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Global warming potential and economic performance of gasification-based chemical recycling and incineration pathways for residual municipal solid waste treatment in Germany

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ABSTRACT

Chemical recycling could facilitate the transition from a linear to a circular carbon economy, where carbon-containing waste is channeled back into the production cycle as a chemical feedstock instead of being incinerated or landfilled. However, the predominant focus on technological aspects of chemical recycling for plastic waste narrows evaluations of its potential in contributing to such a transition. Moreover, it leads to significant controversy about its role in the waste hierarchy as a possible competitor to mechanical recycling. To address these gaps in the literature, this study assesses ecological and economic impacts associated with chemical recycling of residual municipal solid waste in Germany. Combining approaches of life cycle assessment and techno-economic analysis, chemical recycling and conventional incineration-based treatment pathways are comparatively evaluated in terms of global warming potential and economic performance (i.e. fixed capital investment, net present value, dynamic payback period, and levelized cost of carbon abatement). Results indicate that compared to incineration-based conventional pathways, chemical recycling can contribute to reducing greenhouse gas emissions in low-emission energy systems. However, the economic performance of chemical recycling is highly dependent on its scale of operation. Additionally, a price premium for recycling products as well as economic instruments for penalizing CO_2 emissions are identified to play important roles in the economic performance of chemical recycling.

1. Introduction

"Chemical recycling" (CR) involves the decomposition of carbonaceous waste – either thermally or by solvents – into synthesis gas, monomers, oligomers, or higher hydrocarbons to produce chemical products such as plastics, fuels, fertilizers, or pharmaceuticals in conventional quality (Mamani et al., 2020). CR could potentially support the transition from a linear to a circular carbon economy, where carbon content in waste is recirculated into the production cycle as secondary feedstock instead of being incinerated or landfilled (Keller et al., 2020; Lee, 2019; Lee and Scheibe, 2020; Seidl et al., 2020). Especially in recent years, increasing interest in CR is observable from science, industry, and civil society (e.g. BASF, 2020; INEOS, 2020; LB, 2020; Mamani et al., 2020; Ragaert et al., 2017; ZWE, 2019b). However, the predominant attention – academic and otherwise – is focused on plastic waste as CR input, as well as CR's contribution to meeting higher recycling quotas for packaging waste and resolving the plastic crisis (BCG, 2019; CRE, 2020; PlasticsEurope, 2020; PRE, 2020; Ragaert et al., 2017; Solis and

Abbreviations: AE, Aggregated effect; *CEPCI*, Chemical Engineering Plant Costs Index; *CHP*, Combined heat and power; *CNEP*, Climate neutral energy provision; *CR*, Chemical recycling; *DPP*, Dynamic payback period; *EPS*, Energy provision scenario; *EREG*, Environmental regulation; *ES*, Economic scenario; *FCI*, Fixed capital investment; *GWP*, Global warming potential; *LCA*, Life cycle assessment; *LCCA*, Levelized cost of carbon abatement; *MBT*, Mechanical-biological treatment; *MC*, Market conditions; *MSW*, Municipal solid waste; *MSWI*, Municipal solid waste incinerator; *NPV*, Net present value; *PREP*, Predominantly renewable energy provision; *PS*, Plant scaling; *RDF*, Refuse-derived fuel; *rMSW*, Residual municipal solid waste; *RTO*, Regenerative thermal oxidation; *SI*, Supplementary Information; *TEA*, Techno-economic analysis; *UBA*, German Environment Agency.

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Silveira, 2020). This restricted focus has resulted in controversy surrounding the discussion about CR's opportunities and challenges and its role in the waste hierarchy.

The waste hierarchy – from reduce, reuses, recycle, recover to landfill – is part of the European Waste Framework Directive 2008/98/EC to guide waste legislations/policies of EU member countries (EC, 2019a, 2008). With growing global awareness and concerns about problems associated with waste disposal, the EU Commission and EU members have focused their attention at the top of the waste hierarchy, enacting diverse policies/measures such as prevention/reduction of single-use plastics, promotion of eco-design for products, and targeted waste separation to increase recycling quotas via multiple reuse and mechanical recycling (BfJ, 2020b; BMU, 2018; EC, 2020a, 2020b, 2019b). In this context, concerns arise especially among civil society organizations that CR – in particular processes targeting pure plastic streams as inputs – will distract from the goals of prevention and eco-design to promote reuse, in addition to posing a competition to mechanical recycling (Casey, 2019; ChemicalWatch, 2019; GAIA, 2020; NABU, 2020). That the CR discussion is not only targeting "Plastics-to-Plastics" but also "Plastics-to-Fuels" adds to the controversy. Unlike the former where carbon is retained in the production cycle to promote circularity, carbon in the later will be emitted as CO_2 following combustion (Opray, 2017; Tamma, 2019; ZWE, 2018).

The focus on plastics narrows assessments about CR's potential to contribute to the transition towards a circular carbon economy. While academic publications and public discourse have primarily focused on CR of plastic waste, a review of CR technologies by Mamani et al. (2020) identified the applicability of CR to other carbon-containing waste inputs (e.g. unsorted household waste, paper sludges, bituminous materials, oily materials, carbon-fiber composites, and glass-fiber composites). Hence, CR could contribute to a circular carbon economy via recirculating heterogenous and "dirty" carbon-containing waste materials back into the production cycle as secondary carbon feedstock to generate value-added products. By focusing on heterogenous waste, it will not compete with mechanical recycling for pure plastic waste streams. Rather, it will be complementary as it presents a recycling alternative for waste that is not recyclable via conventional recycling techniques. However, despite the relevance and potential applicability for different types of carbon-containing waste, to the best of our knowledge, no published studies have assessed the climate potential and economic performance of CR technologies for mixed and residual waste. Moreover, CR studies and evaluations generally take an isolated view of CR technologies and processes, without consideration of their integration into existing value chains/production lines (Materazzi et al., 2016; Miandad et al., 2016; Quicker et al., 2017; Rollinson and Oladejo, 2020; Zhou et al., 2015; ZWE, 2019a).

To address these gaps in the waste management literature, a case study approach is utilized to assess the global warming potential (GWP) and economic performance of CR for residual municipal solid waste (rMSW) in Germany. rMSW describes the remaining fraction of municipal solid waste (MSW) – following source separation of hazardous and recyclable MSW fractions in households or offices – that is mainly incinerated in Germany today (Beylot and Villeneuve, 2013; Sahimaa et al., 2015). Aim is to provide first insights into the GWP – determined via life cycle assessment (LCA) – and economic performance – examined via techno-economic analysis (TEA) of fixed capital investment (FCI), net present value (NPV), dynamic payback period (DPP), and levelized cost of carbon abatement (LCCA) – of CR for rMSW in comparison to conventional incineration-based treatment pathways.

As presented in Fig. 1, the study evaluates three pathways: i) direct incineration of rMSW, ii) indirect incineration of refuse-derived fuel (RDF) from rMSW, and iii) gasification-based CR of RDF from rMSW. RDF refers to high calorific value fractions such as mixed plastics which are extracted and conditioned in mechanical-biological treatment (MBT)

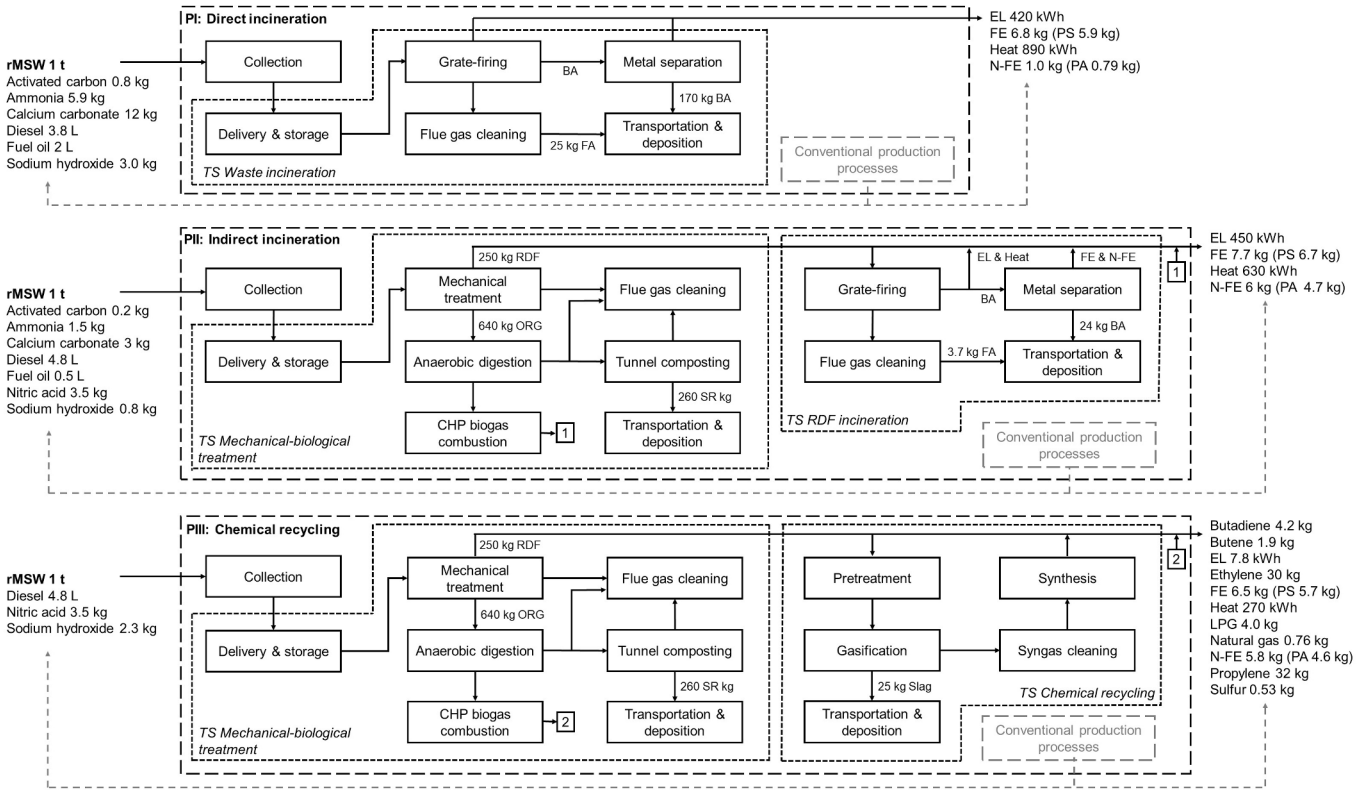

Fig. 1. Illustration of the LCA system environments. BA: Bottom ash. CHP: Combined heat and power generation. EL: Electricity. FA: Fly ash. FE: Ferrous scrap. LPG: Liquefied petroleum gas. N-FE: Non-ferrous scrap. ORG: Organics. PA: Primary aluminum. PS: Primary steel. RDF: Refuse-derived fuel. rMSW: Residual municipal solid waste. SR: Stabilized residues. TS: Treatment step. Values rounded to two significant digits. Energy figures are reported net of energy self-use. Substituted market products at substitution ratios < 100% are indicated in brackets.

plants for energetic purposes. The RDF produced from MBT can be subsequently incinerated in RDF power plants. Alternatively, it can be used as chemical feedstock for gasification-based CR, which represents an alternative rMSW treatment pathway (proven on an industrial scale in Germany) to incineration-based pathways (cf. Section 2.2). This study thus extends the scope of waste management research from isolated technological evaluations of CR for plastic waste to a system-integrated assessment of its potential and opportunities for rMSW. The case study focuses on Germany where conventional incineration-based pathways are well-established (UBA, 2018b).

Next, Section 2 presents the study context, while Section 3 summarizes the approaches utilized for calculating GWP and economic performance. Section 4 presents the results before concluding with key insights, limitations of the present study, and suggestions for future research in Section 5.

2. Study context

2.1. Residual municipal solid waste in Germany

In 2018, 15.2 million tonnes of rMSW are generated in Germany, of which 10.6 million tonnes, i.e. 70%, are incinerated directly in municipal solid waste incinerators (MSWI) for electricity and heat production. Another 2.4 million tonnes, i.e. 16%, are treated in MBT plants, where – following the separation of organics, metal components and inert materials – RDF is produced and subsequently incinerated for energy and/or heat recovery in incineration facilities ranging from RDF power plants, cement works to coal-fired power plants (DESTATIS, 2021a; Pinasseau et al., 2018; UBA, 2018a, 2018b). In Germany, RDF is utilized extensively in RDF power plants that are engineered specifically for recovering energy from RDFs with high calorific values (UBA, 2018b).

Due to its heterogeneous composition, significant proportion of composite materials, and the low quality of potentially recyclable materials, it is challenging to recycle rMSW using conventional mechanical recycling processes. As rMSW-derived RDF contains significant amounts of carbon, CR represents a potential alternative treatment pathway to incineration (Keller et al., 2020; Mamani et al., 2020). Specifically, rMSW can be recirculated via CR into the production cycle as raw material to produce organic chemicals instead of being fully oxidized, i.e. incinerated, as a one-time energy source (Lee et al., 2017b).

2.2. Chemical recycling of residual municipal solid waste

Since the 1970s, interest in CR development and deployment ranging from experimental, pilot- to industrial-scale facilities is observable (Mamani et al., 2020; Schmalfeld and Arendt, 2008). CR processes can be categorized into solvolysis, pyrolysis, and gasification. However, not all CR processes are suitable for processing rMSW due to its heterogeneous composition and the low quality of potentially recyclable materials (Lee et al., 2020). Whereas applications of CR processes such as pyrolysis and gasification have been reported (Keller et al., 2020; Malkow, 2004; Mamani et al., 2020; Mazzoni and Janajreh, 2017), only gasification has been implemented on an industrial-scale in the context of rMSW. Specifically, different gasification technologies are utilized in Berrenrath (between 1993 and 1997) and in Schwarze Pumpe (between 1995 and 2007) in Germany for the chemical conversion of rMSW-derived RDF with coal into syngas for subsequent synthesis into methanol (Lee et al., 2017a; Lee et al., 2017b; Schmalfeld and Arendt, 2008). Today, pilot-scale developments are also observable for rMSW gasification. These are namely ENERKEM in Alberta, Canada where a gasification plant is operated since 2017 to generate methanol from MSW (Enerkem, 2019), and the Institute of Energy Process Engineering and Chemical Engineering at the Technische Universität (i.e. Technical University) Bergakademie Freiberg, Germany where different gasification pilot plants are operated for converting i.a. rMSW-derived RDF into syngas (IEC, 2018; Lee and Meyer, 2019).

Previous and current developments thus point to gasification's potential in handling large amounts of mixed waste in the form of residual MSW. As such, the evaluation of CR as an alternative pathway to incineration-based treatment pathways in this paper is based on waste gasification technology. During the waste gasification process, RDF is broken down into CO- and H_2-rich syngas under high pressure and temperature conditions with oxygen and/or steam as gasification agents. Subsequently, syngas can be converted to platform chemicals such as methanol, which can then be further synthesized via a methanol-to-olefins process to olefins, a raw material for a broad spectrum of chemical products including virgin plastics (Lee et al., 2018b; Lee et al., 2018a; Meyer et al., 2018). These process steps for producing olefins – as described by Keller et al. (2020) – are used for the assessment of CR of rMSW in this study.

3. Methodology

This section presents the LCA method applied to determine GWP and the TEA approach to assess economic performance. Note that elementary and product flows determined for the LCA (e.g. utilization of supplies or energy provision) are also utilized for the subsequent TEA.

3.1. Assessment of global warming potential

The ISO standard 14040:2006 for LCA is utilized to assess the ecological impact of alternative waste treatment pathways in terms of GWP (DIN, 2009).

3.1.1. Goal and scope

Goal is to provide a sound basis for a comparative GWP evaluation of two conventional rMSW treatment pathways – direct incineration (PI) and indirect incineration (PII) – with CR (PIII). The functional unit is defined as the treatment of one tonne of wet rMSW with fractional composition and chemical characteristics as described in Section 3.1.4. The temporal focus of the assessment is set from 2021 until 2028 as assumptions about technological efficiencies and waste composition are based on current knowledge (Montejo et al., 2013). As the geographical scope of the assessment affects assumptions regarding waste composition and process configuration, the investigation focuses on Germany as a case study.

3.1.2. System boundaries

Fig. 1 illustrates the LCA system environments of all three investigated treatment pathways. System boundaries extend from the point of rMSW collection to treatment product provision. Upstream impacts of waste generation are excluded as a zero-burden approach is applied (Montejo et al., 2013). Upstream impacts of supply utilization and downstream impacts of market product substitution are accounted for by system expansion with a background system. Background system data is displayed in Table 1 and refers to specific or market average values indicating the attributional mode of this assessment. Ferrous and non-ferrous scraps are assumed to substitute primary steel and aluminum at substitution ratios of 87% and 79% as suggested by Montejo et al. (2013). Based on Keller et al. (2020), electricity, heat, and chemical recycling products are assumed to substitute conventional products at a substitution ratio of 100% as equivalent quality can be expected. Note that impacts associated with plant construction are beyond the scope of this study.

3.1.3. Life cycle assessment model EASETECH

The LCA-model EASETECH V3.1.7. (Clavreul et al., 2014) – developed at the Technical University of Denmark (DTU) and well established in research (Cobo et al., 2018; Maalouf and El-Fadel, 2019; Morais Lima et al., 2019) – is utilized to assess GWP and to provide data for subsequent TEA. In the following sections, the approach utilized for characterizing rMSW and waste treatment technologies for integration into the

Table 1
Assumed GWP100 and market prices for relevant processes, products, and supplies. *: CHP selling price. EUR: euros. All prices assumed for year 2021. Values rounded to two significant digits.

	Per	GWP100 [kg CO_2eq.]	Price [EUR]	Source [GWP100; price]
Supplies				
Activated carbon	kg	5.1	2.5	GaBi (2019); UN (2021)
Ammonia	kg	2.3	0.3	GaBi (2019); UN (2021)
Calcium carbonate	kg	0.064	0.11	GaBi (2019); UN (2021)
Diesel	kg	0.5	0.54	GaBi (2019); UN (2021)
Fuel oil	L	0.43	0.56	GaBi (2019); UN (2021)
Nitric acid	kg	1.8	0.13	GaBi (2019); UN (2021)
Sodium hydroxide	kg	0.95	1.1	GaBi (2019); UN (2021)
Products				
Butadiene	kg	1.5	0.73	GaBi (2019); UN (2021)
Butene	kg	1	0.73	GaBi (2019); UN (2021)
Electricity	kWh	0.40	0.084*	UBA (2020a); BfJ (2020a); EEX (2021)
Ethylene	kg	1.4	1.08	GaBi (2019); VCI (2020)
Fuel gas	kg	0.61	0.45	GaBi (2019); UN (2021)
Heat	kWh	0.26	0.025*	BDI (2018), UBA (2007)
Liquefied petroleum gas	kg	0.79	0.49	GaBi (2019); UN (2021)
Primary aluminum	kg	6.7	3.3	GaBi (2019); UN (2021)
Primary steel	kg	2.2	0.63	GaBi (2019); UN (2021)
Propylene	kg	1.4	0.99	GaBi (2019); VCI (2020)
Sulfur	kg	0.66	0.11	GaBi (2019); UN (2021)

EASETECH model is introduced.

3.1.4. Characterization of residual municipal solid waste

Similar to the German authorities, this study defines rMSW as a subgroup of mixed MSW (20 03 01) in the European Waste Classification System that comprises waste codes EAV-20030100-U for "not differentiable mixed municipal waste" and EAV-20030101-U for "household waste, household-type commercial waste" (DESTATIS, 2020a; EC, 2014a). Fractional composition of rMSW (see SI, Table S1) is assumed according to data by the German Environment Agency (UBA, 2020b). Reference data for chemical composition of each sub-fraction provided by the software EASETECH (Riber et al., 2009) is adapted based on Weigand and Marb (2006). The generated chemical characteristics of rMSW are available in Table S2 in the SI.

3.1.5. Database for waste treatment technologies

Data on waste treatment technologies is obtained from technical reports by the German government, scientific publications, LCA databases, and process modeling. For conventional waste treatment technologies – i.e. collection, transportation, waste incineration, MBT, and RDF incineration – inventory data is oriented towards information by UBA, 2018a, 2018b) for conventional treatment techniques in Germany. Data on CR is obtained via chemical process modeling based on the approach by Keller et al. (2020). Assumptions are documented below.

Collection: rMSW collection is modeled based on Larsen et al. (2009), assuming the combustion of 3.1 L diesel tonne^{-1} rMSW in a waste collection truck.

Transportation: Transportation of bottom ash, fly ash, stabilized residues, and CR slag is modeled based on data by Olesen (2013) for transporting waste in a semitrailer truck. The truck has a cargo volume of 10 tonnes and travels fully loaded on a motorway at an average speed of 80 km/h consuming 0.021 L diesel tonne^{-1} waste km^{-1}. One-way transportation is considered as the truck is assumed to be used for multiple logistical purposes.

Waste incineration: Similar to UBA (2018a), rMSW is incinerated in an MSWI consisting of the following units: i) delivery and storage, ii) grate-firing, iii) gas treatment, and iv) metal separation.

Delivery and storage is accounted for with the combustion of 0.5 L diesel tonne^{-1} in a wheel loader operated at 33% average load factor (Boldrin, 2012b). For the grate-firing unit, cogeneration is assumed at energy efficiencies of 17% for electricity and 32% for heat relative to the lower heating value of the feedstock (Wang et al., 2020). Furthermore, grate-firing consumes 2 L fuel oil tonne^{-1} rMSW as auxiliary fuel (EC, 2006). Electricity consumption for plant operation is assumed at 62 kWh tonne^{-1} rMSW (Wang et al., 2020). Excess electricity is fed into the German electricity grid, while excess heat is fed into a local heat grid.

For airborne emissions from incineration, a wet gas treatment unit in line with UBA (2018a) is assumed. This comprises a dry electrostatic precipitator unit for dust, a two-stage water scrubber unit for acid pollutants, a selective catalytic reforming unit for nitrogen oxides, and a fly-flow adsorption for heavy metals and dioxins. Total determined supply consumption for flue gas cleaning amounts to 12 kg calcium carbonate, 3 kg sodium hydroxide, 5.9 kg ammonia, and 0.8 kg activated carbon tonne^{-1} rMSW (EC, 2006; UBA, 2018a). Assumed process-specific air emissions of treatment include 140 g CO, 4.3 g N_2O, and 2.5 g CH_4 tonne^{-1} rMSW (Harris et al., 2015; Neuwahl et al., 2019). 260 kg fossil CO_2 tonne^{-1} rMSW is accounted for based on fossil carbon content in rMSW.

Based on Fruergaard et al. (2010), fly ash is assumed to be transported 700 km to a salt mine, where it is utilized as backfilling material consuming 1.8 L diesel and 28 kWh electricity tonne^{-1} fly ash. Bottom ash is processed in a metal-separation unit comprising of two overbelt magnets, a bar sizer, an impact crusher, a flip-flop sieve, and an eddy current separator with recovery efficiencies of 80% for ferrous and 19.5% for non-ferrous scraps as per Šyc et al. (2020). Remaining bottom ash is transported 2 km to a landfill, modeled with the combustion of 1 L diesel tonne^{-1} bottom ash in an 89 kW drivable machine charged on the average to 36% (Boldrin, 2012a; Montejo et al., 2013).

MBT of rMSW: Based on UBA (2018a), rMSW is treated in a MBT plant consisting of the following units: i) delivery and storage, ii) mechanical treatment, iii) anaerobic digestion, iv) tunnel composting, v) biogas combustion, and vi) gas treatment for flue gases.

Delivery and storage are assumed to be similar to waste incineration. In the mechanical treatment unit, rMSW is separated into i) ferrous scrap, ii) non-ferrous scrap, iii) RDF, and iv) an organic sorting fraction lower in calorific value (see SI, Table S3). Sorting efficiencies for metals are based on Montejo et al. (2013). RDF separation is based on Montejo et al. (2011). The organic fraction is modeled as the residual fraction after metal and RDF separation. Electricity consumption of sorting is estimated at 20 kWh tonne^{-1} rMSW according to UBA (2018a). Before RDF is transported to the next treatment step, i.e. RDF incineration or CR, it is dried to 5% moisture in a drum dryer consuming 60 kWh electricity and 850 kWh heat tonne^{-1} water as per Shu and Shu (2007). For RDF transportation to the second treatment step (situated at the same plant site), combustion of 0.5 L diesel tonne^{-1} RDF in a wheel loader operated at 33% average load factor is assumed (Boldrin, 2012b).

The organic sorting fraction is utilized in a subsequent wet anaerobic digestion step to generate biogas. Based on Naroznova et al. (2016), four assumptions are made for biogas generation: i) CH_4 yield of 70% of the degradable carbon, ii) product with 63% CH_4, iii) biogas leakage of

2.7%, and iv) consumption of 30.6 kWh heat and 0.9 L diesel tonne^{-1} organic sorting fraction. Biogas produced in the anaerobic digestion unit is utilized in a CHP stationary engine for energy provision with energy efficiencies of 50% for heat and 42% for electricity (Münster and Lund, 2010; UBA, 2018a). Note that the engine supplies heat and power requirements of other treatment steps. Similar to waste incineration, excess electricity and heat are fed into the German electricity and local heat grids respectively.

Besides biogas, anaerobic digestion produces a residual digestate fraction that is deposited as a stabilized residue following post-treatment via post-composting as per accepted practice in Germany (UBA, 2018a; van Praagh et al., 2009). For post-composting, a tunnel concept consuming 0.45 L diesel tonne^{-1} and 10 kWh electricity tonne^{-1} residual digestate fraction is assumed based on Boldrin et al. (2009). Reduction of methane formation potential during composting is modeled based on Cumulative Respiration Index (AT_4) data reported by van Praagh et al. (2009) for a German tunnel composting facility. Specifically, the composting process is assumed to reduce biodegradable carbon content to 5%. This is proportional to the decrease in degradation potential as indicated by AT_4. Besides biodegradable carbon, the total mass reduction during post-composting encompasses additional elements. Hence, based on Montejo et al. (2013), it is modeled as the percentage of versatile solid content in input waste namely 70% for organic fractions, 30% for mixed fractions with organic share, 15% percent for wood or paper fractions, and 5% for composites, textiles, or disposable sanitary products. Furthermore, moisture is reduced to 5% (Boldrin et al., 2011).

Subsequently, stabilized residue is deposited after 2 km transportation at an adjacent landfill modeled with the combustion of 1 L diesel tonne^{-1} in a 89 kW drivable machine charged on the average to 36% (Boldrin, 2012a). Minor methane emissions from deposited biodegradable carbon during a landfilling period of 100 years are modeled based on Manfredi and Christensen (2009). Specifically, landfill gas generation under moderate landfilling conditions is assumed at a state-of-art landfill with well monitored gas collection, flaring, and biocover. Gas collection is modelled with i) 45% efficiency until year 6, ii) 80% for years 6 to 11, iii) 95% for years 11 to 41, and iv) 0% for years 41 to 100. Carbon decay rates are assumed based on La Cruz and Barlaz (2010), eventually leading to an assumed total emission of 20 kg CO_2eq. tonne^{-1} stabilized residue.

Based on Rose (2002) and Pinasseau et al. (2018), MBT flue gas cleaning for the mechanical step consists of a fabric filter unit for dust, an acidic water scrubber for water-soluble organic components & ammonia, and a bio-filter unit for volatile organic compounds. Flue gas cleaning of the anaerobic digestion step consists of an acidic water scrubber for water-soluble organic components & ammonia and a regenerative thermal oxidation (RTO) unit for volatile organic compounds as well as potential methane leakage during anaerobic digestion and post-composting (Kvist and Aryal, 2019; Pinasseau et al., 2018). Water scrubbers use 3.5 kg nitric acid, while RTO unit requires 45 kWh biogas tonne^{-1} rMSW that is derived from anaerobic digestion (Thomé-Kozmiensky, 2002; UBA, 2018a). Electricity consumption for the entire flue gas cleaning system is estimated at 12 kWh tonne^{-1} rMSW (Rose, 2002). Based on Brinkmann et al. (2017) and Tonini and Astrup (2012), 41 g CO, 0.16 g N_2O, and 103 g CH_4 per m^3 CH_4 input are considered as emitted climate relevant air emissions from biogas utilization.

RDF incineration: Based on UBA (2018a), RDF is assumed to be incinerated in an RDF power plant consisting of the following units: i) grate-firing, ii) gas treatment, and iii) metal separation.

Based on extrapolation of assumptions for waste incineration according to UBA (2018b), RDF incineration is assumed to achieve energy efficiencies of 23% for electricity and 35% for heat. The internal plant consumption is assumed at 62 kWh electricity tonne^{-1} RDF (Wang et al., 2020). Excess electricity and heat are fed into the German electricity grid and a local heat grid respectively. The flue gas treatment unit and process-specific air emissions are assumed to be similar to waste incineration. 810 kg fossil CO_2 tonne^{-1} RDF is emitted based on fossil carbon content in RDF. Finally, management of incineration residues – i.e. metal separation, backfilling, and landfilling – are assumed to be similar to waste incineration.

CR of RDF: The utilized CR technology is assumed to be gasification. The CR plant consists of the following units: i) pretreatment, ii) gasification, iii) gas treatment, iv) synthesis, and v) auxiliary facilities.

Pretreatment unit includes shredding and compacting of RDF consuming 3.7 kWh electricity tonne^{-1} (Poganietz et al., 2019). Inventory data for CR is produced via process modeling with ASPEN Plus V10 for olefin production from RDF as presented in Keller et al. (2020). Specifically, the gasification unit includes a fixed-bed gasification with liquid slag extraction (BGL type). The syngas treatment unit consists of a water scrubbing unit, a two-stage adiabatic CO shift unit, a selective acid gas removal unit, and a three-stage Claus plant. The synthesis unit includes a syngas-to-methanol synthesis, a methanol-to-olefins synthesis, and an olefins recovery stage.

Total process consumptions are 7.3 kg sodium hydroxide and 400 kWh electricity tonne^{1} RDF. From the process, 240 kg olefins tonne^{-1} RDF (120 kg ethylene, 130 kg propylene) is recovered for utilization, i.a. for plastic production. Furthermore, 0.76 kg fuel gas, 11 kg LPG, 4.2 kg butadiene, 1.9 kg butene, 710 kWh heat as process steam, and 2.08 kg sulfur tonne^{-1} RDF are obtained for versatile applications at chemical plants and/or for export. Assumed air emissions are 405 kg fossil CO_2 tonne^{-1} RDF and 42 g CO. For solid emissions, 99 kg slag is transported 2 km to an adjacent landfill, modeled with the combustion of 1 L diesel tonne^{-1} in a 89 kW drivable machine charged on the average to 36% (Boldrin, 2012a).

3.1.6. Impact assessment

GWP is assessed as per Fifth Assessment Report (AR5) of IPCC (Pachauri and Mayer, 2015). Further impacts (e.g. acidification, eutrophication, and land use) commonly addressed in LCA are beyond the current scope. Additionally, biogenic carbon emissions are assumed to have a global warming emissions factor of zero. Time horizon for GWP is set to 100 years, i.e. GWP100.

3.1.7. Sensitivity analysis for GWP

The German energy supply is currently undergoing massive transformation under the Energiewende, i.e. energy transition (BMWI, 2020). As emission credits for the substitution of conventional energy sources (cf. Section 3.1.2) can significantly impact GWP100 of waste treatment practices, a corresponding sensitivity analysis is conducted. Specifically, besides the basic energy provision scenario (BASIC), two additional energy scenarios namely predominantly renewable energy provision (PREP) and climate neutral energy provision (CNEP) are generated based on EC (2014b) data for electricity footprint and BDI (2018) data for heat footprint. In BASIC scenario, 0.4 kg CO2eq. kWh^{-1} for electricity and 0.26 kg CO2eq. kWh^{-1} for heat are assumed (BDI, 2018; UBA, 2020a). In contrast, 0.18 kg CO2eq. kWh^{-1} for electricity (33% natural gas, 67% renewables) and 0.17 kg CO2eq. kWh^{-1} for heat (22% gas, 78% renewables including waste incineration and power-to-heat) are assumed for PREP scenario, while 0.011 kg CO2eq. kWh^{-1} for electricity (100% wind energy) and 0.012 kg CO2eq. kWh^{-1} for heat (100% power-to-heat from wind energy) are assumed for CNEP scenario. Note that indirect impacts of energy provision such as diffusion of green technologies for the production of metals or chemicals are beyond the scope of this research, i.e. GWP100 values for supplies or other treatment products reported in Table 1 are assumed to be constant.

3.2. Techno-economic analysis

Based on Peters et al. (2004), Sinnott and Towler (2020), and Friedman et al. (2020), FCI, NPV, DPP, and LCCA are utilized to assess the economic performance of the three treatment pathways considered.

Specifically, as per typical sizes of modern waste treatment plants in Germany, each pathway is calculated based on an assumed treatment capacity of 300,000 tonnes rMSW year^{-1} (UBA, 2018b). All calculations assume 2021 as the base year, i.e. year of plant investment.

3.2.1. Fixed capital investment

As illustrated in Equation (1), a power factor applied to capacity ratio approach is utilized to calculate FCIs for the individual treatment plants (Peters et al., 2004). Investment data for individual plant components including reference capacities (see SI, Table S4) is mainly drawn from UBA (2018a) for conventional waste treatment processes and from Poganietz et al. (2019) for CR. These are multiplied by the ratio of targeted capacity to reference capacity, raised to a power factor of 0.7 and adjusted by a cost index ratio (Peters et al., 2004). For temporal price indexing, the Chemical Engineering Plant Costs Index (CEPCI) is applied and extrapolated to 2021 (see SI, Table S5) based on linear regression (Chemical Engineering, 2020). Note that assumed corrected FCI values for the individual plant components are provided in Table 2.

$$FCI = \sum_{pc} FCI_{pc} \bullet \left(\frac{\text{cap}}{\text{cap}_{\text{rp}}}\right)^{\kappa^{\text{inv}}} \bullet \left(\frac{\text{pindex}}{\text{pindex}_{\text{ry}}}\right) \quad (1)$$

with
cap … Capacity [t]
cap_{rp}… Capacity of reference plant [t]
FCI … Fixed capital investment [€]
FCI_{pc} … Fixed capital investment for plant component [€]
pc … Plant component index [-]
$pindex$… Price index in year of construction [-]
$pindex_{ry}$ … Price index in reference year [-]
κ^{inv} … Scaling power factor [-]

3.2.2. Net present value

NPV is the difference between the total present value of all cash flows and the present value of all capital investment as defined in Equation (2) (Peters et al., 2004). For NPV calculation, FCI in the year of construction is considered, i.e. 2021. Furthermore, a construction period of three years is assumed (Bator et al., 2018). After plant commissioning in 2024, cash flows for i) local taxes and insurances, ii) maintenance and repair, iii) supplies, iv) labor, v) overhead costs, vi) administration costs, vii) environmental expenses, viii) transportation, ix) revenues, and x) income tax are calculated over the entire plant operating time, assumed unified at 30 years for all treatment plants (Fei et al., 2018). Cash flow calculations are based on Peters et al. (2004) as presented in Equations (3) to (13). Moreover, 90% plant availability is assumed for all treatment plants (SWB, 2020; Wolfersdorf et al., 2017). Discount rate for NPV calculation is 8% (Fei et al., 2018). Process-specific labor requirements in Table 2 are based on reported requirements for German plants (Poganietz et al., 2019; Lauta, 2021; Veolia, 2021; VGB e.V., 2009). General economic assumptions such as tax/inflation rates are presented in Table 3.

Table 2
Economic plant characteristics for treatment steps. CR: Chemical recycling. FCI: Fixed capital investment. MBT: Mechanical-biological treatment. MEUR: Million euros. RDF: Refuse-derived fuel. RDFi: Refuse-derived fuel incineration. rMSW: Residual municipal solid waste. TEH: Thousand employee hours. WI: Waste incineration. Values rounded to two significant digits.

	Unit	WI	MBT	RDFi	CR
Input	–	rMSW	rMSW	RDF	RDF
Capacity	kt/a	300	300	76	76
Unit FCI values					
Anaerobic digestion	MEUR	–	32	–	–
Auxiliary facilities	MEUR	–	–	–	4.8
Biogas combustion	MEUR	–	6.8	–	–
Delivery and storage	MEUR	4.8	4.8	–	–
Gas treatment	MEUR	53	7.5	20	17
Gasification	MEUR	–	–	–	84
Grate-firing	MEUR	100	–	37	–
Mechanical treatment	MEUR	–	19	–	–
Metal separation	MEUR	1.8	–	0.67	–
Pretreatment	MEUR	–	–	–	6.8
Synthesis	MEUR	–	–	–	47
Tunnel composting	MEUR	–	14	–	–
Labor requirements					
Operating labor (skilled)	TEH/a	14	16	9	7.4
Operating labor (unskilled)	TEH/a	78	91	55	42

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

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

$$CF_1 = FCI \bullet (\text{insur} + \text{lTax}) \quad (3)$$

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

$$CF_2 = FCI \bullet \text{maint} \bullet (1 + \text{consum}) \quad (4)$$

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

$$CF_3 = \sum_s requi_s \bullet \text{price}_s \quad (5)$$

with
CF_3 … Supplies [€]
$price_s$ … Price for s [€/t]
$requi_s$ … Total requirements s [t]
s … Supply [-]

$$CF_4 = (sEmploy^* sRate + usEmploy^* usRate) \bullet (1 + \text{supervis}) \quad (6)$$

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

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

with
CF_4 … Labor [€]
CF_5 … Overhead [€]
$overhd$ … Overhead percentage [%]

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

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

Table 3
General economic assumptions for treatment cost calculation. CR: Chemical recycling. I: Incineration. MBT: Mechanical-biological treatment. RDF: Refuse-derived fuel. All prices assumed for year 2021. Values rounded to two significant digits.

	Unit	Value	Source
Administration percentage	%	25	(Peters et al., 2004)
CO_2 certificate price	€/t CO_2	25	(BMU, 2019)
CO_2 certificate price inflation	%	5	(Edenhofer et al., 2019)
Consumables percentage	%	20	(Peters et al., 2004)
Corporate income tax rate	%	15	(BfJ, 2020c)
CR catalyst cost	€/t RDF	7.1	(Poganietz et al., 2019)
Depreciation period (linear)	a	10	(BdF, 2000)
Insurances percentage	%	1	(Peters et al., 2004)
Labor cost inflation rate	%	2.6	(DESTATIS, 2020b)
Labor power factor	–	0.25	(Peters et al., 2004)
Local taxes percentage	%	2	(Peters et al., 2004)
Maintenance & repair percentage (I/CR/MBT)	%	3/4/7	(Peters et al., 2004; Thanopoulos et al., 2020; Yassin et al., 2009)
Overhead percentage	%	50	(Peters et al., 2004)
Price inflation	%	1.4	(DESTATIS, 2021b)
rMSW gate fee	€/t	140	(EUWID, 2020)
Scaling power factor	–	0.7	(Peters et al., 2004)
Skilled employee hour	€/h	80	(Stepstone, 2021)
Supervision percentage	%	15	(Peters et al., 2004)
Transportation cost rate	€/t/km	0.07	(UBA, 2018a)
Unskill. employee hour	€/h	63	(Stepstone, 2021)

$$CF_7 = \sum_{em} outp_{em} \bullet \text{costs}_{\text{em}} \tag{9}$$

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

$$CF_8 = \sum_{str} outp_{str} \bullet \text{dist}_{\text{str}} \bullet \text{tRate} \tag{10}$$

with
CF_8 … Transportation [€]
$dist_{str}$ … Transportation distance for *str* [km]
$outp_{str}$ … Output of *str* [t]
str … Solid treatment residue [-]
tRate … Transportation cost rate [€/t/km]

$$CF_9 = \sum_{prod} outp_{prod} \bullet \text{price}_{\text{prod}} \tag{11}$$

with
CF_9 … Revenues [€]
$outp_{prod}$ … Output of *prod* [t]
$price_{prod}$ … Selling price for *prod* [€/t]
prod … Product [-]

$$tinc = \sum_{k=1}^{9} CF_k - depr \tag{12}$$

$$CF_{10} = \begin{cases} tinc^{*}itax & tinc > 0 \\ 0 & tinc \leq 0 \end{cases} \tag{13}$$

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

3.2.3. Dynamic payback period

DPP refers to the period required to recover the investment considering the time value of money (Chen et al., 2020). It is calculated from FCI and discounted net annual cash flows (cf. Equation 14). Due to the assumption of cash flows occurring at the end of each year, DPP can only have integer values.

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

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

3.2.4. Levelized cost of carbon abatement

LCCA refers to an investment on the basis of euros per tonne of emissions reduced (Friedman et al., 2020). Discounted additional costs for a plant investment are divided by the total carbon reduction over the entire plant operating time (cf. Equation 15). Note that waste incineration i.e. PI – the current standard treatment for rMSW in Germany – is defined as the reference process for LCCA calculations (DESTATIS, 2021a).

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

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

3.2.5. Sensitivity analysis for costs

Three factors are identified as having significant impacts on NPV, DPP, and LCCA namely i) plant scaling (PS), ii) environmental regulation (EREG), and iii) market conditions (MC) influencing prices of CR products. As these factors are associated with significant uncertainty, the basic scenario is extended to include a sensitivity analysis. Specifically, capacity of all treatment plants is doubled (+100%) to assess for PS effects. Note that power factors of 0.7 for FCI and 0.25 for labor are applied (Peters et al., 2004). For EREG effects, conventional MSWI – so

far exempted in Germany – is assumed to be subjected to emissions trading (UBA, 2019). Furthermore, CR is assumed to be exempted from emissions trading. To evaluate MC effects for CR products, prices for CR outputs are assumed to be doubled (+100%), i.e. price premium. Finally, the aggregated effect (AE) of all factors is examined in an additional scenario.

4. Results

4.1. Global warming potential

To facilitate a basic contribution analysis, emissions and substitution effects are differentiated according to i) process emission including use of supplies, transportation, and deposition, ii) heat substitution, iii) electricity substitution, iv) primary steel substitution, v) primary aluminum substitution, and vi) chemical product substitution (see Fig. 2). Total impacts are derived via subtracting substitution effects as avoided impacts from induced impacts, i.e. process emissions.

In the BASIC scenario, while all treatment pathways are observed to contribute to reducing CO_2eq. emissions, the potential for GWP100 reduction is observed to be highest for PII, followed by PI and PIII (-150, -110, and -87 kg fossil CO_2eq. tonne^{-1} rMSW respectively). As incineration-based pathways for energy production, both PI and PII benefit significantly from heat and electricity substitution effects. However, PII exhibits lower process emissions at comparable electricity production due to high efficiencies of biogas and RDF utilization. Moreover, the higher primary aluminum substitution further contributes to reducing its GWP100 potential. Hence, although PII exhibits lower heat substitution effects compared to PI due to its internal heat consumption for anaerobic digestion and RDF drying during MBT (see SI, Table S6 for energy balance), its combined GWP100 reduction potential is considerably higher than that of direct incineration via PI.

Considering PIII, it exhibits the lowest process emissions of the considered pathways. Not only is 28 kg of fossil carbon bonded into CR products, another 15 kg of fossil carbon is deposited as part of the stabilized composting residues from MBT treatment due to mis-sorting (see SI, Table S7 for fossil carbon balance). However, PIII benefits from lower heat substitution effects compared to the incineration-based pathways and enjoys almost no electricity substitution effect. This is because electricity production from RDF is missing and CR acts as a significant consumer of electricity produced via MBT (see SI, Table S6).

In the PREP scenario, substitution effects for heat and electricity are reduced. This significantly impacts the GWP100 of incineration-based pathways. At 61 kg and 8.4 kg fossil CO_2eq. tonne^{-1} rMSW, PI and PII now contribute to an increase in CO_2eq. emissions. In the CNEP scenario where substitution effects for heat and electricity approach zero, CO_2eq. emissions for PI and PII will further increase to 270 and 180 kg fossil CO_2eq. tonne^{-1} rMSW respectively. In contrast, CR is not so strongly impacted by changes in the reference energy system as its focus is on substituting chemical production and not energy/heat production. Hence, even with increasing renewable energy supplies in scenarios PREP and CNEP, PIII continues to contribute to the reduction of CO_2eq.

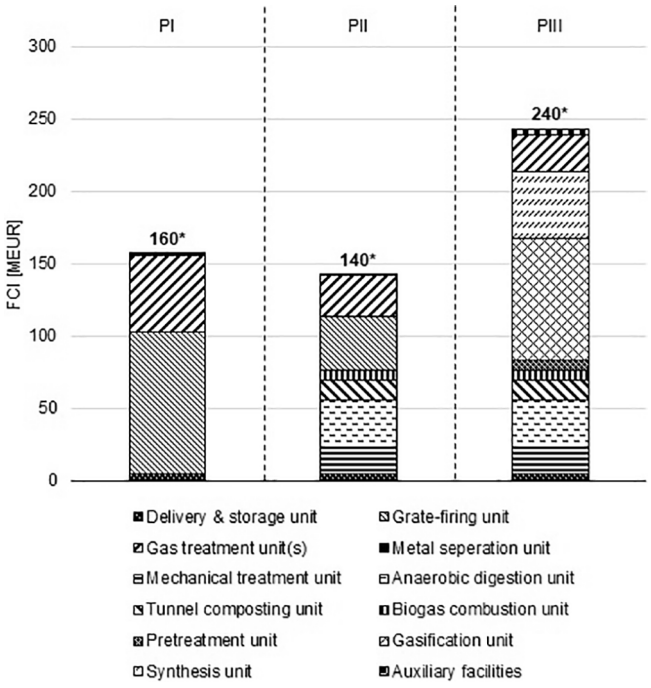

Fig. 3. Fixed capital investment (FCI). MEUR: Million euros. *: Resulting sum. Values rounded to two significant digits.

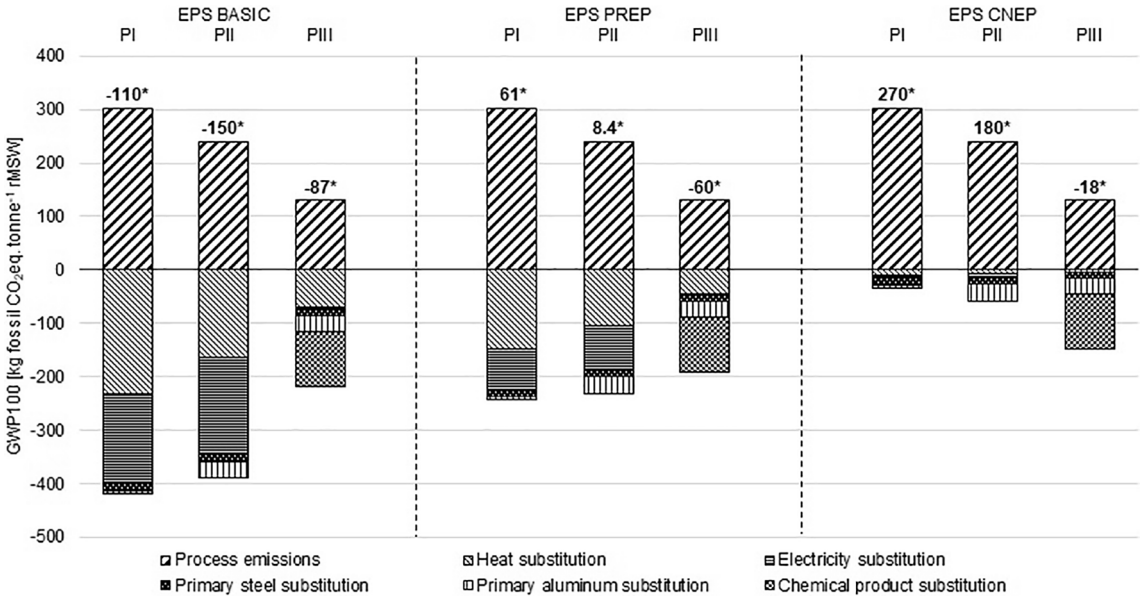

Fig. 2. Global warming potential for all treatment pathways. BASIC: Basic scenario. CNEP: Climate neutral energy provision. EPS: Energy provision scenario. PREP: Predominantly renewable energy provision. *: Total impacts. Values rounded to two significant digits.

emissions, albeit at lower levels.

4.2. Economic performance

4.2.1. Fixed capital investment

As presented in Fig. 3, PIII exhibits the highest FCI, followed by PI and PII at 240, 160 and 140 MEUR respectively. For incineration-based routes, grate-firing and gas treatment units are the main cost drivers. An additional cost driver for PII is the anaerobic digestion unit. In contrast, FCI for PIII is heavily affected by investments for CR plant components, particularly the gasification and synthesis units. Altogether, findings point to comparable investments for PI and PII for a capacity of 300,000 tonnes rMSW year^{-1}, while FCI for PIII is about 60% higher.

The calculated FCI for PI and RDF incineration in PII are in line with data reported in extant literature (Neuwahl et al., 2019). However, higher values are obtained for MBT compared to literature references of average MBT plant investments in Germany (UBA, 2018a). This is due to the complex process configuration including biogas utilization and RDF drying (UBA, 2018b) which is assumed for PII and PIII in the current investigation. Note that an overestimation of FCI would result in an underestimation of NPV for these treatment routes. For CR, the calculated FCI agrees with figures reported e.g. by Koukouzas et al. (2008) for small-scale gasification plants.

4.2.2. Net present value

NPV ranges in the economic scenario BASIC from -160 MEUR to -45 and 79 MEUR for PIII, PII, and PI respectively (see Fig. 4a). Hence, PI is profitable, while PII and PIII exhibit financial losses. In scenario PS, NPV increases all around to 70, 220, and 360 MEUR for PIII, PII, and PI respectively. This suggests that plant scaling is a crucial factor in determining economic performance of treatment pathways for rMSW, incineration-based or otherwise.

In economic scenario EREG, the inclusion of rMSW incineration in emission trading decreases the profitability of PI in scenario BASIC significantly. This suggests that regulations to penalize CO_2 emissions from direct waste incineration could function as an indirect economic incentive to motivate developing alternative waste treatment for rMSW. While the exclusion of CR is observed to have a slight positive impact in

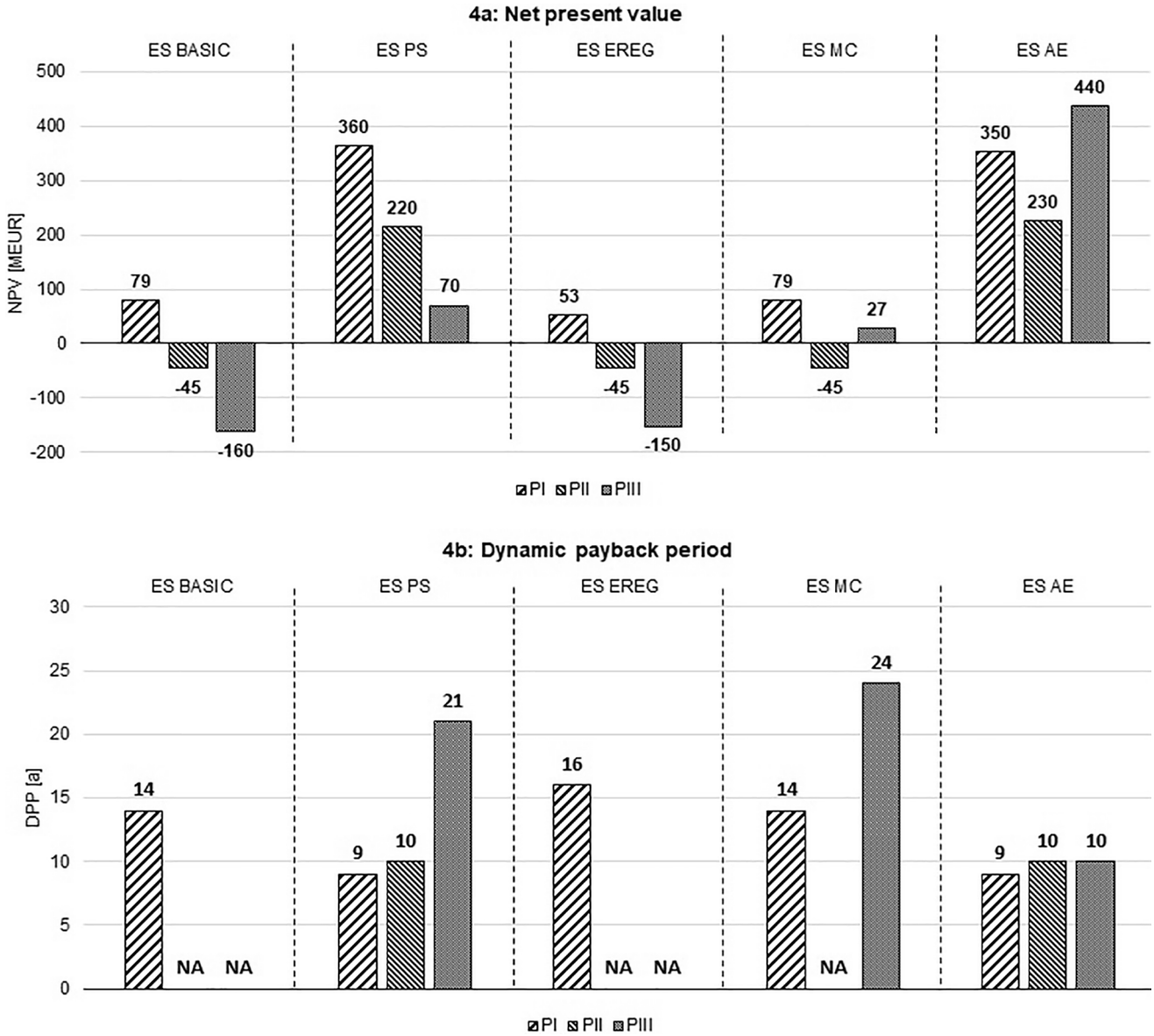

Fig. 4. Net present value (NPV) and dynamic payback period (DPP). AE: Aggregated effect. BASIC: Basic scenario. EREG: Environmental regulation. ES: Economic scenario. MC: Market conditions. MEUR: Million euros. NA: Not applicable. PS: Plant scaling. Values rounded to two significant digits.

reducing PIII's financial losses, due to low process emissions, its effectiveness as an economic incentive is lower than penalizing CO_2 emissions from MSWI.

Economic scenario MC indicates that a price premium for CR products will significant impact the profitability of PIII. A doubling of CR product prices translates into a profitability for PIII. In integrating the aggregated effects of all factors in scenario AE, results suggest that a multi-pronged approach integrating upscaling as well as both indirect economic incentive via penalizing of MSWI's emissions and direct economic incentive via price premium for CR products could increase CR's profitability such that it would be economically more attractive than incineration-based treatment pathways.

4.2.3. Dynamic payback period

For DPP (see Fig. 4b), note that it cannot be calculated for pathways and scenarios with negative NPV. DPP for PI in economic scenario BASIC is 14 years. In scenario PS, DPP ranges from 9 to 10 and 21 years for PI, PII, and PIII respectively. PI thus exhibits the best project practicability. In economic scenario EREG, DPP for PI at 16 years is slightly higher than in scenario BASIC. This suggests that inclusion in emission trading has a measurable impact on the project practicability of PI. In scenario MC, similar to scenario BASIC, a DPP of 14 years is observed for PI, while a DPP of 24 years is observed for PIII. In integrating all effects in scenario AE, a similar DPP is observed for all treatment pathways indicating similar project practicability.

4.2.4. Levelized cost of carbon abatement

LCCA is calculated for PII and PIII based on reference pathway MSWI (c.f. Section 3.2.4). As LCCA is dependent on carbon emissions, which is in turn dependent on the reference energy scenario described in Section 3.1.7, the economic scenarios BASIC, PS, EREG, MC, and AE are assessed for different energy provision scenarios, i.e. BASIC, PREP and CNEP as illustrated in Fig. 5.

Under energy provision scenario BASIC, LCCA for PII ranges from 890 euros tonne^{-1} CO_2 eq. in economic scenario AE to 1700 euros tonne^{-1} CO_2 eq. in economic scenarios BASIC and MC. Note that under energy provision scenario BASIC, LCCA for PIII cannot be calculated as there is no carbon saving in this scenario (cf. Section 4.1). With increasing renewable energy provision, i.e. energy provision scenarios PREP and CNEP, LCCA associated with PIII is significantly lower than for PII. Interestingly, in contrast to PII, PIII exhibits negative LCCA in the economic scenario AE for energy provision scenarios PREP and CNEP. This is because PIII is more profitable and produces less CO_2eq. compared to PI, hence negative costs for carbon abatement are observed.

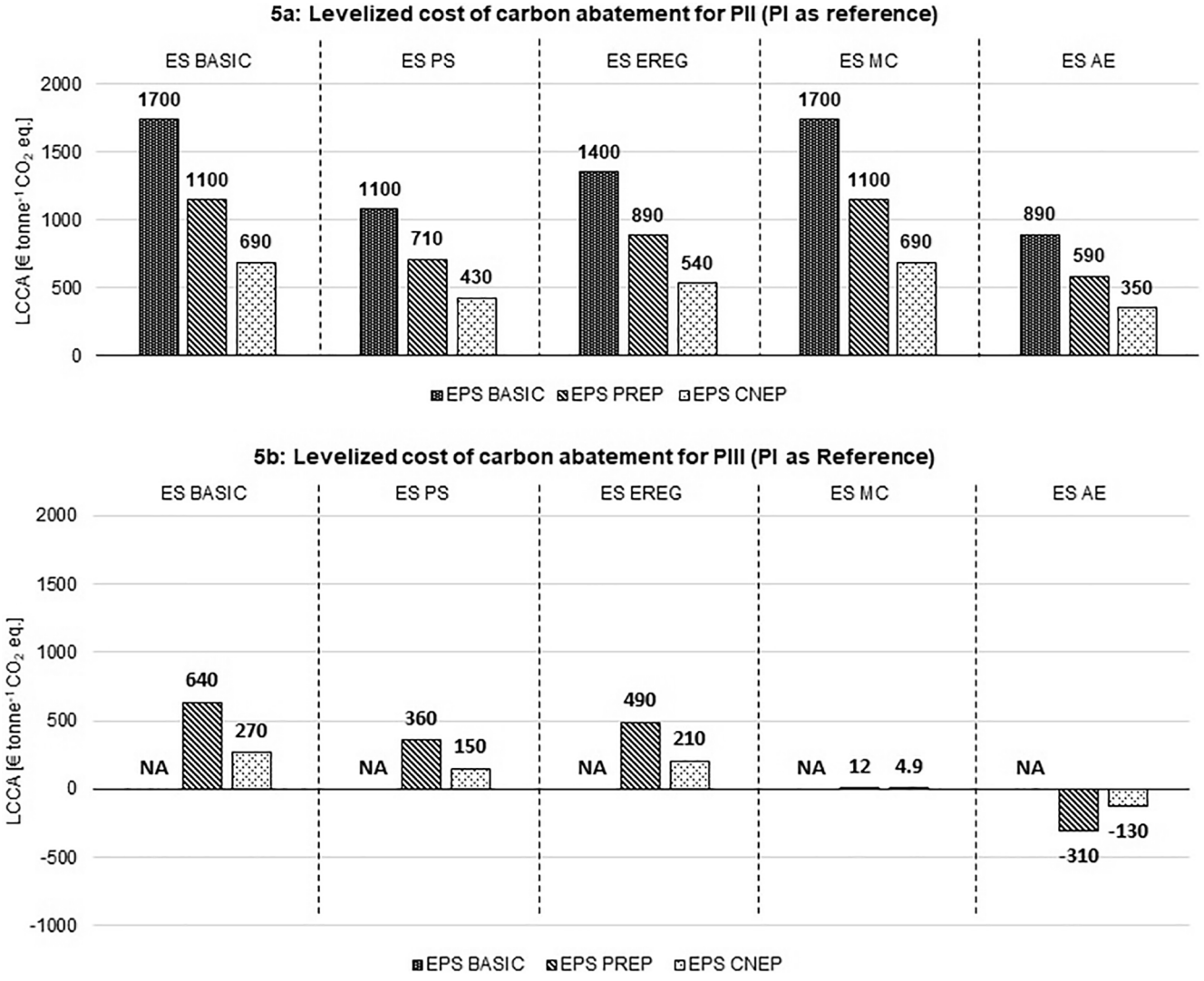

Fig. 5. Levelized cost of carbon abatement (LCCA). AE: Aggregated effect. BASIC: Basic scenario. CNEP: Climate neutral energy provision. EPS: Energy provision scenario. ES: Economic scenario. EREG: Environmental regulation. MC: Market conditions. PREP: Predominantly renewable energy provision. PS: Plant scaling.

5. Discussion and conclusion

This study comparatively assesses global warming potential and economic performance associated with chemical recycling, direct, and indirect incineration treatment pathways of residual municipal solid waste in Germany. From the perspective of carbon emissions, while all treatment pathways are associated with negative global warming potential, that of incineration-based treatment pathways (both direct and indirect) is observed to be strongly dependent on the reference energy system. An increased proportion of renewables in the energy system significantly reduces the global warming reduction potential of incineration-based pathways. In contrast, chemical recycling continues to exhibit – albeit reducing – global warming reduction potential. In view of the announced goals of carbon neutrality by 2050 and 2060 by the EU and China respectively (EC, 2018; Mallapaty, 2020), the relevance of chemical recycling as a climate friendly treatment pathway for residual municipal solid waste is therefore anticipated to increase.

From an economic perspective, chemical recycling requires higher fixed capital investment compared to direct and indirect incineration. However, considering the net present value variations due to changes in plant scaling, environmental regulations, and market conditions, findings suggest that a multi-pronged approach integrating upscaling with both an indirect economic incentive via penalizing CO_2 emissions from municipal solid waste incinerators and a direct economic incentive via price premium for chemical recycling products would increase the profitability of chemical recycling such that it would be comparable or even more attractive than incineration-based treatment pathways. In terms of dynamic payback period, such a multi-pronged approach is also necessary for chemical recycling investments to exhibit a similar project practicability as incineration investments. Levelized cost of carbon abatement calculations based on direct incineration as reference pathway suggest that costs for chemical recycling will be significantly lower than that of indirect incineration with increasing renewable energy provision in the energy system.

Altogether, results suggest that chemical recycling has a potential to contribute to reducing CO_2 emission from the treatment of residual municipal solid waste, and that its contribution and thus relevance will increase as the proportion of renewable energy increases in the energy system. However, to realize this positive ecological impact, higher fixed capital investment compared to conventional incineration-based pathways is required. Currently, under the existing energy system where CO_2 emissions from direct incineration of residual municipal solid waste are not penalized, the economic attractiveness of chemical recycling is limited. However, up-scaling coupled with regulatory changes to enable direct and indirect economic incentives for chemical recycling will increase its profitability compared to conventional incineration-based treatment pathways.

In executing an analysis and assessment of chemical recycling in comparison to established incineration-based treatment pathways for residual municipal solid waste, this study contributes to extant literature on LCA and TEA of chemical recycling technologies by providing comprehensive data on different treatment pathways for residual municipal solid waste. Additionally, it identifies the conditions – and the associated costs – under which chemical recycling could make a significant ecological contribution.

However, although the analyses – in line with proven practices of LCA and TEA – account for upstream and downstream effects, the question regarding what should be included beyond the system boundaries remains open. The integration of chemical recycling into existing complex and established value chains of waste treatment, chemical production, and energy provision in Germany or other nations requires a deep understanding of integration possibilities as well as insights into dynamic developments in these sectors. For instance, a large-scale implementation of chemical recycling could lead to gaps in local or national energy provision for nations where waste-to-energy contributes to baseload and/or is used to stabilize fluctuating input from renewable energy sources. Furthermore, the economic performance of chemical recycling could be improved with concepts integrating multiple mechanical–biological treatment with a single large-scale chemical recycling plant. Comparable systems are i.a. discussed for second generation biofuel production (Rudi et al., 2017; Zimmer et al., 2017). To model such perspectives and possibilities, future research could benefit by integrating data/results from the current investigation with systemic modeling approaches that are capable of mapping relevant waste, energy, and chemical systems more comprehensively to include their infrastructural, regulatory, and financial dynamics.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supplementary material

Supplementary data to this article can be found online at https://doi.org/10.1016/j.wasman.2021.07.040.

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