

# Economic and environmental assessment of automotive plastic waste end-of-life options

## Energy recovery versus chemical recycling

Christoph Stallkamp<sup>1</sup>  | Malte Hennig<sup>2</sup>  | Rebekka Volk<sup>1</sup>  | Frank Richter<sup>2</sup> |  
 Britta Bergfeldt<sup>2</sup> | Salar Tavakkol<sup>2</sup> | Frank Schultmann<sup>1</sup> | Dieter Stapf<sup>2</sup>

<sup>1</sup>Karlsruhe Institute of Technology (KIT), Institute for Industrial Production (IIP), Karlsruhe, Germany

<sup>2</sup>Karlsruhe Institute of Technology (KIT), Institute for Technical Chemistry (ITC), Karlsruhe, Germany

### Correspondence

Christoph Stallkamp, Karlsruhe Institute of Technology (KIT), Institute for Industrial Production (IIP), Karlsruhe, Germany.  
 Email: [christoph.stallkamp@kit.edu](mailto:christoph.stallkamp@kit.edu)

Malte Hennig, Karlsruhe Institute of Technology (KIT), Institute for Technical Chemistry (ITC), Karlsruhe, Germany.  
 Email: [malte.hennig@kit.edu](mailto:malte.hennig@kit.edu)

Editor Managing Review: Lynette Cheah

### Funding information

Ministry of the Environment, Climate Protection, and the Energy Sector of the state of Baden-Württemberg in Germany and industry partners, "THINKTANK Industrial Resource Strategies"; AUDI AG

### Abstract

Most automotive plastic waste (APW) is landfilled or used in energy recovery as it is unsuitable for high-quality product mechanical recycling. Chemical recycling via pyrolysis offers a pathway toward closing the material loop by handling this heterogeneous waste and providing feedstock for producing virgin plastics. This study compares chemical recycling and energy recovery scenarios for APW regarding climate change impact and cumulative energy demand (CED), assessing potential environmental advantages. In addition, an economic assessment is conducted. In contrast to other studies, the assessments are based on pyrolysis experiments conducted with an actual waste fraction. Mass balances and product composition are reported. The experimental data is combined with literature data for up- and downstream processes for the assessment. Chemical recycling shows a lower net climate change impact (0.57 to 0.64 kg CO<sub>2</sub>e/kg waste input) and CED (3.38 to 4.41 MJ/kg waste input) than energy recovery (climate change impact: 1.17 to 1.25 kg CO<sub>2</sub>e/kg waste input; CED: 6.94 to 7.97 MJ/kg waste input), while energy recovery performs better economically (net processing cost of −0.05 to −0.02€/kg waste input) compared to chemical recycling (0.05 to 0.08€/kg waste input). However, chemical recycling keeps carbon in the material cycle contributing to a circular economy and reducing the dependence on fossil feedstocks. Therefore, an increasing circularity of APW through chemical recycling shows a conflict between economic and environmental objectives.

### KEYWORDS

automotive plastics recycling, carbon management, chemical recycling, circular economy, climate change impact, industrial ecology

This is an open access article under the terms of the [Creative Commons Attribution](https://creativecommons.org/licenses/by/4.0/) License, which permits use, distribution and reproduction in any medium, provided the original work is properly cited.

© 2023 The Authors. *Journal of Industrial Ecology* published by Wiley Periodicals LLC on behalf of International Society for Industrial Ecology.

## 1 | INTRODUCTION

The amount of plastic in automobiles increased considerably in the last decades to realize the advantages of plastic components, such as lower costs and weight reduction (Wilts et al., 2016). Therefore, more automotive plastic waste (APW) will have to be handled in the future. Defect plastic components are replaced during the use phase generating mixed APW. At the end-of-life (EoL), a highly heterogeneous automotive shredder residue (ASR) contains mixed plastics of different origins. Currently, landfilling and energy recovery are the dominating waste-handling options for APW and ASR in Europe (Cossu & Lai, 2015; Mehlhart et al., 2018), resulting in high environmental burdens and a loss of valuable resources likewise. Enhancing the circular economy with advanced recycling technologies is a strategy to keep carbon from waste available as feedstock for the chemical industry and reduce its dependency on fossil carbon feedstock (Meys et al., 2021).

Mechanical recyclers already face the challenge of complex waste mixtures and composite materials handling, for example, lightweight packaging waste. However, APW additionally contains high shares of non-standard functionalized engineering thermoplastics, thermosets, and elastomers resulting in polymer cross-contamination and non-polymer impurities within recovered waste fractions (Cossu & Lai, 2015; Pivnenko et al., 2015) that pose an additional challenge for mechanical recycling processes. Therefore, mechanical recycling of automotive plastics can alter the recyclates' mechanical properties (Bernasconi et al., 2007; Colucci et al., 2017; Pietrolungo et al., 2020; Yang et al., 2018). New waste-handling options are needed to reduce environmental impacts, decrease the dependence on fossil feedstocks by increasing circularity while at the same time being cost competitive in comparison to current waste-handling practices.

Chemical recycling processes can close the material and carbon loop by handling heterogeneous waste streams unsuitable for mechanical recycling (Dogu et al., 2021) and providing feedstock for virgin plastics production (Meys et al., 2020; Solis & Silveira, 2020). Pyrolysis as a chemical recycling option is currently assessed for multiple waste streams (Davidson et al., 2021; Kusenberget al., 2022a; Volk et al., 2021; Zeller et al., 2021), with promising results for waste streams primarily consisting of polyolefins.

The literature on pyrolysis can be divided into technology research and technology assessments. Technology-focused research investigates the pyrolysis process in detail regarding feedstock material, reactor type, product composition, and process parameters. This includes studies on the pyrolysis of ASR (Cossu et al., 2014; Galvagno et al., 2001; Notarnicola et al., 2017). However, no studies on the pyrolysis of APW are known. Studies of the second category focus on economic feasibility and environmental impact. Due to the lack of data from established processes or a low technology readiness level (TRL), assessments are made either based on thermodynamic considerations (Meys et al., 2020), experiments with different waste fractions (Cardamone et al., 2022), or laboratory-scale experiments of virgin instead of post-consumer material (Gracida-Alvarez et al., 2019a, 2019b). Jeswani et al. (2021) used experimental data from larger-scale experiments. However, the data is not publicly available due to confidentiality and cannot be validated. While these studies can provide valuable input regarding the potential of different chemical recycling processes, there is the potential for misleading results when experimental data from technology research-oriented studies not specifically designed for process assessment are used. This may be the case if data from a waste stream is used to approximate another waste fraction that behaves differently in pyrolysis.

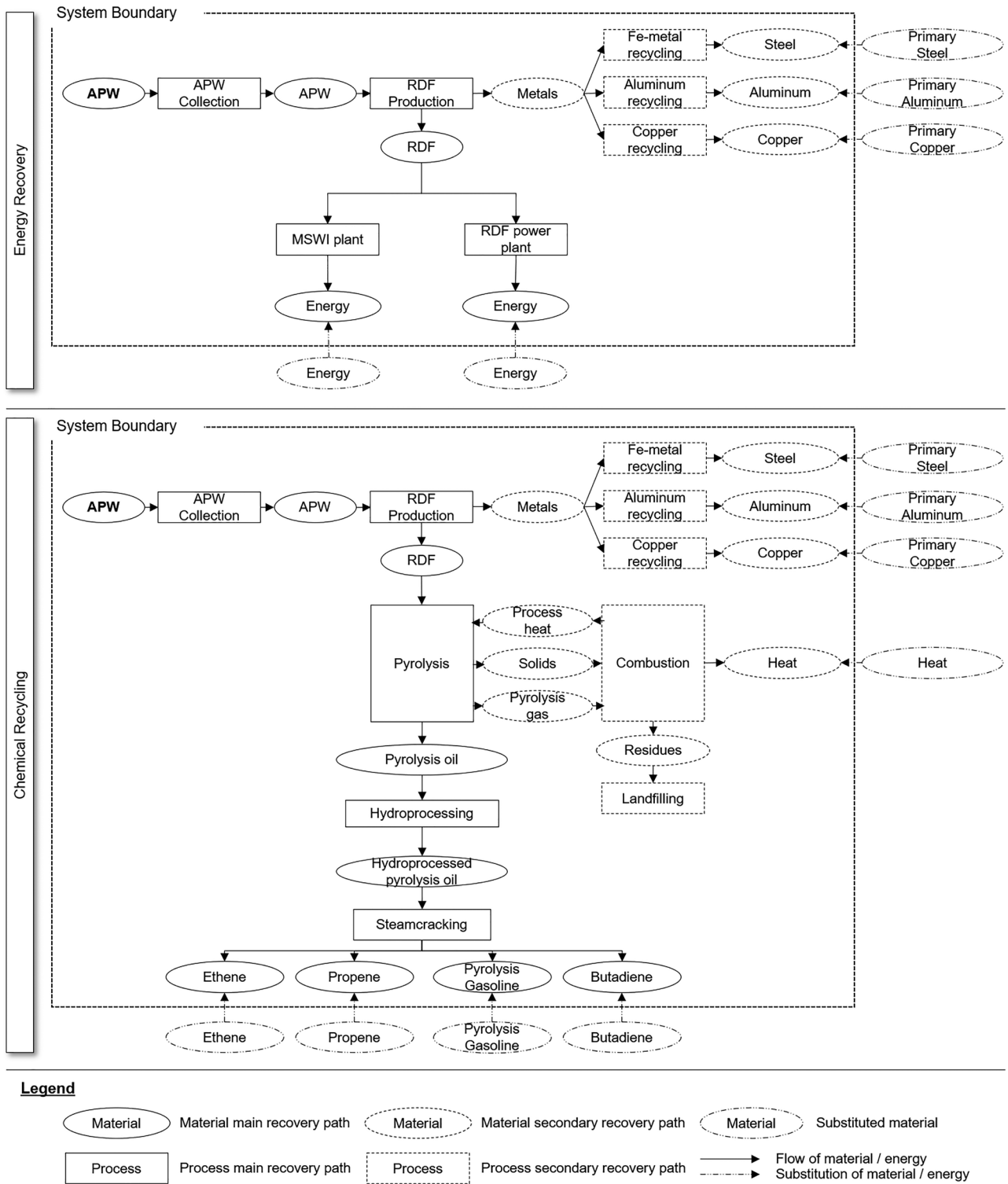
Therefore, this study follows an integrated approach by conducting pyrolysis experiments according to the subsequent process assessment's needs. A sample from an actual APW is taken, thoroughly characterized, and pyrolyzed in a pilot-scale continuous reactor with a throughput of approximately 1 kg/h. Mass balances and pyrolysis gas and oil product composition are reported for transparency. The chemical recycling process chain includes upstream mechanical pre-treatment of the APW, pyrolysis, and downstream upgrading of the liquid pyrolysis product to the specifications of steam cracker feedstock (naphtha substitute) followed by high-value chemical (HVC) production through steam cracking. Literature data for up- and downstream processes are combined with experimental data from the pyrolysis of APW to create a life cycle inventory (LCI) and enable life cycle assessment (LCA) before the commercialization of pyrolysis technology at a large scale. The economic indicator of process costs and the environmental indicators of climate change impact and cumulative energy demand (CED) are compared with the current waste-handling practice of energy recovery. Consequently, this study contributes to understanding different waste treatment options for automotive plastics and their contribution to a sustainable circular economy.

## 2 | METHODOLOGY

Besides the experimental analyses, this study combines material flow cost analysis (MFCA) with LCA methods (Rieckhof & Guenther, 2018). Due to missing data, for example, elemental flows, the LCA is streamlined by narrowing the considered environmental impacts to climate change impact and CED (Gradin & Björklund, 2021).

### 2.1 | Scope

The assessed chemical recycling path is derived from the current energy recovery route for mixed APW collected in automotive workshops. The system boundaries of both considered processes include metal recycling and energy recovery (Figure 1). Table S1-1 in supporting information (S1) summarizes them. The functional unit of the assessment is the treatment of 1 kg of APW collected from automotive workshops.



**FIGURE 1** System boundaries for the energy recovery and chemical recycling paths, including primary material and energy substitution.

### 2.1.1 | Energy recovery path

APW is sent to refuse-derived fuel (RDF) production. In RDF production, the APW is shredded, and steel, aluminum, and copper are separated and sent to respective recycling processes. The RDF is incinerated in municipal solid waste incineration (MSWI) and RDF power plants. Energy is recovered through heat and electricity based on average efficiencies for MSWI and RDF power plants.

**TABLE 1** Overview of scenarios

Scenario no.	Scenario description	Description
1.1	Energy recovery (baseline)	Incineration path of produced RDF: 30% MSWI plant, 70% RDF combustion plant
1.2	Energy recovery (optimized)	Incineration path of produced RDF: 100% RDF combustion plant
2.1	Chemical recycling (baseline)	Yield of pyrolysis products according to conducted experiments <sup>a</sup> : 50% pyrolysis oil, 20% pyrolysis gas, 28% pyrolysis residue, 2% aqueous condensate.
2.2	Chemical recycling (lower yield)	Yield of pyrolysis products adapted: 44% pyrolysis oil, 20% pyrolysis gas, 34% pyrolysis residue, 2% aqueous condensate. <sup>a</sup>

<sup>a</sup>Pyrolysis product distribution converted to a feedstock free of metals (experiments did not have metal-free feedstock).

## 2.1.2 | Chemical recycling (pyrolysis) path

Following RDF production, RDF is fed into an integrated pyrolysis unit producing pyrolysis oil as the main product and pyrolysis gas, an aqueous condensate, and a solid pyrolysis residue fraction as by-products. Gas and oil fractions mainly consist of hydrocarbons; the pyrolysis residue fraction consists of mineral residues and carbonaceous char. Based on their calculated net calorific values, it is assumed that high-temperature process heat for the pyrolysis can be provided by the incineration of pyrolysis by-products, that is, pyrolysis residues, pyrolysis gas, and aqueous condensate. The plant design assumes excess low-temperature heat can be sold to customers, such as a district heating network. Ashes are landfilled. Furthermore, the pyrolysis oil is assumed to be upgraded in a hydroprocessing step, replacing naphtha as a steam-cracking feedstock. By steam cracking, HVC ethene, propene, butadiene, and pyrolysis gasoline are produced, which are raw materials for synthesizing chemical products, including virgin plastics.

## 2.2 | Impact assessment

An economic and environmental assessment of both recovery paths (Figure 1) is performed where each process step is assessed individually, and all process steps along the recycling path are summed up. The APW is not associated with production or use phase burdens following the zero-burden approach (Nakatani, 2014). Products and marketable by-products are rewarded with the impact of primary material production following the avoided burden approach (Nakatani, 2014). Impacts are allocated based on mass. Intermediate products are associated with the burdens of the previous treatment steps.

The economic assessment includes fixed operational costs<sup>1</sup> based on the plant investments and variable utility operating costs. The base year of the economic assessment is 2021. Products and marketable by-products result in revenues that are realized. Costs of CO<sub>2</sub> emissions are internalized based on direct CO<sub>2</sub> emissions of each process step and CO<sub>2</sub> emission prices of the EU emission trading system. The avoided CO<sub>2</sub> emissions from the primary production of products and by-products are rewarded with a negative emission price.

The streamlined LCA assesses the climate change impact over 100 years as defined by the IPCC (2013). The CED, as defined by VDI (2012), is evaluated to assess the energy impacts of the recovery paths (Iacovidou et al., 2017).

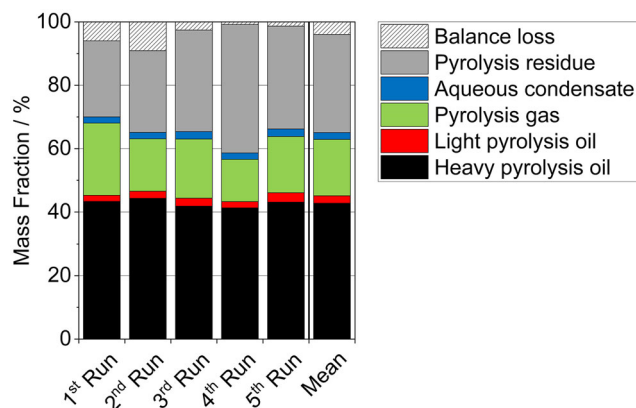
## 2.3 | Scenario definition

Scenarios analyze the influence of underlying data and assumptions on the assessment (Table 1). The energy recovery baseline scenario (scenario 1.1) is the incineration of RDF in MSWI plants (30%) and RDF power plants (70%). The incineration shares reflect the current practice in European countries (Jeswani et al., 2021; van Eygen et al., 2018). Scenario 1.2 assesses the RDF combustion in more efficient RDF power plants only.

The baseline scenario for chemical recycling (scenario 2.1) uses the conducted pyrolysis experiments' data (Section 3). Pyrolysis oil is used to produce HVC, while pyrolysis gas and solids are used for energy recovery. Scenario 2.2 considers a lower pyrolysis oil yield than scenario 2.1, assigning 10% of pyrolysis oil yield to the pyrolysis residue.

## 3 | EXPERIMENTAL ASSESSMENT OF PYROLYSIS

The pyrolysis technology assessment and LCI data for the pyrolysis are based on pilot-scale experiments using collected APW material. The material was thoroughly characterized. Mass balances and energy consumption for pyrolysis were determined from experiments, and pyrolysis oil product quality was assessed based on heteroatom content. The pyrolysis experiments included metals in the feedstock. Therefore, this section reports pyrolysis product distribution based on the full feedstock. For process assessment, the product distribution is converted to metal-free feedstock based on a metal content of 9% in the feedstock material as the metal content is recovered in RDF production in both paths. Details are provided in the Supporting Information (Section A3, Supporting Information S1).



**FIGURE 2** Mass distribution of pyrolysis products in repetitive experimental runs under equal conditions. The underlying data used to create this figure can be found in Supporting Information S2.

### 3.1 | Characterization of automotive plastic waste sample

The APW sample was obtained from waste collected at automotive workshops around Stuttgart, Germany. First, all parts were inventoried. Large metal parts, such as bumper cross beams, were excluded from the sample. Smaller metal parts remained in the sample and were not removed before the experiments. From the automobile manufacturer's database, sample composition was estimated to be polyolefins (57%), polycarbonates and blends thereof (19%), polyamides (5%), other polymers (12%), and other non-polymer materials (e.g., metals, minerals, biomass) (7%). However, manual sorting of a shredded sample showed a significant deviation of the metal content (3.5% ferrous and 5.5% non-ferrous metals), making deviations of other components' shares likely. For elemental balances of the pyrolysis experiments, the pre-sorted material's elemental composition and ash content were determined (Table S1-4, Supporting Information S1).

### 3.2 | Pyrolysis experimental set-up

The sample was shredded to a particle size of 10 mm. The experiments were conducted in an electrically heated continuous pilot-scale screw reactor with a throughput of plastic waste of approximately 1 kg/h and a solids residence time of 45 min at a temperature of 450°C. A moderator medium (sand) improves heat and mass transfer properties at a 1:4 mass ratio (feedstock:moderator). The reactor has already been described elsewhere (Tomasi Morgano et al., 2018; Zeller et al., 2021). The supporting information describes analytical techniques used for material and product characterization (Section A2, Supporting Information S1). Five repetitive experiments with 5 kg of waste material each determined the mean pyrolysis product distribution.

### 3.3 | Pyrolysis products

Approximately 45 wt% of the waste is converted to pyrolysis oil (cf. Figure 2); 2 wt% is converted to an aqueous condensate. Pyrolysis gas makes up 18 wt%, while 31 wt% is retained as pyrolysis residue, consisting of mineral fillers, glass fibers, and metals. On average, 4 wt% of the sample weight is lost (balance loss) due to encrustations within the reactor and measuring inaccuracies. Elemental analysis of condensates shows that pyrolysis oil contains approximately 80 wt% carbon and 12 wt% hydrogen (cf. Table S1-7, Supporting Information S1). The pyrolysis oil contains 8 wt% of heteroatoms, the main share being oxygen and 1 wt% water. The chlorine content in the pyrolysis oil is below 0.1 wt%. Pyrolysis gas consists of carbon dioxide as a principal component. Additionally, it contains significant amounts of C1 to C4 hydrocarbons and a minor amount of hydrogen. A detailed overview of pyrolysis product distribution, composition, and element mass balances of pyrolysis is given in section A2 of the supporting information.

### 3.4 | Utilization of pyrolysis products

Pyrolysis oil can be used as feedstock for steam cracking to produce ethene and propene (Kusenberget al., 2022b). However, steam cracker specifications regarding heteroatom and olefin content require an additional hydroprocessing step for upgrading pyrolysis oil (Kusenberget al., 2022c). The gas from the pyrolysis of APW contains valuable C3 and C4 olefins, but the low gas yield and a high share of carbon dioxide prevent the material

usage. Instead, it can be used as fuel for the pyrolysis process heat required having a net caloric value of 19 MJ/kg. Pyrolysis residue (9 MJ/kg) and aqueous condensate (−2 MJ/kg) must be disposed of and can also serve as fuel for process heat generation.

## 4 | LIFE CYCLE INVENTORY DATA

General assumptions and parameters for the assessments are summarized in Table S1-2 in Supporting Information S1, and additional detailed LCI data is also provided in (Section A3, Supporting Information S1). The following subsections describe the subprocesses and LCI data for the different end-of-life options.

### 4.1 | APW collection

A disposal company collects APW from automotive workshops and supplies it to RDF producers. The economic assessment of the disposal company is limited to the material-specific treatment costs considered in the MFCA. The cost associated with a transfer station is up to 0.02€/kg input (Bilitewski et al., 2018). Environmental impacts are not assessed as they are assigned to transport burdens only (Section 4.8).

### 4.2 | RDF production

APW is shredded and sorted. A share of 9% metals can be recovered and separated into ferrous (39%) and non-ferrous metals (61%). An analysis of the non-ferrous metals in the feedstock sample for the pyrolysis experiments demonstrates the domination of aluminum and copper (>90%) with a balanced mass ratio between them. Thus, we calculate with a share of 50% aluminum and 50% copper for all non-ferrous metal by-products. Process costs include electricity costs and fixed operating costs of 10% (Larrain et al., 2020; Riedewald et al., 2021) of the investment. Investments, electricity costs, and environmental impacts for an RDF production plant with a capacity of 20,000 Mg/year and operating 7500 h/year are derived from an exemplary production process based on manufacturer specifications that are scaled up to the assessed capacity, and by employing a plant factor (Stapf et al., 2019). The investment adds up to 880,000€/year; operation expenses result in 0.06€/kg input. Environmental impacts are derived from the process's electrical energy demand of 1.17 MJ/kg input multiplied by the German CO<sub>2</sub> and CED factors of the electricity grid (Table S1-2, Supporting Information S1). The CO<sub>2</sub> factor describes the CO<sub>2</sub> emissions associated with producing 1 kWh of electricity. In contrast, the CED factor describes the primary energy needed to generate 1 kWh of usable energy. Both factors also exist for the German heat mix. The mechanical pre-treatment has environmental impacts of 2.80 MJ/kg input (CED) and 0.14 kg CO<sub>2</sub>/kg input (climate change impact). The LCI data is summarized in Table S1-8 (Supporting Information S1).

### 4.3 | Metal recycling

Separated metals are sent to conventional recycling processes. Their environmental impacts are derived from ecoinvent datasets,<sup>2</sup> and processing costs are based on energy demand. The environmental rewards for substituting iron scrap in steel production are also based on an ecoinvent dataset<sup>3</sup>; the financial compensation is assumed with 0.025€/kg (Stapf et al., 2019). For aluminum and copper, a compensation of 0.25€/kg (Stapf et al., 2019) is assumed, and environmental rewards are derived from respective ecoinvent datasets<sup>4</sup>.

### 4.4 | Energy recovery

Relevant data for the incineration in MSWI plants and RDF power plants is provided in Table S1-9 (Supporting Information S1). Processing costs are assumed to be 0.12€/kg input (Bilitewski et al., 2018), and the environmental impacts are calculated to be 26.3 MJ/kg input (CED) and 2.47 kg CO<sub>2</sub>/kg input (climate change impact) based on the APW's elemental composition. Rewards are calculated based on the efficiency of the energy recovery facilities, the recovered energy (electricity and heat), and respective CO<sub>2</sub> and CED factors of the recovered energy.

### 4.5 | Pyrolysis

The pyrolysis plant converts RDF to pyrolysis oil, pyrolysis gas, and pyrolysis residues. Mass and elemental balances are closed by adding balance losses to the pyrolysis residue fraction. Adjusted experimental data for a metal-free feedstock is used for product distribution and composition (Section A3.3, Supporting Information S1).



Pyrolysis oil is sent to a hydroprocessing unit for upgrading to steam cracker specifications (Section 4.6). The incineration of pyrolysis by-products supplies the high-temperature heat demand of the pyrolysis unit. Excess low-temperature heat is supplied to a district heating network. The remaining ashes are landfilled. Climate change impact and CED of pyrolysis are calculated based on the amount of carbon dioxide released from the incineration of pyrolysis by-products (i.e., pyrolysis gas, pyrolysis residue, and aqueous condensates) (climate change impact) and their respective net calorific value (CED). Rewards are granted for excess heat substituting conventionally derived district heating. This results in a net climate change impact of 0.51 kg CO<sub>2</sub>/kg input (Section A3.3, Supporting Information S1) and a net CED of 4.49 MJ/kg input. Total processing costs are estimated to be 0.16 €/kg input.

## 4.6 | Hydroprocessing

Raw pyrolysis oil has to be upgraded to fulfill the demands of the petrochemistry processes designed for fossil feedstocks (Kusenberget al., 2022c). This includes removing heteroatoms and saturation of double bonds by hydrotreating and adjusting the pyrolysis oil boiling curve by hydrocracking.

Pyrolysis oil differs from conventional feedstocks mainly in terms of oxygen content and the occurrence of double bonds. Only little research has been conducted on the hydroprocessing of pyrolysis oil from plastic waste. However, for pyrolysis oil from polyolefinic plastic waste, it has been demonstrated on a laboratory scale that upgrading pyrolysis oil is feasible (Neuner et al., 2022). Therefore, the composition of hydroprocessed pyrolysis oil and the demand for hydrogen is estimated based on general assumptions (Section A3.4, Supporting Information S1). Nevertheless, further investigation of the hydroprocessing process for pyrolysis oil is necessary to validate the assumptions made.

A part of the estimated amount of hydrogen needed for hydroprocessing can be separated from steam cracker product gas (Section 4.7). Required additional hydrogen is assumed to be purchased. Here, we calculate with hydrogen produced as a by-product in an oil refinery. Associated environmental impacts are based on an ecoinvent dataset (Brunner, 2021), and prices are assumed to be 1575€/Mg H<sub>2</sub> (Bundestag, 2020). The hydroprocessed pyrolysis oil is sent to the steam cracker. Due to lacking information, a detailed hydroprocessing simulation is impossible. Thus, no energy demands for pumps and gas compression can be calculated, and estimating climate change impact and CED is not possible. In addition, no credits or burdens are considered for treating hydroprocessing by-products and utilization of excess hydrogen.

Hydroprocessing costs are considered in terms of fixed operating expenses and costs for hydrogen, as no energy demands are known. The investment for hydroprocessing is based on a hydroprocessing unit for vegetable oil (Marker et al., 2005) with costs for hydroprocessing of 0.10 €/kg input.

## 4.7 | Steam cracking

It is assumed that hydroprocessed pyrolysis oil replaces naphtha as steam-cracking feedstock. The LCI for producing primary HVC is used (PlasticEurope, 2012a); no further detailed assessment of steam cracking is performed. The assumed product yield is based on the steam-cracking process with naphtha feedstock (Figure S1-2, Supporting Information S1).

## 4.8 | Transportation

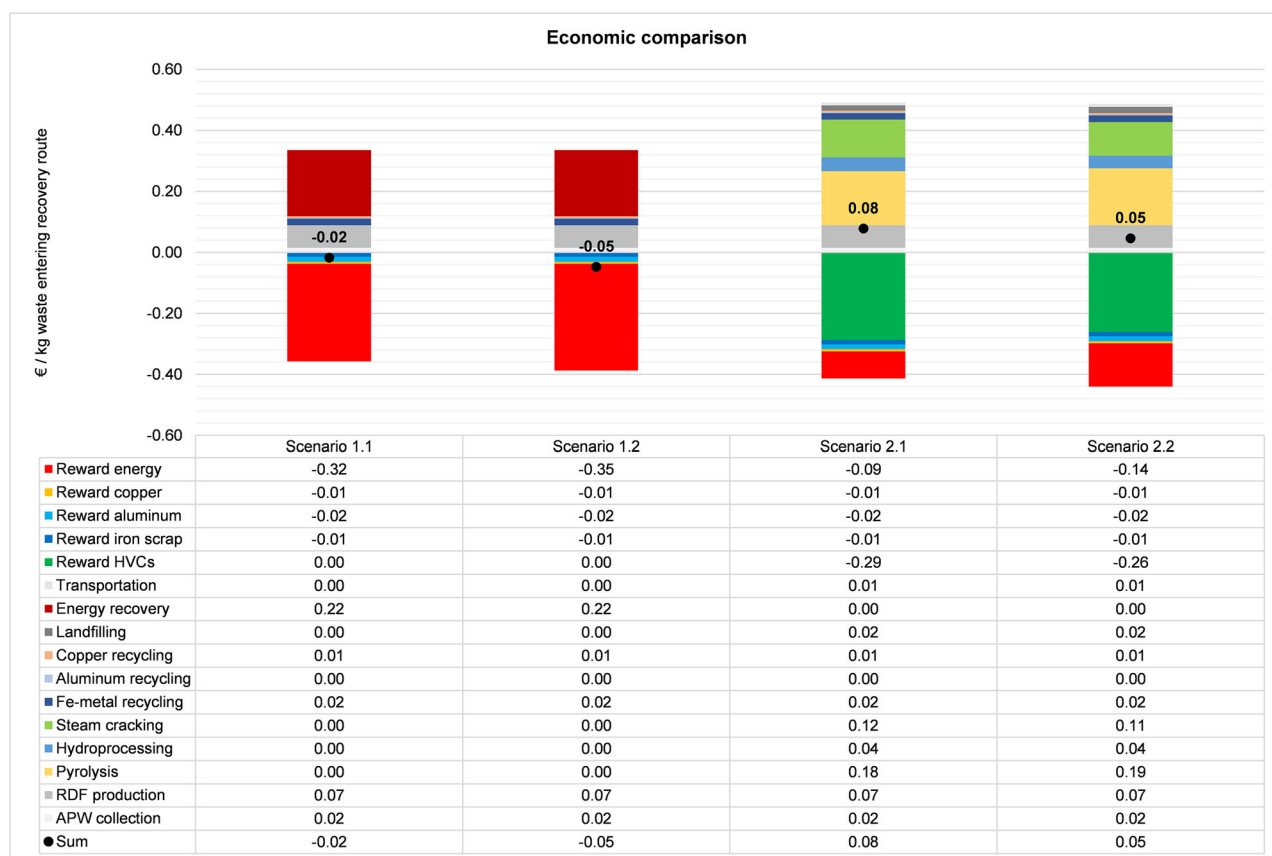
Assumptions and references for the transport distances are summarized in Table S1-12 (Supporting Information S1). Based on a spatial analysis for Germany, total transportation distances between automotive workshops, disposal companies, RDF production, and energy recovery are 115 km. For chemical recycling, instead of MSWI or RDF power plants, the average distance between RDF production and chemical plants with German steam crackers is considered. The spatial analysis results in an average distance of 247 km. Transportation for pyrolysis products or hydroprocessed pyrolysis oil is not considered since we assume the integration of pyrolysis and hydroprocessing into existing chemical plants. Environmental impacts and costs are provided in the Supporting Information (Section A3.6, Supporting Information S1).

## 5 | RESULTS AND DISCUSSION

The environmental impacts and economic assessment of energy recovery and chemical recycling are calculated based on the experimental and LCI data collected.

### 5.1 | Energy recovery scenarios

The baseline scenario of the energy recovery results in gross costs of 0.34€/kg input and induces gross values of 2.73 kg CO<sub>2</sub>e/kg input (climate change impact) and 31.80 MJ/kg input (CED). All RDF is incinerated in either an MSWI or RDF power plant. Incineration accounts for 83% of the



**FIGURE 3** Economic assessment of end-of-life paths and scenarios with costs above the x-axis and revenues beneath it. Assessment for 1 kg of input waste. The underlying data used to create this figure can be found in Supporting Information S2.

climate change impact and 75% of the CED impact. The net values consider substitution rewards for energy and primary metals, resulting in an economic assessment of  $-0.02$  €/kg input (Figure 3),  $1.25$  CO<sub>2</sub>e/kg input (climate change impact, Figure 4), and  $7.97$  MJ/kg input (CED, Figure 5).

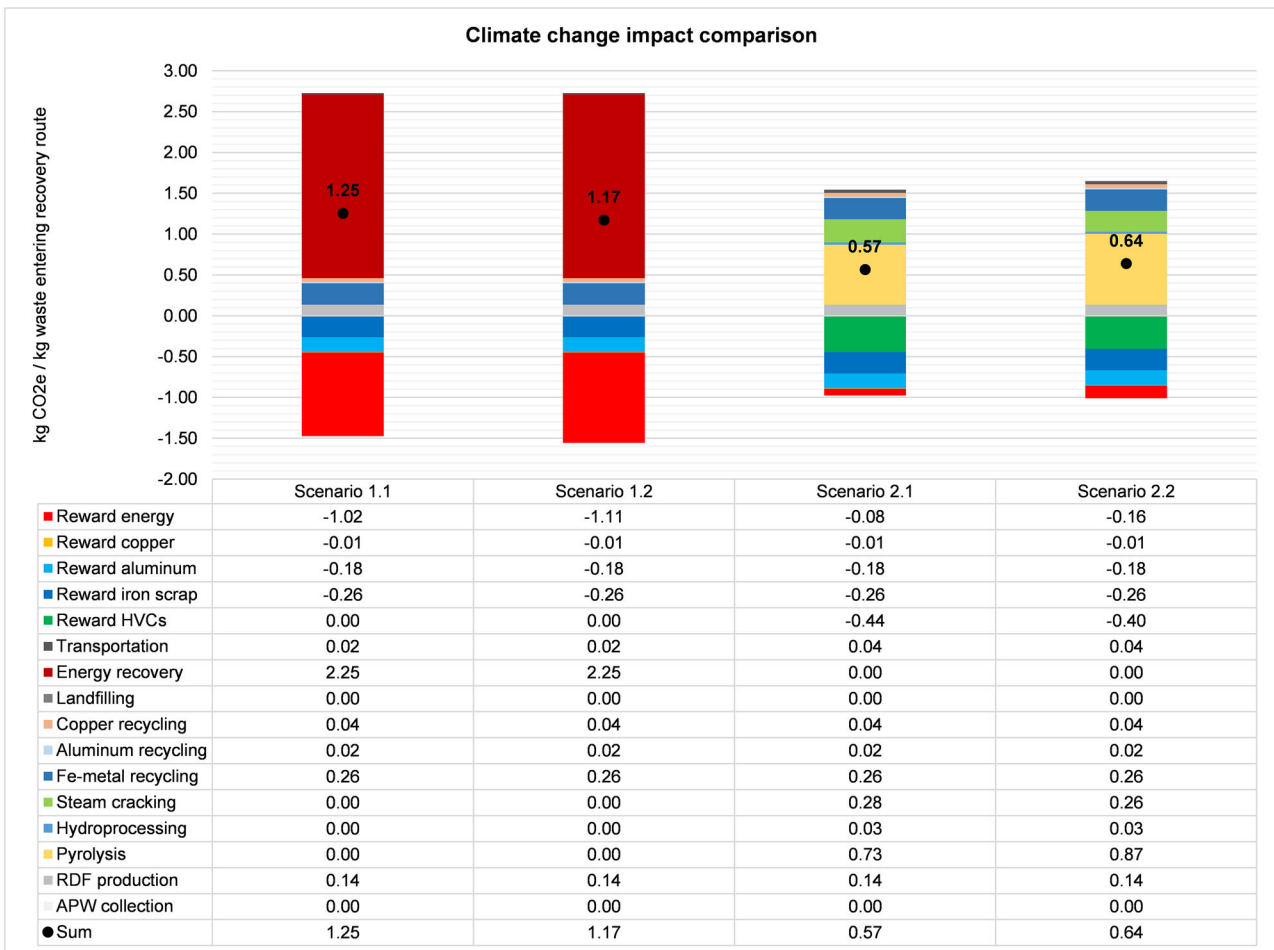
The results (Figures 4 and 5) show that the energy recovery in RDF power plants (scenario 1.2) leads to net reductions of 7% (climate change impact) and 13% (CED) compared to the baseline scenario 1.1. The economic assessment improves by 89% (Figure 3). The higher energy efficiency of RDF power plants and associated substitution rewards achieve the improvements.

## 5.2 | Chemical recycling (pyrolysis) scenarios

Chemical recycling of APW (scenario 2.1) results in gross processing costs of  $0.49$ €/kg input and induces gross values of  $1.54$  kg CO<sub>2</sub>e/kg input (climate change impact) and  $33.69$  MJ/kg input (CED). It leads to HVCs production of  $0.13$  kg ethene,  $0.06$  kg propene,  $0.04$  kg butadiene, and  $0.08$  kg pyrolysis gasoline (Figure 6). Figure 6 also outlines the carbon flow of the chemical recycling route, indicating the share of carbon that can potentially be recovered and contribute to a circular economy. Steam cracking significantly impacts all indicators<sup>5</sup> due to its high energy demand and emissions. Pyrolysis substantially influences the cost of the chemical recycling path (36%) due to increased investments and a high impact on the environmental indicators<sup>6</sup> due to the incineration of the pyrolysis by-products providing process heat. The net values of scenario 2.1 are an economic assessment of  $0.08$  €/kg input,  $0.57$  kg CO<sub>2</sub>e/kg input (climate change impact), and a CED of  $3.38$  MJ/kg input.

In scenario 2.2, a decrease in pyrolysis oil yield is assumed, and the mass balance difference is allocated to the solid fraction. Thus, more solids are incinerated, increasing climate change impact and CED. Higher rewards for recovered energy do not fully compensate for this increase, leading to a net climate change impact of  $0.64$  kg CO<sub>2</sub>e/kg input. The CED impact increases to  $4.41$  MJ/kg input. The net processing costs along the value chain decrease to  $0.05$ €/kg input because the high energy prices make the revenues from recovered heat higher than those from produced HVCs. The reduced pyrolysis oil yield results in fewer produced HVCs and less carbon potentially available for a circular economy.





**FIGURE 4** Comparison of end-of-life paths and scenarios regarding their net climate change impact. Assessment for 1 kg of input waste. The underlying data used to create this figure can be found in Supporting Information S2.

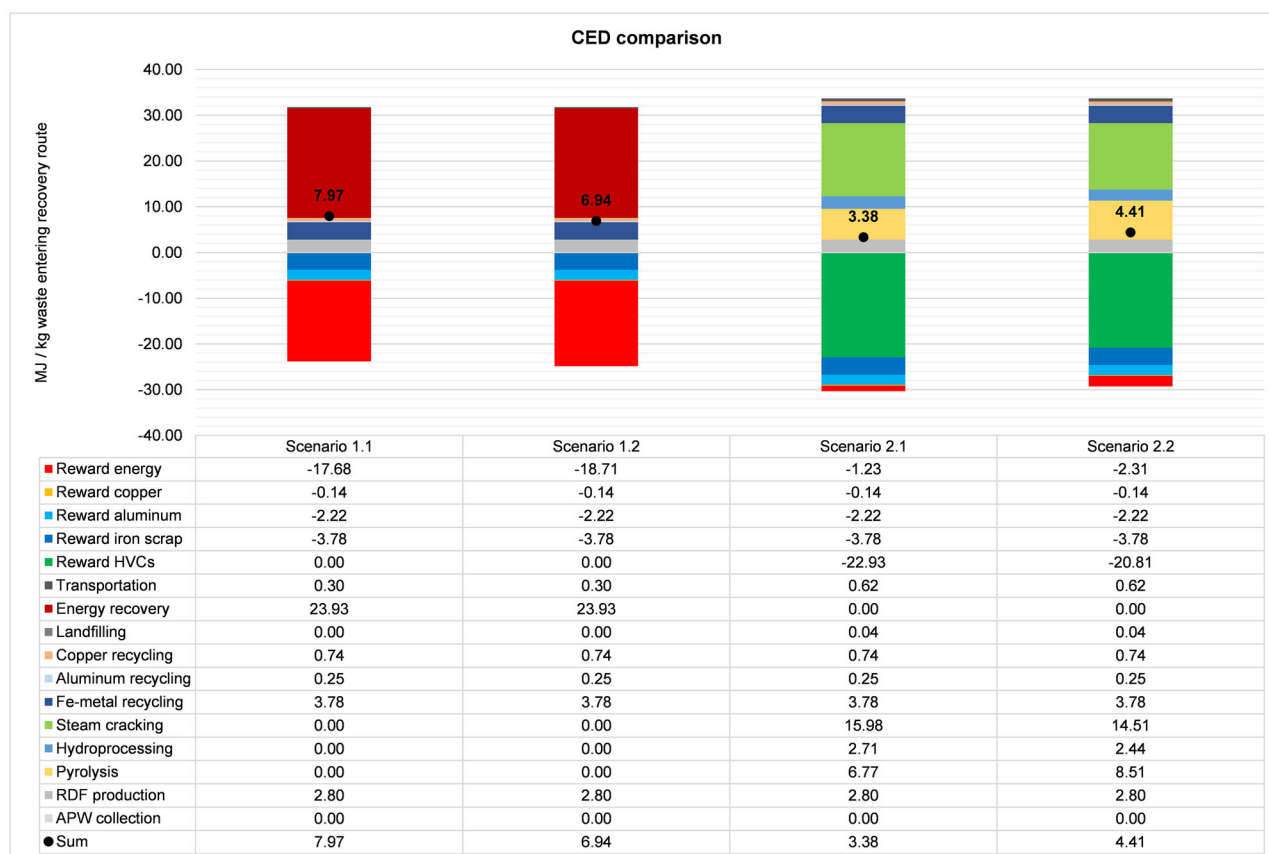
### 5.3 | Comparison of energy recovery and chemical recycling (pyrolysis)

Economically, energy recovery performs better than chemical recycling. This is due to high rewards for producing and substituting energy and lower gross processing costs due to fewer processing steps along the value chain. Energy recovery performs even better using more efficient RDF power plants with higher rewards for recovered energy and lower emissions. Economically, chemical recycling performs better with a lower pyrolysis oil yield due to an increased share of incinerated material and associated rewards for providing excess heat. Lower pyrolysis oil yields also result in lower processing costs at the steam-cracking plant, and the higher rewards for substitute district heating compensate for lower rewards for the avoided primary production of HVCs. This is an effect of the high energy prices in 2021 used in the assessment.

The climate change impact of the energy recovery scenarios is influenced by the incineration paths and their efficiencies (Figure 4). Incineration in efficient RDF power plants reduces the climate change impact compared to MSWI plants due to higher substitution rewards for generated heat and electricity. Regarding climate change impact, chemical recycling performs considerably better than energy recovery. Rewards for substituting primary HVCs counterbalance the high impacts of steam cracking and pyrolysis. In the chemical recycling scenarios, the impact of climate change increases with decreasing pyrolysis oil yield.

The CED impact of energy recovery decreases with more efficient incineration paths as more electricity and heat can be recovered (Figure 5). Both chemical recycling scenarios show lower CED impacts compared to the energy recovery. The CED savings decrease with decreasing pyrolysis oil yield as fewer HVCs are produced. This leads to lower rewards for the avoided primary production that cannot be compensated by increasing rewards for recovered energy.

In addition to the environmental benefits, chemical recycling can potentially contribute to a circular economy closing the carbon cycle. However, mass and elemental balances for carbon recovery demonstrate that a high pyrolysis oil yield is advantageous when the goal is to contribute to a circular economy.



**FIGURE 5** Comparison of considered end-of-life paths and scenarios regarding their net cumulative energy demand impact (MJ/kg input). Assessment for 1 kg of input waste. The underlying data used to create this figure can be found in Supporting Information S2.

## 5.4 | Sensitivity analysis

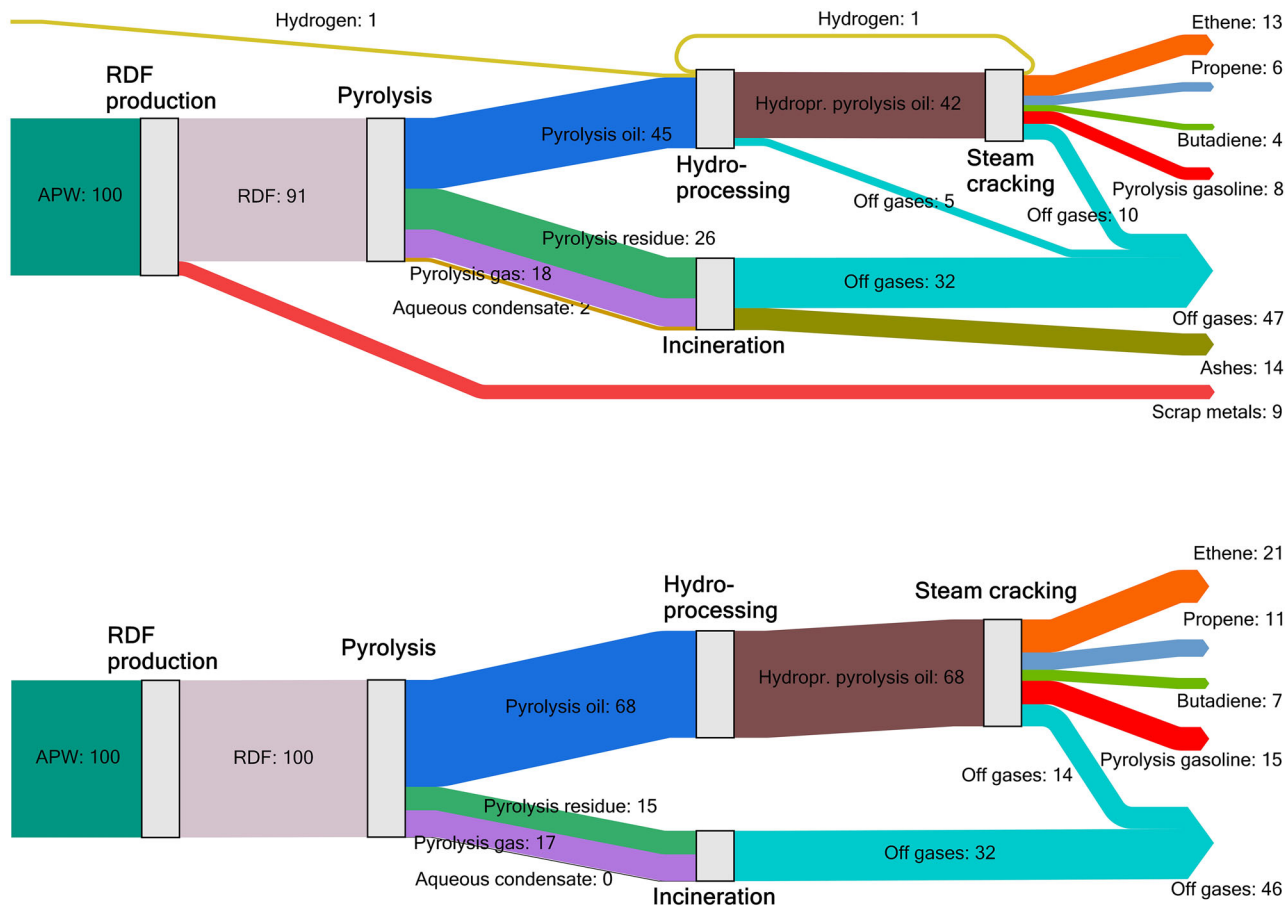
The sensitivity analysis focuses on processing costs and the global warming potential and was conducted for both baseline scenarios. An overview of the analyses is provided in Figure 7.

### 5.4.1 | Energy costs and electricity mix

Sensitivity analysis Sen1 and Sen2 analyze the effect of electricity prices and electricity revenues with an increase (Sen1) and a decrease of 10% (Sen2). Sen1 results in a 15% decrease in the total costs of energy recovery (Figure 7) as revenues for recovered energy from incineration are higher than increased electricity costs at the RDF producer. The electricity cost increase of RDF production increases the total costs for chemical recycling by 9%. Sen2 leads to a 15% increase in the total costs of energy recovery and a 9% decrease in chemical recycling.

Sen3 assumes an electric power supply mix that includes more fossil energy sources by a 10% increased CO<sub>2</sub> factor of the electricity mix. Sen4 considers a more decarbonized electricity supply by a 10% decreased CO<sub>2</sub> factor. While the carbon dioxide emissions from APW incineration remain unaltered, financial rewards for substituting electricity change due to the CO<sub>2</sub> emission fees associated with conventional electricity generation. In Sen3, this causes a decrease in the total net cost by 13%. The total cost of chemical recycling is not impacted, as only heat and no electricity is substituted. Higher emissions associated with the electricity used for RDF production lead to a 3% higher climate change impact on energy recovery and a 2% higher climate change impact for chemical recycling compared to their respective baselines. Sen4 results in a 13% increase in the costs of energy recovery, while the costs for chemical recycling do not change. The climate change impact of energy recovery decreases by 3%, while the climate change impact for chemical recycling decreases by 2%.

Sen5 and Sen6 analyze the influence of a 10% increase and decrease in the CO<sub>2</sub> emission price. A 10% increase (Sen5) causes a 31% increase in energy recovery costs since incineration costs increase more than the rewards for the recovered energy. For chemical recycling, a cost increase for CO<sub>2</sub> emissions from steam cracking is compensated by increasing rewards for recovered energy resulting in a 2% increase in total processing costs. The variations are symmetrical and do not influence the climate change impact.



**FIGURE 6** Material flow (top) and carbon flow (bottom) of the chemical recycling (scenario 2.1). Numbers in percent (rounded) of automotive plastic waste input. The underlying data used to create this figure can be found in Supporting Information S2.

## 5.4.2 | Chemical recycling

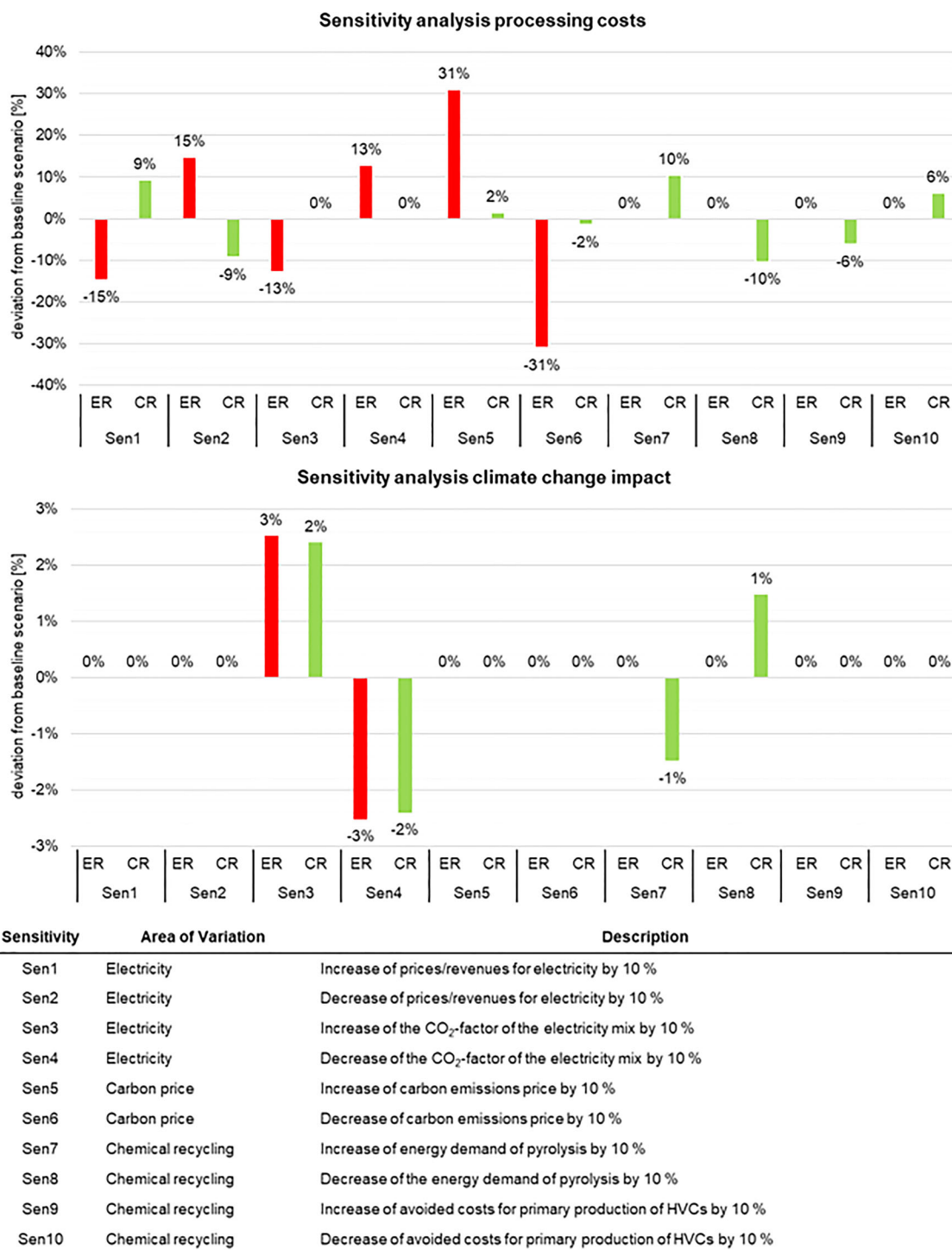
Sen7 and Sen8 estimate the effects of changing the energy demand of the pyrolysis process that may occur when it is scaled up to an industrial scale. The energy demand is increased by 10% (Sen7) and decreased by 10% (Sen8). The variations impact the processing costs of chemical recycling that increase or decrease by 10% as revenues from district heat vary. The climate change impact increases (Sen7) and decreases (Sen8) by 1% due to variations in the amount of heat substituted and associated burdens.

In Sen9 and Sen10, the avoided costs of the primary production of HVCs are varied. When conventional HVC production cost increase by 10% (Sen9), the chemical recycling cost decrease by 6% due to higher financial rewards for HVC. A 10% decrease in conventional HVC production costs (Sen10) leads to a 6% increase in chemical recycling costs.

## 5.5 | Comparison with other studies

There is no literature on APW pyrolysis. Therefore, the established data and results cannot be directly compared with other experimental data. The best comparison is the ASR pyrolysis on a similar experimental scale conducted by Notarnicola et al. (2017) and Galvagno et al. (2001). At similar temperatures (450–500°C), ASR yields almost double the amount of char (ca. 50%) and considerably less pyrolysis oil (20%–30%) in comparison to the pyrolysis of APW (31% char, 49% pyrolysis oil). The same is observed in laboratory-scale experiments (Joung et al., 2007; Santini et al., 2012; Zolezzi et al., 2004). Despite variations in the reported composition of ASR and pyrolysis product distribution in different studies, it can still be concluded that APW generally yields higher amounts of pyrolysis oil due to a higher share of polymers in the feedstock material. However, ASR is available in much higher quantities.

Only a few studies assess recycling paths for automotive plastics regarding environmental or economic indicators. Ciacci et al. (2010) and Passarini et al. (2012) compare ASR treatment processes in an LCA. They both include chemical recycling or feedstock recycling. However, their



**FIGURE 7** Sensitivity analysis results for energy recovery (ER) and chemical recycling (CR). The underlying data used to create this figure can be found in Supporting Information S2.

results are incomparable since they assess an open-loop gasification scenario. The thermal treatment with energy recovery is designed as co-combustion with municipal solid waste (Ciacci et al., 2010; Passarini et al., 2012). Therefore, the results are also not comparable because of the different characteristics of the waste streams in MSWI plants.

However, Ciacci et al. (2010) and Passarini et al. (2012) indicate that advanced recycling technologies, such as chemical recycling, show better environmental performance than energy recovery. This is consistent with this study. Arena and Ardolino (2022) analyze the recovery of polymers from hard-to-handle plastic waste streams such as EoL vehicles. In their environmental assessment, they also include catalytic pyrolysis and energy recovery. Their assessment is based on a feedstock of single polymers and not an actual waste fraction. Therefore, the results are not entirely

comparable, but they also indicate that catalytic pyrolysis performs better than energy recovery regarding the climate change impact. Cardamone et al. (2022) environmentally assess plastic recycling options for ASR with an extruder and pyrolysis combination. This waste-handling option is part of a recycling scheme that outperforms the current energy recovery option (Cardamone et al., 2022). They indicate that pyrolysis could be more environmentally beneficial than energy recovery, assuming a very optimistic pyrolysis oil yield. However, due to the chosen feedstock and process design of the chemical recycling, the results are only partially comparable to the results of this study. Other studies, for example, Li et al. (2016) and Chen et al. (2019), conduct an LCA for automobile recycling but do not include the chemical recycling of ASR.

## 5.6 | Limitations

This study has limitations and faces uncertainties due to (1) data and methodology, (2) model limits, and (3) assumptions. First, regarding (1) data and methodology: This study analyses a waste fraction with a comparatively low volume compared to ASR. However, the waste fraction of APW can become highly relevant if the dismantling of large plastic components becomes part of automobiles' EoL treatment processes. According to Wilts et al. (2016), dismantling could increase the amount of separated plastics available for recycling by a factor of six. Also, mixed plastic waste fractions separated from ASR by post-shredder-treatment processes have shown similar behavior in pyrolysis as the APW used in this study (Zeller et al., 2021).

The experiments demonstrate that the pyrolysis of engineering thermoplastics is technically feasible, and there is an environmentally beneficial alternative to their incineration. However, data for the subsequent hydroprocessing is unavailable yet, and very general assumptions had to be used. Also, this study does not consider other chemical recycling technologies besides pyrolysis. Mechanical recycling options and a combined mechanical and chemical recycling approach are also excluded. However, mechanical recycling processes for automotive plastics, like Sparenberg (2021) or the VW Sicon Process (Krinke et al., 2008), focus on separating standard thermoplastics and cannot handle engineering thermoplastics.

Second, this study has model limitations (2): The study assesses a defined waste stream of APW from workshop repair jobs with a specific composition to which the results are limited. The assessment is based on calculations for Germany. However, they are generic and can be transferred to countries with similar conditions. Nevertheless, this study does not consider dynamics such as changing waste compositions.

Third, assumptions (3) introduce uncertainties that are partly covered by scenarios (Section 2.3) and sensitivity analysis (Section 5.4). However, assumptions regarding hydroprocessing (Section 4.6) are not covered due to missing data. All processing facilities with direct CO<sub>2</sub> emissions are assumed to be covered within the EU emission trading system. No national emission trading systems are included in the assessment, indicating a possible extension. In reality, a mix of scenarios and sensitivities is likely, due to differing plant efficiencies, variable waste compositions, yields, and qualities of the pyrolysis products.

## 6 | CONCLUSIONS AND OUTLOOK

The pyrolysis experiments demonstrate the technical feasibility of the pyrolysis of APW containing engineering thermoplastics. Based on the experimental results, energy recovery and chemical recycling of APW to HVCs via pyrolysis are assessed, compared, and analyzed in different scenarios regarding their climate change impact, CED, and processing costs. The results show that chemical recycling has lower net environmental impacts than energy recovery, while, under current market conditions, energy recovery performs better economically. Therefore, this study identifies a conflict between the economic and environmental objectives of EoL options for APW. The sensitivity analysis shows that the price and revenue associated with electricity and the carbon emission price significantly impact the assessment. Here, the assessed indicators for energy recovery correlate stronger with the varied parameters than for chemical recycling.

Chemical recycling also has the potential to keep carbon in the material cycle. The gained pyrolysis oil is a valuable petrochemical feedstock and can be used in different processes, for example, to produce new primary plastics. Therefore, the chemical recycling of APW can contribute to a circular economy by closing the carbon and the automotive plastic loop. Additional research is needed to provide experimentally validated data for the hydroprocessing of the produced pyrolysis oil. Further research should also address the potential of mechanical recycling processes for engineering thermoplastics, alternative feedstock utilization paths of pyrolysis oil with lower quality demands than steam cracking, and scenarios when technologies are combined. This can contribute to designing and optimizing recycling systems for complex mixed plastic wastes.

### AUTHOR CONTRIBUTIONS

**Christoph Stallkamp:** Data curation; formal analysis; investigation; methodology; writing—original draft; writing—review and editing. **Malte Hennig:** Data curation; formal analysis; investigation; methodology; writing—original draft; writing—review and editing. **Rebekka Volk:** Conceptualization; supervision; resources; writing—review and editing. **Frank Richter:** Investigation. **Britta Bergfeldt:** Investigation. **Salar Tavakkol:** Resources. **Frank Schultmann:** Supervision; writing—review and editing. **Dieter Stapf:** Conceptualization; supervision; resources; writing—review and editing.



## ACKNOWLEDGMENTS

This study was carried out within the research project “Leuchtturm Kreislaufwirtschaft—Schwerpunkt Chemisches Recycling” funded by the “THINKTANK Industrial Resource Strategies” and AUDI AG. The THINKTANK was financed by the Ministry of the Environment, Climate Protection, and the Energy Sector of the state of Baden-Württemberg in Germany and industry partners. The Volkswagen Original Teile Logistik GmbH & Co. KG provided the automotive plastic waste sample.

## CONFLICT OF INTEREST STATEMENT

The authors declare no conflict of interest.

## DATA AVAILABILITY STATEMENT

The data underlying the findings of this study are available in the supporting information of this article. Additionally, datasets from the ecoinvent database version 3.7 were used and cited in the manuscript.

## ORCID

Christoph Stallkamp  <https://orcid.org/0000-0001-8260-2889>

Malte Hennig  <https://orcid.org/0000-0002-0289-5608>

Rebekka Volk  <https://orcid.org/0000-0001-9930-5354>

## NOTES

<sup>1</sup> Maintenance, insurance, and general plant overhead including human resources, research and development, information technology, finance, and legal (Larrain et al., 2020).

<sup>2</sup> For aluminum recycling, the datasets from Müller (2020) and Lesage (2020) are used while Classen (2020) is applied for copper recycling.

<sup>3</sup> Ecoinvent dataset from Wernet (2020).

<sup>4</sup> For aluminum production, the dataset from Jungbluth (2020) and for copper production, the dataset from Turner (2020) is applied.

<sup>5</sup> Steam cracking accounts for 29% of the cost, 19% of the climate change impact, and 50% of the CED.

<sup>6</sup> climate change impact: 48% and CED: 21%.

## REFERENCES

- Arena, U., & Ardolino, F. (2022). Technical and environmental performances of alternative treatments for challenging plastics waste. *Resources, Conservation and Recycling*, 183, 106379. <https://doi.org/10.1016/j.resconrec.2022.106379>
- Bernasconi, A., Rossin, D., & Armanni, C. (2007). Analysis of the effect of mechanical recycling upon tensile strength of a short glass fibre reinforced polyamide 6.6. *Engineering Fracture Mechanics*, 74(4), 627–641. <https://doi.org/10.1016/j.engfracmech.2006.10.002>
- Bililewski, B., Wagner, J., & Reichenbach, J. (2018). Bewährte Verfahren zur kommunalen Abfallbewirtschaftung. *UBA Texte*, 2018(39), 320. [https://www.umweltbundesamt.de/sites/default/files/medien/1410/publikationen/2018-05-30\\_texte\\_39-2018-verfahren-kommunale-abfallwirtschaft\\_0.pdf](https://www.umweltbundesamt.de/sites/default/files/medien/1410/publikationen/2018-05-30_texte_39-2018-verfahren-kommunale-abfallwirtschaft_0.pdf)
- Brunner, F. (2021). Hydrogen production, gaseous, petroleum refinery operation - Europe without Switzerland - hydrogen, gaseous. Ecoinvent database version 3.7. <https://v38.ecoquery.ecoinvent.org/Details/LCIA/732fc37d-ef34-4727-92f3-b943efdc5ba6/290c1f85-4cc4-4fa1-b0c8-2cb7f4276dce>
- Bundestag. (2020). Wasserstoff: Produktionskosten nach Typ bis 2050 [Hydrogen: production costs by type by 2050]. Statista. <https://de.statista.com/statistik/daten/studie/1195863/umfrage/produktionskosten-von-wasserstoff-nach-wasserstofftyp-in-deutschland/>
- Cardamone, G. F., Ardolino, F., & Arena, U. (2022). Can plastics from end-of-life vehicles be managed in a sustainable way? *Sustainable Production and Consumption*, 29, 115–127. <https://doi.org/10.1016/j.spc.2021.09.025>
- Chen, Y., Ding, Z., Liu, J., & Ma, J. (2019). Life cycle assessment of end-of-life vehicle recycling in China: A comparative study of environmental burden and benefit. *International Journal of Environmental Studies*, 76(6), 1019–1040. <https://doi.org/10.1080/00207233.2019.1618670>
- Ciacchi, L., Morselli, L., Passarini, F., Santini, A., & Vassura, I. (2010). A comparison among different automotive shredder residue treatment processes. *The International Journal of Life Cycle Assessment*, 15(9), 896–906. <https://doi.org/10.1007/s11367-010-0222-1>
- Classen, M. (2020). Treatment of copper scrap by electrolytic refining, RER, undefined. Ecoinvent database version 3.7. <https://v37.ecoquery.ecoinvent.org/Details/UPR/7B47EADD-7695-4088-870F-DFC9963D3EF9/8B738EA0-F89E-4627-8679-433616064E82>
- Colucci, G., Simon, H., Roncato, D., Martorana, B., & Badini, C. (2017). Effect of recycling on polypropylene composites reinforced with glass fibres. *Journal of Thermoplastic Composite Materials*, 30(5), 707–723. <https://doi.org/10.1177/0892705715610407>
- Cossu, R., Fiore, S., Lai, T., Luciano, A., Mancini, G., Ruffino, B., Viotti, P., & Zanetti, M. C. (2014). Review of Italian experience on automotive shredder residue characterization and management. *Waste Management*, 34(10), 1752–1762. <https://doi.org/10.1016/j.wasman.2013.11.014>
- Cossu, R., & Lai, T. (2015). Automotive shredder residue (ASR) management: An overview. *Waste Management*, 45, 143–151. <https://doi.org/10.1016/j.wasman.2015.07.042>
- 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>
- Dogu, O., Pelucchi, M., Van De Vijver, R., Van Steenberge, P. H. M., D'hooge, D. R., Cuoci, A., Mehl, M., Frassoldati, A., Faravelli, T., & Van Geem, K. M. (2021). The chemistry of chemical recycling of solid plastic waste via pyrolysis and gasification: State-of-the-art, challenges, and future directions. *Progress in Energy and Combustion Science*, 84, 100901. <https://doi.org/10.1016/j.pecs.2020.100901>
- Galvagno, S., Fortuna, F., Cornacchia, G., Casu, S., Coppola, T., & Sharma, V. K. (2001). Pyrolysis process for treatment of automobile shredder residue: Preliminary experimental results. *Energy Conversion and Management*, 42(5), 573–586. [https://doi.org/10.1016/S0196-8904\(00\)00089-3](https://doi.org/10.1016/S0196-8904(00)00089-3)



- Gracida-Alvarez, U. R., Winjobi, O., Sacramento-Rivero, J. C., & Shonnard, D. R. (2019a). System analyses of high-value chemicals and fuels from a waste high-density polyethylene refinery. Part 1: Conceptual design and techno-economic assessment. *ACS Sustainable Chemistry & Engineering*, 7(22), 18254–18266. <https://doi.org/10.1021/acssuschemeng.9b04763>
- Gracida-Alvarez, U. R., Winjobi, O., Sacramento-Rivero, J. C., & Shonnard, D. R. (2019b). System analyses of high-value chemicals and fuels from a waste high-density polyethylene refinery. Part 2: Carbon footprint analysis and regional electricity effects. *ACS Sustainable Chemistry & Engineering*, 7(22), 18267–18278. <https://doi.org/10.1021/acssuschemeng.9b04764>
- Gradin, K. T., & Björklund, A. (2021). The common understanding of simplification approaches in published LCA studies—A review and mapping. *The International Journal of Life Cycle Assessment*, 26(1), 50–63. <https://doi.org/10.1007/s11367-020-01843-4>
- Iacovidou, E., Velis, C. A., Purnell, P., Zwiner, O., Brown, A., Hahladakis, J., Millward-Hopkins, J., & Williams, P. T. (2017). Metrics for optimising the multi-dimensional value of resources recovered from waste in a circular economy: A critical review. *Journal of Cleaner Production*, 166, 910–938. <https://doi.org/10.1016/j.jclepro.2017.07.100>
- IPCC. (2013). Climate change 2013: The physical science basis; summary for policymakers, a report of Working Group I of the IPCC, report of the Intergovernmental Panel on Climate Change. New York. <http://www.ipcc.ch/>
- 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>
- Joung, H. T., Seo, Y. C., Kim, K. H., Hong, J. H., & Yoo, T. W. (2007). Distribution and characteristics of pyrolysis products from automobile shredder residue using an experimental semi-batch reactor. *Korean Journal of Chemical Engineering*, 24(6), 996–1002. <https://doi.org/10.1007/s11814-007-0110-y>
- Jungbluth, N. (2020). Aluminium alloy production, AlMg<sub>3</sub>, RER, undefined. Ecoinvent database version 3.7. <https://v36.ecoquery.ecoinvent.org/Details/UPR/22B2A4E5-E99C-444B-8AD0-03039959B7DA/8B738EA0-F89E-4627-8679-433616064E82>
- Krinke, S., Boßdorf-Zimmer, B., & Goldmann, D. (2008). The Volkswagen-SiCon Process: Eco-efficient solution for future end-of-life vehicle treatment. <https://www.mech.kuleuven.be/lce2006/157.pdf>
- Kusenbergh, M., Eschenbacher, A., Djokic, M. R., Zayoud, A., Ragaert, K., De Meester, S., & Van Geem, K. M. (2022a). Opportunities and challenges for the application of post-consumer plastic waste pyrolysis oils as steam cracker feedstocks: To decontaminate or not to decontaminate? *Waste Management*, 138, 83–115. <https://doi.org/10.1016/j.wasman.2021.11.009>
- Kusenbergh, M., Fausson, G. C., Thi, H. D., Roosen, M., Grilc, M., Eschenbacher, A., De Meester, S., & Van Geem, K. M. (2022b). Maximizing olefin production via steam cracking of distilled pyrolysis oils from difficult-to-recycle municipal plastic waste and marine litter. *The Science of the Total Environment*, 838(2), 156092. <https://doi.org/10.1016/j.scitotenv.2022.156092>
- Kusenbergh, M., Roosen, M., Zayoud, A., Djokic, M. R., Thi, H. D., De Meester, S., Ragaert, K., Kresovic, U., & Van Geem, K. M. (2022c). Assessing the feasibility of chemical recycling via steam cracking of untreated plastic waste pyrolysis oils: Feedstock impurities, product yields and coke formation. *Waste Management*, 141, 104–114. <https://doi.org/10.1016/j.wasman.2022.01.033>
- Larrain, M., Van Passel, S., Thomassen, G., Kresovic, U., Alderweireldt, N., Moerman, E., & Billen, P. (2020). Economic performance of pyrolysis of mixed plastic waste: Open-loop versus closed-loop recycling. *Journal of Cleaner Production*, 270, 122442. <https://doi.org/10.1016/j.jclepro.2020.122442>
- Lesage, P. (2020). Treatment of aluminium scrap, post-consumer, prepared for recycling, at refiner, RER, undefined. Ecoinvent database version 3.7. <https://v37.ecoquery.ecoinvent.org/Details/UPR/88392A35-EB59-4588-B65C-06979D0839A0/8B738EA0-F89E-4627-8679-433616064E82>
- Li, W., Bai, H., Yin, J., & Xu, H. (2016). Life cycle assessment of end-of-life vehicle recycling processes in China—Take Corolla taxis for example. *Journal of Cleaner Production*, 117, 176–187. <https://doi.org/10.1016/j.jclepro.2016.01.025>
- Marker, T., Petri, J., Kalnes, T., McCall, M., Mackowiak, D., Jerosky, B., Reagan, B., Nemeth, L., Krawczyk, M., Czernik, S., Elliott, D., & Shonnard, D. (2005). *Opportunities for biorenewables in oil refineries*. <https://doi.org/10.2172/861458>
- Mehlhart, G., Möck, A., & Goldmann, D. (2018). Effects on ELV waste management as a consequence of the decisions from the Stockholm Convention on decaBDE. Darmstadt. Oeko-Institut e.V. <https://www.oeko.de/fileadmin/oekodoc/ACEA-DecaBDE-final-report.pdf>
- 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., Käthelhön, 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*, 374(6563), 71–76. <https://doi.org/10.1126/science.abg9853>
- Tomasi Morgano, M., Leibold, H., Richter, F., Stapf, D., & Seifert, H. (2018). Screw pyrolysis technology for sewage sludge treatment. *Waste Management*, 73, 487–495. <https://doi.org/10.1016/j.wasman.2017.05.049>
- Müller, J. (2020). Steel production, electric, chromium steel 18/8, RER, undefined. Ecoinvent database version 3.7. <https://v37.ecoquery.ecoinvent.org/Details/UPR/04805BFA-BE14-4D64-817F-AF9A0AD7C879/8B738EA0-F89E-4627-8679-433616064E82>
- Nakatani, J. (2014). Life cycle inventory analysis of recycling: Mathematical and graphical frameworks. *Sustainability*, 6(9), 6158–6169. <https://doi.org/10.3390/su6096158>
- Neuner, P., Graf, D., Netsch, N., Zeller, M., Herrmann, T.-C., Stapf, D., & Rauch, R. (2022). Chemical conversion of Fischer–Tropsch waxes and plastic waste pyrolysis condensate to lubricating oil and potential steam cracker feedstocks. *Reactions*, 3(3), 352–373. <https://doi.org/10.3390/reactions3030026>
- Notarnicola, M., Cornacchia, G., De Gisi, S., Di Canio, F., Freda, C., Garzone, P., Martino, M., Valerio, V., & Villone, A. (2017). Pyrolysis of automotive shredder residue in a bench scale rotary kiln. *Waste Management*, 65, 92–103. <https://doi.org/10.1016/j.wasman.2017.04.002>
- Passarini, F., Ciacci, L., Santini, A., Vassura, I., & Morselli, L. (2012). Auto shredder residue LCA: Implications of ASR composition evolution. *Journal of Cleaner Production*, 23(1), 28–36. <https://doi.org/10.1016/j.jclepro.2011.10.028>
- Pietrolungo, M., Padovano, E., Frache, A., & Badini, C. (2020). Mechanical recycling of an end-of-life automotive composite component. *Sustainable Materials and Technologies*, 23, e00143. <https://doi.org/10.1016/j.susmat.2019.e00143>
- Pivnenko, K., Jakobsen, L. G., Eriksen, M. K., Damgaard, A., & Astrup, T. F. (2015). Challenges in plastic recycling. In S. Margherita di Pula (Ed.), *Proceedings Sardinia 2015: Fifteenth International Waste Management and Landfill Symposium CISA Publisher*.
- PlasticsEurope. (2012). Eco-profiles and environmental product declarations of the European plastics manufacturers – Ethylene, propylene, butadiene, pyrolysis gasoline, ethylene oxide (EO), ethylene glycols (MEG, DEG, TEG). [https://www.plasticseurope.org/download\\_file/787/0](https://www.plasticseurope.org/download_file/787/0)
- Rieckhof, R., & Guenther, E. (2018). Integrating life cycle assessment and material flow cost accounting to account for resource productivity and economic-environmental performance. *The International Journal of Life Cycle Assessment*, 23(7), 1491–1506. <https://doi.org/10.1007/s11367-018-1447-7>

- Riedewald, F., Patel, Y., Wilson, E., Santos, S., & Sousa-Gallagher, M. (2021). Economic assessment of a 40,000 t/y mixed plastic waste pyrolysis plant using direct heat treatment with molten metal: A case study of a plant located in Belgium. *Waste Management*, 120, 698–707. <https://doi.org/10.1016/j.wasman.2020.10.039>
- Santini, A., Passarini, F., Vassura, I., Serrano, D., Dufour, J., & Morselli, L. (2012). Auto shredder residue recycling: Mechanical separation and pyrolysis. *Waste Management*, 32(5), 852–858. <https://doi.org/10.1016/j.wasman.2011.10.030>
- Solis, M., & Silveira, S. (2020). Technologies for chemical recycling of household plastics - A technical review and TRL assessment. *Waste Management*, 105, 128–138. <https://doi.org/10.1016/j.wasman.2020.01.038>
- Spärgen, B. (2021). *From linear to circular economy Raas*. [https://circulareconomy.europa.eu/platform/sites/default/files/bsp\\_albis\\_wipag\\_open\\_loop\\_closed\\_loop\\_raas\\_10\\_19s.pdf](https://circulareconomy.europa.eu/platform/sites/default/files/bsp_albis_wipag_open_loop_closed_loop_raas_10_19s.pdf)
- Stapf, D., Wexler, M., Ciceri, G., Johansson, I., & Whitty, K. (2019). Pre-treatment of municipal solid waste (MSW) for gasification (Biomass pre-treatment for bioenergy – Case study 3). *IEA Bioenergy Report*, <https://doi.org/10.13140/RG.2.2.27035.39205>
- Turner, D. (2020). *Electrorefining of copper, anode*, GLO, undefined. Ecoinvent database version 3.7. <https://v37.ecoquery.ecoinvent.org/Details/UPR/4DF92F61-FECA-44E0-B82A-D9BE71F9507E/8B738EA0-F89E-4627-8679-433616064E82>
- Van Eygen, E., Laner, D., & Fellner, J. (2018). Circular economy of plastic packaging: Current practice and perspectives in Austria. *Waste Management*, 72, 55–64. <https://doi.org/10.1016/j.wasman.2017.11.040>
- VDI. (2012). *Cumulative energy demand (KEA) - Terms, definitions, methods of calculation (VDI 4600)*. Verlag des Vereins Deutscher Ingenieure.
- Volk, R., Stallkamp, C., Steins, J. J., Yogish, S. P., Müller, R. C., Stapf, D., & Schultmann, F. (2021). Techno-economic assessment and comparison of different plastic recycling pathways: A German case study. *Journal of Industrial Ecology*, 25, 1318–1337. <https://doi.org/10.1111/jiec.13145>
- Wernet, G. (2020). *Market for iron scrap, unsorted*, GLO, undefined. Ecoinvent database version 3.7. <https://v37.ecoquery.ecoinvent.org/Details/UPR/F495D4CC-326A-45F2-96B0-E73DB41E2280/8B738EA0-F89E-4627-8679-433616064E82>
- Wilts, C. H., von Gries, N., Dehne, I., Oetjen-Dehne, R., Buschow, N., & Sanden, J. (2016). Entwicklung von Instrumenten und Maßnahmen zur Steigerung des Einsatzes von Sekundärrohstoffen: mit Schwerpunkt Sekundärkunststoffe. <https://epub.wupperinst.org/frontdoor/index/index/docId/6523>
- Yang, S., Bai, S., Duan, W., & Wang, Q. (2018). Preparation of composites based on recycled polypropylene and automotive shredder residue. *Polymer International*, 67(7), 936–945. <https://doi.org/10.1002/pi.5591>
- 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, 1763–1770. <https://doi.org/10.1002/cite.202100102>
- Zolezzi, M., Nicoletta, C., Ferrara, S., Iacobucci, C., & Rovatti, M. (2004). Conventional and fast pyrolysis of automobile shredder residues (ASR). *Waste Management*, 24(7), 691–699. <https://doi.org/10.1016/j.wasman.2003.12.005>

## SUPPORTING INFORMATION

Additional supporting information can be found online in the Supporting Information section at the end of this article.

**How to cite this article:** Stallkamp, C., Hennig, M., Volk, R., Richter, F., Bergfeldt, B., Tavakkol, S., Schultmann, F., & Stapf, D. (2023). Economic and environmental assessment of automotive plastic waste end-of-life options: Energy recovery versus chemical recycling. *Journal of Industrial Ecology*, 27, 1319–1334. <https://doi.org/10.1111/jiec.13416>