFDP and TAP categories, whereas, due to the high electricity demand, the FEP and MEP impact categories are higher than those of the incineration process. Thus, implementing chemical recycling technologies involves a trade-off: reduced fossil dependency and lower climate impact, but more serious potential damage to aquatic ecosystems.

Compared to the fossil production of the respective recycled products, the chemical recycling process shows 12 - 40 % lower GWP, which is mainly due to the substitution of caprolactam, which accounts for up to 82 % of the fossil production emissions. As caprolactam is derived from PA in the waste feedstock, the results show that the GWP reduction is highly dependent on the polymer type in the feedstock. Also, powering the chemical recycling process with renewable energy could reduce the GWP by up to 59 %. Still, even with fully renewable electricity and a green hydrogen supply, emissions cannot be reduced to zero. Unavoidable emissions from product upgrading and, particularly, the waste treatment of the side streams remain. In contrast, a sensitivity analysis showed that the ecological benefit of the recycling process over virgin production is highly dependent on the data and assumptions. Slightly lowering the product yield, powering the process with fossil-based electricity and supplying grey hydrogen, the GWP can increase by 114 %. This would exceed the comparable virgin production emissions by a factor of nearly two, being twice as harmful to the environment as virgin production.

When integrating chemical recycling into circular economy strategies, the focus should be on processes involving monomer-yielding feedstocks, such as PA, to maximise substitution credits. Regional factors such as power supply and hydrogen availability need to be considered when conducting chemical recycling LCAs, as they can decide if the recycling process is ecologically preferable compared to virgin production. In addition to economic feasibility, further research should therefore examine how chemical recycling processes can be implemented in existing industrial infrastructure at a global and regional level.

Acknowledgements The authors want to thank CARBOLIQ GmbH for the provision of the process data and the support for planning and executing the data generation trials. Also, the authors want to thank Carbon Minds for the provision of their database cm.chemicals for the life cycle inventory calculations.

Author Contributions CRediT authorship contribution statement: Darius Steegborn: Data curation, Investigation, Methodology, Writing – original draft, Visualisation. Manfred Renner: Writing – review and editing. Philip Biessey: Conceptualisation, Project administration, Supervision, Validation, Writing – review and editing.

Declarations

Competing Interests The authors declare no competing interests.

References

Almohamadi, H., Alamoudi, M., Ahmed, U., Shamsuddin, R., & Smith, K. (2021). Producing hydrocarbon fuel from the plastic waste: Techno-economic analysis. *Korean Journal of Chemical Engineering*, *38*(11), 2208–2216. https://doi.org/10.1007/s11814-021-0876-3

Antelava, A., Damilos, S., Hafeez, S., Manos, G., Al-Salem, S. M., Sharma, B. K., Kohli, K., & Constantinou, A. (2019). Plastic Solid Waste (PSW) in the Context of Life Cycle Assessment (LCA) and Sustainable Management. *Environmental Management*, *64*(2), 230–244. https://doi.org/10.1007/s00267-019-01178-3

Ardolino, F., Palladini, A., & Arena, U. (2023). Social life cycle assessment of innovative management schemes for challenging plastics waste. *Sustainable Production and Consumption*, *37*, 344–355. https://doi.org/10.1016/j.spc.2023.03.011

Awino, F. B., & Apitz, S. E. (2024). Solid waste management in the context of the waste hierarchy and circular economy frameworks: An international critical review. *Integrated Environmental Assessment and Management*, *20*(1), 9–35. https://doi.org/10.1002/ieam.4774

Biessey, P., Vogel, J., Seitz, M., & Quicker, P. (2023). Plastic Waste Utilization via Chemical Recycling: Approaches, Limitations, and the Challenges Ahead. *Chemie Ingenieur Technik*, *95*(8), 1199–1214. https://doi.org/10.1002/cite.202300042

Boie, W. (1957). *Vom Brennstoff zum Rauchgas: Feuerungstechnisches Rechnen mit Brennstoffkenngrössen und seine Vereinfachung mit Mitteln der Statistik*. Teubner.

BusinessAnalytiq. (2025a). *Caprolactam price index*. https://businessanalytiq.com/procurementanalytics/index/caprolactam-price-index/ (accessed 03.11.2025)

BusinessAnalytiq. (2025b). *Naphtha price index*. https://businessanalytiq.com/procurementanalytics/index/naphtha-price-index/ (accessed 03.11.2025)

Carbon Minds. (2024). *https://www.carbon-minds.com/products/data/*.

Chari, S., Paulillo, A., & Materazzi, M. (2025). Exploring the potential of chemical recycling using a distributed model in the UK - A life cycle assessment perspective. *Waste Management (New York, N.Y.)*, *199*, 13–24. https://doi.org/10.1016/j.wasman.2025.02.052

Chaudhari, U. S., Johnson, A. T., Reck, B. K., Handler, R. M., Thompson, V. S., Hartley, D. S., Young, W., Watkins, D., & Shonnard, D. (2022). Material Flow Analysis and Life Cycle Assessment of Polyethylene Terephthalate and Polyolefin Plastics Supply Chains in the United States. *ACS Sustainable Chemistry & Engineering*, *10*(39), 13145–13155. https://doi.org/10.1021/acssuschemeng.2c04004

Civancik-Uslu, D., Nhu, T. T., van Gorp, B., Kresovic, U., Larrain, M., Billen, P., Ragaert, K., Meester, S. de, Dewulf, J., & Huysveld, S. (2021). Moving from linear to circular household plastic packaging in Belgium: Prospective life cycle assessment of mechanical and thermochemical recycling. *Resources, Conservation and Recycling*, *171*, 105633. https://doi.org/10.1016/j.resconrec.2021.105633

Da Monteiro, A. R. D., Miranda, D. M. V. de, Da Silva Pinto, J. C. C., & Soto, J. J. (2022). Life Cycle Assessment of the Catalytic Pyrolysis of High-Density Polyethylene (HDPE) and High-Impact Polystyrene (HIPS). *Macromolecular Reaction Engineering*, *16*(6), Article 2200037. https://doi.org/10.1002/mren.202200037

Dai, L., Zhou, N., Lv, Y., Cheng, Y., Wang, Y., Liu, Y., Cobb, K., Chen, P., Lei, H., & Ruan, R. (2022). Pyrolysis technology for plastic waste recycling: A state-of-the-art review. *Progress in Energy and Combustion Science*, *93*, 101021. https://doi.org/10.1016/j.pecs.2022.101021

Das, S., Liang, C., & Dunn, J. B. (2022). Plastics to fuel or plastics: Life cycle assessment-based evaluation of different options for pyrolysis at end-of-life. *Waste Management (Oxford, United Kingdom)*, *153*, 81–88. https://doi.org/10.1016/j.wasman.2022.08.015

Davidson, M. G., Furlong, R. A., & McManus, M. C. (2021). Developments in the life cycle assessment of chemical recycling of plastic waste – A review. *Journal of Cleaner Production*, *293*, 126163. https://doi.org/10.1016/j.jclepro.2021.126163

Dębek, C., & Walendziewski, J. (2015). Hydrorefining of oil from pyrolysis of whole tyres for passenger cars and vans. *Fuel*, *159*, 659–665. https://doi.org/10.1016/j.fuel.2015.07.024

Djandja, O. S., Wang, Z., Duan, P., Wang, F., & Xu, Y. (2021). Hydrotreatment of pyrolysis oil from waste tire in tetralin for production of high-quality hydrocarbon rich fuel. *Fuel*, *285*, 119185. https://doi.org/10.1016/j.fuel.2020.119185

Doka, G. (2013). *Updates to Life Cycle Inventories of Waste Treatment Services: Part II "Waste incineration"*.

Domo Chemicals GmbH. (2022). *Sicherheitsdatenblatt Caprolactam: Inklusive Spezifikationen*.

ecoinvent. (2022). *ecoinvent database v3.9.1*.

Egli, S. (2005). *Anleitung mit Checkliste zur Energieoptimierung von Kehrichtverbrennungsanlagen: Schlussbericht*.

Ellen MacArthur Foundation. (2017). *The new plastics economy: catalysing action*.

Erakca, M., Baumann, M., Helbig, C., & Weil, M. (2024). Systematic review of scale-up methods for prospective life cycle assessment of emerging technologies. *Journal of Cleaner Production*, *451*, 142161. https://doi.org/10.1016/j.jclepro.2024.142161

European Commission. (2019). *The European Green Deal: communication from the commission to the european parliament, the european council, the council, the european economic and social committee and the committee of the regions*.

Faisal, F., Rasul, M. G., & Jahirul, M. I. (2023). Upgrading MSW pyrolysis oil through distillation and hydrotreatment for automobile engine applications: A short review. *Energy Reports*, *9*, 555–560. https://doi.org/10.1016/j.egyr.2023.12.008

Fibrant. (2024). *Product Data Sheet - EcoLactam® AP Quality liquid*.

Gabrielli, P., Rosa, L., Gazzani, M., Meys, R., Bardow, A., Mazzotti, M., & Sansavini, G. (2023). Net-zero emissions chemical industry in a world of limited resources. *One Earth*, *6*(6), 682–704. https://doi.org/10.1016/j.oneear.2023.05.006

Garcia-Garcia, G., Martín-Lara, M. Á., Calero, M., & Blázquez, G. (2024). Environmental impact of different scenarios for the pyrolysis of contaminated mixed plastic waste. *Green Chemistry*, *26*(7), 3853–3862. https://doi.org/10.1039/D3GC04396G

Gargiulo, V., Alfe, M., Ruoppolo, G., Cammarota, F., Rossi, C. O., Loise, V., Porto, M., Calandra, P., Pochylski, M., Gapinski, J., & Caputo, P. (2023). How char from waste pyrolysis can improve bitumen characteristics and induce anti-aging effects. *Colloids and Surfaces a: Physicochemical and Engineering Aspects*, *676*, 132199. https://doi.org/10.1016/j.colsurfa.2023.132199

Genuino, H. C., Pilar Ruiz, M., Heeres, H. J., & Kersten, S. R. A. (2023). Pyrolysis of mixed plastic waste: Predicting the product yields. *Waste Management (Oxford, United Kingdom)*, *156*, 208–215. https://doi.org/10.1016/j.wasman.2022.11.040

GESTIS-Stoffdatenbank. (2024a). *Characterisation of diesel fuel*. https://gestis.dguv.de/data?name=536303&lang=en

GESTIS-Stoffdatenbank. (2024b). *Characterisation of naphtha; Low boiling point naphtha*. https://gestis.dguv.de/data?name=035130&lang=en

Geyer, R. (2020). Production, use, and fate of synthetic polymers. In *Plastic Waste and Recycling* (pp. 13–32). https://doi.org/10.1016/B978-0-12-817880-5.00002-5

Gutierrez-Lopez, J., McGarvey, R. G., Noble, J. S., Hall, D. M., & Costello, C. (2024). Quantification of social metrics for use in optimization: An application to solid waste management. *Journal of Cleaner Production*, *480*, 144111. https://doi.org/10.1016/j.jclepro.2024.144111

Hermanns, R., Kraft, A., Hartmann, P., & Meys, R. (2023). Comparative Life Cycle Assessment of Pyrolysis – Recycling Germany's Sorted Mixed Plastic Waste. *Chemie Ingenieur Technik*, *95*(8), 1259–1267. https://doi.org/10.1002/cite.202300041

Hita, I., Gutiérrez, A., Olazar, M., Bilbao, J., Arandes, J. M., & Castaño, P. (2015). Upgrading model compounds and Scrap Tires Pyrolysis Oil (STPO) on hydrotreating NiMo catalysts with tailored supports. *Fuel*, *145*, 158–169. https://doi.org/10.1016/j.fuel.2014.12.055

Holtkamp, M., Renner, M., Matthiesen, K., Wald, M., Luinstra, G. A., & Biessey, P. (2024). Robust downstream technologies in polystyrene waste pyrolysis: Design and prospective life-cycle assessment of pyrolysis oil reintegration pathways. *Resources, Conservation and Recycling*, *205*, 107558. https://doi.org/10.1016/j.resconrec.2024.107558

Hong, J., & Xu, X. (2012). Environmental impact assessment of caprolactam production – a case study in China. *Journal of Cleaner Production*, *27*, 103–108. https://doi.org/10.1016/j.jclepro.2011.12.037

Hugo Fuchs, L. (1994). PURIFICATION OF CAPROLACTAM(5,350,847).

Icha, P., & Lauf, T. (2024). *Entwicklung der spezifischen Treibhausgas-Emissionen des deutschen Strommix in den Jahren 1990 - 2023*.

Iribarren, D., Dufour, J., & Serrano, D. P. (2012). Preliminary assessment of plastic waste valorization via sequential pyrolysis and catalytic reforming. *Journal of Material Cycles and Waste Management*, *14*(4), 301–307. https://doi.org/10.1007/s10163-012-0069-6

ISO. (2006a). *DIN ISO EN 14040:2006 Environmental management – Life cycle assessment – Principles and framework: German version*.

ISO. (2006b). *DIN ISO EN 14044:2006 Environmental management – Life cycle assessment – Requirements and guidelines: German version*.

Jeswani, H., Krüger, C., Russ, M., Horlacher, M., Antony, F., Hann, S., & Azapagic, A. (2021). Life cycle environmental impacts of chemical recycling via pyrolysis of mixed plastic waste in comparison with mechanical recycling and energy recovery. *Science of the Total Environment*, *769*, 144483. https://doi.org/10.1016/j.scitotenv.2020.144483

Kleijne, K. de, Huijbregts, M. A. J., Knobloch, F., van Zelm, R., Hilbers, J. P., Coninck, H. de, & Hanssen, S. V. (2024). Worldwide greenhouse gas emissions of green hydrogen production and transport. *Nature Energy*. Advance online publication. https://doi.org/10.1038/s41560-024-01563-1

Klotz, M., Oberschelp, C., Salah, C., Subal, L., & Hellweg, S. (2024). The role of chemical and solvent-based recycling within a sustainable circular economy for plastics. *The Science of the Total Environment*, *906*, 167586. https://doi.org/10.1016/j.scitotenv.2023.167586

Kulas, D. G., Zolghadr, A., Chaudhari, U. S., & Shonnard, D. R. (2023). Economic and environmental analysis of plastics pyrolysis after secondary sortation of mixed plastic waste. *Journal of Cleaner Production*, *384*, 135542. https://doi.org/10.1016/j.jclepro.2022.135542

Kusenberg, M., Langhe, S. de, Parvizi, B., Abdulrahman, A. J., Varghese, R. J., Aravindakshan, S. U., Kurkijärvi, A., Gandarillas, A. M., Jamieson, J., Meester, S. de, & van Geem, K. M. (2024). Characterization and impact of oxygenates in post-consumer plastic waste-derived pyrolysis oils on steam cracking process efficiency. *Journal of Analytical and Applied Pyrolysis*, *181*, 106571. https://doi.org/10.1016/j.jaap.2024.106571

Kusenberg, M., Zayoud, A., Roosen, M., Thi, H. D., Abbas-Abadi, M. S., Eschenbacher, A., Kresovic, U., Meester, S. de, & van Geem, K. M. (2022). A comprehensive experimental investigation of plastic waste pyrolysis oil quality and its dependence on the plastic waste composition. *Fuel Processing Technology*, *227*, 107090. https://doi.org/10.1016/j.fuproc.2021.107090

Lase, I. S., Tonini, D., Caro, D., Albizzati, P. F., Cristóbal, J., Roosen, M., Kusenberg, M., Ragaert, K., van Geem, K. M., Dewulf, J., & Meester, S. de (2023). How much can chemical recycling contribute to plastic waste recycling in Europe? An assessment using material flow analysis modeling. *Resources, Conservation and Recycling*, *192*, 106916. https://doi.org/10.1016/j.resconrec.2023.106916

Lau, W. W. Y., Shiran, Y., Bailey, R. M., Cook, E., Stuchtey, M. R., Koskella, J., Velis, C. A., Godfrey, L., Boucher, J., Murphy, M. B., Thompson, R. C., Jankowska, E., Castillo Castillo, A., Pilditch, T. D., Dixon, B., Koerselman, L., Kosior, E., Favoino, E., Gutberlet, J., . . . Palardy, J. E. (2020). Evaluating scenarios toward zero plastic pollution. *Science (Washington, DC, United States)*, *369*(6510), 1455–1461. https://doi.org/10.1126/science.aba9475

Lechleitner, A., Schwabl, D., Schubert, T., Bauer, M., & Lehner, M. (2020). Chemisches Recycling von gemischten Kunststoffabfällen als ergänzender Recyclingpfad zur Erhöhung der Recyclingquote. *Oesterreichische Wasser- Und Abfallwirtschaft*, *72*(1-2), 47–60. https://doi.org/10.1007/s00506-019-00628-w

Losier, T. P. (1995). Process for the purification of crude caprolactam(0 670 307 A2).

Mallapragada, D. S., Dvorkin, Y., Modestino, M. A., Esposito, D. V., Smith, W. A., Hodge, B.-M., Harold, M. P., Donnelly, V. M., Nuz, A., Bloomquist, C., Baker, K., Grabow, L. C., Yan, Y., Rajput, N. N., Hartman, R. L., Biddinger, E. J., Aydil, E. S., & Taylor, A. D. (2023). Decarbonization of the chemical industry through electrification: Barriers and opportunities. *Joule*, *7*(1), 23–41. https://doi.org/10.1016/j.joule.2022.12.008

Maniscalco, M. P., Longo, S., Cellura, M., Miccichè, G., & Ferraro, M. (2024). Critical Review of Life Cycle Assessment of Hydrogen Production Pathways. *Environments*, *11*(6), 108. https://doi.org/10.3390/environments11060108

Merkel, A., Plessing, L., Luinstra, G. A., Grünewald, M., & Biessey, P. (2023). Development of a separation sequence for the integration of a styrene recyclate into polystyrene production. *Chemie Ingenieur Technik, 95*(8), 1323–1331. https://doi.org/10.1002/cite.202100210

Meys, R., Frick, F., Westhues, S., Sternberg, A., Klankermayer, J., & Bardow, A. (2020). Towards a circular economy for plastic packaging wastes – the environmental potential of chemical recycling. *Resources, Conservation and Recycling*, *162*, 105010. https://doi.org/10.1016/j.resconrec.2020.105010

Meys, R., Kätelhön, A., Bachmann, M., Winter, B., Zibunas, C., Suh, S., & Bardow, A. (2021). Achieving net-zero greenhouse gas emission plastics by a circular carbon economy. *Science (Washington, DC, United States)*, *374*(6563), 71–76. https://doi.org/10.1126/science.abg9853

Moni, S. M., Mahmud, R., High, K., & Carbajales-Dale, M. (2020). Life cycle assessment of emerging technologies: A review. *Journal of Industrial Ecology*, *24*(1), 52–63. https://doi.org/10.1111/jiec.12965

OECD. (2022). *Global Plastics Outlook: Economic Drivers, Environmental Impacts and Policy Options*. OECD Publishing. https://doi.org/10.1787/de747aef-en

Pires Costa, L., Vaz de Miranda, D. M., & Pinto, J. C. (2022). Critical Evaluation of Life Cycle Assessment Analyses of Plastic Waste Pyrolysis. *ACS Sustainable Chemistry & Engineering*, *10*(12), 3799–3807. https://doi.org/10.1021/acssuschemeng.2c00265

Plastics Europe. (2022a). *"Plastics - The Facts 2022"*.

Plastics Europe. (2022b). *Eco-profiles program and methodology: Version 3.1*. https://plasticseurope.org/wp-content/uploads/2024/03/PlasticsEurope-Ecoprofiles-program-and-methodology_V3.1.pdf?utm_source=chatgpt.com

Ragaert, K., Delva, L., & van Geem, K. (2017). Mechanical and chemical recycling of solid plastic waste. *Waste Management (Oxford, United Kingdom)*, *69*, 24–58. https://doi.org/10.1016/j.wasman.2017.07.044

Santos, A., Barbosa-Póvoa, A., & Carvalho, A. (2019). Life cycle assessment in chemical industry – a review. *Current Opinion in Chemical Engineering*, *26*, 139–147. https://doi.org/10.1016/j.coche.2019.09.009

Schwartz, E., He, H., Ogunseitan, O. A., & Schoenung, J. M. (2025). Phenol production from pyrolysis of waste printed circuit boards: life cycle and techno-economic assessment. *Resources, Conservation and Recycling*, *215*, 108019. https://doi.org/10.1016/j.resconrec.2024.108019

Snowden-Swan, L. J., Spies, K. A., Lee, G. J., & Zhu, Y. (2016). Life cycle greenhouse gas emissions analysis of catalysts for hydrotreating of fast pyrolysis bio-oil. *Biomass and Bioenergy, 86*, 136–145. https://doi.org/10.1016/j.biombioe.2016.01.019

Sphera. (2025). *Guidelines for LCA in Chemical Recycling: v1.0*. Prepared for Chemical Recycling Europe. Available at: https://chemicalrecyclingeurope.eu/wp-content/uploads/2025/05/CRE-LCA-Guideline-Final-Clean-signed-1.pdf

Tinge, J., Groothaert, M., op het Veld, H., Ritz, J., Fuchs, H., Kieczka, H., & Moran, W. C. (2000). Caprolactam. In *Ullmann's Encyclopedia of Industrial Chemistry* (Vol. 30, pp. 1–31). Wiley. https://doi.org/10.1002/14356007.a05_031.pub3

Tsangas, M., Papamichael, I., Loizia, P., Voukkali, I., Salman Raza, N., Vincenzo, N., & Zorpas, A. A. (2024). Life cycle assessment of electricity generation by tire pyrolysis oil. *Process Safety and Environmental Protection*, *186*, 376–387. https://doi.org/10.1016/j.psep.2024.04.038

Wang, H., Wang, L., & Shahbazi, A. (2015). Life cycle assessment of fast pyrolysis of municipal solid waste in North Carolina of USA. *Journal of Cleaner Production*, *87*, 511–519. https://doi.org/10.1016/j.jclepro.2014.09.011

Yadav, G., Singh, A., Dutta, A., Uekert, T., DesVeaux, J. S., Nicholson, S. R., Tan, E. C., Mukarakate, C., Schaidle, J. A., Wrasman, C. J., Carpenter, A. C., Baldwin, R. M., Román-Leshkov, Y., & Beckham, G. T. (2023). Techno-economic analysis and life cycle assessment for catalytic fast pyrolysis of mixed plastic waste. *Energy & Environmental Science*, *16*(9), 3638–3653. https://doi.org/10.1039/D3EE00749A

Zeb, W., Roosen, M., Knockaert, P., Janssens, S., Withoeck, D., Kusenberg, M., Hogie, J., Billen, P., Tavernier, S., van Geem, K. M., & Meester, S. de (2023). Purification and characterisation of post-consumer plastic pyrolysis oil fractionated by vacuum distillation. *Journal of Cleaner Production*, *416*, 137881. https://doi.org/10.1016/j.jclepro.2023.137881

Zeller, M., Netsch, N., Richter, F., Leibold, H., & Stapf, D. (2021). Chemical Recycling of Mixed Plastic Wastes by Pyrolysis – Pilot Scale Investigations. *Chemie Ingenieur Technik*, *93*(11), 1763–1770. https://doi.org/10.1002/cite.202100102