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RESEARCH ARTICLE

# How do political steering instruments influence the integration of chemical recycling into plastic recycling networks?

## A case study for Germany

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## Abstract

A steadily increasing plastic production requires the treatment of ever greater amounts of plastic waste. Plastic waste that is unsuitable for mechanical recycling is incinerated in Europe, generating large amounts of $CO_2$. Chemical recycling of such plastic waste offers an alternative to reduce the climate change impact (CCI) of plastic waste treatment and contributes toward closing the plastic loop. This paper presents a strategic location optimization model designed to effectively integrate chemical recycling facilities into existing waste treatment networks. The model optimizes material flows for both operational costs and CCI and is extended by using a goal programming approach to balance both objectives. The study focuses on two significant waste streams: (1) lightweight packaging, which in Germany accounts for 59% of post-consumer plastic waste, with 34% of it being unsuitable for mechanical recycling, and (2) automotive plastic waste, which as engineering plastics from the automotive sector presents a challenge for recycling due to its complex composition. Moreover, the study explores scenarios to evaluate political instruments and quantifies their impacts on the German plastic waste treatment network. It quantifies the impact of an increase of a legal national recycling rate and the extension of emission trading systems to the German waste sector as it was implemented in early 2024. Raising the recycling rate to 65% or introducing a $CO_2$ emission fee of 45€/t $CO_2$ could reduce the waste treatment network's CCI by up to 64% while increasing the circularity and decarbonization of plastics.

**KEYWORDS**
chemical plastic recycling, industrial ecology, pyrolysis of mixed post-consumer waste, reverse logistics, strategic network design, waste management

# 1 INTRODUCTION

Global plastic production reached 367 million metric tons[1] in 2022 (Plastics Europe, 2022) and continues to grow (IEA, 2018), amplifying its environmental impact due to energy-intensive and fossil-based production processes (Cabernard et al., 2021; IPCC, 2022). The European Waste Directive promotes the efficient use of natural resources and circular economy principles (European Parliament & Council, 2008), like post-consumer recycling technologies (Arena & Ardolino, 2022). In this study, we introduce a novel, spatially explicit optimization framework that employs mixed-integer linear programming (MILP) to identify cost- and $CO_2$-optimal waste treatment networks and applies goal programming to explore trade-offs between these conflicting objectives under different policy scenarios.

Mechanical recycling repurposes plastics by breaking them down and remolding them, often yielding lower-quality products (Schyns & Shaver, 2021). However, these technologies face challenges in dealing with mixed or polluted waste, resulting in downcycling and low recycling rates (Cossu & Lai, 2015; Pivnenko et al., 2015). Alternative waste treatment options, like chemical recycling, are being explored to address these limitations. Despite its higher energy consumption and material loss (Möck et al., 2022), chemical recycling is increasingly recognized as an essential solution for plastic waste that cannot be processed mechanically, particularly to achieve circular economy goals (Jeswani et al., 2021). Chemical recycling breaks plastics into their basic components or monomers, enabling high-quality material recovery. Pyrolysis, a key technology, decomposes plastic waste at high temperatures without oxygen (Qureshi et al., 2020).

The research underscores pyrolysis' potential to reduce the carbon footprint of plastic waste treatment (Hermanns et al., 2023; Jeswani et al., 2021; Klotz et al., 2024). Comparative studies highlight its economic and environmental advantages for lightweight packaging (LWP) and automotive plastic waste (APW). Combining mechanical and chemical recycling optimizes outcomes for LWP, while pyrolysis outperforms incineration for APW (Stallkamp et al., 2023; Volk et al., 2021). However, economic challenges persist, as energy recovery remains the more cost-effective option of plastic recycling in certain scenarios, highlighting a conflict of objectives (Fivga & Dimitriou, 2018; Gracida-Alvarez et al., 2019).

Technical hurdles further complicate the adoption of pyrolysis. Feedstock contamination or variations in feedstock composition, for example, caused by seasonal or local differences in waste collection and waste composition, lead to variations in pyrolysis oil quality, affecting its potential applications and complicating its integration into existing production processes (Kusenberg, Eschenbacher, Delva et al., 2022; Stallkamp et al., 2024). These issues emphasize the importance of ongoing research and optimization to enhance the scalability and efficiency of chemical recycling systems.

LWP and APW have emerged as focal points of current research due to their substantial contribution to post-consumer plastic waste[2] and their inherent challenges for mechanical recycling. Contamination in LWP and the complex material compositions of APW make them difficult to recycle mechanically.

Given the waste sector's critical role in closing the circularity of plastics, economic challenges need to be addressed and studied jointly, for example, by implementing targeted subsidies and regulatory reforms (Stallkamp et al., 2024; Voss et al., 2022). Possible levers to support the circularity of plastics are increasing mandatory recycling rates or expanding $CO_2$ pricing instruments to the waste sector.

To systematically assess the efficiency of such policy instruments, integrated system models serve as essential tools to evaluate their impacts, allowing for the exploration of interactions between processes, material flows, and steering effects of policy instruments. Unlike isolated assessments of individual technologies, such integrated models can capture the complexities of the waste sector, including the steering effects of recycling rates, $CO_2$ pricing, and the potential role of chemical recycling. While some existing studies explore process and material interactions (Liu et al., 2019; Oliveux et al., 2015; Sommer et al., 2022; Vo Dong et al., 2015, 2018), they fall short of analyzing the full scope of policy instruments and their infrastructural, economic, and logistical implications, such as economies of scale and transportation costs.

Therefore, this study develops a strategic location and capacity optimization model for a future waste management network in Germany, mutually integrating (novel) technologies and material flows while considering economic and environmental effects and demonstrating the critical role of policy instruments in designing a waste treatment network. With this model, we offer both a methodological and contextual contribution to the field of waste management. Methodologically, we combine MILP and a goal programming (GP) approach to examine trade-offs between costs and climate impact. Contextually, the model provides a novel quantification of how policy instruments affect infrastructure and value chain design for chemical recycling technology integration in Germany. This addresses a key gap in the literature, where policy instruments are often discussed in theory but rarely operationalized within spatially explicit optimization models.

# 2 METHODS

This section outlines the end-of-life (EoL) treatment options, defined scenarios, and model framework, followed by a multi-objective decision-making (MODM) model formulation for designing a future recycling system. Figure 1 depicts the modeled system for managing LWP (black lines) and APW (blue lines) waste flows, processed at facilities along two primary EoL paths: energy recovery and chemical recycling. Life cycle stages include sorting, dismantling, pretreatment, and recovery, culminating in one of two recovery paths. For clarity, Figure 1 shows flows treated sepa-

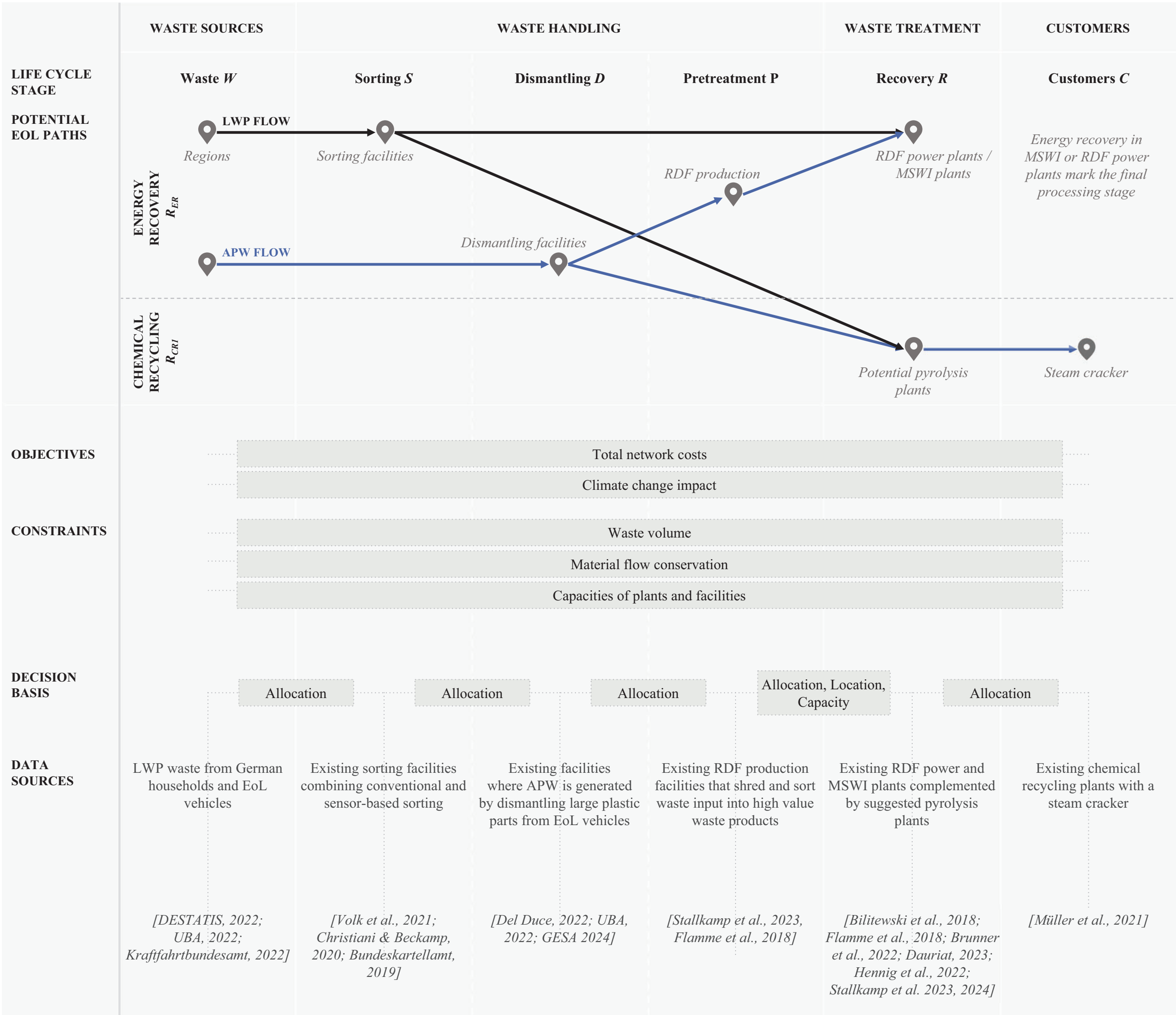


**FIGURE 1** Model framework for optimizing waste treatment networks: Potential end-of-life (EoL) paths, objectives, constraints, and data used for lightweight packaging and automotive plastic waste. The pins mark key stages of the waste management network, while the arrows represent material flows of LWP and APW between the processing facilities.

rately before the waste treatment stage. However, in practice and in the mathematical model, facilities may process both waste types, leading to multiple end nodes.

## 2.1 | Waste treatment network scenarios

In this study, we develop two complementary optimization models to evaluate both technological configurations and policy-driven strategies discussed within Germany's waste management network. **Model 1** optimizes the waste treatment route for LWP sorting residues and APW, considering both the status-quo energy recovery route and a pyrolysis-based chemical recycling route, identifying optimal plant locations.[3] It functions as a single-objective optimization model designed to either minimize costs or CCI.

We assume that LWP waste accumulates in district centers and is processed using existing German infrastructure. APW is dismantled in dismantling facilities.[4] It is assumed that the EoL vehicles accumulate in the district centers and are allocated to the closest facility with open capacity. The dismantled APW is sent to refuse-derived fuel (RDF) producers. Currently, RDF and LWP sorting residues are used as feedstock for energy recovery. Alternatively, pyrolysis plants of different sized plant capacities can be constructed (see Section 1.2 of Supporting Information S1; Stallkamp et al., 2024) to handle both waste fractions either individually or jointly. We assume that each district in Germany is a potential location for a pyrolysis plant. This single-objective model is optimized to minimize either network costs or CCI. Economic optimization focuses on network costs, avoiding revenue uncertainties from fluctuating prices and demand.

Building on Model 1's cost- and CCI-optimal solutions, **Model 2** implements a bi-objective goal-programming (GP) approach that integrates policy constraints directly into the optimization. This allows simultaneous consideration of economic and environmental objectives, revealing the trade-offs and quantifying how specific policy instruments can steer the network away from the cost-minimal configuration toward reduced emissions.

To assess the impact of regulatory measures, we construct 10 scenarios centered on two policy levers: recycling mandates and the expansion of the emissions trading system (ETS). These instruments were selected due to their well-established legal frameworks in the EU and Germany, as well as the availability of quantitative targets and performance data, as well as their broad political acceptance and relative ease of implementation in the German context. Recycling mandates are modeled as binding share constraints aligned with the EU Packaging Directive and proposals under the German Circular Economy Act. Examples include mandatory recycling rates of 55% by 2025 and 65% by 2035, with additional scenarios testing more stringent targets up to 75%. ETS expansion scenarios follow the projected price trajectories in Germany and the wider EU ETS.[5] Alternative measures, such as deposit-return schemes for packaging, landfill taxes tiered by material type, eco-modulated product fees, mandatory producer take-back obligations, or variable pay-as-you-throw tariffs were excluded due to limited historical pricing data or their current pilot status, which would introduce significant uncertainty.

By comparing the spatial configurations, technology mixes, and cost–emissions outcomes across these scenarios, our models provide insights into the effectiveness and practical implications of policy instruments for guiding Germany's transition toward a more circular and low-carbon plastic economy.

## 2.2 | Methodological framework

Our optimization model integrates techno-economic assessments for handling costs, life cycle assessments (LCA) for estimating environmental impacts of end-of-life processing steps, and geo-information for evaluating transportation costs and emissions. Model inputs for the linear optimization models are costs, life cycle impacts per process unit, and transportation matrices.

The primary objective is to develop a MILP model that considers regional factors such as waste generation volumes in districts, proximity to customer demand for pyrolysis oil, logistics, capacities, and existing infrastructures. This model identifies the optimal waste treatment network for LWP sorting residues and APW in terms of costs and CCI. The MILP model determines material flows, transportation activities, and treatment capacities.

Similar optimization models have been widely used for planning logistic infrastructures (Dekker et al., 2004; Govindan et al., 2015; Komkova & Habert, 2023; Melo et al., 2009; Soleimani et al., 2017; Sommer et al., 2022). Unlike conventional models, our approach addresses LWP and APW specifically, incorporates bi-objective optimization, combines distinct modeling approaches, and evaluates the impact of political steering instruments.

## 2.3 | Optimization approach

Employing the two mathematical optimization models enables the analysis of the influence of political steering instruments on rolling out a chemical plastic recycling network. The main aim is to compare the cost-optimal solution with a network configuration that minimizes greenhouse gas emissions as well as trade-offs. Detailed information about sets, variables, and parameters is provided in Table 1. A comprehensive list of referenced equations and constraints can be found in Table 2.

### 2.3.1 | Model objective

The baseline scenarios aim to minimize either the total network costs, including waste handling and transportation (cf. Equation 1), or the CCI, with the latter achieved by substituting costs with CCI terms to minimize environmental impact. Equation (2) calculates the transport costs via the shortest distance between facilities and material flows, using stage-specific cost factors, which, again, are replaced by emission factors respectively to calculate the CCI. Costs associated with waste and product handling are calculated in Equation (3). Waste handling costs are determined by multiplying the material flow to a processing stage by a material- and location-specific cost factor. These costs are location dependent due to variations in labor costs and facility operation costs. They also include fixed operating costs per placed pyrolysis plant, which depend on the plant's input capacities. The environmental impact of waste handling can be calculated by replacing the cost factor with the respective emission factor. For the environmental impact of opening a pyrolysis plant, the emission factor is based on a reference plant (Dauriat, 2023) and is calculated using a linear annualization over 20 years (German Federal Ministry of Finance, 1995) to ensure that high initial infrastructure investments do not distort the results.

**TABLE 1** Notation of sets, parameters, and variables used in the optimization models. For the material flows through the waste handling system and network structure see Figure 1.

| | |
|---|---|
| **Sets** | |
| *Nodes of waste generation* | |
| $W$ | Set of districts with waste sources ($w \in W$) |
| *Nodes of waste handling and treatment* | |
| $S$ | Set of sorting facilities for LWP waste ($s \in S$) |
| $D$ | Set of dismantling facilities for EoL vehicles ($d \in D$) |
| $P$ | Set of RDF producers treating APW for energy recovery or pyrolysis ($p \in P$) |
| $R_{\mathrm{ER}}$ | Set of energy recovery facilities ($r_{er} \in R_{\mathrm{ER}}$) |
| $R_{\mathrm{CR}}$ | Set of districts with potential locations for pyrolysis plants ($r_{cr} \in R_{\mathrm{CR}}$) |
| *Nodes of customers for products* | |
| $C$ | Set of customers with demand for pyrolysis oil ($c \in C$) |
| *Sets for problem description* | |
| $K$ | Set of input capacities for new pyrolysis plants ($k \in K$) |
| $M$ | Set with waste material flows LWP and APW ($m \in M$) |
| **Parameters** | |
| *Network distances and transportation efforts* | |
| $d_{i\to j}$ | Distance between node $i$ and node $j$ (km)<br>$(i,j) = \{(w,s),(w,d),(d,p),(p,r_{er}),(p,r_{cr}),(s,r_{er}),(s,r_{cr}),(r_{cr},c)\}$ |
| $c^t_{i\to j}$ | Transportation costs for 1 ton waste per km (€)<br>$(i,j) = \{(w,s),(w,d),(d,p),(p,r_{er}),(p,r_{cr}),(s,r_{er}),(s,r_{cr}),(r_{cr},c)\}$ |
| $cci^t_{i\to j}$ | Transportation CCI for 1 ton waste per km (kg $CO_2$e)<br>$(i,j) = \{(w,s),(w,d),(d,p),(p,r_{er}),(p,r_{cr}),(s,r_{er}),(s,r_{cr}),(r_{cr},c)\}$ |
| *Amount of waste and product demand* | |
| $waste^m_w$ | Amount of waste material flow $m$ at location $w$ (tons) |
| $demand_c$ | Demand of pyrolysis oil at customer location c (tons) |
| *Industrial plants and material flow processing* | |
| $c^m_j$ | Processing costs per ton input at a facility for waste material flow $m$ (€)<br>$(j) = \{s,d,p,r_{er}\}$ |
| $cci^m_j$ | Processing CCI per ton input at a facility for waste material flow $m$ (kg $CO_2$e)<br>$(j) = \{s,d,p,r_{er}\}$ |
| $\gamma^m_j$ | Product yield per ton input at a facility for waste material flow $m$ (%)<br>$(j) = \{s,d,p\}$ |
| $capacity^m_j$ | Input capacity of a facility for waste material flow $m$<br>$(j) = \{s,d,p\}$ |
| $capacity_{r_{er}}$ | Input capacity at facilities $r_{er}$ |
| $c^{k,m}$ | Variable operating costs per ton input at pyrolysis plant of capacity $k$ for waste material flow $m$ (€) |
| $c^{\mathrm{fix}}_k$ | Fix operating costs for pyrolysis plant of capacity $k$ (€) |
| $cci^{k,m}$ | Processing CCI per ton input at pyrolysis plant of capacity $k$ for waste material flow $m$ (kg $CO_2$e) |
| $cci^{\mathrm{fix}}_k$ | Annualized CCI for placing a pyrolysis plant of capacity $k$ (kg $CO_2$e) |
| $\gamma^m_k$ | Product yield per ton input at pyrolysis plant of capacity $k$ for waste material flow $m$ (%) |
| $capacity^k_{r_{cr}}$ | Input capacity at facilities $r_{cr}$ depending on capacity class $k$ |
| *Model extensions* | |
| $F_1(x,y)$ | Costs minimization as a function of the decision variables $x$ and $y$ in GP programming approach |
| $F_2(x,y)$ | CCI minimization as a function of the decision variables $x$ and $y$ in the GP approach |
| $F^E_1$ | Costs single-objective optimum, relevant for the maximum distance in the GP approach |
| $F^E_2$ | CCI single-objective optimum, relevant for the maximum distance in the GP approach |
| $rr_{\mathrm{LWP}}$ | Recycling rate for plastic packaging in LWP waste that must be achieved |
| $rr^{\mathrm{MR}}_{\mathrm{LWP}}$ | Mechanical recycling rate of plastic packaging in LWP waste |
| $rr_{\mathrm{APW}}$ | Recycling rate for APW that must be achieved |

(Continues)

**TABLE 1** (Continued)

| Variables | |
|---|---|
| $x^m_{w\to s}, x^m_{w\to d}, x^m_{d\to p} x^m_{p\to r_{er}}, x^m_{s\to r_{er}}, x^m_{r\to c} x^{m,k}_{s\to r_{cr}}, x^{m,k}_{p\to r_{cr}}$ | Amount of transported quantity between facilities of waste material flow $m$ |
| $y^k_r$ | $\{$ 0, *if no pyrolysis plant of capacity class k is opened in district r*<br>1, *if a pyrolysis plant of capacity class k is opened in district r* |
| *dist* | Maximum distance from each normalized single-objective optimum in the GP approach |

**TABLE 2** Equations and constraints used in the models.

| Equation/constraint | Number |
|---|---|
| $c^{total} = c^{transport} + c^{handling}$ | (1) |
| $c^{transport} = \sum_{(i,j)\in\varepsilon} \sum_{m\in M} x^m_{i\to j} \cdot c^t_{i\to j} \cdot d_{i\to j} + \sum_{(i,j)\in F} \sum_{m\in M} \sum_{k\in K} x^{m,k}_{i\to j} \cdot c^t_{i\to j} \cdot d_{i\to j} +$ | (2) |
| $F = \{(p, r_{cr}), (s, r_{cr})\}$<br>$\varepsilon = \{(w, s), (w, d), (d, p), (p, r_{er}), (s, r_{er}), (r_{cr}, c)\}$ | |
| $c^{handling} = \sum_{(i,j)\in\varepsilon} \sum_{m\in M} x^m_{i\to j} \cdot c^m_j + \sum_{(i,j)\in F} \sum_{m\in M} \sum_{k\in K} x^{m,k}_{i\to j} \cdot c^{k,m} + \sum_{r_{cr}\in R_{CR}} \sum_{k\in K} y^k_{r_{rc}} \cdot c^{fix}_k$ | (3) |
| $\varepsilon = \{(w, s), (w, d), (d, p), (p, r_{er}), (s, r_{er}), (r_{cr}, c)\}$<br>$F = \{(p, r_{cr}), (s, r_{cr})\}$ | |
| $\sum_{s\in S} x^m_{w\to s} = waste^m_w, \quad \forall w \in W, m = 0$ | (4) |
| $\sum_{d\in D} x^m_{w\to d} = waste^m_w, \quad \forall w \in W, m = 1$ | (5) |
| $\sum_{r_{cr}\in R_{RC}} \sum_{k\in K} x^{m,k}_{s\to r_{cr}} + \sum_{r_{er}\in R_{ER}} x^m_{s\to r_{er}} = \gamma^m_s \cdot \sum_{w\in W} x^m_{w\to s}, \quad \forall s \in S, m = 0$ | (6) |
| $\sum_{p\in P} x^m_{d\to p} = \gamma^m_d \cdot \sum_{w\in W} x^m_{w\to d}, \quad \forall d \in D, m = 1$ | (7) |
| $\sum_{r_{rc}\in R_{RC}} \sum_{k\in K} x^{m,k}_{p\to r_{cr}} + \sum_{r_{er}\in R_{ER}} x^m_{p\to r_{cr}} = \gamma^m_p \cdot \sum_{d\in D} x^m_{d\to p}, \quad \forall p \in P, m = 1$ | (8) |
| $\sum_{c\in C} x_{r\to c} = \sum_{s\in S} \sum_{k\in K} x^{m,k}_{s\to r_{cr}} \cdot \gamma^m_k + \sum_{p\in P} \sum_{k\in K} x^{m,k}_{p\to r_{cr}} \cdot \gamma^m_k, \quad \forall r_{cr} \in R_{CR}, \forall m \in M$ | (9) |
| $\sum_{r_{rc}\in R_{CR}} x_{r\to c} \le demand_c, \quad \forall c \in C$ | (10) |
| $\sum_{w\in W} x^m_{w\to s} \le capacity^m_s, \quad \forall s \in S, m = 0$ | (11) |
| $\sum_{w\in W} x^m_{w\to d} \le capacity^m_d, \quad \forall d \in D, m = 1$ | (12) |
| $\sum_{d\in D} x^m_{d\to p} \le capacity^m_p, \quad \forall p \in P, m = 1$ | (13) |
| $\sum_{s\in S} \sum_{m\in M} x^m_{s\to r_{cr}} + \sum_{p\in P} \sum_{m\in M} x^m_{p\to r_{cr}} \le capacity_{r_{er}}, \quad \forall r_{er} \in R_{ER}$ | (14) |
| $\sum_{s\in S} \sum_{m\in M} x^{m,k}_{s\to r_{cr}} + \sum_{p\in P} \sum_{m\in M} x^{m,k}_{p\to r_{cr}} \le capacity^k_{r_{cr}} \cdot y^k_r, \quad \forall r_{cr} \in R_{CR}, \forall k \in K$ | (15) |
| $\sum_{k\in K} y^k_r \le 1, \quad \forall r_{cr} \in R_{CR}$ | (16) |
| $x^m_{w\to s}, x^m_{w\to d}, x^m_{d\to p} \ge 0, \quad \forall w \in W, \forall s \in S, \forall d \in D, \forall p \in P, \forall m \in M$ | (17) |
| $x^m_{p\to r_{er}}, x^m_{s\to r_{er}}, x^m_{r\to c} \ge 0, \quad \forall p \in P, \forall s \in S, \forall r_{er} \in R_{ER}, \forall c \in C, \forall m \in M$ | (18) |
| $x^{m,k}_{s\to r_{cr}}, x^{m,k}_{p\to r_{cr}} \ge 0, \quad \forall s \in S, \forall p \in P, \forall r_{cr} \in R_{CR}, \forall m \in M, \forall k \in K$ | (19) |
| $y^k_r \in \{0, 1\}, \quad \forall r_{cr} \in R_{CR}, \forall k \in K$ | (20) |
| minimize $\quad dist$ | (21) |
| $\frac{F_1(x,y)-F^E_1}{F^E_1} - dist \le 0$ | (22) |
| $\frac{F_2(x,y)-F^E_2}{F^E_2} - dist \le 0$ | (23) |
| $dist \ \ge \ 0$ | (24) |
| $\sum_{s\in S} \sum_{r_{cr}\in R_{CR}} \sum_{k\in K} (x^{m,k}_{s\to r_{cr}} \cdot \gamma^m_k) \quad \ge (rr_{LWP} - rr^{MR}_{LWP}) \sum_{w\in W} \sum_{s\in S} x^m_{w\to s} \quad , m = 0$ | (25) |
| $\sum_{p\in P} \sum_{r_{cr}\in R_{CR}} \sum_{k\in K} (x^{m,k}_{p\to r_{cr}} \cdot \gamma^m_k) \quad \ge rr_{APW} \cdot \sum_{d\in D} \sum_{p\in P} x^m_{d\to p,m} \quad , m = 1$ | (26) |

### 2.3.2 | Model constraints

The optimization model (Model 1) is subject to the following constraints. Constraints (4) to (9) describe the flow conservation, ensuring that a material flow entering a facility must exit it while considering the process yields at each stage and facility, while constraint (10) ensures that customers of pyrolysis oil are not supplied beyond their maximum demand. Moreover, the waste handling network and plant-specific capacities must not be exceeded (constraints 11 to 14). For the pyrolysis plants, this maximum capacity is determined by the capacity class selected and must also not be exceeded by the incoming material flow (constraint 15). Furthermore, constraint (16) specifies that only one pyrolysis plant can be placed in each district. Finally, constraints (17) to (20) define the range of values of the decision variables value range.

### 2.3.3 | MODM extensions

The single-objective optimization model is extended to a MODM model (Model 2) where both objectives (minimizing network costs in € and minimizing CCI in kg $CO_2$e) are simultaneously optimized. GP is particularly useful for addressing multi-objective optimization problems involving conflicting objectives, as it allows for explicit weighting and normalization of deviations from target values, ensuring a balanced solution (Jones & Tamiz, 2010).

Following Stallkamp et al. (2022), a GP approach is adapted to balance network costs and CCI by treating both objectives with equal weight. This ensures that the model minimizes the combined deviation from the target goals for costs and CCI, representing a compromise between economic and environmental objectives. Using a min–max approach minimizes the maximum relative deviation from the single-objective optima, ensuring neither objective dominates the solution (Jones & Tamiz, 2010).

Normalization is applied to ensure comparability between objectives, which are measured in different units[6] (Jones & Tamiz, 2010). The deviation of each objective is expressed as a percentage of its respective single-objective optimum. Equal weighting is used to identify a reference solution that prioritizes economic and environmental concerns equally. This reference solution serves as a benchmark for evaluating the effectiveness of political steering instruments, such as recycling rate and $CO_2$ pricing.

To achieve this balance, the MODM model introduces *dist* as a decision variable that represents the maximum normalized deviation from the single-objective optima. The objective function (21) minimizes *dist*, ensuring that neither cost nor CCI is disproportionately affected. Constraints (22) and (23) define the normalization approach by expressing deviations in percentage terms relative to the single-objective optima, $F_1{}^E$ (total network costs) and $F_2{}^E$ (total network CCI). Constraint (24) further defines the feasible range of *dist*, ensuring that the model remains consistent with the individual optimization results.

### 2.3.4 | Extensions of political steering instruments

To incorporate political steering instruments, enhanced recycling rate demands (*RR scenarios*) are modeled as constraints to ensure compliance for each material flow. Specifically, constraint (25) enforces the required recycling rate for LWP by ensuring that the sum of mechanically and chemically recycled waste meets or exceeds the recycling target. Similarly, constraint (26) ensures that the recycling rate for APW meets the target based on the dismantling of plastic parts from EoL vehicles. These constraints reflect the nature of mandatory targets set by law, which are treated as mandatory targets rather than optional goals.

In contrast, the extension of the national ETS to the waste management sector is modeled as an economic instrument, not as a constraint. The $CO_2$ price is incorporated into the cost factor for waste incineration by multiplying the national $CO_2$ emission fee by the emission factor for incinerating 1 ton of material. This economically penalizes incineration without explicitly restricting this treatment option.

## 3 | CASE STUDY

This case study examines the impact of political steering instruments on Germany's waste treatment network for LWP sorting residues and APW. EoL options include energy recovery in MSWI, RDF power plants, a chemical recycling network, or a hybrid solution. The reference year for modeling is 2021, with input data summarized in Table 3.

### 3.1 | Waste generation in Germany

In Germany, LWP is collected separately and sorted in facilities. Plastic composites (17%) and sorting residues (25%), difficult to recycle mechanically, are potential pyrolysis feedstock but are currently incinerated (Christiani & Beckamp, 2020). In 2021, 2681 kt of LWP household waste was sorted, yielding 1126 kt of feedstock for energy recovery or chemical recycling, representing 20% of Germany's plastic waste (Conversio, 2022). The spatial distribution of LWP waste in Germany is derived from DESTATIS (2022).

APW originates from EoL vehicle treatment or vehicle repair workshops. Although no specific data exists for the latter, APW volume is derived from the statistics on annual EoL vehicle statistics. Recycling is limited by functionalized thermoplastics and diverse polymers, leading to most APW being incinerated (Cossu & Lai, 2015). In Germany, around 3 kg of plastic is dismantled per EoL vehicle (UBA, 2022), with 337,135 vehicles treated in 2021, yielding 1011 t of APW (Kraftfahrtbundesamt, 2022). Spatial distribution is based on deregistration statistics by district (Kraftfahrtbundesamt, 2022).

**TABLE 3** Waste handling costs and climate change impact of energy recovery and chemical recycling route for a pyrolysis plant with a 50,000 t input/year capacity.

| | LWP | | | APW | | | Ref. |
|---|---|---|---|---|---|---|---|
| | Costs ([€/t] | Yield [-] | CCI [kg $CO_2$e/t] | Costs [€/t] | Yield [-] | CCI [kg $CO_2$e/t] | |
| **LWP sorting facility** | 84 | 0.58 | 335 | – | – | – | [1] |
| **RDF power plants** | 248 | – | 2350 | 254 | – | 2472 | [1,2] |
| **MSWI plants** | 123 | – | 2350 | 123 | – | 2472 | [1,2] |
| **Dismantling facilities** | – | – | – | 4 | 0.003 | 9 | [3] |
| **RDF producer** | – | – | – | 60 | 0.91 | 140 | [2] |
| **Pyrolysis plant*** | 74 | 0.4 | 264 | 80 | 0.5 | 166 | [4,5,6] |
| **Transport LWP waste**** | 0.18 | – | 1.4 | – | – | – | [7] |
| **Transport EoL vehicles**** | – | – | – | 0.02 | – | 0.1 | [8] |
| **Transport intermediates**** | 0.04 | – | 0.19 | 0.04 | – | 0.19 | [9] |
| **Transport pyrolysis oil**** | – | – | – | 0.02 | – | 0.1 | [8] |

[1]: (Volk et al., 2021), [2]: (Stallkamp et al., 2023), [3]: (Del Duce, 2022), [4]: (Stallkamp et al., 2024), [5]: (Zeller et al., 2021), [6]: (Ecoinvent dataset, 2021), [7]: (Doka, 2022), [8]: (Valsasina, 2022a) [9]: (Valsasina, 2022b)
*For an input capacity of 50,000 t/year.
**Per ton and kilometer.

Considering production losses at RDF facilities (Hennig et al., 2022; Stallkamp et al., 2023), APW feedstock is minimal compared to LWP sorting residues. However, this estimate is conservative since it excludes repair job APW, and a potential higher dismantling of significant plastic components in the automobiles' EoL treatment. Consistently dismantling and separately storing such components could increase the potential of APW sixfold (Wilts et al., 2016).

## 3.2 | Waste collection and transportation

In Germany, LWP waste is collected separately and transported to sorting facilities via lorries. Transport distances are adjusted using a $\sqrt{2}$ tortuosity factor to reflect road conditions (Delivand, 2011; Diehlmann et al., 2019). Emissions and costs, based on Ecoinvent data, are 0.18€/ton-km and 1.40 kg $CO_2$e (Doka, 2022).

The APW process chain begins with EoL vehicle transport to dismantling facilities, using >32 t EURO6 lorries with costs of 0.02€/ton-km and 0.10 kg $CO_2$e/ton-km (Valsasina, 2022a). Intermediate products are transported using 16–32 t EURO6 lorries with costs of 0.04€/ton-km and 0.19 kg $CO_2$e/ton-km (Valsasina, 2022b). Pyrolysis oil transport uses >32 t EURO6 lorries with equivalent costs and emissions (Valsasina, 2022a).

## 3.3 | End-of-life options

### 3.3.1 | LWP

Costs and emissions from sorting facilities using conventional and sensor-based technologies are included. Facility locations are based on Bundeskartellamt (2019) complemented by manual research of the facilities' capacities. Volk et al. (2021) report processing costs of 84€/t and 335 kg $CO_2$e/t input, separating 42% into mechanically recyclable plastics and 58% into sorting residues. The focus is on sorting residues, valued for their high calorific content, while other streams (e.g., metals) are assumed to be handled similarly across networks.

Sorting residues are either incinerated with energy recovery at RDF or MSWI plants or chemically recycled. Energy recovery at MSWI has processing costs of 123€/t input and 2350 kg $CO_2$e/t input, while RDF power plants incur higher costs of 248€/t input due to EU ETS regulations requiring $CO_2$ certificates (Bilitewski et al., 2018; EIB, 2024). Locations and capacities of these plants in Germany are detailed by Flamme et al. (2018).

### 3.3.2 | APW

After the EoL vehicles are transported to the dismantling facilities, APW is generated. Dismantling large plastic parts requires equipment and personnel, with costs 4€/t input and emissions of 9 kg $CO_2$e/t input (Del Duce, 2022; Lander, 2004). Dismantling facility locations in Germany are reported by GESA (2024). RDF producers separate metals from APW and shred the remaining plastic, a process costing 60€/t input and 140 kg $CO_2$e/t input (Stallkamp et al., 2023). RDF is either incinerated with energy recovery or chemically recycled. MSWI incineration costs 123€/t input and emits 2472 kg $CO_2$e/t input, while RDF power plants incur 254€/t input with equivalent emissions.

RDF from APW and LWP sorting residues serves as feedstock for pyrolysis, yielding pyrolysis oil and by-products like pyrolysis gas, aqueous condensate, and solid residues (Hennig et al., 2022; Stallkamp et al., 2023). Pyrolysis gas powers a combined heat and power unit, supplying part of the process energy, while aqueous condensate and solid residues are treated as waste for disposal (Stallkamp et al., 2024).

Stallkamp et al. (2024) provide costs structures for APW pyrolysis plants of different capacities. They are used to calculate the fixed operating costs that are independent of the handled material[7] and the input specific variable operating costs. Based on this, the model is extended to estimate the variable CCI of waste handling for the assessed feedstocks, incorporating the carbon intensity of the technical processes in the plant. Additionally, a fixed CCI is calculated for the placement of pyrolysis plants with multiple input capacities, based on emissions from plant construction.[8] The five plant capacities introduced by Stallkamp et al. (2024) range from an input capacity of 3750 t up to 120,000 t input/year (cf. Section 2.1 of Supporting Information S2).

Variable operating costs include $CO_2$e emission fees, as pyrolysis plants are assumed to fall under the EU ETS as petrochemical industrial processes (see Section 1.1 of Supporting Information S1). Table 3 summarizes costs and CCI for a pyrolysis plant with a 50,000 t/year capacity.

Fixed operating costs for a 50,000 t/year capacity are 9.66 Mio. €/year, with construction emitting 1.47 t $CO_2$e annually. Pyrolysis yields 40% pyrolysis oil from LWP sorting residues and 50% from APW feedstock (Stallkamp et al., 2023; Zeller et al., 2021).

After hydro-processing (Kusenberg, Eschenbacher, Djokic et al., 2022), pyrolysis oil becomes a potential feedstock for steam cracking (Kusenberg, Roosen, Zayoud et al., 2022). Refineries with steam crackers are integrated into the model as potential demand points for pyrolysis plants, with site locations and capacities detailed by Müller et al. (2021). Hydro-processing is assumed to occur on-site at refineries, though capacity and placement decisions are excluded due to data unavailability.

## 4 RESULTS

The models used to conduct a German case study focus on three optimized baseline scenarios: minimizing the overall costs of the waste treatment network (B1), minimizing $CO_2$e emissions (B2), and achieving an optimal balance between costs and emissions (GP). Following these scenarios, trade-offs for political instruments are explored, focusing on how changes in recycling rates and $CO_2$ prices affect the parameters. An overview of the scenarios is presented in Table 4. The resulting costs and $CO_2$e emissions of the waste treatment network designs are shown in Figure 2, with a further breakdown of the results provided in Section 2.2 of Supporting Information S2.

### 4.1 Model 1: Single-objective optimization

Minimizing total network costs, no pyrolysis plants are placed as they are more expensive than energy recovery (cf. Figure 2a) because municipal solid waste incineration (MSWI) plants have lower costs than RDF power plants since they are not part of the EU ETS (Section 1.1 of Supporting Information S1). Consequently, in the cost-minimizing baseline scenario (*B1*), waste is exclusively incinerated in MSWI plants, with total system costs of 598 Mio. €/year, where waste handling accounts for 531 Mio. €/year and transportation for 67 Mio. €/year.

The CCI of this cost-optimized waste treatment network amounts to 6200 kt $CO_2$e/year. Here, the main emission impact is also contributed by waste handling. In this scenario, transportation accounts for 497 kt $CO_2$e. Accordingly, a geographic depiction of facility locations and LWP and APW flows is shown in Figure 3a.

When minimizing CCI (scenario *B2*), all waste is chemically recycled due to the lower CCI of pyrolysis compared to incineration (cf. Figure 2b). Twenty pyrolysis plants are placed near LWP sorting plants: 5 plants in capacity class 2, 4 plants in capacity class 3, and 11 plants in the largest capacity class 4. Since larger plants are preferred, centralized processing is favored (see Section 2.1 of Supporting Information S2). Financing these plants costs 302 Mio. €/year within the CCI-minimizing scenario represents 37% of its total yearly waste treatment network costs.

The *B2* network's emissions total 2201 kt $CO_2$e/year, of which 75% come from waste treatment, 23% from transportation, and 2% from constructing pyrolysis plants. Figure 3b illustrates the regional flow in the CCI-optimized scenario.


**FIGURE 2** Impact of different scenarios on (a) climate change impact (in kt $CO_2$e) and (b) costs (in Mio. €). All results are displayed in tabular form in Section 2.2 of Supporting Information S2.

**TABLE 4** Description of all scenarios assessed, including their key results.

| Model | Scenario | Description | Recycling rate | Costs (in Mio. €) | CCI (in kt $CO_2$e) |
|---|---|---|---|---|---|
| Single-objective optimization model | B1 | Model optimized to minimize financial costs | 42% | 598 | 6200 |
| | B2 | Model optimized to minimize climate change impact (CCI) | 65% | 806 | 2201 |
| Multi-objective optimization model | GP | Model optimizing both financial costs and climate change impact (CCI) with equal weight assigned to both objectives | 62% | 761 | 2736 |
| | RR 50 | Model with a requirement for at least a 50% recycling rate | 50% | 667 | 4712 |
| | RR 55 | Model with a requirement for at least a 55% recycling rate | 55% | 707 | 3917 |
| | RR 63 | Model with a requirement for at least a 63% recycling rate | 63% | 774 | 2492 |
| | RR 65 | Model with a requirement for at least a 65% recycling rate | 65% | 792 | 2245 |
| | $CO_2$ 25 | Model applying Germany's national $CO_2$ ETS to the waste sector with a carbon price of €25 p4er ton of $CO_2$ | 42% | 826 | 6200 |
| | $CO_2$ 35 | Model applying Germany's national $CO_2$ ETS to the waste sector with a carbon price of €35 per ton of $CO_2$ | 42% | 872 | 6200 |
| | $CO_2$ 45 | Model applying Germany's national $CO_2$ ETS to the waste sector with a carbon price of €45 per ton of $CO_2$ | 65% | 790 | 2246 |
| | $CO_2$ 55 | Model applying Germany's national $CO_2$ ETS to the waste sector with a carbon price of €55 per t$CO_2$ | 65% | 791 | 2245 |
| | $CO_2$ 65 | Model applying Germany's national $CO_2$ ETS to the waste sector with a carbon price of €65 / t$CO_2$ | 65% | 791 | 2246 |
| | $CO_2$ EU | Model applying the average EU ETS carbon price of 2021 to the waste sector (€53 per ton of $CO_2$). | 65%2 | 791 | 2246 |

## 4.2 | Model 2: Multi-objective optimization

Scenario *GP* assigns equal weight to both costs and CCI, optimizing them simultaneously. In the optimal multi-objective network, 13% of the available waste feedstock is incinerated, and 87% is chemically recycled. Promising locations for pyrolysis plants are distributed across Germany (cf. Figure 4).

The total costs of the *GP* waste treatment network amount to 761 Mio. €, which is approximately 27% more than *B1*. The annualized financing costs for the placed pyrolysis plants is 244 Mio. €, accounting for 32% of the total costs. The waste handling costs are 449 Mio. €/year (59%), while the transportation costs 69 Mio. €/year (9%).

The total emissions in the GP scenario amount to 2736 kt $CO_2$e/year and are, thereby, 56% lower than in the cost-minimal solution (*B1*) and only 24% higher than the emission-minimal solution (*B2*). The construction of new pyrolysis plants contributes 35 kt $CO_2$e/year (1%) and is, therefore, neglectable. Waste handling emits 2197 kt $CO_2$e/year (80%). The transportation of waste and materials emits 505 kt $CO_2$e/year (18%).

## 4.3 | Political steering instruments

The following section explores two approaches to achieving a more balanced solution by implementing political steering instruments. These approaches were previously outlined in the model setup as constraints (for recycling rate requirements) and economic adjustments (for $CO_2$ pricing).

The first approach is evaluated in the *RR scenarios*, where recycling rates of 50%, 55%, 63%, and 65% for LWP waste are considered. These align with EU targets for recycling packaging materials and Germany's Circular Economy Act (UBA, 2023). We set a base mechanical recycling rate of 42%[9] (Volk et al., 2021),[10] though Germany's 63% target for plastic packaging remains unmet due to mechanical limitations (UBA, 2023). For APW, no official recycling rate exists; it is primarily incinerated (Stallkamp et al., 2024). Thus, APW recycling, limited by technology yields, is capped at 35%. LWP achieves higher recycling rates by combining mechanical and chemical recycling but are capped by today's pyrolysis yields.

The second approach extends the national $CO_2$ ETS to the waste management sector for the model reference year 2021, which was implemented in Germany in 2024. In this approach, variations in $CO_2$ prices are considered, with levels of 25€/t $CO_2$, 35€/t $CO_2$, 45€/t $CO_2$, 55€/t $CO_2$, 65€/t $CO_2$ corresponding to the German national $CO_2$ price of the years 2021, 2023, 2024, 2025, and 2026, respectively, as well as the EU ETS price of the year 2021.

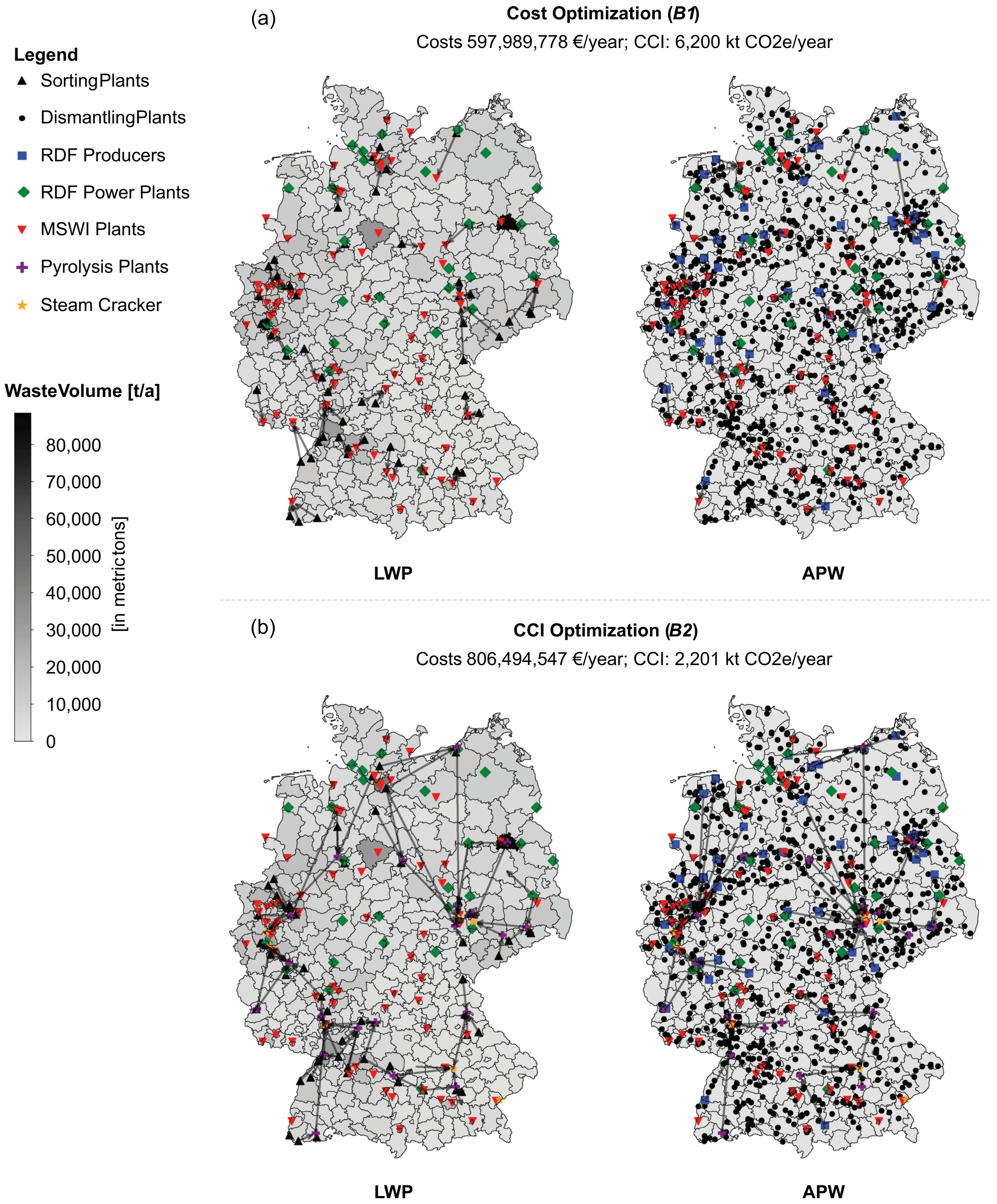


**FIGURE 3** Material flow in waste treatment networks optimized for (a) costs (scenario B1) and for (b) climate change impact (scenario B2).

While Model 2 and the GP approach serve as references for how an optimal balance of economic and environmental outcomes could look like, the scenarios aim to evaluate how closely political steering strategies can approximate that optimal solution. The results of all scenarios are also illustrated in Figure 2.

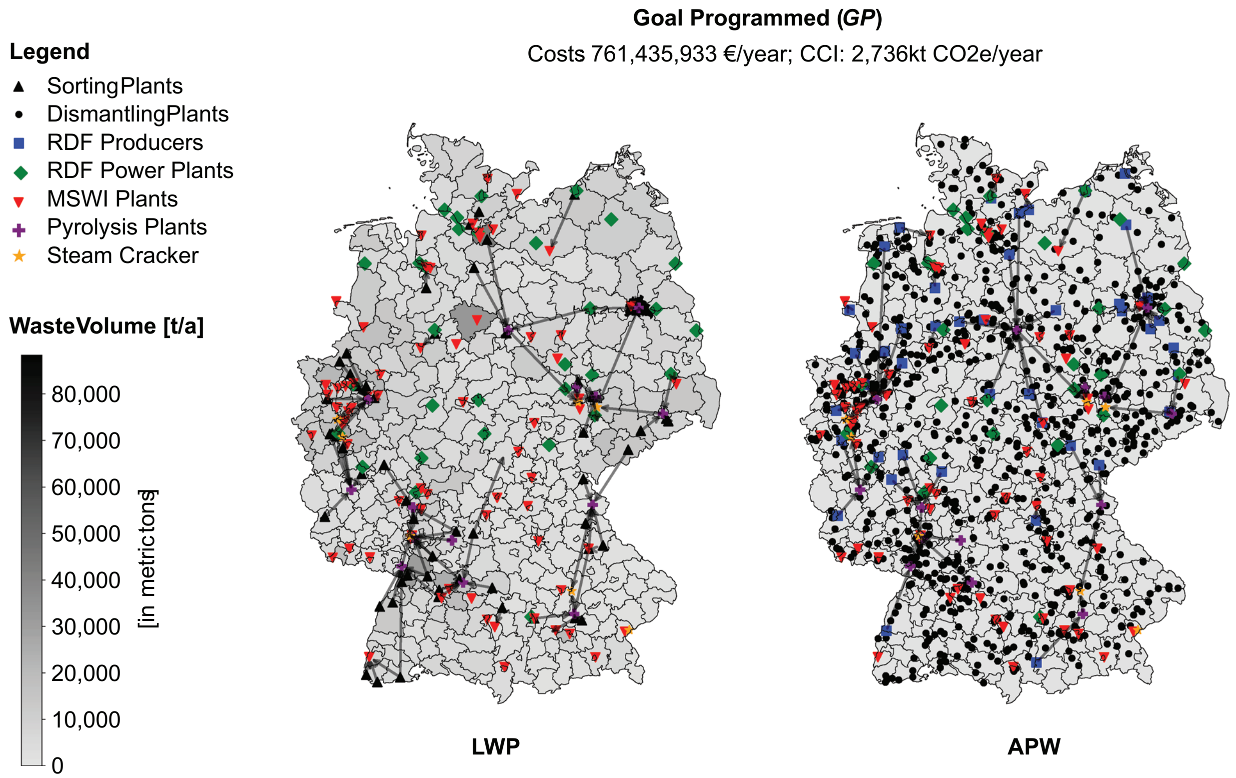


**FIGURE 4** Material flow in goal programmed waste treatment network optimized for both costs and for climate change impact (scenario GP).

### 4.3.1 | Recycling rates

Including the recycling target of 50% in scenario *RR 50* by using additional constraints and optimizing for costs, six pyrolysis plants are implemented (see Section 2.1 of Supporting Information S2). For this scenario, the total network costs are 667 Mio. €/year, while the $CO_2$e emissions are set to 4712 kt $CO_2$e/year. Compared to scenario *B1*, the network costs increase by 12%. However, a reduction of 24% in CCI can be achieved in comparison to scenario *B1*. With a recycling target of 55% (scenario *RR 55)*, the total network costs rise to 707 Mio. €/year, primarily due to implementing 10 pyrolysis plants, while total $CO_2$e emissions for this waste treatment scenario drop to 3917 kt $CO_2$e/year. This results in an 18% increase in costs and a 37% decrease in $CO_2$e compared to *B1*. Raising the recycling target to 63% in scenario *RR 63* leads to 15 pyrolysis plants, with total network costs of 774 Mio. €/year while $CO_2$e emissions decrease to 2492 kt $CO_2$e/year. This scenario shows 29% costs increase compared to scenario *B1* and a 60% lower emission in comparison to scenario *B1*. At a 65% recycling target (scenario *RR 65*), the total network costs reach 792 Mio. €/year, $CO_2$e emissions fall to 2245 kt $CO_2$e/year, and a total of 17 pyrolysis plants are built. This scenario includes one pyrolysis plant in the smallest capacity class (1), another one in capacity class 3, and 15 plants in the largest capacity class (4). This corresponds to a 32% increase in costs and a 32% decrease in emissions when compared to *B1*.

### 4.3.2 | Extension of an emission trading scheme to the waste management sector

Assuming the 2021-level certificate price of 25€/t $CO_2$ (scenario *$CO_2$ 25*), the extension of the national ETS does not change the environmentally advantageous network configuration in comparison to scenario *B1* as the total network costs increase by 38%, leading to costs of 826 Mio. €/year while the CCI remains at 6200 kt $CO_2$. Thus, 25€/t $CO_2$e is not sufficient to have a positive impact on the network design. A price of 35€/t $CO_2$e shows the same issue (scenario *$CO_2$ 35*), resulting in 46% higher network costs (872 Mio. €) in comparison to scenario *B1* but still without any environmental advantages. Neither scenario *$CO_2$ 25* nor scenario *$CO_2$ 35* involves the construction of pyrolysis plants. However, a national $CO_2$ price of 45€/t $CO_2$e (scenario *$CO_2$ 45*) significantly changes costs and environmental impact. In this scenario, the CCI is reduced to 2246 kt $CO_2$e, representing a 64% decrease compared to scenario *B1*. The total costs in *$CO_2$ 45* amount to 790 Mio. €, which is 32% higher than in scenario *B1* but 9% lower than scenario *$CO_2$ 35*. This cost reduction is primarily due to decreased waste handling expenses and total emission fees. Although emission fees are still present, they are significantly lower than in the *$CO_2$ 35* scenario. This is because the *$CO_2$ 45* scenario includes the construction of 16 pyrolysis plants of capacity class 4, thus, lowering the emission impact and fees paid. The scenarios with higher $CO_2$ prices (*$CO_2$ 55* and *$CO_2$ 65*) and the *$CO_2$ EU* scenario yield nearly identical economic and environmental results as scenario *$CO_2$ 45*.

## 5 | DISCUSSION

Offering a novel location optimization model for integrating pyrolysis plants into existing waste treatment networks, this study provides transferable decision models for sustainable waste management. The findings from the German case study contribute to the discussion on the large-scale rollout of chemical recycling technologies, demonstrating how policy instruments can steer the integration of pyrolysis plants to reduce CCI and increase carbon circularity.

In scenario *B1*, minimizing network costs results in the exclusive use of MSWI plants, leading to higher $CO_2$e emissions. Compared to the cost-optimal network configuration, 64% of the total $CO_2$e emissions could be saved in *B2* when optimizing the CCI of the waste management network employing chemical recycling via pyrolysis. This result highlights the necessary trade-off between economic and environmental objectives. Under current framework conditions, however, energy recovery is favored as it minimizes costs.

The *GP* scenario demonstrates that when both economic and environmental objectives are prioritized equally, chemical recycling (via pyrolysis) becomes a favorable solution for 87% of the waste stream(s), even though it results in higher operational costs. Fourteen pyrolysis plants with a maximum capacity of 120,000 t input per year are built and fully utilized, which suggests a preference for centralized processing.

Analyzing various recycling rates (*RR scenarios*) shows that this mechanism generates a steering effect above 65% toward a balanced network and the environmentally advantageous EoL alternative, chemical recycling. Increasing recycling targets leads to higher network costs but also reduces in $CO_2$e emissions. However, this steering strategy requires legal certainty regarding the crediting of chemically recycled plastics to the legally binding recycling rates. This crediting would support the development of chemical recycling and a circular economy for plastics. It should, however, be noted that there is no internalization of costs; instead, recycling capacity is only built up to meet the predetermined quota, with no incentive to exceed this quota.

Another strategy to steer a network toward balancing costs and CCI, is the introduction of ETS to the waste sector (*$CO_2$ scenarios*). A price of 25€/t and 35€/t $CO_2$e, it is still too low to lead to any environmental benefits or changes in the network design, while a price of 45€/t $CO_2$e significantly reduces the CCI by 64%, despite increasing overall network costs. This indicates that once a certain $CO_2$ price threshold is reached, further optimization of the network through carbon pricing instruments becomes limited as the waste handling option with a lower CCI is preferred for the waste treatment (cf. Figure 2). The threshold is crossed at a $CO_2$ price of 39.50€/t $CO_2$e. At that level, all waste is subjected to chemical recycling rather than energy recovery due to reduced $CO_2$e emissions and, consequently, lower emission fees. This analysis illustrates that the current price level of 55€/t $CO_2$e is sufficiently high to influence the waste treatment network design.

The results indicate that significant increases in recycling rates and reductions in CCI could be achieved through strategic policy instruments and the integration of pyrolysis plants into waste treatment networks. Based on these findings, we recommend that German policymakers consider the inclusion of chemical recycling into national recycling rates and raising the national recycling rate to 65% besides extending the ETS to include the waste sector.

## 6 | LIMITATIONS

While this study examines two key political steering instruments, it excludes other measures, such as the German Federal Climate Protection Act, which sets legally binding national greenhouse gas reduction targets to achieve climate neutrality by 2045. Furthermore, the model is limited to energy recovery and pyrolysis-based chemical recycling, excluding by-product treatment and other waste streams and waste treatment technologies (e.g., dissolution, solvolysis, and gasification). It is also constrained to one pyrolysis plant per district, potentially restricting capacity for waste processing. Expanding to other waste sources, such as plastic waste from construction, agricultural, or electrical equipment, is possible but requires further technological validation. The model can be adapted to and extended with compatible data for other waste streams, though the applicability is still hypothetical for pyrolysis plants. Furthermore, the case study uses data specific to Germany, which, when compared to other nations, may export a higher volume of used cars than it disposes of, potentially limiting APW availability. The model can, however, be applied to other regional data.

Future research should move beyond the presented greenfield placement assumption by integrating existing infrastructure like refineries and chemical parks. Furthermore, additional environmental impacts and the interactions between national and international ETS could be explored. In parallel, the impact analysis of alternative policy instruments were excluded for data and maturity reasons, yet hold relevance for future analyses.

Finally, this study utilizes a static optimization model with data for 2021, excluding variable waste and energy stream volumes and compositions, as well as market dynamics. Future model improvements should consider dynamic factors like energy price fluctuations to capture energy market dynamics more accurately. Additionally, incorporating dynamic and stochastic factors will enhance the model's applicability. Despite these constraints, our findings clearly demonstrate that binding recycling targets of at least 65% and $CO_2$ prices above €60/t can substantially incentivize the deployment of low-emission chemical recycling technologies and waste networks, if outputs from chemical recycling are legally recognized and attributed toward official recycling rates to ensure regulatory certainty. By embedding realistic policy levers directly into a spatially explicit optimization model, we offer a practical tool for policymakers and industry stakeholders to design more circular and climate-aligned plastic waste networks in Germany and beyond.

## AUTHOR CONTRIBUTIONS

*Conceptualization*: Christoph Stallkamp. *Methodology*: Christoph Stallkamp. *Software*: Christoph Stallkamp. *Validation*: Christoph Stallkamp and Malte Hennig. *Formal analysis*: Christoph Stallkamp. *Investigation*: Christoph Stallkamp. *Resources*: Rebekka Volk and Frank Schultmann. *Data curation*: Christoph Stallkamp and Teresa Oehlcke. *Writing—original draft preparation*: Christoph Stallkamp, Teresa Oehlcke, Rebekka Volk, and Frank Schultmann. *Writing—review and editing*: Christoph Stallkamp, Teresa Oehlcke, Rebekka Volk, Malte Hennig, and Frank Schultmann. *Visualization*: Christoph Stallkamp and Teresa Oehlcke. *Supervision*: Rebekka Volk and Frank Schultmann. *Project administration*: Rebekka Volk and Frank Schultmann. *Funding acquisition*: Rebekka Volk and Frank Schultmann. All authors have read and agreed to the published version of the manuscript.

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## CONFLICT OF INTEREST STATEMENT

The authors declare no conflicts of interest.

## DATA AVAILABILITY STATEMENT

Data will be made available upon request.

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## NOTES

[1] Metric tons are referred to as tons throughout the article.
[2] In Germany, LWP waste accounts for 59% of post-consumer plastic waste (Conversio, 2022), with 34% of plastic packaging not being mechanically recyclable (Christiani & Beckamp, 2020).
[3] See Section 2.3 for details on the mathematical formulation of objective functions and constraints.
[4] The location of the dismantling facilities is based on the Gemeinsame Stelle Altfahrzeuge of the German federal states (https://fachbetrieberegister.gadsys.de/fachbetrieberegister/Altfahrzeugverwertung/dst=10&is=1&e=1).
[5] See Section 1.1 of Supporting Information S1 for details on facility allocations in EU and German ETS pricing scenarios.
[6] Euros for costs and kg $CO_2$e for CCI.
[7] Include the costs for financing (capital payments and working capital) the pyrolysis plants per year and over the operational lifetime of 20 years (Stallkamp et al., 2024).
[8] Based on a reference plant (Dauriat, 2023) and annualized over 20 years using a linear annualization method (German Federal Ministry of Finance, 1995) to avoid distortions from high initial investment costs.
[9] The mechanical plastic recycling rate reflects the current performance of mechanical recycling processes. This study focuses on optimizing overall recycling rates through the integration of chemical recycling and does not specifically address improving mechanical recycling rates.
[10] Based on Volk et al. (2021), it is assumed that the sorting yield of high-grade recyclable plastics in the LWP waste stream is suitable for mechanical plastic recycling processes. Other plastic waste fractions are not suitable due to intermixing and impurities.

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## SUPPORTING INFORMATION

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

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