DOI: 10.1111/jiec.13145

JOURNAL OF INDUSTRIAL ECOLOGY WILEY

RESEARCH AND ANALYSIS

Techno-economic assessment and comparison of different plastic recycling pathways

A German case study

Rebekka Volk[1] | Christoph Stallkamp[1] | Justus J. Steins[1] | Savina Padumane Yogish[2] | Richard C. Müller[1] | Dieter Stapf[2] | Frank Schultmann[1]

[1] Karlsruhe Institute of Technology (KIT), Institute for Industrial Production (IIP)

[2] Karlsruhe Institute of Technology (KIT), Institute for Technical Chemistry (ITC)

Correspondence
Rebekka Volk, Karlsruhe Institute of Technology, Department of Economics and Management, Institute for Industrial Production (IIP), Hertzstr. 16, 76187 Karlsruhe, Germany.
Email: rebekka.volk@kit.edu

Editor Managing Review: Lynette Cheah

Funding information
This study was carried out within the research projects "Leuchtturm Kreislaufwirtschaft – Schwerpunkt Chemisches Recycling" and "Kreislaufwirtschaft für Kunststoffe" funded by the "THINKTANK Industrielle Ressourcenstrategien" (Industrial Resource Strategies) which is financed by the Ministry of the Environment, Climate Protection, and the Energy Sector of the state of Baden-Württemberg in Germany and industry partners.

Abstract

Greenhouse gas (GHG) emissions need to be reduced to limit global warming. Plastic production requires carbon raw materials and energy that are associated today with predominantly fossil raw materials and fossil GHG emissions. Worldwide, the plastic demand is increasing annually by 4%. Recycling technologies can help save or reduce GHG emissions, but they require comparative assessment. Thus, we assess mechanical recycling, chemical recycling by means of pyrolysis and a consecutive, complementary combination of both concerning Global Warming Potential (GWP) [CO_2e], Cumulative Energy Demand (CED) [MJ/kg], carbon efficiency [%], and product costs [€] in a process-oriented approach and within defined system boundaries. The developed techno-economic and environmental assessment approach is demonstrated in a case study on recycling of separately collected mixed lightweight packaging (LWP) waste in Germany. In the recycling paths, the bulk materials polypropylene (PP), polyethylene (PE), polyvinylchloride (PVC), and polystyrene (PS) are assessed. The combined mechanical and chemical recycling (pyrolysis) of LWP waste shows considerable saving potentials in GWP (0.48 kg CO_2e/kg input), CED (13.32 MJ/kg input), and cost (0.14 €/kg input) and a 16% higher carbon efficiency compared to the baseline scenario with state-of-the-art mechanical recycling in Germany. This leads to a combined recycling potential between 2.5 and 2.8 million metric tons/year that could keep between 0.8 and 2 million metric tons/year additionally in the (circular) economy instead of incinerating them. This would be sufficient to reach both EU and German recycling rate targets (EC 2018). This article met the requirements for a gold-silver *JIE* data openness badge described at http://jie.click/badges.

KEYWORDS
carbon management, chemical/feedstock recycling, circular economy, environmental accounting, GHG emissions, plastics recycling

1 INTRODUCTION

Plastic production requires predominantly crude oil and energy while emitting GHG emissions that need to be reduced to combat climate change (IPCC, 2013). The German chemical sector, the largest in Europe, accounts for 6% of annual GHG emissions (UBA, 2018a; Destatis, 2018; VCI, 2019; Wyns et al., 2018). In Europe, 49 million tons[1] of plastics are produced annually for packaging (40%), construction (20%), automotive parts (9%) and electronics (6%). 25.8 million tons of plastic waste are generated annually (EC, 2018). Plastic waste can be recycled mechanically or chemically, or processed to chemical or physical energy carriers. Currently, less than 30% is collected for recycling; a significant part is exported to non-EU countries with lower environmental standards (EC, 2018). In Germany, 46 wt% of the plastic waste (including production and processing wastes) is recycled mechanically; 1 wt% is recycled chemically (Conversio, 2018).

The EU demands a recycling rate for plastics packaging waste of 55% in 2030, while Germany demands 63% until 2022 (EC, 2018). But, mechanical recycling only can hardly fulfil the imposed recycling rates due to technical restrictions and economic limitations (Pivnenko et al., 2015; Ragaert et al., 2017).

Mechanical recycling processes plastic waste fractions without significant changes to their chemical composition, while chemical recycling processes it to intermediate chemicals by changing its chemical structure (Conversio, 2018; Stapf et al., 2019b). The latter can be used as a secondary or renewable feedstock (Meran et al., 2008; Sommerhuber et al., 2016; Gu et al., 2016a,b) in plastics' or other chemicals' production and can significantly reduce GHG emissions (Makuta et al., 2000; Dormer et al., 2013). Mechanical recycling cannot produce high- or virgin-quality plastics, but up to 20—50% cheaper plastics compared to virgin plastics (Gu et al., 2016a,b; 2017). Thus, it is challenging to identify the environmentally or economically best recycling option, depending on locally available technologies, capacities, process efficiencies, specific waste compositions, and conditions (Van Eygen et al., 2018a).

Life Cycle Assessment (LCA) (DIN EN ISO 14040:2006) is widespread to assess environmental impacts during a products' lifecycle (Rieckhof & Guenther, 2018; Rebitzer, 2002; Klöpffer & Renner, 2008). Multiple LCA and techno-economic analyses of olefin production from oil, coal, methane, and ethane (e.g., Ren et al., 2006, 2008; Ren, 2009; Xiang et al., 2014a,b; 2015; Amghizar et al., 2017; Zhao et al., 2017) and plant-specific approaches (e.g., Patel, 2003; Pereira et al., 2013; Kanchanapiya et al., 2015) were conducted. LCA was also applied to mechanical recycling, incineration with energy recovery, and landfilling of plastic waste to compare disposal alternatives (e.g., by Lazarevic et al., 2010; Wäger et al., 2011; Al-Maadeed et al., 2012; Turner et al., 2015; Wäger & Hischier, 2015; Gu et al., 2017; Van Eygen et al., 2018b). Separately collected waste fractions (Perugini et al., 2005; Achilias et al., 2007; Turner et al., 2015; Van Eygen et al., 2018a), post-industrial plastic waste (Huysman et al., 2017), and post-consumer electronic waste (Achilias et al., 2009; Wäger et al., 2011; Wäger & Hischier, 2015) were assessed. Mechanical plastics recycling and re-granulate performance was extensively researched (Chen et al., 2011; Turner et al., 2015; Gu et al., 2016a,b; 2017; Van Eygen et al., 2018a,b). Few works considered chemical recycling and only recent works address mixed post-consumer packaging waste (e.g., Perugini et al., 2005; Achilias et al., 2007,2009; Lazarevic et al., 2010; BKV & Plastics Europe, 2019; Bergsma, 2019a,b; Meys et al., 2020; Russ et al., 2020). However, there is not enough pyrolysis data for other use cases than mixed household waste (Vogel et al., 2020).

Some works assess national waste management systems or compliance to future regulations (Chen et al., 2011; Van Eygen et al., 2018a,b; Bergsma, 2019a,b). Van Eygen et al. (2018a,b) assessed single-polymer and mixed-polymer recycling of Austrian plastic packaging waste. They highlight the importance of high-quality single-polymer recycling and lower environmental benefits of mixed-polymer recycling compared to the status quo. Chen et al. (2011) assessed (LCA) plastic waste recycling and energy-recovery technologies versus landfilling in China and found highest GHG reductions in low-grade plastics production from mechanical recycling and highest fossil fuel-savings in refuse-derived fuel (RDF) production and combustion. Huysman et al. (2017) developed a quality indicator to measure circular economy performance of post-industrial plastic waste.

Perugini et al. (2005) assessed and compared mechanical and chemical recycling (low-temperature fluidized bed pyrolysis and high-pressure hydrogenation) of plastic containers and highlighted the good environmental performance of coupling feedstock and mechanical recycling. Achilias et al. (2007) assessed chemical recycling (dissolution/reprecipitation and catalytic pyrolysis on laboratory fixed bed reactors) of single-polymer model plastics, commercial plastics, and plastic wastes and received a polymer recovery of >90%. BKV and Plastics Europe (2019) analyzed the technology readiness of chemical recycling processes (pyrolysis, gasification) of plastic waste focusing on data availability, necessary pretreatments, and economic competitiveness. Bergsma (2019a,b) assessed potential material inputs for chemical recycling from unrecycled Dutch waste (e.g., recycling losses, PET-trays, mixed plastics) and compared chemical (pyrolysis, hydropyrolysis, gasification) and mechanical recycling. Meys et al. (2020) developed a theoretical chemical recycling model to identify the best possible performance of five environmental impacts in 75 scenarios compared to existing recycling processes. Russ et al. (2020) assessed pyrolysis of mixed plastic waste (LCA) to understand favorable conditions and influencing factors.

Sensor-based sorting of plastic wastes is not new (Allen et al., 1999; Feldhoff et al., 1997; Murase & Sato, 1999; Scott, 1995; Wan et al., 1994). Recent publications address hyperspectral (Serranti et al., 2011, 2015; Habich & Beel, 2014) or black plastics sorting (Huang et al., 2017). Today, near-infrared sensors are widespread, but can neither separate HDPE and LDPE nor extract colored/black plastics.

[1] Tons refer to metric tons throughout the article.

Literature gaps exist regarding the assessment of single-/dual-commingled waste plastics from collection systems (Turner et al., 2015), real-waste fractions with minerals, metals, and other contaminations, and new recycling technologies (e.g., pyrolysis). Notably, there is a lack of high-quality Life Cycle Inventory (LCI) data on material reprocessing/recycling and more case-specific LCA studies on recycled plastics are highly desirable (Turner et al., 2015; Gu et al., 2017). Most works investigate new sorting and recycling technologies under lab conditions only (e.g., Achilias et al., 2007). Assessments and primary data of pilot or industrial-scale chemical recycling plants (pyrolysis, gasification, solvolysis) are often missing and transparent LCI data is missing almost entirely (Gu et al., 2017), except for recent works (BKV & Plastics Europe, 2019; Meys et al., 2020). Also, a comparative study of sorting and recycling technologies for mixed plastic packaging waste is missing.

This study develops a method to assess primary plastics production (see Annex A3 of Supporting Information S2), post-consumer plastic packaging waste sorting (Section 2.2.1), and recycling (mechanical, chemical, and combined) (Section 2.3) concerning costs, carbon efficiency, cumulative energy demand (CED), and global warming potential (GWP) (Section 2.4) for polyethylene (PE), polypropylene (PP), polyvinylchloride (PVC), and general purpose polystyrene (GPPS). We assess packaging waste provided by the German packaging collection systems (see Section 2.3 for its composition). This study differs from previous studies in three main aspects:

1. It provides a transparent and comparative techno-economic and environmental assessment of primary plastic production and different plastic recycling paths comprising GWP, CED, carbon efficiency, and costs. The underlying data is provided and includes details for a theoretical sensor-based sorting plant. Therefore, we address the lacking high-quality LCI data on bespoke processes.
2. It develops a techno-economic assessment of recycling paths on industrial scale based on literature data.
3. It considers a real waste composition from separately collected mixed-polymer lightweight packaging (LWP) waste instead of mono-fractions (Perugini et al., 2005), contaminated mono-fractions (Meys et al., 2020), or specified mixed plastic fractions (Russ et al., 2020). The assessment of real mixed plastic waste is complex since multiple materials' treatments (of minerals, metals, fine fraction, or organic material) have to be allocated.

Our approach enables producers and customers of plastic packaging to integrate GWP, CED, cost, and carbon efficiency into multi-objective procurement and investment decision-making. This is particularly valuable when facing CO_2e prices or tax, stricter regulation and volatile landfilling, incineration and co-combustion prices. This study also supports policymakers regarding the promotion and regulation of favorable recycling options for mixed plastics packaging waste.

2 | METHODOLOGY

2.1 | Goal and scope

This study combined mass flow analysis (MFA) with LCA data of the considered recycling technologies (mechanical, chemical, and combined recycling) of mixed LWP waste. Mass and energy balances for the assessment of sensor-based sorting, mechanical recycling, chemical (feedstock), and combined recycling of mixed LWP were established in a partial LCA (attributional approach). The model data is literature-based and process-oriented where data was not available, that is, value chains were disaggregated into relevant unit processes that are assessed (i) based on simulation or (ii) measured data where physic-chemical models do not exist.

The assessment covers a mechanical pretreatment step to separate metals and minerals from refuse-derived fuel (RDF) as recycling feedstock, the state-of-the-art re-granulate production from sorted plastics for medium- and low-quality plastic products (like waste bins, formwork panels, or park benches), and the chemical feedstock production via chemical recycling (here: pyrolysis). Other chemical recycling processes are not assessed (see Section 4.2). The mass balances (including amounts and material types) and the impact assessment of CED, GWP, carbon efficiency, and cost of each recycling path were calculated (Section 3). Then, the different recycling paths were compared including compensation for substituting primary material and for energy co-generation through incineration of by-products.

The functional unit was 1 kg of mixed LWP waste that is collected separately and recycled in Germany. In the assessment, input waste was not associated with any GWP, CED, or cost.

Mechanical recycling includes mechanical pretreatment, sensor-based sorting and regranulation[2]. The main product is plastic re-granulate, often with a reduced quality compared to the original plastics (input) and with few possible recycling cycles (EMAF, 2016). Main fields of application are road construction (100–150 kilotons/a), window and door profiles (100–150 kilotons/a), pipes (50–70 kilotons/a), landscaping, agriculture, electronics, packaging, and plastic sheets (Conversio, 2018, p. 67ff.).

[2] Costs of all regranulation substeps (post-sorting, cleaning/washing, melting) are considered. Regarding CED and GWP, only melting is considered due to its dominance and missing data for the other substeps.

Chemical recycling processes mechanically pre-treated waste to chemical intermediates. In both recycling paths, the same mechanical pretreatment process was applied that is suitable to provide feedstock for maximum sensor-based plastics sorting yield and that is also standard for producing RDF from mixed waste (Stapf et al., 2019a). In the combined recycling assessment, the recovered pure plastics from sensor-based sorting were mechanically regranulated, while the sorting residues were chemically recycled.

For all recycling paths, all economic and environmental burdens and rewards associated with the process steps, the by-products and their handling were assigned to the respective main products as the treatment of plastic waste is the focus of this study. The substitution of primary plastic material was rewarded in all paths by multiplying the amount of produced re-granulate or virgin plastic by the impact of the substituted primary plastic (Section 2.4).

The system boundaries (see Table S2-1 of Supporting Information S2) exclude energy inputs for plant construction and machine production, the plastics use phase, the transportation[3], and cleaning/washing[4] processes. Likewise, transportation emissions are excluded, as they are expected to be relatively low (Chen et al., 2011). In the following subsections, technologies and recycling paths are assessed in detail. Different scenarios (Section 2.5) and sensitivities (Section 3.4) demonstrate the variability of the results.

2.2 | Technology assessment

2.2.1 | Mechanical recycling and sensor-based sorting assessment

Prior to sensor-based sorting or pyrolysis, LWP waste is sorted in a mechanical pretreatment step using conventional technologies, including comminution, classification, sifting, and metal separation and it is processed to RDF (Stapf et al., 2019a). Then, the produced RDF is sorted in a sensor-based sorting plant to separate fractions with distinct qualities for further recycling and processing. Plastic types often cannot be separated by conventional sorting technologies, and mixed fractions cannot be recycled to high-quality products as small ratios of cross-contamination can lead to unusable batches (Masoumi et al., 2012; EMAF, 2016). Since data on sensor-based sorting is not publicly available, we assessed different sorting technologies and modelled a theoretical sorting plant (see Annex A2 of Supporting Information S2).

Sensor-based sorting produces sorted plastics and sorting residues. Sorted plastics are the input for a regranulation process, while sorting residues are valuable fuel for different thermal recycling paths due to their high calorific value. As the modelled mechanical pretreatment is more refined than the usual mechanical steps in mechanical-biological treatment plants, further treatments of sorting residues are omitted. Burdens associated with incineration of the sorting residues and rewards for substituting other fuels were included and allocated to the sensor-based sorted plastics. Within the scenario analysis, sorting residues were allocated to different thermal recycling paths (Section 2.5).

The sorted plastic types are regranulated in an extruder. Additional sorting processes and associated mass reductions at the re-granulator were considered in the sorting plant yield. A separate assessment was not carried out.

2.2.2 | Chemical recycling

Chemical recycling processes mechanically pretreated waste to monomers or other chemicals (Conversio, 2018) that can be used as secondary feedstock in plastics' or other chemicals' production. In the assessed process, the produced RDF is fed into an integrated pyrolysis unit producing two useful co-products (pyrolysis liquid and pyrolysis gas), and a solid fraction (by-product). The pyrolysis liquid consists of oily and watery parts that are separated by condensation for further use. Gas and liquid fractions mainly consist of hydrocarbons; the solid fraction consists of minerals, char, and hydrocarbons. Pyrolysis oil replaces naphtha as the feedstock of the hot part of the steam cracking process, where its components are thermally cracked under the presence of steam. In contrast, the pyrolysis gas enters only the cold downstream part of the steam cracking process for separation of the individual gaseous components. The solid fraction from the pyrolysis is combusted to provide its process heat and to condition it for deposit. The excess heat is sold for district heating purposes and is considered in all impact categories. In the steam cracking process, monomers are produced that are subsequently polymerized. The produced plastics have virgin quality and are rewarded with primary plastics' substitution.

2.3 | Inventory data of technologies and other data

The assessment of the mechanical, chemical, and combined recycling paths was demonstrated for LWP waste in a case study for Germany. The separately collected German LWP sums up to ca. 2.5 million tons/year. Its average composition is shown in Figure S2–5 of Supporting Information S2. The environmental impact factors (Table S2–5 of Supporting Information S2) specific to Germany were used to calculate the GWP

[3] Transportation between production gate and customer, post-consumer collection and transport to sorting, recycling, re-processing or incineration plants should be included in future research.
[4] Contamination usually does not impair the sorting result (Safavi et al., 2010).

[CO_2e/kg input] and CED [MJ/kg input] per process step based on its energy demand. Together with energy efficiencies of the combustion systems, they were also used to calculate associated rewards and burdens. Uncertainties and their impacts on the results are considered in a sensitivity analysis (see Annex A7 of Supporting Information S2). For the calculation of the substitution effects of primary material, LCAs based on the "cradle-to-gate" approach for primary plastic production in Europe were used (PlasticsEurope, 2018b) (see Annex A3 of Supporting Information S2).

2.3.1 | Mechanical pretreatment

LWP waste is sorted with conventional technologies and can be separated into metals (1.2% ferrous and 1.8% non-ferrous metals), heavy components (10%), a low-calorific fine components (20%), and water (2%) (Stapf et al., 2019a). The variable sorting costs include the gate fee of incoming waste, compensation for metal recycling, landfilling of heavy components, and the gate fees for energetic utilization (Annex A4 of Supporting Information S2). The fixed costs for a waste pretreatment plant with a capacity of 100,000 tons/year and operating 8,000 h/year added up to 2.07 million €/year.

The environmental impact was derived from the electrical energy demand of 0.40 MJ/kg treated waste. This was measured at an exemplary treatment process and is higher than literature values (Bilitewski et al., 2018) due to higher processing demands for the RDF production (see Annex A9 of Supporting Information S2). The energy demand of the mechanical pretreatment had a net[5] impact of 1.84 MJ/kg (CED) and 0.32 kg CO_2e/kg (GWP). We reward electricity and heat production from incineration of the low-calorific fine components based on their calorific value, the emission-factor for household waste, and the efficiency of waste incineration plants. Ferrous metals replace primary metal in cast iron production; non-ferrous metals substitute primary raw aluminum (as a non-ferrous representative).

2.3.2 | Mechanical recycling and sensor-based sorting

A theoretical sorting plant (Figure 1) was assessed to quantify its impact on the mechanical recycling path regarding costs, CED, and GWP. The combination of sorting technologies results in operational costs of 31.84 €/(ton input), in 0.09 MJ/(kg input) (CED), and in 0.006 kg CO_2e/(kg input) (GWP) (Annexes A2 and A4 of Supporting Information S2).

In mechanical recycling, sensor-based sorting residues are incinerated[6]. Thus, incineration gate fees and emissions, as well as compensation for produced heat and electricity were included. In the baseline scenario, all sorting residues were incinerated in RDF power plants and municipal solid waste incineration (MSWI) plants (Section 3.1). Other thermal recycling paths were calculated and discussed in further scenarios (Sections 2.5, 3.2). The efficiency rates and environmental impact factors used for the impact assessment are displayed in Table S2–5 of Supporting Information S2 and available as data tables in Supporting Information S3.

In the regranulation, a carbon efficiency of 98% was assumed due to additional sorting steps and material losses (Dehoust et al., 2016). For different carbon efficiencies see Section 2.5. As regranulation costs are not available in literature, the cumulated anterior processing costs were subtracted from the available re-granulates' market prices resulting in 0.4 €/kg[7]. Additionally, we assumed an energy consumption of the regranulation of 0.21 kWh per kg re-granulate that is ranging between 0.18 and 0.24 kWh/kg specified by Großmann (2011).

2.3.3 | Chemical recycling

Adequate experimental data of reasonable quality for scalable mixed plastics pyrolysis is only provided by Andreas et al. (1981). They pyrolyzed a waste composition similar to LWP sorting residues in a rotary kiln reactor at 650°C and yielded gas (38 wt.-%), liquids (30 wt.-%) and solids (32 wt.-%). Applying this data, we integrate the pyrolysis process into the process route given in Figure 2. The underlying data of Figure 2 and all following figures can be found in Supporting Information S1. RDF from mechanical pretreatment is fed into the integrated pyrolysis unit and its impacts are allocated to the desired pyrolysis products (gas, oil) by mass. Pyrolysis oil replaces naphtha as steam cracking feedstock, while the pyrolysis gas components are separated in the cold part of the steam cracking process. The integrated pyrolysis process consists of the pyrolysis reactor, consecutive oil and gas upgrading steps, the solids' incineration and heat recovery system, solids' transport and mixing, and necessary auxiliary units. It is designed to separate all additives, pollutants, and impurities from pyrolysis oil and gas to meet the feedstock specifications of downstream chemical processing of virgin quality material. Process simulation determines both GWP and CED of the pyrolysis process of German LWP waste with 0.993 kg CO_2e/kg (GWP) and 14.99 MJ/kg (CED).

[5] Net environmental impacts, carbon efficiencies and costs include rewards; their gross values only include burdens.
[6] In chemical recycling, the sensor-based sorting residues are chemically recycled.
[7] We compare bale and re-granulate prices to assess regranulation costs, since the regranulation is the primary process executed between these qualities. Price differences are 0.33 €/kg (PE-HD), 0.36 €/kg (PE-LD), 0.54 €/kg (PP), and 0.24 €/kg (PET) (Plasticker 2019) leading to 0.4 €/kg on average.

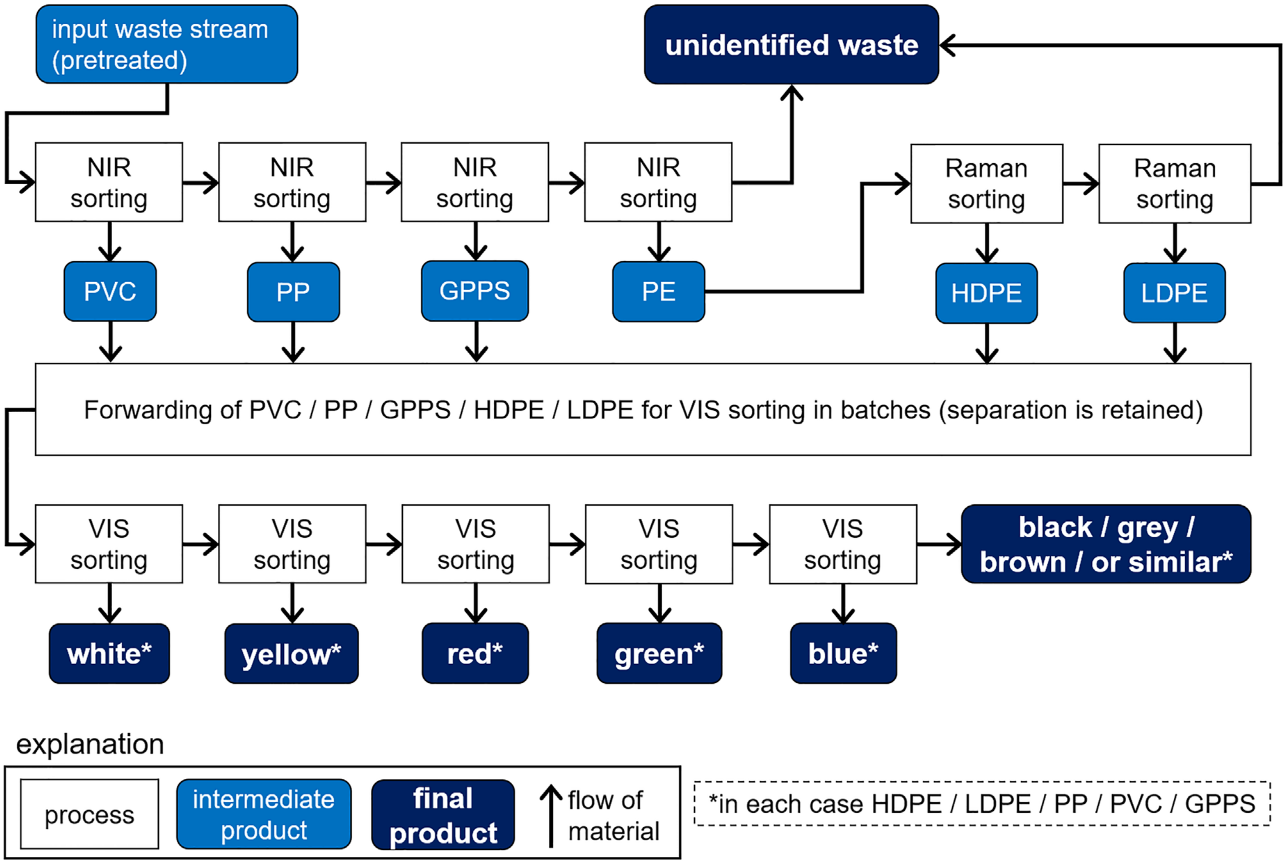


FIGURE 1 Material flow in a theoretical integrated sensor-based sorting plant, designed for maximum color-wise material recovery of plastic fractions PE, PP, PVC, and GPPS, respectively. Sorting residues are unidentified waste and non-sorted plastic colors. The input waste stream is conventionally sorted and mechanically pretreated

The solid fraction of the pyrolysis unit is combusted; the minerals are deposited. Excess heat is sold for district heating and considered in all assessment categories; it was credited with 0.155 kg CO_2e/kg (GWP) and with 2.63 MJ/kg (CED).

Following BKV and Plastics Europe (2019), the integrated process scheme was developed for an 8.1 ton/h pyrolysis process operating 8,000 h/year. The scale-up is derived from the mass and energy balances from Andreas et al. (1981) and operational and capital investment cost were calculated. The cost assessment is based on the investment of a rotary kiln waste treatment plant of similar complexity, which was scaled down to the considered pyrolysis plant size according to BKV and Plastics Europe (2019). Fixed costs added up to 14.2 million €/year for this plant size which fits to the mechanical pretreatment process specified in Section 2.3.1. Variable costs are 8.21 €/ton of the main product and include electricity of the combustion air-compressor (4 bar at 5 €/1,000 m^3), landfill fees for combustion residues (100 €/ton), and the compensations generated from district heating (0.03 €/kWh). Our calculations led to total pyrolysis costs of around 320 €/ton of mixed plastic waste. For steam cracking, existing data on primary ethylene and propylene production is used (PlasticsEurope 2012a). Therefore, no further detailed steam cracking assessment is made; however, the impact of the inputs is altered.

2.4 | Impact assessment

GWP[8], CED[9], cost, and carbon efficiency are assessed. The carbon efficiency is the recycling rate in terms of the ratio of total carbon mass of the desired products divided by the total carbon mass of the feed(s) per conversion or separation step or per recycling path. Finally, the recycling paths are compared concerning their product costs.

Each process step was assessed individually; the recycling paths sum up all process steps along the path. Also, the above mentioned rewards and burdens were considered. Downstream process steps include the impact of upstream processes of the value chain. Equations (1)–(3) show the

[8] Here, GWP100 is assessed as defined in the Kyoto Protocol (IPCC 2013).
[9] CED is the "total quantity of primary energy which is necessary to produce, use and dispose of a product" (VDI 2012).

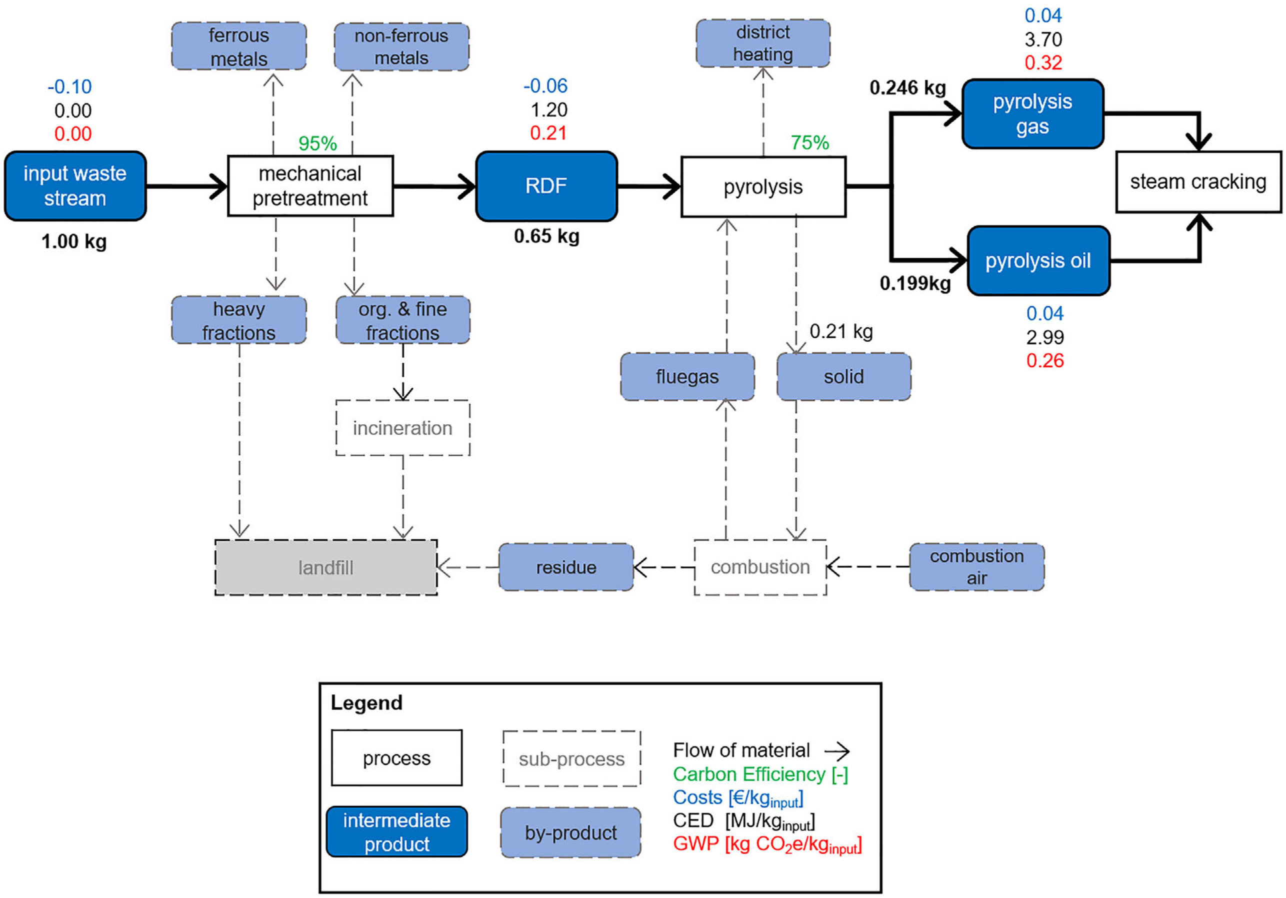


FIGURE 2 Detailed pyrolysis route of mixed plastics per 1 kg of input waste. Refuse-derived fuel (RDF) is the intermediate product of mechanical pretreatment. Underlying data used to create this figure can be found in Supporting Information S1

exemplary calculation of CED, GWP, and cost of mechanical pretreatment to produce RDF:

$$\begin{aligned} CED_{RDF} = \; & CED_{input\ waste} + CED_{mechanical\ pretreatment} + CED_{landfilling\ of\ heavy\ content} + CED_{incineration\ of\ fine\ fraction} - \\ & CED_{compensation\ for\ electricity\ generation\ from\ incineration} - CED_{compensation\ for\ heat\ generation\ from\ incineration} - \\ & CED_{compensation\ for\ recycling\ of\ ferrous\ metals} - CED_{compensation\ for\ recycling\ of\ non-ferrous\ metals} \end{aligned} \tag{1}$$

$$\begin{aligned} & GWP_{RDF} = \\ & GWP_{input\ waste} + GWP_{mechanical\ pretreatment} + GWP_{landfilling\ of\ heavy\ content} + GWP_{incineration\ of\ fine\ fraction} - \\ & GWP_{compensation\ for\ electricity\ generation\ from\ incineration} - GWP_{compensation\ for\ heat\ generation\ from\ incineration} - \\ & GWP_{compensation\ for\ recycling\ of\ ferrous\ metals} - GWP_{compensation\ for\ recycling\ of\ non-ferrous\ metals} \end{aligned} \tag{2}$$

$$\begin{aligned} & costs_{RDF} = \\ & costs_{mechanical\ pretreatment\ plant} + costs_{electricity\ for\ mechanical\ pretreatment} + costs_{landfilling\ of\ heavy\ contents} + \\ & costs_{incineration\ of\ fine\ fraction\ (gate\ fee)} - revenue_{ferrous\ metals} - revenue_{non-ferrous\ metals} \end{aligned} \tag{3}$$

The carbon efficiency is based on stoichiometric mass balances. Only in the mechanical pretreatment, we assume 5% carbon in the fine fraction so that it has to be combusted (Stapf et al., 2019a).

2.5 | Scenario definition

Multiple scenarios were developed on varying sensor-based sorting yields and different incineration paths for sorting residues to analyze the influence of underlying data and assumptions (Table 1). Also, a combination of mechanical and chemical recycling is considered. Further assumptions and parameters are varied in a sensitivity analysis (Section 3.4, Annex A7 of Supporting Information S2).

TABLE 1 Overview of scenarios

Scenario no.	Scenario description	Sorting yield	Incineration paths of sorting residues
1.1.1	Mechanical recycling	42%	100% MSWI plant
1.1.2	Mechanical recycling	42%	25% MSWI plant, 75% RDF combustion plant
1.1.3	Mechanical recycling	42%	18% MSWI plant, 58% RDF combustion plant, 13% cement plant, 11% coal-powered plant
1.2.1	Mechanical recycling	22%	100% MSWI plant
1.2.2	Mechanical recycling	22%	25% MSWI plant, 75% RDF combustion plant
1.2.3	Mechanical recycling	22%	18% MSWI plant, 58% RDF combustion plant, 13% cement plant, 11% coal-powered plant
2	Chemical recycling	–	–
3.1	Combined recycling	42%	–
3.2	Combined recycling	22%	–

Note. "–" indicates parameter does not apply.

Christiani and Beckamp (2020) state that 32% of the LWP provided by the German collection systems are high-grade recyclable plastics. After conventional sorting, 35% of the mass is discarded and the relative amount of high-grade recyclable plastics rises to 49%. A buffer of 6% for non-identifiable plastics leads to a resulting sensor-based sorting yield of 43% for mechanical recycling. This buffer results from the share of other polyolefins in the high-quality recyclable plastic composition (Figure S2–5, Supporting Information S2). Furthermore, additional material losses between 2% and 50% for sorting steps at the regranulation plant are assigned to the sorting plant (Dehoust et al., 2016). The best and worst-case scenarios for mechanical recycling the sensor-based sorting yield are multiplied by the extremes of additional regranulation losses. Thus, we distinguished a sorting yield of 42% (Scenario 1.1) and 22% (Scenario 1.2). Due to missing data, we assume sorting yields only per polymer type that is then further processed. So, after the sensor-based sorting, we assess a mono stream of one specific polymer type that is then processed in a regranulation facility. In this step, we do not consider a mix of polymers.

Sorting residues are incinerated or co-combusted in cement kilns, coal-fired or RDF combustion plants or MSWI plants (in decreasing efficiency order) depending on its calorific value and chlorine content.

Scenario 1.1.1 reflects the worst case where sorting residues are incinerated in MSWI plants. Scenario 1.2.1 differs from it in a lower sorting yield and thus a higher sorting residue incineration. Scenario 1.1.2 is the baseline scenario with co-combustion of sorting residues in efficient RDF combustion plants (75%) and MSWI plants (25%). This reflects the current practice in Germany and other European countries (Ketelsen & Kanningen, 2016; Van Eygen et al., 2018b; UBA, 2018c; Russ et al., 2020). Scenario 1.2.2 has a lower sorting yield than the baseline scenario. Scenarios 1.1.3 and 1.2.3 include all four thermal recycling paths following current German combustion shares for RDF (Ketelsen & Kanningen, 2016) assuming that the sorting residues undergo further treatment (e.g., sensor-based sorting steps) to separate PVC and meet the strict chlorine limits for co-combustion in coal and cement power plants (5–15 g/kg in coal-powered plants and <10 g/kg in cement kilns) (UBA, 2015).

Scenario 2 examines the chemical recycling (Section 2.3.3). As the variation of the process parameters would result in unknown pyrolysis gas, oil and solid yields and compositions, sub-scenarios can neither be defined nor analyzed. Scenarios 3.1 and 3.2 combine mechanical and chemical recycling and examine high and low sensor-based sorting yields. Here, sorting residues are fed into the chemical recycling process[10].

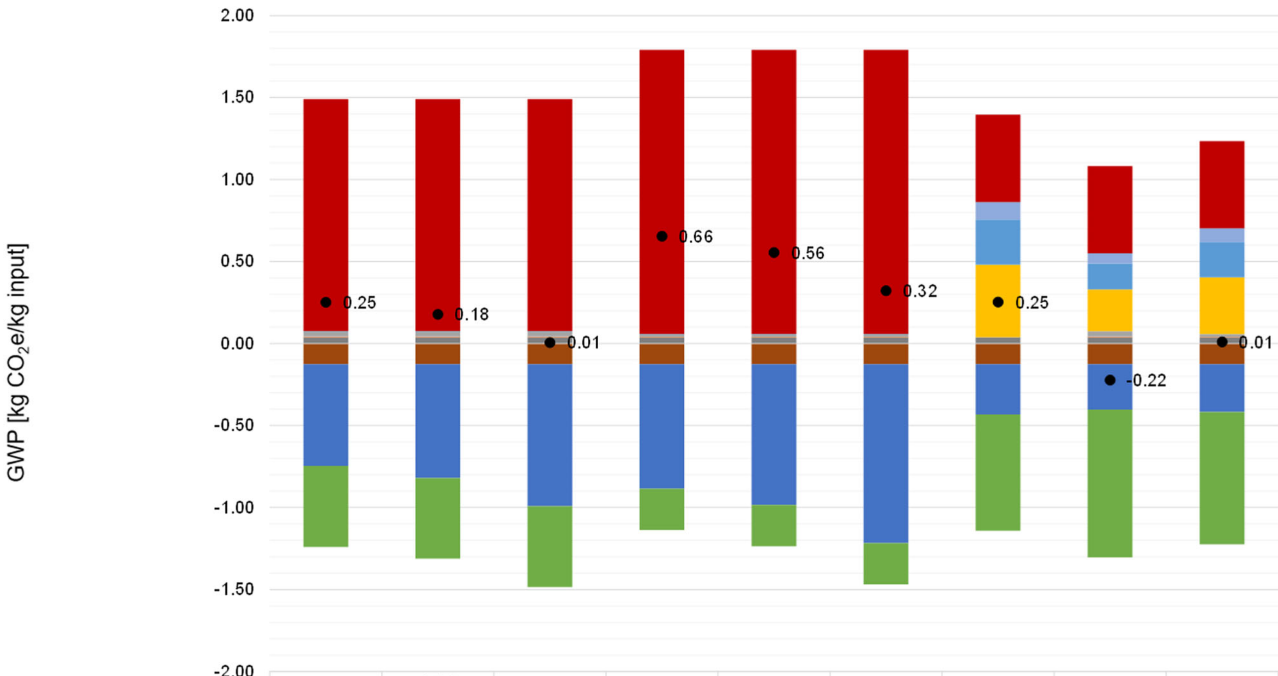
3 RESULTS

The impacts of the recycling paths were calculated based on the collected data and the defined scenarios in a case study for Germany. The results are illustrated for HDPE, but are also available for LDPE, PP, PVC, and GPPS (Annex A6 of Supporting Information S2). The impacts are described per kilogram of waste input (Figure S2–6 of Supporting Information S2).

3.1 Baseline scenario

The mechanical recycling results in gross sorting and regranulation costs of 0.10 €/kg and induces gross values of 0.67 kg CO_2e/kg (GWP) and 3.83 MJ/kg (CED). The net values take full (100%) substitution of primary material into account (Figure S2-1 to Figure S2-4 of Supporting Information S2) and result in costs of −0.16 €/kg (=revenues), 0.18 CO_2e/kg (GWP), and −18.14 MJ/kg (CED). The differences between net and gross values result from avoided costs and avoided energy for primary material production and highlight the calculatory impact of primary material

[10] Pyrolysis inputs differ in chemical recycling (scenario2) (RDF) and combined recycling (scenarios3.1 and 3.2) (sorting residues). Therefore, the pyrolysis products could be different.



	1.1.1	1.1.2 Baseline	1.1.3	1.2.1	1.2.2	1.2.3	2	3.1	3.2
Substitution Primary Material	-0.49	-0.49	-0.49	-0.25	-0.25	-0.25	-0.71	-0.90	-0.81
Rewards Incineration	-0.62	-0.69	-0.87	-0.76	-0.86	-1.09	-0.31	-0.28	-0.29
Burdens Incineration	1.41	1.41	1.41	1.73	1.73	1.73	0.53	0.53	0.53
Rewards Metall Recycling	-0.13	-0.13	-0.13	-0.13	-0.13	-0.13	-0.13	-0.13	-0.13
Landfilling Heavy Components	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Polymer Synthesis	0.00	0.00	0.00	0.00	0.00	0.00	0.11	0.06	0.08
Steam Cracking	0.00	0.00	0.00	0.00	0.00	0.00	0.28	0.16	0.22
Pyrolysis	0.00	0.00	0.00	0.00	0.00	0.00	0.44	0.26	0.35
Regranulation	0.03	0.03	0.03	0.02	0.02	0.02	0.00	0.03	0.02
Sensor-Based Sorting	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Mechanical Pretreatment	0.04	0.04	0.04	0.04	0.04	0.04	0.04	0.04	0.04
Sum	0.25	0.18	0.01	0.66	0.56	0.32	0.25	-0.22	0.01

Scenario

FIGURE 3 Comparison of considered recycling processes and scenarios regarding their net GWP impact [kg CO_2e/kg input]. Assessment for 1 kg of input waste. In Scenarios 2, 3.1, and 3.2, "Burdens Incineration" results from the incineration of the fine fraction of the mechanical pretreatment. Underlying data used to create this figure can be found in Supporting Information S1

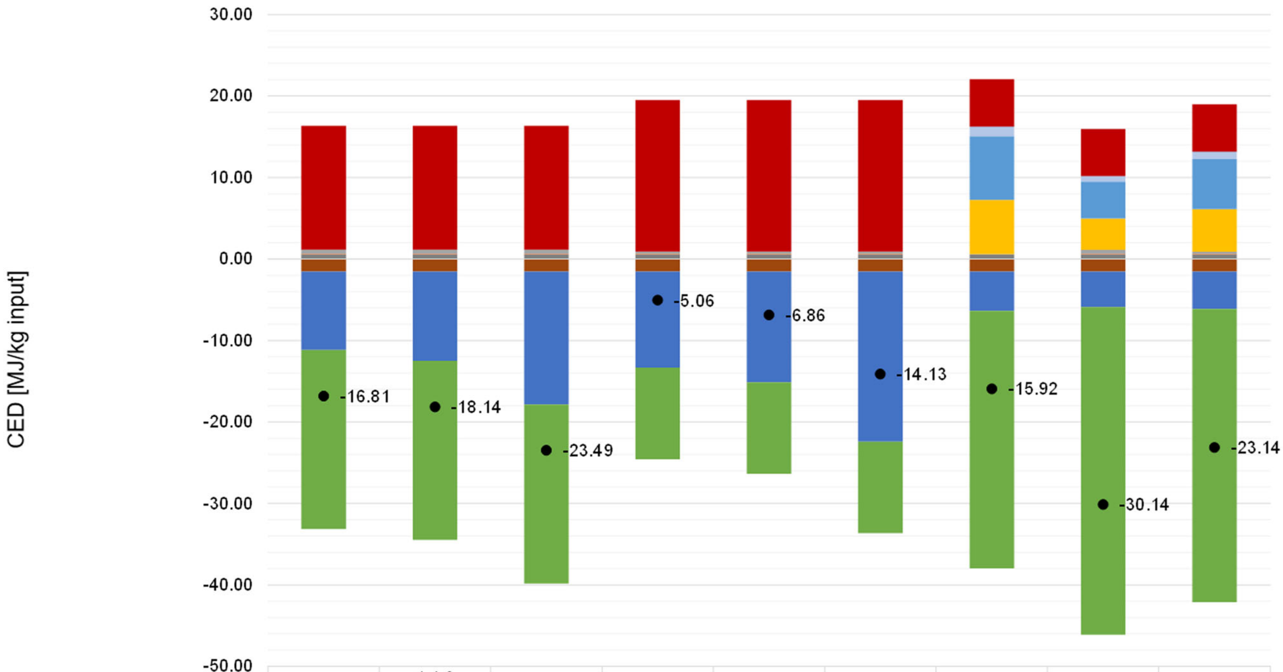
substitution on costs, GWP and CED. However, in this scenario, the substitution cannot compensate for the high GWP burdens that result mainly from incineration of sorting residues.

In the whole recycling path, incineration accounts for 95% of the GWP and 93% of the CED impact. Incineration of the sensor-based sorting residues accounts for around 60% of the CED and GWP. The regranulation process and its electrical energy demand have a high impact on CED and costs.

The results presume that the sensor-based sorting process' outcome is a single plastic fraction that is processed further (Section 2.5). We do not break down the sensor-based sorted plastic by type due to missing data regarding the plastic composition after the sorting.

3.2 | Other scenarios

The other scenarios of mechanical recycling consider different thermal recycling paths for sorting residues and higher material losses. The results (Figures 3 and 4) show that including additional industrial co-combustion (Scenario 1.1.3) leads to net reductions by 95% (GWP) and 30% (CED) compared to the baseline (Scenario 1.1.2). In the worst-case (Scenario 1.1.1), the net GWP increases by 15% and the net CED increases by 7%. This is also valid for the lower yield scenarios. It results from the higher efficiency of industrial co-combustion compared to MSWI and highlights the impact of the considered thermal recycling. Scenarios with efficient cement kilns and coal-powered plants particularly show better environmental performance than the baseline scenario.



	1.1.1	1.1.2 Baseline	1.1.3	1.2.1	1.2.2	1.2.3	2	3.1	3.2
Substitution Primary Material	-21.96	-21.96	-21.96	-11.22	-11.22	-11.22	-31.58	-40.22	-35.98
Rewards Incineration	-9.66	-10.98	-16.33	-11.80	-13.60	-20.87	-4.86	-4.36	-4.60
Burdens Incineration	15.21	15.21	15.21	18.59	18.59	18.59	5.80	5.80	5.80
Rewards Metall Recycling	-1.53	-1.53	-1.53	-1.53	-1.53	-1.53	-1.53	-1.53	-1.53
Landfilling Heavy Components	0.02	0.02	0.02	0.02	0.02	0.02	0.02	0.02	0.02
Polymer Synthesis	0.00	0.00	0.00	0.00	0.00	0.00	1.14	0.66	0.90
Steam Cracking	0.00	0.00	0.00	0.00	0.00	0.00	7.82	4.52	6.14
Pyrolysis	0.00	0.00	0.00	0.00	0.00	0.00	6.67	3.86	5.23
Regranulation	0.46	0.46	0.46	0.24	0.24	0.24	0.00	0.46	0.24
Sensor-Based Sorting	0.06	0.06	0.06	0.06	0.06	0.06	0.00	0.06	0.06
Mechanical Pretreatment	0.58	0.58	0.58	0.59	0.59	0.59	0.58	0.58	0.58
● Sum	-16.81	-18.14	-23.49	-5.06	-6.86	-14.13	-15.92	-30.14	-23.14

Scenario

FIGURE 4 Comparison of considered recycling processes and scenarios regarding their net CED impact [MJ/kg input]. Assessment for 1 kg of input waste. Underlying data used to create this figure can be found in Supporting Information S1

Higher material losses lead to higher GWP and CED impacts regardless of the thermal recycling mix. Additional sorting residues are incinerated, and less primary material is substituted and rewarded. Regarding costs, there are no differences between the incineration mixes due to the same assumed gate fees. However, additional material losses lead to rising costs of mechanical recycling due to additional gate fees for incineration. Also, the amount of produced re-granulate decreases and reduces the substitution effect.

Scenario 2 (chemical recycling) (Figure S2–7 of Supporting Information S2) results in −0.24 €/kg net sorting and reprocessing costs (=revenues) and induces −15.92 MJ/kg (CED) and 0.25 kg CO_2e/kg (GWP). Steam cracking has a significant impact on all impact categories, mainly because of its high energy demand. Gross values of Scenario 2 are 0.33 €/kg sorting and recycling costs, 15.66 MJ/kg (CED), and 0.96 kg CO_2e/kg (GWP). The high yield of recycled material leads to a high reward for primary material substitution.

The combined approach (Scenario 3.1) (Figure S2–8 of Supporting Information S2) results in net values of −0.29 €/kg sorting and reprocessing costs (=revenues), −30.14 MJ/kg (CED), and −0.22 kg CO_2e/kg (GWP) in a combined assessment of virgin plastics and re-granulate output. With higher material losses at the regranulation plant (Scenario 3.2), GWP and CED impacts increase because more sorting residues are chemically recycled. Costs also increase due to a lower total yield. In both scenarios, a high yield leads to high rewards for primary plastic substitution.

3.3 | Comparison of recycling processes and scenarios

The assessed recycling paths are compared for HDPE, LDPE, PP, PVC, and GPPS (Figure 6, and Annex A6 of Supporting Information S2). The study results of HDPE discussed in the following are representative for the other assessed plastic types (except PVC). The results for PVC are different, although the trend is consistent.

GWP impacts (Figure 3) of mechanical recycling are influenced by the incineration paths and their efficiencies (Section 3.2). More efficient industrial co-combustion reduces the impacts due to higher substitution rewards for generated heat and electricity. GWP is also influenced by material yield, as higher material losses lead to additional sorting residues that are incinerated, and less primary material is substituted.

Chemical recycling performs slightly worse compared to the baseline scenario for most fractions (except for GPPS, where it is significantly worse). Reduced incineration burdens compensate the GWP impact of the steam cracking and polymer synthesis as no sorting residues are incinerated. The non-existent incineration rewards from the combustion of sorting residues counterbalance higher rewards for substituting primary material. Even though chemical recycling performs slightly worse than mechanical recycling regarding GWP, differences in impurities, additives, and quality aspects of the final products are not included. Including these has a substantial effect on the recycling paths' performance (Section 3.4). In the combined approaches, lower material losses at the regranulation plants lead to considerably lower GWP and net GWP credits resulting from omitted incineration burdens from sorting residues and higher total yields due to additional virgin plastic produced by chemical recycling.

All assessed scenarios show CED savings (Figure 4). Within mechanical recycling, CED savings increase with a decrease in material losses due to decreasing incineration burdens and increasing primary material substitution rewards. CED savings of mechanical and chemical recycling are comparable. Although chemical recycling has a higher energy demand than mechanical recycling, this is compensated by higher rewards for substituting primary material. Thus, the CED advantageousness of specific mechanical and chemical recycling processes cannot be clearly stated, but depends on different factors, for example, sorting yield, incineration of sorting residues, and specific plant efficiencies (Sections 3.4 and 4). Combined recycling has the highest CED savings, combining low incineration burdens and high rewards for substituting primary material. With less yield in mechanical recycling, Scenarios 1.1.3 and 3.2 are comparable regarding GWP and CED, but with considerably higher revenues in Scenario 3.2.

For all considered plastics, mechanical recycling obtains net revenues (Figure 5). For HDPE, the revenues are 0.16 €/kg waste input decreasing to 0.08 €/kg with lower sorting yield. Chemical recycling has higher revenues (0.24 €/kg waste input) and the combined recycling yield maximum revenues (between 0.16 and 0.29 €/kg waste input depending on the plastics type [see Annex A6 of Supporting Information S2]). The reason for this is higher (virgin) product qualities and market prices, and a higher overall yield. The revenue amount depends on the re-granulate price, the substituted amount, cost of primary plastic production, and the specific polymer synthesis cost. Combined recycling shows the highest carbon efficiency (74%) in Scenario 3.1, outperforming chemical (59%) and mechanical recycling (20–40%) (Figure 6).

In Germany, 5.2 million tons of post-consumer plastic wastes waste from packaging, construction, vehicles, electrical and electronic equipment, household, agriculture, and others are collected annually (Conversio, 2018), including 3.8 million tons of the considered plastics. Depending on the material losses, between 0.8 (Scenario 1.2.1) and 1.5 (Scenario 1.1.1) million tons re-granulate per year could be kept in use. Assuming combined recycling, between 2.5 (Scenario 3.2) and 2.8 (Scenario 3.1) million tons/year could be recycled. Thus, 1–2 million tons/year could be kept in the economy additionally, instead of incinerating them. This would suffice to achieve both the EU and German plastics packaging recycling targets (Section 1). Moreover, the additional plastic yield from chemical recycling would be of virgin quality.

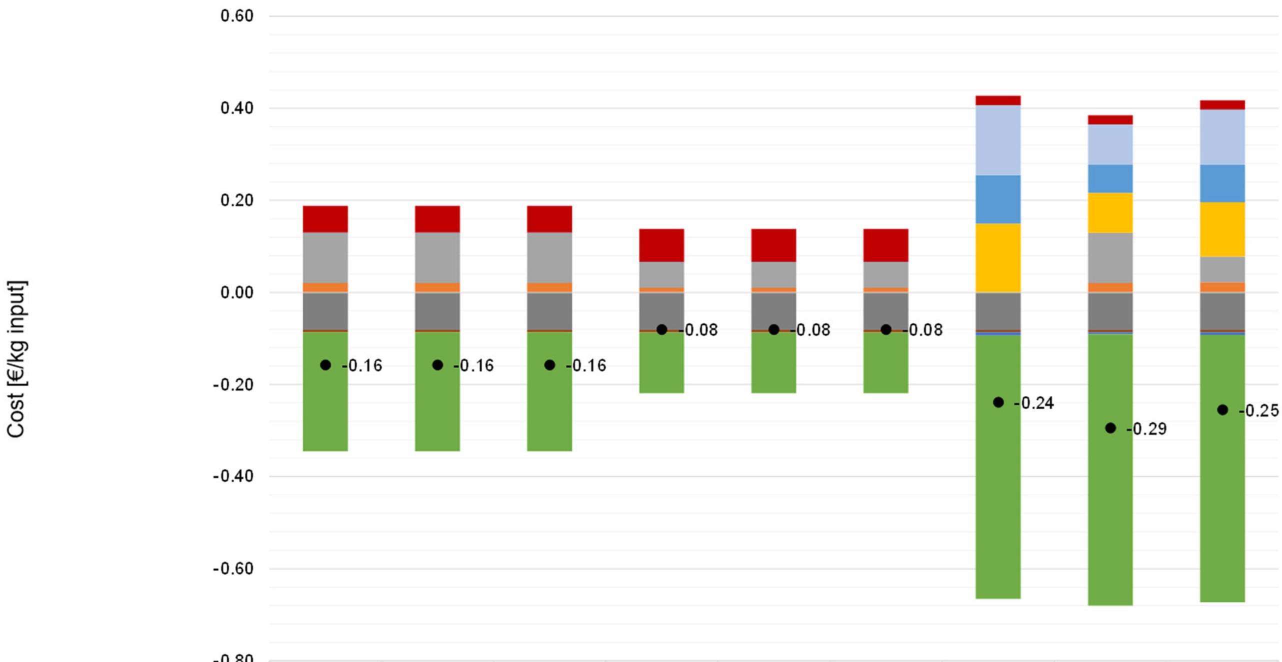
Today, around 1.9 million tons/year re-granulates are reclaimed from the total plastic waste in Germany (=ca. 30%) annually by mechanical recycling (Conversio, 2018). However, re-granulates often have lower quality because of limited material purity, degradation of material properties, or color impurities. Thus, they cannot be used for specific applications (e.g., food packing, medical products) (UBA, 2018b). This is discussed within the sensitivity analysis (Section 3.4). In other EU countries, mechanical recycling rates are lower than in Germany (PlasticsEurope, 2018a), and application of the assessed recycling technologies could realize significant GWP and CED reductions.

3.4 | Quality of re-granulate from mechanical recycling

Virgin material substitution ratio and organic contamination can significantly influence assessments of mechanical plastics recycling (Lazarevic et al., 2010; Turner et al., 2015; Gu et al., 2017; Van Eygen et al., 2018a). Thus, a substitution ratio of 1:0.81 is recommended by Rigamonti et al. (2009) and Turner et al. (2015) for mechanical plastics recycling to reflect lower material qualities of re-granulates. The baseline scenario does not consider reduced re-granulate qualities; these are subject to the following sensitivity analysis (see Annex A8 of Supporting Information S2 for details).

The sensitivity analysis shows that lower substitution ratios lead to significantly higher environmental impacts of mechanical recycling due to lower rewards for substituting primary material (Figure 7). If more re-granulate than virgin material is needed to produce a specific product, chemical recycling is more advantageous concerning CED and GWP. For substitution ratios of 1:0.4 and higher, the combined mechanical and chemical recycling is advantageous concerning CED. For GWP, combined recycling is advantageous independently of the substitution ratio. Below a substitution ratio of 1:0.4, chemical recycling leads to higher CED savings than combined recycling. Associated costs do not change due to assumed constant market prices and market clearance (=all re-granulates are sold for the given price).

The sensitivity analysis emphasizes the need to assess the re-granulate quality. In chemical recycling, all or most additives, pollutants and impurities are captured in the solid fraction and extracted from further processing. Thus, it is particularly useful to handle "difficult" plastic wastes.



	1.1.1	1.1.2 Baseline	1.1.3	1.2.1	1.2.2	1.2.3	2	3.1	3.2
Substitution Primary Material	-0.26	-0.26	-0.26	-0.13	-0.13	-0.13	-0.57	-0.59	-0.58
Rewards Incineration	0.00	0.00	0.00	0.00	0.00	0.00	-0.01	0.00	-0.01
Burdens Incineration	0.06	0.06	0.06	0.07	0.07	0.07	0.02	0.02	0.02
Rewards Metall Recycling	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Landfilling Heavy Components	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Polymer Synthesis	0.00	0.00	0.00	0.00	0.00	0.00	0.15	0.09	0.12
Steam Cracking	0.00	0.00	0.00	0.00	0.00	0.00	0.10	0.06	0.08
Pyrolysis	0.00	0.00	0.00	0.00	0.00	0.00	0.15	0.09	0.12
Regranulation	0.11	0.11	0.11	0.06	0.06	0.06	0.00	0.11	0.06
Sensor-Based Sorting	0.02	0.02	0.02	0.01	0.01	0.01	0.00	0.02	0.02
Mechanical Pretreatment	-0.08	-0.08	-0.08	-0.08	-0.08	-0.08	-0.08	-0.08	-0.08
Sum	-0.16	-0.16	-0.16	-0.08	-0.08	-0.08	-0.24	-0.29	-0.25

FIGURE 5 Comparison of considered recycling processes and scenarios regarding their net product costs [€/kg input]. Assessment for 1 kg of input waste. Underlying data used to create this figure can be found in Supporting Information S1

4 | DISCUSSION

Here, the study results are discussed regarding distinctions to existing studies (Section 4.1) and limitations of this study are specified (Section 4.2).

4.1 | Comparison of results with literature

For mixed plastics, Turner et al. (2015) calculated lower GWP impacts than this study. However, their data quality is rather poor and comparability can be questioned due to different assessed plastics (source-segregated plastics vs. real waste with mixed plastics, paper, metals, and other materials) and due to other national energy mixes. A higher carbon intensity of the energy mix results in higher rewards for energy generation, and better performance of mechanical recycling. Gu et al. (2017) found that mechanical PE recycling has only around 76% of the GWP impact of its virgin production. We can confirm these results for PE. In this study, the average GWP impact of all assessed plastics constitutes 70% of their virgin production.

Both Chen et al. (2011) and Perugini et al. (2005) highlight synergies between various technologies, especially a cascade utilization of mechanical and chemical recycling regarding a good environmental performance. These advantages of combining mechanical and chemical recycling can be confirmed. However, a direct comparison with Chen et al. (2011) is impossible, due to differences in the methodology.

In Bergsma (2019a), pyrolysis of the waste fraction led to lower GWP savings than mechanical recycling, which is consistent with our results. Meys et al. (2020) favor mechanical recycling over refinery feedstock production for the mono waste streams of PET, HDPE, LDPE, PP, and GPPS

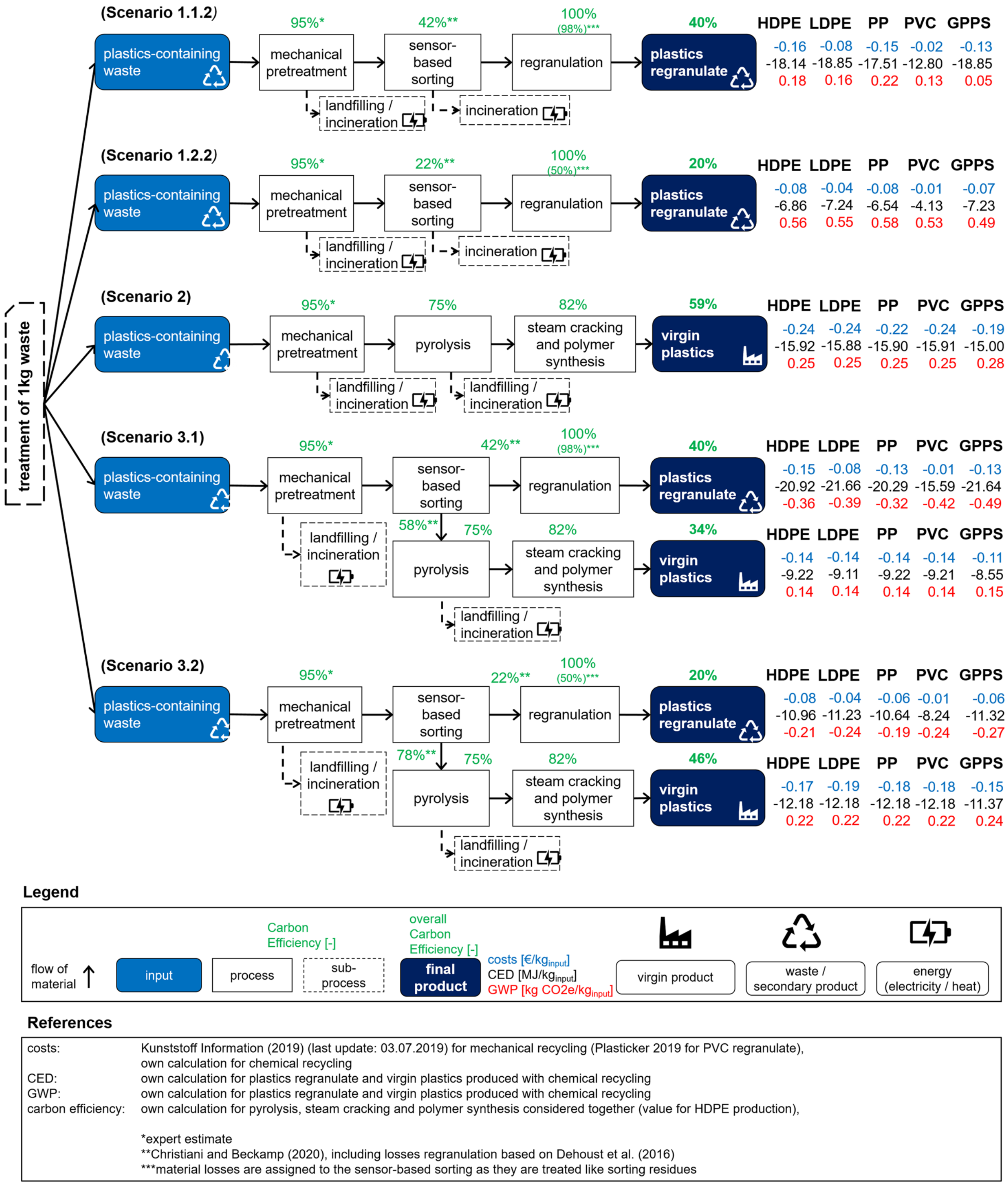
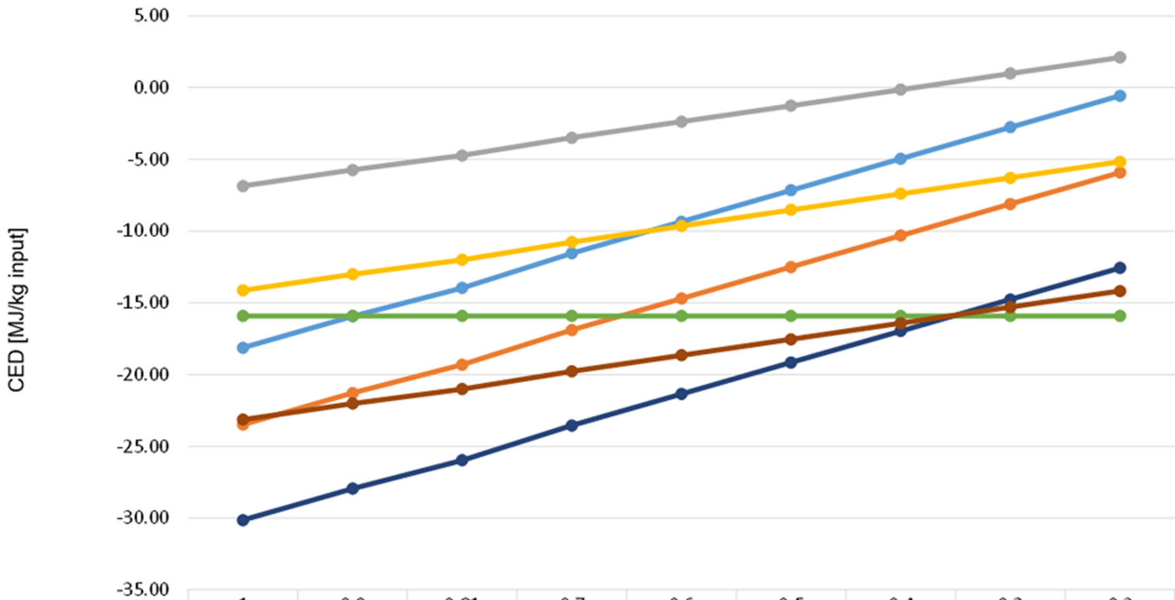
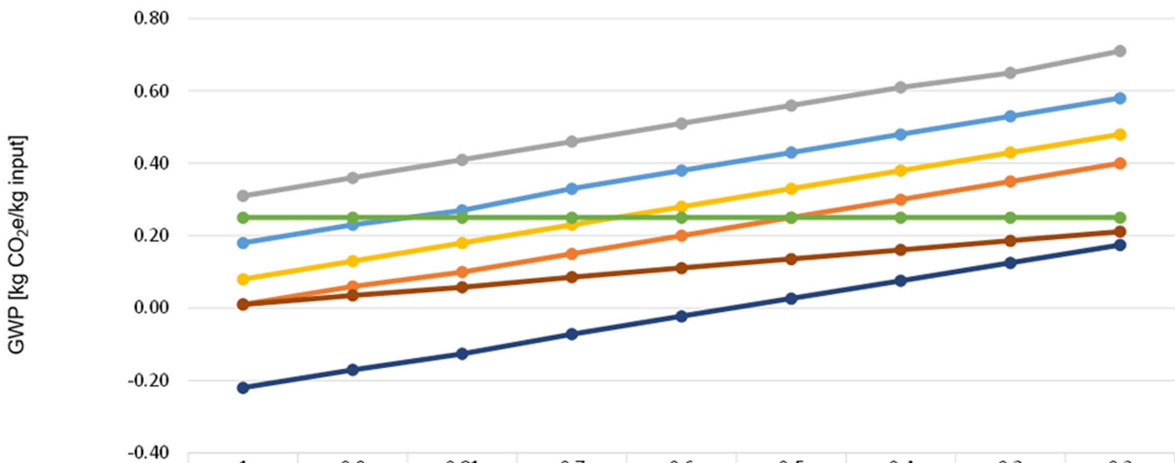


FIGURE 6 Comparison of different plastic recycling paths including rewards for substituted primary plastic production (assessed in Annex A3 of Supporting Information S2), power and heat gains, as well as burdens or rewards, respectively, for metal byproducts, incineration, and landfilling. Values are given for the treatment of 1 kg of waste (see FigureS2–5 of Supporting Information S2 for its composition). The percentages above the process boxes indicate the carbon efficiency of the respective process while those at the final products indicate the carbon efficiency of the whole recycling option. Underlying data used to create this figure can be found in Supporting Information S1



	1	0.9	0.81	0.7	0.6	0.5	0.4	0.3	0.2
Scenario 1.1.2	-18.14	-15.94	-13.96	-11.55	-9.35	-7.16	-4.96	-2.77	-0.57
Scenario 1.1.3	-23.49	-21.29	-19.31	-16.90	-14.70	-12.51	-10.31	-8.12	-5.92
Scenario 1.2.2	-6.86	-5.74	-4.73	-3.50	-2.38	-1.26	-0.14	0.98	2.10
Scenario 1.2.3	-14.13	-13.01	-12.00	-10.77	-9.65	-8.53	-7.41	-6.29	-5.17
Scenario 2	-15.92	-15.92	-15.92	-15.92	-15.92	-15.92	-15.92	-15.92	-15.92
Scenario 3.1	-30.14	-27.94	-25.97	-23.55	-21.36	-19.16	-16.97	-14.77	-12.57
Scenario 3.2	-23.14	-22.02	-21.01	-19.78	-18.66	-17.54	-16.42	-15.30	-14.18

Material Substitution Ratio (HDPE)



	1	0.9	0.81	0.7	0.6	0.5	0.4	0.3	0.2
Scenario 1.1.2	0.18	0.23	0.27	0.33	0.38	0.43	0.48	0.53	0.58
Scenario 1.1.3	0.01	0.06	0.10	0.15	0.20	0.25	0.30	0.35	0.40
Scenario 1.2.2	0.31	0.36	0.41	0.46	0.51	0.56	0.61	0.65	0.71
Scenario 1.2.3	0.08	0.13	0.18	0.23	0.28	0.33	0.38	0.43	0.48
Scenario 2	0.25	0.25	0.25	0.25	0.25	0.25	0.25	0.25	0.25
Scenario 3.1	-0.22	-0.17	-0.13	-0.07	-0.02	0.03	0.08	0.13	0.17
Scenario 3.2	0.01	0.04	0.06	0.09	0.11	0.14	0.16	0.19	0.21

Material Substitution Ratio (HDPE)

FIGURE 7 CED and GWP impact of the material substitution ratio of HDPE re-granulate and virgin material. The material substitution rate does not change for virgin material produced by chemical recycling (Scenario 2). Underlying data used to create this figure can be found in Supporting Information S1 regarding GWP impact, which is also consistent with our results. Russ et al. (2020) found no significant difference regarding GWP and CED impact between mechanical and chemical recycling (pyrolysis), but state a high impact of thermal recycling paths and carbon intensities of the electricity mixes on the results, due to their implications for the incineration rewards/burdens. This is consistent with our results. A combined recycling path is not considered in Russ et al. (2020).

Although similar results are found, the comparison with other studies is difficult as the considered plastic fractions differ. Particularly, different input waste leads to different pyrolysis results and impacts the assessment of chemical recycling.

4.2 | Shortcomings/limitations

Despite reasonable results, this study has limitations and the results face uncertainties due to (1) data and methodological limits, (2) model limits, or (3) assumptions:

(1) First, the measured LWP composition is quite old and might not reflect the current LWP composition. PET is excluded as it is separately collected and recycled in Germany. Furthermore, pyrolysis data is limited to specific pilot plants. The theoretical pyrolysis modelling (Section 2.3.3) based on this data requires future validation in experiments with current waste compositions. Potential changes in the plants' operational modes are not covered.

Polymer deterioration and additives (e.g., Gu et al., 2017) usually reduce material quality in mechanical recycling (Turner et al., 2015). Like Van Eygen et al. (2018a), our mechanical recycling assessment neither covers fillers/additives nor hazardous/interfering materials like phthalates (e.g., Pivnenko et al., 2016); only different substitution ratios are addressed in the sensitivity analysis (Section 3.4). Furthermore, pollutants or impurities enter the material flow (into and onto the plastics) due to contamination during use or by other wastes. Particularly, additives like bromine, chlorine, and phthalates need to be included in LCA datasets following EU regulation 1907/2006 (REACH). Compliance with this regulation is problematic for mechanical recycling, but less challenging for chemical recycling.

Furthermore, this study does not consider chemical recycling processes other than pyrolysis, such as gasification, or partial solvolytic recycling. It should be emphasized that pyrolysis results strongly depend on the applied technology, process temperature, and process pressure. We used data of rotary kiln pyrolysis technology; different pyrolysis technologies might lead to varying results. This is particularly important when comparing different studies. Also, we only assessed the pyrolysis of LWP waste and transferred the results to the pyrolysis of LWP waste sorting residues. A more detailed analysis of the examined scenario must be carried out, since the data of Andreas et al. (1981) does not allow data variation and scenario analysis. Providing this data is part of ongoing research by the authors to provide pyrolysis LCI data and enable sensitivity analysis of chemical recycling. Detailed investigations of the pyrolysis of different waste compositions are necessary and will be part of future research.

Chemical recycling technologies are less mature than mechanical recycling technologies. The lower technology readiness level of chemical recycling has not been considered in the sensitivity analysis of this study. Moreover, transportation was excluded, as its impact is relatively low; 900 km account for 10% of the treatments (Chen et al., 2011).

(2) Second, this study assesses a defined, representative LWP waste in Germany. However, it is a snapshot and thus not reflecting carbon emission-factors of other national energy mixes or its timely change in Germany (e.g., coal exit). Thus, the study results are specific for Germany. Furthermore, the study is static and does not cover dynamics, for example, of feedback loops, changing LWP waste compositions,[11] changing market demand, cost/price variability, trends, or changing substitution ratios over time.

(3) Third, assumptions introduce uncertainties that were partly covered by scenarios (Section 2.5) and sensitivity analyses (Section 3.4, Annex A7 of Supporting Information S2). However, assumptions might not reflect real market behavior or re-granulate and secondary plastics applications, but might impact the assessment (particularly the calculated rewards). Market effects have been discussed and considered in a reduced substitution ratio (Section 3.4). In reality, a mix of scenarios or sensitivities is likely, due to differing effectiveness of collection and recycling networks, plant efficiencies, and variable waste compositions.

A similar approach to assess plastics production, sorting, and recycling paths is not known to the authors. It can support decision-makers from academia, industry, and politics to make better-informed choices for optimal recycling and treatment strategies of mixed plastic wastes. Furthermore, it provides performance benchmarks for existing and new processes.

Finally, it should be noted that numerous limitations listed here are not unique to this study, but are ubiquitous in the LCA studies on waste recycling (e.g., Turner et al., 2015; Chen et al., 2011).

[11] Real waste might vary in composition and amount across regions, seasons, and due to other factors, e.g. design-for-recycling or regulations (e.g. EC 2019a,b, regulations 94/62/EG and 2018/852).

5 CONCLUSIONS

Mechanical, chemical, and a combined recycling of mixed plastics waste is transparently assessed, compared, and analyzed in different scenarios with respect to GWP, CED, carbon efficiency, and costs. Rewards and burdens are integrated for primary plastics substitution, metal by-products, incineration of non-recyclable residues, and landfilling of mineral non-combustible recycling residues, as well as GWP and CED gains from heat and power recovery. The developed assessment model is applied in a case study on separately collected LWP waste in Germany.

The results show that mechanically or chemically recycled plastics are advantageous compared to virgin plastics produced from fossil feedstock. Mechanical and chemical recycling perform similarly regarding GWP and CED, depending on sorting yield, thermal recycling paths of sorting residues and substitution ratio. Chemical recycling performs better than mechanical recycling concerning cost and carbon efficiency. Both chemical recycling and mechanical recycling are outperformed by a combination of both in all four assessed categories; particularly for GWP and CED this is valid for substitution ratios as low as 1:0.4. In the baseline, the costs are insensitive to differing substitution ratios.

Recycling potentials of LWP waste in Germany are 2.8 million tons/year when considering a combined mechanical and chemical recycling of the considered plastics. This would be sufficient to reach both EU and German recycling targets (EC, 2018). However, additional measures like design-for-recycling, CO_2-taxes, higher incineration prices (EC, 2018), improved packaging performance (EMAF, 2016) or management (Federal Ministry of Justice and Consumer Protection, 2019) are required to reach mechanical recycling targets (Conversio, 2018; Christiani, 2017).

The study results strongly depend on local or national circumstances such as waste composition, local processing plants, or energy mix (carbon intensity). Thus, they are only partly transferable. Calculated economic benefits might also not be generated, if organizational barriers, lacking stakeholder cooperation, or market aloofness for recycled products persist.

Research is necessary to provide experimentally validated data specifically for chemical recycling of real mixed waste. Standardization[12] efforts aim at including mass balancing, allocation rules, and chemical characteristics (e.g., calorific value) to improve data consistency, comparability, and traceability of recycled feedstock into new and certified recycled products (EMAF, 2019; PlasticsEurope, 2017). Future research should address the quality assessment within mechanical recycling. And, a detailed study of the pyrolysis process and process control for different waste compositions (beyond household waste) is needed (Vogel et al., 2020). The developed assessment model could be applied to other plastic waste or chemical recycling technologies such as gasification (Seidl et al., 2019) or solvolysis (Zhao et al., 2018; Schlummer et al., 2020).

ACKNOWLEDGEMENTS

Thank you to all anonymous reviewers for their valuable suggestions and input which considerably improved the quality of this study. Many thanks also to the editors for their constant support. This study was carried out within the research projects “Leuchtturm Kreislaufwirtschaft – Schwerpunkt Chemisches Recycling” and “Kreislaufwirtschaft für Kunststoffe” funded by the “THINKTANK Industrielle Ressourcenstrategien” (Industrial Resource Strategies), which is financed by the Ministry of the Environment, Climate Protection, and the Energy Sector of the state of Baden-Württemberg in Germany and industry partners.

CONFLICT OF INTEREST

The authors declare no conflict of interest.

ORCID

Rebekka Volk https://orcid.org/0000-0001-9930-5354
Christoph Stallkamp https://orcid.org/0000-0001-8260-2889
Justus J. Steins https://orcid.org/0000-0002-9782-9151
Dieter Stapf https://orcid.org/0000-0001-6499-062X
Frank Schultmann https://orcid.org/0000-0001-6405-9763

REFERENCES

Achilias, D. S., Roupakias, C., Megalokonomos, P., Lappas, A. A., & Antonakou, E. V. (2007). Chemical recycling of plastic wastes made from polyethylene (LDPE and HDPE) and polypropylene (PP). *J. Hazard. Mater*, *149*, 536–542. https://doi.org/10.1016/j.jhazmat.2007.06.076.

Achilias, D. S., Antonakou, E. V., Koutsokosta, E., & Lappas, A. A. (2009). Chemical recycling of polymers from waste electric and electronic equipment. *J. Appl. Polym. Sci*, *114*(1), 212–221. https://doi.org/10.1002/app.30533.

Allen, V., Kalivas, J. H., & Rodriguez, R. G. (1999). Post-consumer plastic identification using Raman spectroscopy. In: *Applied Spectroscopy*, *53*(6), 672–681. https://doi.org/10.1366/0003702991947324.

Al-Maadeed, M. A., Madi, N. K., Kahraman, R., Hodzic, A., & Ozerkan, N. G. (2012). An overview of solid waste management and plastic recycling in Qatar. *J. Polym. Environ*, *20*(1), 186–194. https://link.springer.com/article/10.1007/s10924-011-0332-2

[12] E.g. for mechanical recycling (DIN EN 13430:2004), labelling (DIN EN 6120:2019), tracing (DIN EN 15343:2008, EUCertPlast certificate), technical capabilities (DIN EN 15347:2008; DIN EN 15342:2008 (GPPS); DIN EN 15344:2008 (PE); DIN EN 15345:2008 (PP); DIN EN 15346:2015 (PVC) and DIN EN 15348:2015: (PET)) and testing and quality control (DIN CEN/TS 16010/16011:2013).

Amghizar, I., Vandewalle, L. A., van Geem, K. M., & Marin, G. B. (2017). New trends in olefin production. *Engineering*, *3*(2), 171–178. https://doi.org/10.1016/J.ENG.2017.02.006

Andreas, B., Birckenstaedt, W., Bracker, G. P., Collin, G., Grigoliet, G., & Zander, M. (1981). Pyrolytische Rohstoffrückgewinnung / Pyrolytic material recycling. Forschungsbericht / Research report T81-017, Fachinformationszentrum Energie, Physik, Mathematik GmbH Karlsruhe, Eggenstein-Leopoldshafen.

Bergsma, G. (2019a). Chemical recycling: Opportunities for climate policies and circular economy? CE Delft, January 2019, presentation slides. https://cedelft.eu/en/publications/2173/exploratory-study-on-chemical-recycling-update-2019

Bergsma, G. (2019b). Chemical recycling and its CO2reduction potential. *CE Delft*. https://ce.nl/assets/upload/file/Presentaties/2019/Chemical%20recycling%20an%20environmental%20perspective%20CE%20Delft%2020%20february%202019%20brussel.pdf

Bilitewski, B., Wagner, J., & Reichenbach, J. (2018). Bewährte Verfahren zur kommunalen Abfallbewirtschaftung/Best practices for municipal waste management. https://www.umweltbundesamt.de/sites/default/files/medien/1410/publikationen/2018-05-30_texte_39-2018-verfahren-kommunale-abfallwirtschaft_0.pdf

BKV and Plastics Europe (2019). Thermal processes for feedstock recycling of plastic waste, study conducted by KIT and Conversio. https://www.bkv-gmbh.de/infothek/studien.html and https://www.bkv-gmbh.de/fileadmin/documents/Studien/Thermal_Processes_for_Feedstock_Recycling_of_Plastics_Waste_short.pdf

Chen, X., Xi, F., Geng, Y., & Fujita, T. (2011). The potential environmental gains from recycling waste plastics: Simulation of transferring recycling and recovery technologies to Shenyang, China. *Waste Manage. 2011*, *31*(1), 168–179. https://doi.org/10.1016/j.wasman.2010.08.010

Christiani, J. (2017). Recycling und Recyclingfähigkeit bei Verpackungen – Das neue VerpackG in der Praxis [Recycling and recyclability of packaging - The new VerpackG in practice]. Stand der Technik und Herausforderungen für einzelne Verpackungsmaterialen [State of the art and challenges for particular packaging materials]. HTP Ingenieurgesellschaft.

Christiani, J., & Beckamp, S. (2020). Was können die mechanische Aufbereitung von Kunststoffen und das werkstoffliche Recycling leisten? [Which performance can be achieved by mechanical processing of plastics and material recycling?]. *Energie aus Abfall*, *17*, 139–152.

Conversio Market & Strategy GmbH (2018). Bericht Stoffstrombild Kunststoffe in Deutschland 2017 [Report material flow picture plastics in Germany 2017]. https://www.bkv-gmbh.de/infothek/studien.html

Dehoust, M., & Merz, G. (2016). Umweltpotenziale der getrennten Erfassung und des Recyclings von Wertstoffen im Dualen System. https://www.gruener-punkt.de/de/nachhaltigkeit/studie-oeko-institut.html

Destatis (2018). Umweltnutzung und Wirtschaft - Tabellen zu den UGR, Teil 3: Luftemissionen - Ausgabe 2018. Umweltökonomische Gesamtrechnung [Environmental use and economy - tables of environmental accounting, part 3: Air emissions - edition 2018. Environmental economic accounts / Environment and economy - Tables of environmental accounting]. https://www.destatis.de/DE/Themen/Gesellschaft-Umwelt/Umwelt/Publikationen/Umweltnutzung-Wirtschaft/umweltnutzung-und-wirtschaft-bericht-5850001147004.pdf?__blob=publicationFile&v=3. (last access: 8 April 2021).

DIN 6120:2019-03, Kennzeichnung von Packstoffen und Packmitteln – Packstoffe und Packmittel aus Kunststoff [Labelling of packaging materials and supplies - Plastic packaging materials and supplies], Beuth Verlag.

DIN CEN/TS 16010:2013-04, Kunststoffe – Kunststoff-Rezyklate – Probenahmeverfahren zur Prüfung von Kunststoffabfall und Rezyklaten [Plastics - Plastic recyclates - Sampling methods for testing plastic waste and recyclates], Beuth Verlag.

DIN CEN/TS 16011:2013-04, Kunststoffe – Kunststoff-Rezyklate – Probenvorbereitung [Plastics - Plastic recyclates - Sample preparation], Beuth Verlag.

DIN EN 13430:2004-10, Verpackung – Anforderungen an Verpackungen für die stoffliche Verwertung [Packaging - Requirements for packaging for mechanical recycling], Beuth Verlag.

DIN EN 15342:2008-02, Kunststoffe – Kunststoff-Rezyklate – Charakterisierung von Polystyrol (PS)-Rezyklaten [Plastics - Plastic recyclates - Characterization of polystyrene (PS) recyclates], Beuth Verlag.

DIN EN 15343: 2008-02, Kunststoffe – Kunststoff-Rezyklate – Rückverfolgbarkeit bei der Kunststoffverwertung und Bewertung der Konformität und des Rezyklatgehalts [Plastics - Plastic recyclates - Traceability in plastics recycling and assessment of conformity and recyclate content], Beuth Verlag.

DIN EN 15344:2008-02, Kunststoffe – Kunststoff-Rezyklate – Charakterisierung von Polyethylen (PE)-Rezyklaten [Plastics - Plastic recyclates - Characterization of polyethylene (PE) recyclates], Beuth Verlag.

DIN EN 15345:2008-02, Kunststoffe – Kunststoff-Rezyklate – Charakterisierung von Polypropylen (PP)-Rezyklaten [Plastics - Plastic recyclates - Characterization of polypropylene (PP) recyclates], Beuth Verlag.

DIN EN 15346:2015-01, Kunststoffe – Kunststoff-Rezyklate – Charakterisierung von Polyvinylchlorid (PVC)-Rezyklaten [Plastics - Plastic recyclates - Characterization of polyvinyl chloride (PVC) recyclates], Beuth Verlag.

DIN EN 15347:2008-02, Kunststoffe – Kunststoff-Rezyklate – Charakterisierung von Kunststoffabfällen [Plastics - Plastic recyclates - Characterization of plastic wastes], Beuth Verlag.

DIN EN 15348:2015-01, Kunststoffe – Kunststoff-Rezyklate – Charakterisierung von Polyethylenterephthalat (PET)-Rezyklaten [Plastics - Plastic recyclates - Characterization of polyethylene terephthalate (PET) recyclates], Beuth Verlag.

DIN EN ISO 14040:(2006). Umweltmanagement – Ökobilanz – Grundsätze und Rahmenbedingungen [Environmental management - life cycle assessment - principles and framework].

Dormer, A., Finn, D. P., Ward, P., & Cullen, J. (2013). Carbon footprint analysis in plastics manufacturing. *J. Clean. Prod.*, *51*, 133–141. https://doi.org/10.1016/j.jclepro.2013.01.014.

EC (2018). A European strategy for plastics in a circular economy. Retrieved from http://ec.europa.eu/environment/circular-economy/pdf/plastics-strategy-brochure.pdf

EC (2019a). Einweg-Plastikprodukte verschwinden ab 2021 aus den Regalen [Throw-away plastic products are demonetized in 2021], press release 28/03/2019, Europäische Kommission – Vertretung in Deutschland. https://ec.europa.eu/germany/news/20190328-plastikprodukte_de

EC (2019b). Circular Economy: Commission welcomes European Parliament adoption of new rules on single-use plastics to reduce marine litter, press release 27/03/2019. Retrieved from https://ec.europa.eu/commission/presscorner/detail/en/STATEMENT_19_1873

EMAF. (2016). The new plastics economy: Rethinking the future of plastics. *Ellen MacArthur Foundation*. Retrieved from https://www.ellenmacarthurfoundation.org/assets/downloads/EllenMacArthurFoundation_TheNewPlasticsEconomy_Pages.pdf

EMAF. (2019). Enabling a circular economy for chemicals with the mass balance approach. *Ellen MacArthur Foundation*. Retrieved from https://www.ellenmacarthurfoundation.org/assets/downloads/Mass-Balance-White-Paper.pdf (last access: 19 Nov 2019).

Federal Ministry of Justice and Consumer Protection (2019). *Verpackungsgesetz – VerpackG [German Packaging Act]*. https://www.gesetze-im-internet.de/verpackg/VerpackG.pdf. (last access: 21 July 2019).

Feldhoff, R., Wienke, D., Cammann, K., & Fuchs, H. (1997). Online post-consumer package identification by NIR spectroscopy combined with a FuzzyARTMAP classifier in an industrial environment. In: *Applied Spectroscopy*, *51*(3), 362–368. https://www.osapubli-shing.org/viewmedia.cfm?uri=as-51-3-362

Großmann, M. (2011). Leistungs- und Effizienzsteigerung in der Einschneckenextrusion durch alternative Plastifiziertechnik [Increased performance and efficiency in single-screw extrusion through alternative plasticizing technology]. Dissertation, University Duisburg-Essen. https://core.ac.uk/download/pdf/33798730.pdf (last access: 21 July 2019).

Gu, F., Hall, P., & Miles, M. J. (2016a). Performance evaluation for composites based on recycled polypropylene using principal component analysis and cluster analysis. *J. Clean. Prod.*, *115*, 343–353. https://doi.org/10.1016/j.jclepro.2015.12.062.

Gu, F., Hall, P., & Miles, N. J. (2016b). Development of composites based on recycled polypropylene for injection moulding automobile parts using hierarchical clustering analysis and principal component estimate. *J. Clean. Prod.*, *137*, 632–643. https://doi.org/10.1016/j.jclepro.2016.07.028.

Gu, F., Guo, J., Zhang, W., Summers, P. A., & Hall, P. (2017). From waste plastics to industrial raw materials: A life cycle assessment of mechanical plastic recycling practice based on a real-world case study. *Sci. Total Environ.*, 601–602, 1192–1207. https://doi.org/10.1016/j.scitotenv.2017.05.278

Habich, U.,Beel, H. (2014). *Modifizierung von Recyclingverfahren durch sensorbasierte Sortierung [Modification of recycling processes through sensor-based sorting]*. Recycling und Rohstoffe, *7*, 471–482. http://www.vivis.de/phocadownload/Down-load/2014_rur/2014_RuR_471_482_Habich.pdf

Huang, J., Tian, C., Ren, J., & Bian, Z. (2017). Study on impact acoustic – Visual sensor-based sorting of ELV plastic materials. *Sensors*, *17*(6), 1325–1346. https://www.mdpi.com/1424-8220/17/6/1325/pdf

Huber, N., Eschlböck-Fuchs, S., Scherndl, H., Freimund, A., Heitz, J., & Pedarnig, J. D. (2014). In-line measurements of chlorine-containing polymers in an industrial waste sorting plant by laser-induced breakdown spectroscopy. *Appl. Surface Sci.*, *302*, 280–285. https://doi.org/10.1016/j.apsusc.2013.10.070

Huysman, S., De Schaepmeester, J., Ragaert, K., Dewulf, J., & De Meester, S. (2017). Performance indicators for a circular economy: A case study on post-industrial plastic waste. https://doi.org/10.1016/j.resconrec.2017.01.013

IPCC. (2013). Climate change 2013. *The physical science basis; summary for policymakers, a report of Working Group I of the IPCC, technical summary, a report accepted by Working Group I of the IPCC but not approved in detail and frequently asked questions; part of the Working Group I Contribution to the fifth assessment report of the Intergovernmental Panel on Climate Change*. New York: Intergovernmental Panel on Climate Change. http://www.ipcc.ch/

Kanchanapiya, P., Limphitakphong, N., Pharino, C., & Chavalparit, O. (2015). Evaluation of greenhouse gas emissions and reduction from the petrochemical industry in Thailand. *Greenhouse Gas Measurement and Management*, *4* (2–4), 161–177. https://doi.org/10.1080/20430779.2015.1008362

Ketelsen, K., & Kanningen, K. (2016). ASA-Strategie 2030 - Ressourcen- und Klimaschutz durch eine stoffspezifische Abfallbehandlung; Arbeitsgemeinschaft Stoffspezifische Abfallbehandlung; https://www.asa-ev.de/de/informationen/asa-strategie/

Klöpffer, W., & Renner, I. (2008). Life-cycle based sustainability assessment of products. In S. Schaltegger, M. Bennett, R.L. Burritt, C. Jasch (Eds.): *Environmental management accounting for cleaner production* (pp. 91–102, vol. *24*). Dordrecht, Netherlands: Springer (Eco-Efficiency in Industry and Science)

Lazarevic, D., Aoustin, E., Buclet, N., & Brandt, N. (2010). Plastic waste management in the context of a European recycling society: Comparing results and uncertainties in a life cycle perspective. *Resources, Conservation and Recycling*, *55*(2), 246–259. https://doi.org/10.1016/j.resconrec.2010.09.014

Makuta, M., Moriguchi, Y., Yasuda, Y., & Sueno, S. (2000). Evaluation of the effect of automotive bumper recycling by life-cycle inventory analysis. *Journal of Material Cycles and Waste Management*, *2*, 125–137. https://link.springer.com/article/10.1007%2Fs10163-000-0028-5

Masoumi, H., Safavi, S. M., & Khani, Z. (2012). Identification and classification of plastic resins using near-infrared reflectance. *International Journal of Mechanical and Mechatronics Engineering*, *6*(5), 213–220. https://pdfs.semanticscholar.org/4dac/692f2cd4eb4a62c0f0fc3be994ff58f39ca2.pdf (last access: 21 July 2019).

Meran, C., Ozturk, O., & Yuksel, M. (2008). Examination of the possibility of recycling and utilising recycled polyethylene and polypropylene. *Materials and Design*, *29* (3), 701–705. https://doi.org/10.1016/j.matdes.2007.02.007.

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

Murase, A., & Sato, N. (1999). Rapid identification of plastics by pyrolysis infrared spectroscopy with a new pyrolysis probe. *Applied Spectroscopy*, *53*(6), 745–747. https://www.osapublishing.org/viewmedia.cfm?uri=as-53-6-745&seq=0&casa_to-ken=wYF5PXDmtG4AAAAA:DgtsCr6F5Pj8GdF9a16shCKqRO2JE63NNmGIlXD6BKmfls-BlFvHTFVkvw5tqrDLpP_nVlk3Fw-3xAw

Patel, M. (2003). Cumulative energy demand (CED) and cumulative CO2 emissions for products of the organic chemical industry. *Energy*, *28*(7), 721–740. https://doi.org/10.1016/S0360-5442(02)00166-4

Pereira, C. P., Prata, D. M., Monteiro, L. P. C., & Mainier, F. B. (2013). Evaluation of the CO_2 Emissions in a Petrochemical Plant. *ARPN Journal of Science and Technology*, *3*(8). http://citeseerx.ist.psu.edu/viewdoc/summary?doi=10.1.1.680.5732

Perugini, F., Mastellone, M. L., & Arena, U. (2005). A life cycle assessment of mechanical and feedstock recycling options for management of plastic packaging wastes. *Environmental Progress*, *24*(2), 137—154. https://doi.org/10.1002/ep.10078.

Pivnenko, K., Jakobsen, L. G., Eriksen M., Damgaard A., & Astrup. (2015). Challenges in plastic recycling. *International Waste Management and Ladfill Symposium*. Sardinia: CISA

Pivnenko, K., Eriksen, M. K., Martín-Fernández, J. A., Eriksson, E., & Astrup, T. F. (2016). Recycling of plastic waste: Presence of phthalates in plastics from households and industry. *Waste Management*, *54*, 44–52. https://doi.org/10.1016/j.wasman.2016.05.014

Plasticker. (2019). Echtzeit-Preisspiegel mit Preis- und Mengenstatistik /Real-time price index with price and quantity statistics. Retrieved from https://plasticker.de/preise/pms.php

PlasticsEurope. (2017). View paper on mechanical and organic recycling. Retrieved from https://www.plasticseurope.org/download_file/force/511/315

PlasticsEurope. (2018a). Plastics – the facts 2018: An analysis of European plastics production, demand and waste data. Retrieved from http://www.plasticseurope.org/download_file/force/2387/319.

PlasticsEurope (2018b). Eco-Profiles: PlasticsEurope's Eco-profiles comprise Life Cycle Inventory datasets (LCI) and Environmental Product Declarations (EPD) for plastics. Retrieved from https://www.plasticseurope.org/en/resources/eco-profiles

PlasticsEurope. (2012a). *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. (last access: 21 July 2019).

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

Rebitzer, G. (2002). Integrating life cycle costing and life cycle assessment for managing costs and environmental impacts in supply chains. In S. Seuring, M. Goldbach, (Eds.), *Cost management in supply chains* (pp. 127–146). Heidelberg: Physica-Verlag HD.

Ren, T., Patel, M., & Blok, K. (2006). Olefins from conventional and heavy feedstocks. Energy use in steam cracking and alternative processes. *In Energy*, *31* (4), 425–451. https://doi.org/10.1016/j.energy.2005.04.001.

Ren, T., Patel, M., & Blok, K. (2008). Steam cracking and methane to olefins. Energy use, CO_2 emissions and production costs. *In Energy*. https://doi.org/10.1016/j.energy.2008.01.002.

Ren, T. (2009). Petrochemicals from oil, natural gas, coal and biomass. Energy use, economics and innovation. Utrecht University. Retrieved from https://dspace.library.uu.nl/bitstream/handle/1874/32318/ren.pdf?sequence=2

Rieckhof, R., & Guenther, E. (2018). Integrating life cycle assessment and material flow cost accounting to account for resource productivity and economic-environmental performance. *In Int J Life Cycle Assess*, *23* (7), 1491–1506. https://doi.org/10.1007/s11367-018-1447-7.

Rigamonti, L., Grosso, M., & Giugliano, M. (2009). Life cycle assessment for optimising the level of separated collection in integrated MSW management systems. *Waste Management*, *29*, 934–944. https://doi.org/10.1016/j.wasman.2008.06.005.

Russ, M., Gonzales, M., Horlacher, M., Shonfield, P., & Deimling, S. (2020). Evaluation of pyrolysis with LCA - 3 Case Studies. BASF SE, Leinfelden-Echterdingen https://www.basf.com/global/en/who-we-are/sustainability/we-drive-sustainable-solutions/circular-economy/mass-balance-approach/chemcycling/lca-for-chemcycling.html (last access: 4 Aug 2020)

Safavi, S. M., Masoumi, H., Mirian, S. S., & Tabrizchi, M. (2010). Sorting of polypropylene resins by colour in MSW using visible reflectance spectroscopy. In: *Waste management*, *30*(11), 2216–2222. https://www.sciencedirect.com/science/article/pii/S0956053x10003405 (last access: 21 July 2019).

Schlummer, M., Fell, T., Mäurer, A., & Altnau, G. (2020). Die Rolle der Chemie beim Recycling - Physikalisches und chemisches Kunststoffrecycling im Vergleich. *Kunststoffe*, *6*, (2020), 51–54; https://www.ivv.fraunhofer.de/content/dam/ivv/de/documents/Leistungsangebot/recycling-und-umwelt/I-PDF%20Fraunhofer%20IVV%20Kunststoffe_2020_06_.pdf (last access: 7 Aug. 2020)

Scott, D. M. (1995). A two-colour near-infrared sensor for sorting recycled plastic waste. In: *Measurement Science and Technology*, *6*(2), 156–159. https://i-opscience.iop.org/article/10.1088/0957-0233/6/2/004/pdf (last access: 21 July 2019).

Seidl, L. G., Lee, R. P., Keller, F., & Mayer, B. (2019). Chemical Utilization of Carbonaceous Waste and Lignite – A Case Study of Sustainable Olefin Production in Germany. Circular Economy – A Fresh View on Petrochemistry. DGMK Conference. https://dgmk.de/app/uploads/2019/05/Seidl.pdf (last access: 19 Feb 2020).

Serranti, S., Gargiulo, A., & Bonifazi, G. (2011). Characterization of post-consumer polyolefin wastes by hyperspectral imaging for quality control in recycling processes. In: *Waste Management*, *31*(11), 2217–2227. https://www.sciencedirect.com/science/ar-ticle/pii/S0956053x11002728 (last access: 21 July 2019).

Serranti, S., Luciani, V., Bonifazi, G., Hu, B., & Rem, P. C. (2015). An innovative recycling process to obtain pure polyethylene and polypropylene from household waste. In: *Waste management*, *35*, 12–20. https://www.sciencedirect.com/science/ar-ticle/pii/S0956053x14005017 (last access: 21 July 2019).

Sommerhuber, P. F., Wang, T., & Krause, A., (2016). Wood-plastic composites as potential applications of recycled plastics of electronic waste and recycled particleboard. *J. Clean. Prod.*, *121*, 176–185. https://doi.org/10.1016/j.jclepro.2016.02.036.

Stapf, D., Wexler, M., Ciceri, G., Johansson, I., & Whitty, K. (2019a). Biomass pre-treatment for bio-energy. Case study 3: Pretreatment of municipal solid waste (MSW) for gasification. *IEA Bio-energy*. https://doi.org/10.13140/RG.2.2.27035.39205

Stapf, D., Seifert, H., & Wexler, M. (2019b). Thermische Verfahren zur rohstofflichen Verwertung kunststoffhaltiger Abfälle. *Energie aus Abfall*, *16* (2019), 357–376. https://www.vivis.de/2019/01/energie-aus-abfall-band-16/148/

Turner, D., William, I. D., & Kemp, S. (2015). Greenhouse gas emission factors for recycling of source-segregated waste materials. *Resources, Conservation and Recycling*, *105(A)*, 186–197. https://doi.org/10.1016/j.resconrec.2015.10.026

UBA. (2015). Thermische Nutzung von Abfallstoffen durch industrielle Mitverbrennung [Thermal utilization of waste materials through industrial co-incineration]. Retrieved from https://www.cleaner-production.de/images/BestPractice/data_de/ICC.pdf

UBA. (2018a). Nationale Inventarberichte zum Deutschen Treibhausgasinventar 1990 bis 2016. Emission der von der UN-Klimarahmenkonvention abgedeckten Treibhausgase. [National inventory reports on the German greenhouse gas inventory 1990 to 2016. Emissions of greenhouse gases covered by the UN Framework Convention on Climate Change.]. Retrieved from https://www.umweltbundesamt.de/publikationen/nationaler-inventarbericht-deutschen-1

UBA. (2018b). Einsatz von Kunststoffrezyklaten in Deutschland in 2017 [Use of recycled plastics in Germany in 2017]. Retrieved from https://www.umweltbundesamt.de/sites/default/files/medien/384/bilder/dateien/6_abb_einsatz-kunststoffrezyklaten_2018-12-20.pdf

UBA. (2018c). Bewährte Verfahren zur kommunalen Abfallbewirtschaftung [Best practices in municipal waste management]. Retrieved from https://www.umweltbundesamt.de/sites/default/files/medien/1410/publikationen/2018-05-30_texte_39-2018-verfahren-kommunale-abfallwirtschaft_0.pdf

Van Eygen, E., Laner, D., & Fellner, J. (2018a). Integrating high-resolution material flow data into the environmental assessment of waste management system scenarios: The case of plastic packaging in Austria. *Environmental Science & Technology*, *52*(19), 10934–10945. https://pubs.acs.org/doi/10.1021/acs.est.8b04233

Van Eygen, E., Laner, D.;, & Fellner, J. (2018b). 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

VCI (2019). Chemiewirtschaft in Zahlen 2019 [Data of the chemical industry 2019]. Verband der Chemischen Industrie e.V. Retrieved from https://www.vci.de/vci/downloads-vci/publikation/chemiewirtschaft-in-zahlen-print.pdf

VDI (2012). *VDI 4600 Cumulative energy demand (KEA) - Terms, definitions, methods of calculation.*, Verlag des Vereins Deutscher Ingenieure.

Vogel, J., Krüger, F., & Fabian, M. (2020). Chemisches recycling. Retrieved from https://www.umweltbundesamt.de/sites/default/files/medien/1410/publikationen/2020-07-17_hgp_chemisches-recycling_online.pdf

Wäger, P. A., Hischier, R.;, & Eugster, M. (2011). Environmental impacts of the Swiss collection and recovery systems for waste electrical and electronic equipment (WEEE): a follow-up. *The Science of the Total Environment*, *409*, 1746–1756. https://doi.org/10.1016/j.scitotenv.2011.01.050.

Wäger, P., & Hischier, R. (2015). Life cycle assessment of post-consumer plastics production from waste electrical and electronic equipment (WEEE) treatment residues in a Central European plastics recycling plant. *Science of the Total Environment*, *529*, 158–167. https://doi.org/10.1016/j.scitotenv.2015.05.043.

Wan, J. K. S., Vepsäläinen, K. P., & Ioffe, M. S. (1994). Time-resolved, pulsed laser sensitized acoustic technology: Application to online sorting of waste plastics. *Journal of Applied Polymer Science*, *54*(1), 25–31. https://online-library.wiley.com/doi/epdf/10.1002/app.1994.070540103

Wyns, T., Khandekar, G., & Robson, I. (2018). A bridge towards a carbon neutral Europe: Europe's energy intensive industries contribution to the EU Strategy for long-term EU greenhouse gas emissions reductions. *Institute for European Studies*. Retrieved from https://www.ies.be/files/Industrial_Value_Chain_25sept.pdf

Xiang, D., Qian, Y., Man, Y., & Yang, S. (2014a). Techno-economic analysis of the coal-to-olefins process in comparison with the oil-to-olefins process. *Applied Energy*, *113*, 639–647. https://doi.org/10.1016/j.apenergy.2013.08.013.

Xiang, D., Yang, S., Liu, X., Mai, Z., & Qian, Y. (2014b). Techno-economic performance of the coal-to-olefins process with CCS. *Chemical Engineering Journal*, *240*, 45–54. https://doi.org/10.1016/j.cej.2013.11.051.

Xiang, D., Yang, S., Li, X., & Qian, Y. (2015). Life cycle assessment of energy consumption and GHG emissions of olefins production from alternative resources in China. *Energy Conversion and Management*, *90*, 12–20. https://doi.org/10.1016/j.enconman.2014.11.007.

Zhao, Z., Liu, Y., Wang, F., Li, X., Deng, S., Xu, J., & Wei, W. (2017). Life cycle assessment of primary energy demand and greenhouse gas (GHG) emissions of four propylene production pathways in China. *Journal of Cleaner Production*, *163*, 285–292. https://doi.org/10.1016/j.jclepro.2015.12.099.

Zhao, Y.-B., Lv, X.-D., & Ni, H.-G. (2018). Solvent-based separation and recycling of waste plastics: A review. *Chemosphere*, *209*, 707–720. https://doi.org/10.1016/j.chemosphere.2018.06.095

SUPPORTING INFORMATION

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

How to cite this article: Volk R, Stallkamp C, Steins JJ, Yogish SP, Müller RC, Stapf D, & Schultmann F. Techno-economic assessment and comparison of different plastic recycling pathways: A German case study. *J Ind Ecol*. 2021;1–20. https://doi.org/10.1111/jiec.13145