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Detritus - Multidisciplinary Journal for Waste Resources and Residues - is aimed at extending the "waste" concept by opening up the field to other waste-related disciplines (e.g. earth science, applied microbiology, environmental science, architecture, art, law, etc.) welcoming strategic, review and opinion papers. **Detritus is indexed in Emerging Sources Citation Index (ESCI) Web of Science, Scopus, Elsevier and Google Scholar**. Detritus is an official journal of IRACE / International Research Association on Circular Economy (www.irace.eco) and IWWG / International Waste Working Group (www.iwwg.eu).

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Editorial

WASTE MANAGEMENT IS PIVOTAL IN THE CROSSROAD BETWEEN FOOD AND BIOCHAR PRODUCTION OUTCOMES

Introduction

Increases in farm food waste, like in other parts of our globe, call for long-term sustainable solutions that will help reduce the emergent associated environmental/health risks. Besides, many food waste mitigation efforts largely focus on retail and consumer stages. There remains considerable food waste within the production phase, from growing, harvesting, transportation, to storage/ processing (Srivastava and Tonjes, 2025). However, biochar production appears positioned as a promising problem-solving approach applicable to farm food waste. When integrated with on-farm operations, biochar production can potentially improve farm viability while mitigating food waste challenges. As biomass, organic waste, and agricultural residues remain widely recognized as promising feedstocks for biofuel and energy production, thermo-chemical processes can convert these wastes into valuable energy products such as biochar, eventually becoming a sustainable and environmentally friendly solution (Rasaq et al., 2021).

Biochar and its production

Biochar–a carbon-rich material, is generated by "roasting" organic materials in a "pyrolyzer oven" operating at high temperatures (350-700°C) with limited oxygen, able to remove water and volatile organic compounds. Feedstocks for biochar production can come from diverse sources but need to be of low cost if the final product is to be economically viable (Rasaq et al., 2024). Key biochar production process steps include a) Gathering feedstock to the processing plant; b) Sorting, drying, and grinding (as needed), and c) Feeding into the pyrolyzer. Biochar production, as involving "thermochemical conversion" process (as shown in Figure 1), operates by different process variations, namely: a) Slow pyrolysis; b) Fast pyrolysis; c) Torrefaction; d) Hydrothermal Carbonization (HTC); e) Gasification. HTC is somewhat unique in the above list, as it operates at 180-374°C and 4-22 MPa to produce hydrochar and aqueous-phase products with- out requiring pre-drying. Torrefaction typically operates at 200-300°C with residence times of 15-60 min. Pyrolysis, carried out at 300-700°C,

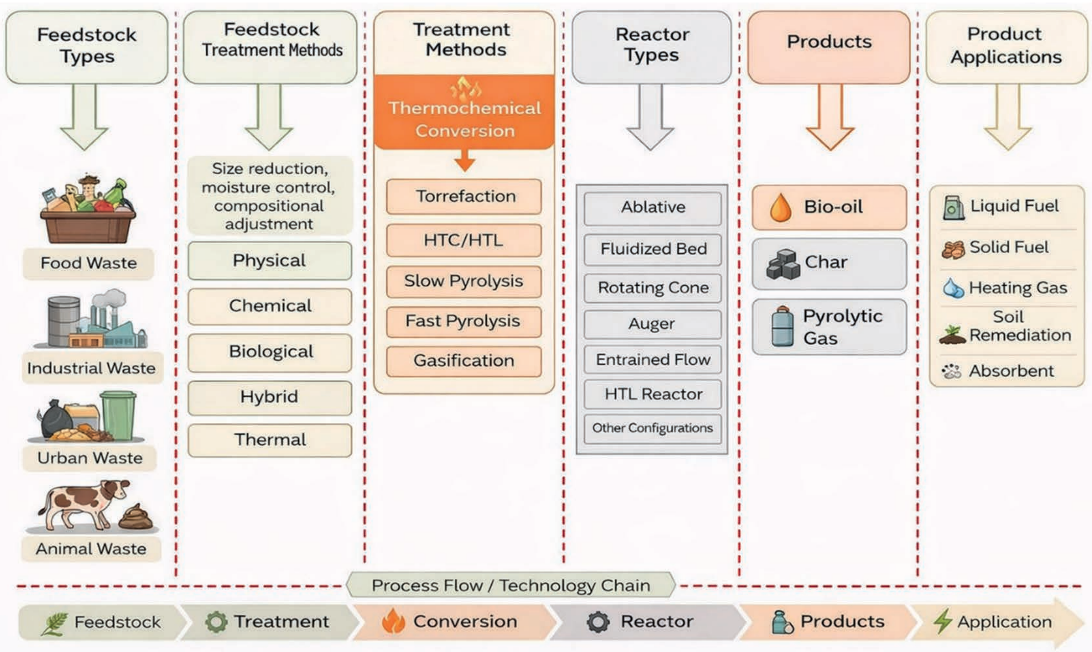

FIGURE 1: Thermochemical conversion pathways for the valorization of diverse waste feedstocks. Feedstock types, pretreatment methods, thermochemical conversion routes, reactor configurations, products, and end-use applications are shown. Red dashed lines separate the main stages of the technology chain. (Modified from Rasaq et al., 2024, Open Access - Creative Commons Attribution (CC BY) license)

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produces biochar, bio-oil, and syngas, while gasification operates above 700°C to generate syngas rich in hydrogen and carbon monoxide (Rasaq et al., 2024).

Farm food waste

Farm-level food waste alone accounts for approximately 16.8% of total U.S. food waste, highlighting the critical role producers play in the overall efficiency of the supply chain. By USDA, over one-third of food produced is either lost or wasted before reaching consumers, with a substantial portion at the farm level. For example, food loss/waste in the U.S. is estimated to cost roughly $218 billion annually, emphasizing the enormous financial impact across production, distribution, and retail (Divert, 2025). Farm food waste can arise for several reasons: a) Crops dam- aged in the field, rendering them unsaleable; b) Produce rejected due to not meeting retailer specifications (i.e. blemishes, color, or even size); c) Farm food waste caused by surplus, where farmers obtain more crops than can be sold on the market; d) Farm losses from unharvested crops due to labor issues. Most food waste at the farm level is either handled by being left in the field or ploughed back into soil or grazed by animals. As such, it might not be completely wasted, although most of its value is lost to the farmer. At consumer end of food supply chain, massive amounts of (food) waste increasingly taken to landfills /incinerators calls for biochar production even more! When food waste gets dumped at a landfill, it decomposes and releases methane and CO_2 into the atmosphere relatively quickly, before (landfill) been capped and gases captured. Thus, by diverting food waste from landfills, biochar production can reduce overall emissions of undesirable gases to the atmosphere. Further, by removing large quantities of food scraps (from homes, institutions, industry) from landfills, biochar production reduces the space (landfills) occupied, including environmental risks that arise from (landfill) leakage.

Biochar utilization

Biochar-amended soils have been shown to result in enhanced crop yields from vegetable crops like lettuce, and staple crops like maize and wheat. When incorporated into the soil, biochar from food waste can provide essential nutrients to the crops, functioning in the capacity of a slow-release fertilizer as shown in Figure 2, together with other promising attributes such as soil water capacity improvement, microbial community enhancement, and long-term carbon sequestration (Blenis and Hue, 2023). In this context, biochar should improve soil quality, support remediation of contaminated soil(s), and reduce the need for fossil-based soil amendments. If additional products can be created during the biochar production process, the systematic recovery of these valuable materials, products, and energy from food waste should support both environmental protection and economic growth - closely aligning with circular (bio)economy and sustainable development principles.

Some biochar examples tackling US farm food waste There are promising examples in the United States where biochar production via pyrolysis has helped manage problematic agrifood waste. By converting crop residues, husks, orchard pruning, and food scraps into a valuable soil amendment, biochar supports a circular economy, sequesters carbon, and enhances soil health. For instance, in California's Central Valley, biochar production from orchard waste has reduced farm waste, created jobs, and contributed to a growing bioeconomy (AFT, 2020). Applied Carbon, a U.S.-based start-up, has developed an agricultural robot that efficiently converts plant waste into biochar, providing a mobile and scalable solution for on-farm carbon capture and soil fertility enhancement (S Foundation, 2024). Similarly, the non-profit Soil Cycle in Montana employs a community-focused model: food scraps are collected via e-bikes, processed into biochar, and returned to local garden, demonstrating both waste reduction and soil impro-

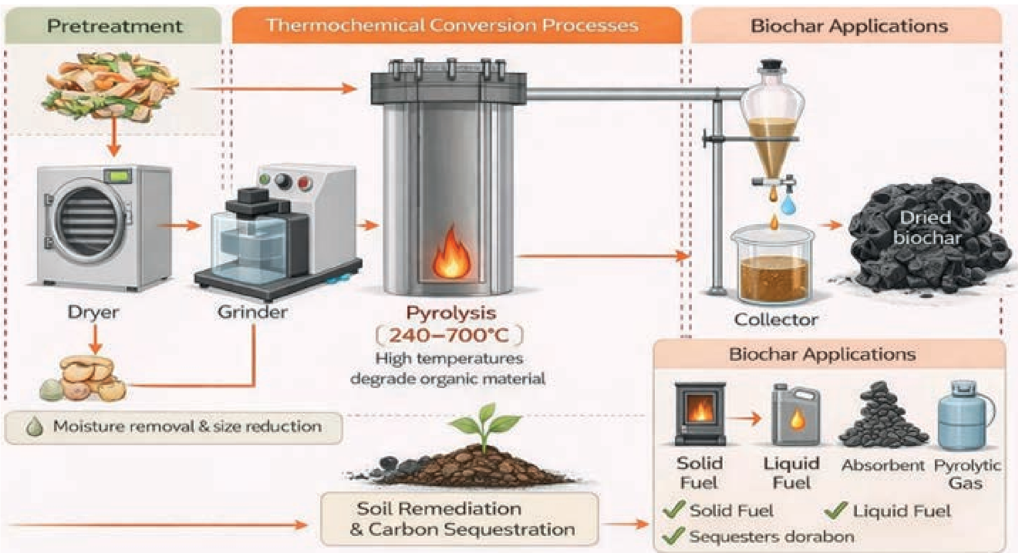

FIGURE 2: Biochar production from food waste via thermochemical conversion, including pretreatment (drying and grinding), pyrolysis (240-700°C), product collection, and applications in fuel, adsorption, and soil remediation (illustration created by Waheed A. Rasaq).

vement at a regional level (National Center for Appropriate Technology). Additionally, the American Farmland Trust has documented the economic and agronomic benefits of biochar production through partnerships with farmers, including almond orchards in California (AFT, 2020). The U.S. Forest Service's Rocky Mountain Research Station (RMRS) also conducts active research on biochar, exploring its applications in soil health, pollution mitigation, and carbon sequestration, often using forest slash as feedstock.

The (above-mentioned) examples reveal situations that demonstrate opportunities of laboratory-scale results translate to large-scale U.S. waste management/soil amendment systems. Furthermore, the diverse nature of food waste, and the challenges associated with biochar production via pyrolysis makes more relevant the need for increased extension outreach efforts to increasingly educate many more US households, foodservice and all relevant (food waste) producing industries. Expanding extension outreach/knowledge base to include agricultural engineering/processing experts specialized in tackling food waste biochar production via pyrolysis will further support the capacities of those tackling environmental health impacts/risks associated with (food waste) disposal/removal.

Challenges and opportunities for biochar from food waste

Despite biochar production's ability to subject organic biomass of water-rich environment to high temperatures/pressures, the food waste still presents distinct limitations (for biochar production application), namely: a) its composition can vary widely; b) it tends to be high in moisture; c) its availability tends to vary over the course of the year; and d) the volume available at one location may not match the size needed for industrial biochar production or for the local biochar market For instance, U.S. food waste are believed to typically contains higher proportions of processed foods, meats, dairy products, oils, and packaged items compared to laboratory-prepared or model food waste streams (EPA, 2023).

Differences in food waste composition can significantly influence pyrolysis performance, including biochar yield, energy recovery, and emissions behavior. Food waste streams rich in lipids and proteins–such as those commonly found in U.S. households–tend to exhibit higher energy potential during thermal conversion but also introduce operational challenges (Su et al., 2022). Beverage, sauce, and fresh produce emerging from U.S. household disposal further elevates the varying moisture content of food waste, which can reduce thermal efficiency and increase pre-treatment energy requirements.. One possible approach to handling these challenges could be to incorporate Anaerobic Digestion as a preprocessing step. Besides, some farms that already have digesters for processing manure also utilize some food waste, with generally positive results. The addition of biochar production to the system could result in greater benefits to the farmer and to the community.

Concluding Remarks

Overall, converting agrifood waste into biochar represents a promising strategy to simultaneously address organic waste streams, soil degradation, and renewable energy needs. The development of effective thermal treatment strategies, with recommendations for improving biochar quality, highlights the potential of thermochemical conversion as a sustainable, circular solution to the food waste challenge in the US.

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REFERENCES

(AFT), A.F.T., n.d. What is Biochar ? Biochar in Agriculture: Case Studies and Best Practices 1-7. https://farmland.org/biochar, Accessed 12 January 2026.

(EPA), U.S.E.P.A., 2023. Advancing Sustainable Materials Management: Facts and Figures Report 2023-2025.

Blenis, N., Hue, N., 2023. Biochar Production , Modification , and Its Uses in Soil Remediation : A Review 1-15.

Divert, I., 2025. The Cost of Wasted Food 1-10, https://divertinc.com/the-cost-of-wasted-food/, Accessed 18 January 2026.

Foundation, S., 2024. https://www.appliedcarbon.com/news/ Accessed 18 January 2026.

Rasaq, W.A., Okpala, C.O.R., Igwegbe, C.A., Białowiec, A., 2024. Navigating Pyrolysis Implementation—A Tutorial Review on Conside- ration Factors and Thermochemical Operating Methods for Bio- mass Conversion. Materials (Basel). 17. doi.org/10.3390/ma17030725

Srivastava, S., & Tonjes, D. (2025). Food Waste Crisis in the United States of America: the Known, The Overlooked and The Undi- sclosed. Detritus, 30, 14-22. doi.org/10.31025/2611- 4135/2025.19468

Su, G., Chyuan, H., Fattah, I.M.R., Sik, Y., Jang, J., Wang, C., 2022. Science of the Total Environment State-of-the-art of the pyrolysis and co-pyrolysis of food waste : Progress and challenges. Sci. Total Environ. 809, 151170. doi.org/10.1016/j.scito- tenv.2021.151170

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A SCORING SYSTEM TO EVALUATE THE PERFORMANCE OF ENTITIES AT VARIOUS LEVELS OF A MSW MANAGEMENT SYSTEM

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ABSTRACT
Waste generators, particularly at the household level, waste collectors, and local governments play pivotal roles in municipal solid waste management (MSWM) systems worldwide, especially in developing countries. In the Indian state of Kerala, waste collection has been formalized through a predominantly female workforce known as the Haritha Karma Sena (HKS), which undertakes door-to-door collection of non-biodegradable waste from households. This initiative has significantly enhanced the efficiency of MSWM in the state. To systematically evaluate the performance of HKS and other key stakeholders in the MSWM system, this study proposes a comprehensive scoring system developed using the Delphi method. The framework encompasses twenty parameters across three primary stakeholders: waste generators, Haritha Karma Sena waste collectors, and Local Self Government Institutions (LSGIs). The scoring system is designed to identify critical performance gaps and guide improvements, ultimately enhancing the overall efficiency of MSWM in Kerala. Moreover, this model holds potential for adaptation in other developing countries with similar waste management challenges.

1. INTRODUCTION

The continuous increase in urban population, along with an improvement in their standard of life have led to a substantial rise in the annual generation of municipal solid waste (MSW) (Singh, 2021; Singh, 2023a, b). This rise in generation has necessitated a paradigm shift in the approach to managing MSW the world over. In many countries, municipal solid waste management (MSWM) systems are currently undergoing a transitional phase aimed at safeguarding public health from waste-related hazards while promoting circularity. However, these efforts are often impeded by a lack of reliable, comprehensive, and high-quality data, which are essential for informed decision-making and effective system improvements (Singh, 2025). This situation underscores the need for continuous enhancements and revisions in MSWM systems (Singh, 2024). A performance evaluation system based on suitable indicators becomes important in this context. Such a system helps track progress during the transition, ensures accountability, supports evidence-based decision-making, facilitates benchmarking and standardization, enhances transparency and public trust, and drives continuous improvement.

A review of 63 works spread across four continents (Africa, Europe, America and Asia) and 32 countries identified various performance indicators (PI) for MSWMs that were classified under different heads: technical, environmental, economical, exogeneous, social and institutional (Colivicchi et al., 2025). A study conducted in Nagpur, India, on the evaluation of MSWM systems using Technical Indicators (TI) provided a valuable resource for establishing ground-level support mechanisms and incentive-based schemes, particularly to strengthen the role of the informal sector in MSW management (Singh, 2025). A well-structured scoring system improves the policy decision-making. A study on the effectiveness of construction and demolition waste management (CDWM) across 11 cities in China based on the performance indicators revealed a lack of coordinated development between CDWM outcomes and supporting guarantee systems (Wu, 2024). In a study conducted in Patras, a city in Western Greece, the performance indicators objectively assessed ‘Circularity’ in MSWM (Emmanouil et al., 2025).The eight indicators, namely, municipal waste generation, municipal waste recycling rate, food waste generation, bio-waste recycling, glass packaging recycling, paper and cardboard recycling, collected quan-

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tities of waste electronics and material reuse rate, were effectively used to measure the extent of circularity in the MSWM of the city. These PIs also showed the bottlenecks in the implementation of Circular Economy (CE)-based MSWM of a city like Patras. Performance indicators offer a good solution to the poor data management that hinders the identification of improvement opportunities and benchmarking of the performance. In a Sri Lankan study of 22 composting stations situated in urban and peri-urban areas, the four composite key performance indicators, General Facility Management Index (GFMI), Composting Process Management Index (CPMI), Compost Quality Index (CQI), and Cost Recovery Index (CRI), helped local authorities to improve the efficiency of MSWM (Manipura et al., 2025).

The Solid Waste Management Rules, 2016, notified by the Government of India, marked a significant policy shift by placing the responsibility for the segregated collection, transportation, treatment, and scientific disposal of solid waste on the Local Self Government Institutions (LSGIs), with a particular focus on urban bodies. In response, several states developed localized waste management frameworks to align with these rules. Among them, the state of Kerala has implemented a notable decentralized and community-centric model through the establishment of the Haritha Karma Sena (HKS).

This study proposes a structured performance assessment framework designed to evaluate the contributions of three principal stakeholder groups within the MSWM system of the state: waste generators (households), waste collectors (HKS personnel), and the regulatory authority (LSGI). The scoring system developed serves not only as a tool for performance benchmarking and accountability but also as an instrument for motivating continuous improvement by identifying best practices and highlighting areas requiring attention.

Unlike many existing scoring systems (Cai et al., 2025, Colivicchi et al., 2025, Manipura et al., 2025) this case-specific scoring system evaluates the performance of an MSWM framework by assessing the efficiency of individual primary stakeholders rather than the system as a whole, thereby enabling more targeted and effective interventions. The framework is designed to be scalable and adaptable across diverse local governance contexts, including panchayats, municipalities, and urban local bodies. Unlike indicators commonly used to assess MSWM systems in developing countries, the proposed method is based on real-time, field-level data. Owing to its reliance on real-time inputs, the scoring system is inherently dynamic and can be effectively applied at different LSGI levels, where responsibilities for waste management and safe disposal are clearly defined under existing legislation. The approach provides a clear, systematic, and replicable methodology that helps in customizing the scoring systems for use in societies with comparable municipal solid waste collection and management practices. By promoting transparency, encouraging behavioural change, and facilitating more effective policy implementation, the proposed model can contribute to the broader objective of achieving sustainable and inclusive solid waste management.

2. METHODOLOGY

2.1 Study area and context of the study

The state of Kerala (Figure 1), laying along the south-west coast of India (8°17’30” N to 12°47’40” N and 74°27’47” E to 77°37’12” E), is divided into 14 major revenue administrative units called Districts. As per census 2011(the last official census to be carried out in India) the total population of the state is 33.41 million with a density of 860/km^2 which is much higher than the national average of 382/km^2. Kerala experiences a tropical climate, characterized by mild winds and heavy monsoon showers. There are three distinct seasons - South-West monsoon (June-September), North-East monsoon (October – December) and summer (March- May). The winter season in Kerala during the months of December to February is not a marked one, compared to other parts of India.

Kerala is among the top states in India with high per-capita Gross State Domestic Product (GSDP) at constant price of ₹1,62,992 in 2021-22, the corresponding national average being ₹1,07,670. The state is known for its success in decentralized planning, devolving power to local democratic entities. The three-tier system of LSGI comprises Gram Panchayats, Block Panchayats, and District Panchayats. In addition, there is separate governing structure for urban centres- Corporations for the major cities and municipalities for the other cities. Presently, Kerala’s local self-government framework encompasses 941 Grama Panchayats, 152 Block Panchayats, 14 District Panchayats, 87 Municipalities, and 6 Corporations with a Ward or Division (consisting of 400-450 households) representing the smallest sub-unit of the these LSGIs. The Department of Local Self-Government shoulders the responsibility of orchestrating the activities of these 1200 local bodies.

Central Pollution Control Board (CPCB), the statutory authority that monitors pollution and waste management in India, has reported the MSW generation in the urban and semi-urban areas in the state of Kerala as 3472 TPD in the year 2021-22. The national figure for the year was 1,70,462 TPD. The gap between generation and treatment in Kerala was 781 TPD for this year(MSW_AnnualReport_2021-22 CPCB, n.d.). The annual waste generated, as per the state nodal agency for waste management, the Suchitwa Mission, is 3.7million tonnes (~300 g/person/day). Biodegradable fraction of the waste is 69% and the remaining 31% is non-biodegradable fraction. 79% of the non-biodegradable fraction has got a good calorific value which encourages the state to segregate the waste properly (Suchitwa mission, 2021)

The Government of Kerala has adopted a policy for solid waste management with two strategies:

- Decentralised waste management;
- Centralised waste management wherever necessary.

Figure 2 shows the process flow diagram of the Waste management in Kerala.

The Decentralized Solid Waste Management (DSWM) as conceived in Kerala, is a system involving the segregation and processing of waste at source to the maximum extent possible and then at the community level. There are

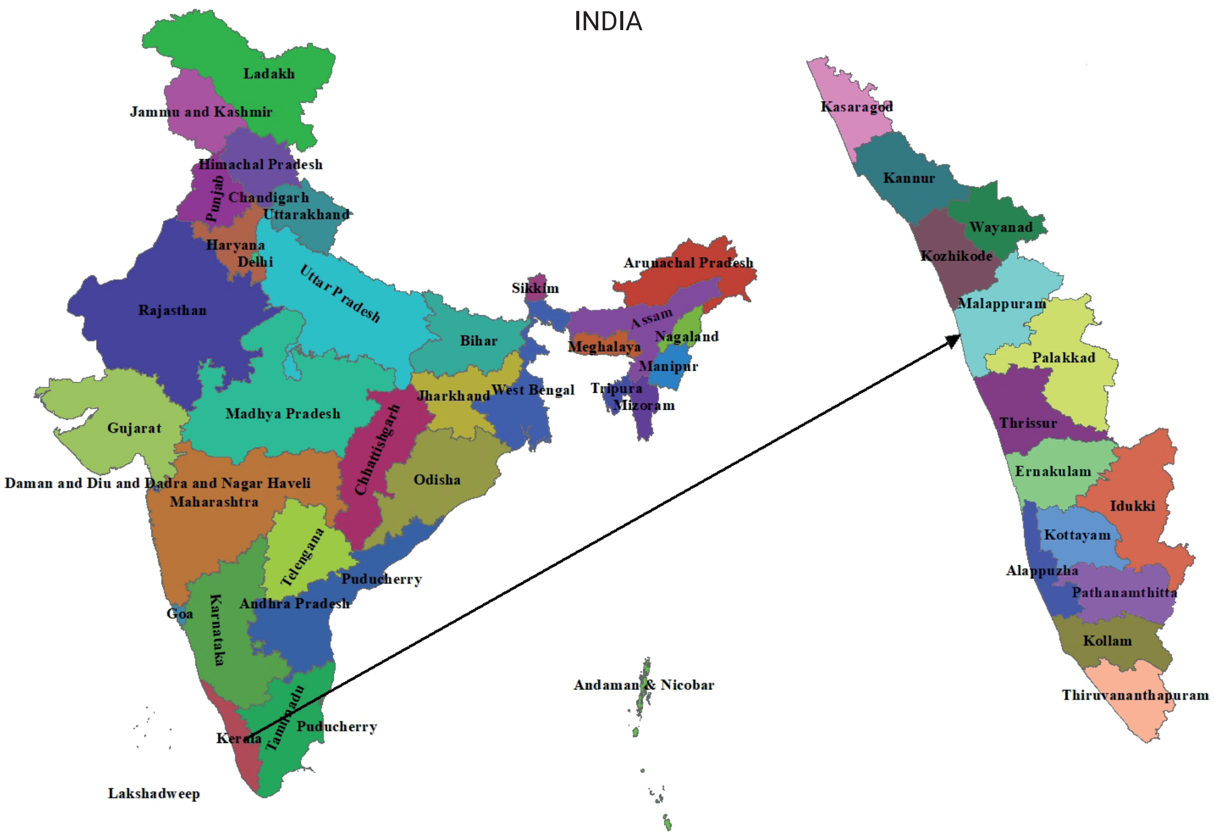

FIGURE 1: State of Kerala in India.

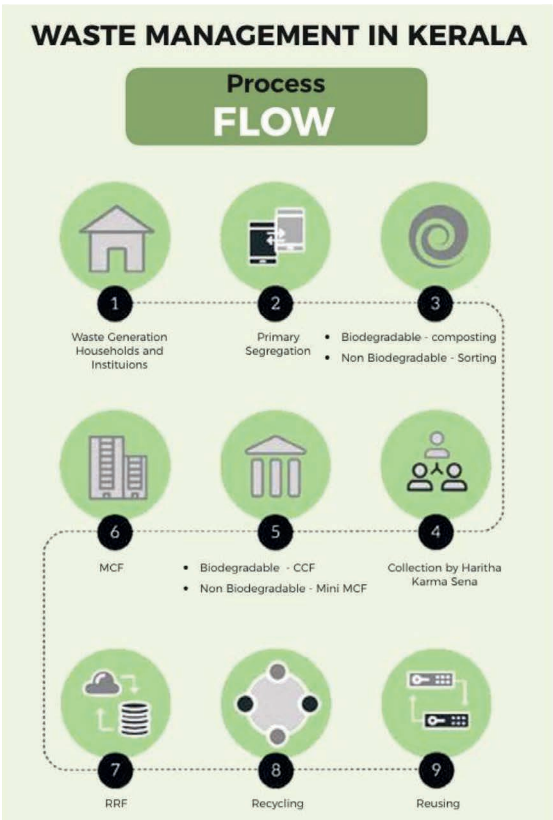

FIGURE 2: Flow diagram of Waste management in Kerala state (Source: Suchitwa Mission, Kerala).

different methods for the treatment of biodegradable and non-biodegradable waste in such a system. While composting and biomethanation are common methods used for treatment of biodegradable waste, non-biodegradable wastes are collected and made available for recycling processes. The aim is to substantially reduce the amount of waste reaching the landfill sites and minimise the associated issues.

2.2 Haritha Karma Sena

The Haritha Karma Sena (HKS, meaning the green work force) in Kerala, is a structured waste management work force operational since 2017 under the Haritha Kerala Mission. It serves as a decentralized, community-driven initiative aimed at enhancing municipal solid waste management (MSWM) at the local level. Comprising primarily women, the HKS operates under the supervision of Local Self Government Institutions (LSGIs). Each LSGI deploys at least a two-member HKS team per ward, managing waste collection from approximately 300 to 500 households monthly.

This model focuses on the door-to-door collection of segregated non-biodegradable waste, followed by the systematic sorting, resource recovery, and channelization of recyclables to authorized units. By promoting the principles of recycling and circular economy, the HKS significantly mitigates landfill dependency and environmental pollution. The HKS model offers valuable insights for replication in other urban and semi-urban contexts across India and other developing countries. Although the initiative has substantially improved waste management practices statewide, the absence of a performance appraisal mechanism impedes a systematic evaluation of the roles and effectiveness of its key stakeholders.

2.3 Procedure for Scoring System development

The stages involved the formulation of the scoring system are:

i. Based on literature survey and discussions with subject experts and officials in charge of the MSWM system, a preliminary scoring system was developed;
ii. The scoring system developed was shared with experts for their inputs through a Delphi process;
iii. Based on the expert inputs, the scoring system was further refined and finalised.

Numerous studies have highlighted that the use of performance indicators can significantly enhance the effectiveness of municipal solid waste management (MSWM) systems by enabling systematic evaluation and continuous improvement. In the present study, a preliminary scoring framework was developed through consultations with 25 experts, including academic researchers and field-level practitioners involved in MSWM in the state of Kerala. The expert panel comprised 13 academics actively engaged in teaching and research on MSWM and 12 field-level professionals involved in the implementation of MSWM systems at the local level. The experts possessed professional experience ranging from 8 to 15 years in the field. The framework aimed to assess the performance of three key stakeholder groups: household-level waste generators, Haritha Karma Sena (HKS)- the waste collectors, and the LSGIs responsible for coordination and oversight.

The initial framework was structured around nine main criteria and 24 sub-criteria, covering various dimensions of stakeholder performance. To validate and refine these indicators, a two-round Delphi method was employed. A panel of 15 waste management experts was invited to evaluate the relevance and applicability of each criterion and sub-criterion for inclusion in the final scoring system. A consensus threshold was applied, whereby any sub-criterion receiving over 50% of responses indicating irrelevance was excluded from the framework. This ensured that the final system retained only those indicators considered meaningful, widely accepted, and practically applicable.

Based on expert feedback, two sub-criteria—namely, “training attendance” under the HKS based waste picker stakeholder group and “infrastructure availability” under the LSGI stakeholder group—were excluded due to limited perceived relevance. The finalized scoring framework thus consisted of 22 sub-criteria distributed under nine main criteria. Each indicator was defined using quantifiable parameters, which were subsequently rated and assigned weights, resulting in a robust and adaptable performance assessment tool for evaluating decentralized MSWM systems.

3. RESULTS AND DISCUSSION

3.1 Scoring System

The scoring system for the key stakeholders- waste generators, waste pickers and LSGIs- was formulated based on clearly defined criteria and sub-criteria that reflect their specific responsibilities, such as waste segregation, service quality, operational efficiency, and resource management. By converting day-to-day practices into measurable indicators, the system provides a transparent and standardized method to monitor, compare, and improve waste management outcomes.

Table 1 gives the identified criteria for evaluating the HKS based MSWM in the state of Kerala after expert opinion survey findings and following a due process in the Delphi analysis. After these criteria identification, the next step was to develop a scoring system based on these criteria.

3.2 Waste Generators

Household-level waste generators are one of the primary stakeholder groups in MSWM. A dedicated scoring system was developed to assess their performance, structured around three main criteria and nine sub-criteria.

3.2.1 Waste Generation

The first main criterion identified for evaluating household-level waste generators was ‘waste generation’. This was assessed using two sub-criteria: Per Capita Generation Score (PGS) and Efforts to Reduce Waste (ERW). The PGS indicator measures the average quantity of non-biodegradable waste (NBW) generated per person per day within a household. It serves as a direct metric to quantify waste generation behaviour. Households were scored based on their PGS values, using the following scoring equation:

$$PGS = \text{Total NBW for N days}/N*\text{household size} \quad (1)$$

The second sub-criterion, Efforts to Reduce Waste (ERW), evaluates the initiatives undertaken by households to minimize waste generation at the source. This includes practices such as reducing packaging, reusing materials, and adopting low-waste lifestyle choices. The Waste Reduction Score (WRS) was developed to quantify these efforts, using the following equation:

$$WRS = \text{Practices adopted by household}/\text{Total applicable practices} \quad (2)$$

The list of practices and its evaluation method is described in the Table 2.

3.2.2 Segregation

Household-level segregation of non-biodegradable waste (NBW) was assessed using four sub-criteria: segregation accuracy, segregation consistency, availability of required bins, and a segregation quiz. The scoring system was developed for the first three sub-criteria, while the segregation quiz is designed to be conducted quarterly at the household level to evaluate improvements in residents’ knowledge of waste management practices. The Segregation Accuracy Score (SAS) measures how accurately

TABLE 1: List of stake holders and the criteria for the evaluation.

Stake holder	Main Criteria	Sub criteria	Score name
Waste Generators	Waste generation	Per Capita Generation	Per Capita Generation Score (PGS)
		Efforts to reduce waste	Waste reduction score (WRS)
	Segregation	Accuracy	Segregation accuracy Score (SAS)
		Consistency in segregation	Consistency score (SCS)
		Presence of required bins	Bin availability score (BAS)
		Segregation quiz	Quiz (SQS)
	Handling Habits	Pre-treatment of Waste	Pre-treatment Score (PTS)
		Avoidance of littering and open dumping	Littering Avoidance Score (LAS)
		Timeliness of Disposal	Timeliness of Disposal score (TDS)
Waste Collectors	Service Quality	Collection Coverage Rate	Collection Coverage Score (CCS)
		Mismatch Error	Quantity mismatch score (QMS)
		On Time Collection Rate	On Time Collection Score (TCS)
	Interactions with Generators	Household Satisfaction	Household Satisfaction score (HSS)
	Interaction with Authorities	Timely execution of duties	Timely execution score (TES)
LSGI authorities	Support for Collectors	Timely disbursement of wages	Wage timeliness score (WTS)
		Provision of equipment	Equipment availability score (EAS)
		Adequate working conditions	Adequate working conditions score (WCS)
	Handling of collected waste	Recycling rate	Recycling rate score (RRS)
		Facility downtimes	Facility Downtime score (FDS)
	Resource management	Budget utilization	Budget utilization Score (BUS)
		Staff strength	Staff strength Score (SSS)
		Issues resolved	Issues resolved score (IRS)

households separate waste into the appropriate categories (e.g., organic, recyclable, non-recyclable). The Segregation Consistency Score (SCS) assesses the regularity with which proper segregation practices are maintained. The Bin Availability Score (BAS) evaluates whether households possess the necessary color-coded bins corresponding to different waste categories. The Segregation Quiz Score (SQS) will be arrived based on the number of correct answers provided by the households.

The scoring equations for these sub-criteria are as follows:

Segregation Accuracy Score (SAS) = Number of categories household segregates into/ Number of waste types generated by the household (3)

TABLE 2: List of practices for waste reduction at source.

Practice	Example	Verification
1) Reduction of Single- use Plastics	Avoiding plastic bags, disposable cups, etc.	Spot check
2) Use of Reusable Bags, Containers & Bottles	Cloth totes, steel lunchboxes, glass containers	Spot check
3) Buying in Bulk to Reduce Packaging Waste	Buying whole grains in bulk instead of packaged items	Spot check
4) Repair and maintenance of Electronics	Reusing old phones, participating in take-back or swap programs	Verify records/physical verification
5) Donation & Reuse of Clothes and Other Items	Donating unused clothing and items to charity	Proof of donation (receipt/photo) / spot check
6) Use of natural cleaning	Using homemade detergents, Vinegar-based cleaners etc.	Spot check
7) Reducing paper usage	Using digital copies instead of printed	Spot check
8) Avoidance of Overpacked & Disposable items	Preferring loose vegetables, avoiding individually wrapped items	Spot check (packaging/unnecessary wrapping)
9) Repurposing & Upcycling Items Instead of Discarding	DIY projects using waste material, converting old clothes into bags	Spot check survey/verification
10) Composting of Organic Waste (Only if Applicable)	Backyard composting	Physical check
11) Participation in Community Waste Reduction Initiatives	Zero-waste workshops, swap shops, repair cafés	Survey

SCS = Days with correct segregation/Total collection days (4)

BAS = Bins available/Bins needed (5)

SQS = Questions answered correctly/Total number questions asked (6)

3.2.3 Handling Habits

The handling habits of households were evaluated using three sub-criteria: (i) pre-treatment of waste, (ii) timely disposal of generated waste, and (iii) avoidance of littering in the surroundings. The Pre-Treatment Score (PTS) assesses whether households perform any preparatory actions—such as washing, drying, or compressing waste—prior to disposal. The Timely Disposal Score (TDS) measures the promptness with which waste is disposed of according to the scheduled collection days. In calculating the TDS, the number of days during which waste was ready for collection is compared against the total number of scheduled collection opportunities within the review period. The third and final sub-criterion, Avoidance of Littering and Open Dumping, evaluates adherence to proper waste disposal practices. The Littering Avoidance Score (LAS) will be calculated based on the littering instances noticed in and around the housing premises during the household visits for the collection of waste.

The scoring equations are as follows:

PTS = Amount of waste Pre-treated/Total amount of pre-treatable waste generated (7)

TDS = Timely disposal/Total disposal opportunities (8)

LAS = No of littering instances/Total number of inspections (9)

The final score for the household level waste generators, Waste Generator Score (WGS) as described in equation 13, will be a proctored fraction score between 0 and 1 and will be arrived based on the three main-criteria score as described in equation 10, 11 and 12.

Waste generation Score = (PGS+WRS)/2 (10)

Segregation Score = (SAS + SCS + BAS + SQS)/4 (11)

Handling habit Score = (PTS +TDS + LAS)/3 (12)

Waste generators(Households) Score (WGS) = (Waste generation score +Segregation score + Handling habits score)/3 (13)

3.3 Waste Collectors

The performance of HKS-based waste collectors was evaluated using three key criteria: service quality, interaction with waste generators, and interaction with local authorities. A scoring system was developed to assess these dimensions comprehensively.

3.3.1 Service quality

Service quality of HKS-based waste collectors was assessed using three sub-criteria: collection coverage, on-time collection rate, and mismatch error. The Collection Coverage Score (CCS) represents the percentage of households receiving waste collection services. The on-time collection will be assessed based on the Timely Collection Score (TCS) which measures the punctuality and adherence of collection activities to the scheduled timings. The mismatch error will be evaluated as Quantity Mismatch Score (QMS) which monitors instances of incorrect waste collection or missed pickups. The scores for these sub-criteria were calculated using the following equations:

Collection coverage score (CCS) = Household served/Total household assigned (14)

Timely collection score (TCS) = Total collections on time/Total schedule collections (15)

Quantity Mismatch Score (QMS) = Total weight of waste from households/Total weight of waste at collection centre (16)

3.3.2 Interaction with Generators

Interaction with household-level waste generators was evaluated using a single sub-criterion, the Household Satisfaction Score (HSS), which gathers feedback from households to assess the quality of service and responsiveness of waste collectors. The HSS was calculated using the following formula:

HSS = (households satisfied/total household surveyed) (17)

3.3.3 Interaction with Authorities

This sub-criterion was evaluated through the Timely Execution of Duties Score (TES), which measures how promptly and effectively collectors perform their assigned responsibilities.

TES = (actual time taken for duty/Time allotted for duty) (18)

The final score for the HKS based waste pickers, Waste Picker Score (WPS) as described in equation 20, will be a proctored fraction score between 0 and 1 and will be arrived based on the three main-criteria score as described in equation 17, 18 and 19.

Service quality Score = (CCS+TCS+QMS)/3 (19)

Generators interaction Score = HSS
Authority interaction Score = TES

Waste Pickers, HKS, Score (WPS) = (Service quality score + Generators interaction score + Authority interaction score)/3 (20)

3.4 LSGI Authorities

The Local Self Government Institutions (LSGIs), which support the Haritha Karma Sena (HKS) and oversee the implementation of municipal solid waste management (MSWM), shall also be evaluated in this system. Their role in relation to both waste generators and waste collectors was assessed using a scoring system developed around three main criteria, further detailed by six sub-criteria.

3.4.1 Support for Collectors

LSGI support for waste collectors was evaluated based on three sub-criteria: timely disbursement of wages, equipment availability, and adequate working conditions. The Wage Timeliness Score (WTS) assesses whether payments to collectors are made punctually, as verified through payroll records and wage receipts. The Equipment Availability Score (EAS) measures the provision of necessary tools and protective gear to collectors. Lastly, the Adequate Working Conditions Score (WCS) evaluates the overall work environment with respect to safety, hygiene, and institutional support. The WCS was calculated using criteria outlined in Table 3.

The score equations are as follows:

WTS = Wages paid on time/Total wages due (21)

EAS = Total equipment provided/Total equipment required (22)

WCS = score for no. of conditions satisfied/maximum score (23)

3.4.2 Handling of Collected Waste

The efficiency of Local Self Government Institutions (LSGIs) in handling collected waste was assessed using two sub-criteria: the Recycling Rate Score (RRS) and the Facility Downtime Score (FDS). The RRS quantifies the percentage of non-biodegradable waste (NBW) that is effectively recycled, reflecting the success of resource recovery efforts. The FDS evaluates the operational efficiency of waste processing facilities by tracking their downtime during the assessment period. The equations used to calculate these scores are provided below:

RRS = Recycled waste/Total Recyclable waste (24)

FDS = No. of days facilities were down/Total no. of operations days (25)

3.4.3 Resource Management

Resource management at the LSGI level, with respect to both waste generators and collectors, was evaluated using four sub-criteria. The Budget Utilization Score (BUS) measures how effectively the allocated funds for waste management are utilized. The Staff Strength Score (SSS) assesses the adequacy of staffing levels and the efficiency of workforce management. Additionally, the effectiveness of addressing operational challenges within the MSWM system was assessed through the Issue Resolving Score (IRS), which compares the number of issues reported to those resolved. The final scoring equations for these sub-criteria are presented below:

BUS= (Fund spent/Total fund allotted in Budget) (26)

SSS= (Active work force/Total work force required) (27)

IRS = (Issues reported/issues resolved) (28)

The final score for the LSGI authority, LSGI Score as described in equation 32, will be a proctored fraction score between 0 and 1 and will be arrived based on the three main-criteria score as described in equation 29,30 and 31

Support Score = (WTS+EAS+WCS)/3 (29)

Handling Score = (RRS+FDS)/2 (30)

Resource management Score = (BUS+SSS+IRS)/3 (31)

Final score of the LSGI Authority, LSGI Score = (Support Score + Handling score + Resource management score)/3 (32)

3.5 Weightage to Scoring system

Each main-criterion for the three key stake holders was then distributed among Seven field level experts for arriving a weightage to each of the main criterion in the implementation at the field level. The weights so assigned is summarised below

3.5.1 Waste Generators

The experts have the opinion of giving equal weightage to the two main criteria namely waste generation and segregation and a lesser weightage to handling habits. Accordingly, the weights assigned by the experts was as follows

Waste generation = 0.4
Segregation = 0.4
Handling habits score = 0.2

Hence the equation given 13 was modified as:

WGS = (0.4*Waste generation Score + 0.4*Segregation Score + 0.2*Handling habits Score)

3.5.2 Waste collectors

The experts were of the opinion that Service quality must be given a higher weightage than the other two criteria and the weights assigned based on the feedback of experts are as follows:

Service quality = 0.50
Generators interaction = 0.25
Authority interaction = 0.25

The final equation given in 20 was modified as below:

WPS = (0.5*Service quality score + 0.25*Generator interaction score + 0.25*Authority interaction score)

3.5.3 LSGI Authorities

The opinion of the experts was that all the three criteria can be given an equal weightage so that the entire system functions effectively and efficiently and hence the equation given in 32 is valid after assigning a weightage.

During the initial phase of implementing the scoring system, equal weightage was assigned to each sub-criterion score. The average of these sub-criteria scores was

TABLE 3: Adequate working conditions.

Factors	Assessment Criteria	Score
Wage & Benefits	Above min wage with benefit	3
	Min wage with no benefit	2
	Below min wage	1
Access to Medical Checkups	Regular health checkups	3
	Irregular checkups	2
	No medical support	1
Workload Feasibility	Fair workload per worker	3
	Slightly overloaded	2
	Highly overloaded	1
Rest Breaks & Facilities	Adequate breaks & rest areas	3
	Few breaks	2
	No rest facilities	1
Ventilation & Air Quality at work	Proper ventilation & temperature	3
	Some ventilation	2
	No ventilation & uncomfortable working temperature	1

calculated to determine the score for each main criterion, which was then averaged to produce a final performance score for each stakeholder group. The scoring system is designed to be dynamic; sub-criteria scores will be monitored regularly, and as improvements are observed in specific sub-criteria, differential weightage will be assigned accordingly. This adaptive weighting approach aims to incentivize continuous improvement and enhance the overall efficiency of the MSWM system.

The performance of the three key stakeholders—namely household-level waste generators, HKS waste collectors, and the LSGI authority—will be qualitatively ranked using a five-level scale, as presented in Table 4, based on their respective final scores.

3.6 Discussion

The current study developed a scoring system for the main MSWM stakeholders, including waste producers, waste collectors, and local governments that carry out and are accountable for the safe disposal of waste in accordance with current national regulations. This decentralized scoring approach, which evaluates each stakeholder using a primary criterion and a sub-criterion, aids in identifying each stakeholder's weaknesses and evaluating the stakeholder's performance based on their functionality.

This strategy encourages long-term environmental, economic, and social sustainability within MSWM systems and enables more informed decisions, as demonstrated by a performance assessment model study carried out in Qassim Province, in the central region of the Kingdom of Saudi Arabia, Alhumid (2019). This scoring system functions as an effective tool for benchmarking municipal solid waste management practices in various local bodies. Additionally, the current study highlighted how these variables allow for comparisons across towns inside and within administrative regions.

These scores in the 22 parameters are useful not only for the local organizations that implement the MSWM but also for analyzing the state's administrative districts' performance. This method will be helpful for identifying the gaps in each district and for a statewide integrative approach to the MSWM system. Waste management operations in the state of Kerala are overseen and guided by the designated nodal agency, Suchitwa Mission (www.suchitwamission.org). The same agency could also oversee the implementation of the proposed scoring system. In this role, Suchitwa Mission may be responsible for conducting evaluations, providing feedback, recommending improvements, and instituting recognition mechanisms, such as awards, to acknowledge best-performing local bodies or stakeholders across various categories.

TABLE 4: Performance assessment of key stake holders.

Score of stake holder	Performance of key stake holders
Greater than 0.85	Excellent Performance
0.7 – 0.85	Good Performance
0.5 – 0.7	Average Performance
0.3 – 0.5	Below Average Performance
Less than 0.3	Poor Performance

The proposed scoring system shall also be useful in addressing the state's restricted landfill capacity as a result of its dense population and in advancing the development of an MSWM system that is more focused on the circular economy. A similar case is addressed in a study carried out in the city of Jakarta which confronts pressing municipal solid waste management (MSWM) issues, with limited landfill capacities and rampant illegal dumping in slum and non-slum areas, exacerbating land consumption challenges (Suryawan & Lee, 2024). The study used twelve indicators across five adaptive capacity dimensions, correlating them with the Importance-Performance (I-P) metrics and the participant demographics. This study helped both slum and non-slum communities to underscore the importance of bolstering the circular economy and waste segregation. While a pronounced awareness regarding waste treatment was observed among non-slum residents, the individuals from slum locales indicated a greater need for focused encouragement and resources to engage in waste segregation initiatives effectively. The insights from this study are pivotal for urban planners, policymakers, and scholars. Using the stakeholders' scores, our study evaluates the performance of important MSWM system participants in the state of Kerala. More involvement in source-level waste management is encouraged by this scoring system, which also makes it possible to identify operational inefficiencies and implement data-driven ward-level improvements. Incentive mechanisms like tax rebates can be devised. The LSGI Score shall serves as a benchmark for assessing LSGIs' performance throughout the state. This score makes it possible to identify systemic issues and provides guidance for focused activities meant to improve Kerala's MSWM framework as a whole.

The state of Kerala's implementation of the HKS system, which formalized waste collection, represents a new change in MSW management that can be embraced by developing nations worldwide. The scoring system established for the HKS can be utilized as measure of the formalisation in the MSWM which is absent in the system when the globe is moving towards a Circular Economy (CE) based MSWM

The present study also has certain limitations. Methodological constraints include the exclusion of stakeholders' perspectives during the development of the grading system and the relatively small number of experts (n = 25, for criteria identification, n=7 for assigning weights to criteria) involved in the Delphi process. In addition, the limited participation of specialists in the system development discussions may have restricted the identification of more comprehensive and contextually relevant parameters in the initial framework. Another limitation is the context-specific nature of the scoring system, which was developed for an MSWM system that rely on manual door-to-door collection of non-biodegradable solid waste in response to a legal requirement. An operational limitation of the study is the absence of rigorous field testing of the scoring system.

Future studies may focus on improving the proposed scoring system by addressing the identified limitations and by incorporating sustainability considerations into key parameters such as waste composition, waste streams, and technical capabilities. Evidence from a case study conducted in Istanbul, Turkey (Ak & Braida, 2015) suggests that such integration can significantly enhance municipal solid waste management. This approach may facilitate the identification of best practices while promoting accountability and continuous improvement among local authorities. Additionally, the applicability and effectiveness of the scoring system as a tool for assessing circular economy components within MSWM warrant further investigation. The evaluation of MSWM systems and their stakeholders could also be strengthened by applying the scoring system within an implementation–feedback framework, followed by iterative revisions based on field-level outcomes.

4. CONCLUSIONS

A scoring system focused on household waste management can provide significant value by identifying low-performing households within a community and establishing clear benchmarks for improvement. Beyond individual assessment, such a system has the potential to foster a competitive spirit among households and communities, motivating collective efforts to enhance waste management practices and achieve higher performance scores. This framework evaluates multiple dimensions of waste management, including waste segregation, collection efficiency, recycling efforts, and community participation. By quantifying key performance indicators, the scoring system offers a transparent and objective means to monitor progress, recognize achievements, and guide targeted interventions to strengthen overall municipal solid waste management.

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REFERENCES

Ajayi, S. O., Oyedele, L. O., Bilal, M., Akinade, O. O., Alaka, H. A., Owolabi, H. A., & Kadiri, K. O., 2015. Waste effectiveness of the construction industry: Understanding the impediments and requisites for improvements. Resources, Conservation and Recycling, 102, 101–112. https://doi.org/10.1016/j.resconrec.2015.06.001

Ak, H., & Braida, W. (2015). Sustainable municipal solid waste management decision making: Development and implementation of a single score sustainability index. Management of Environmental Quality: An International Journal, 26(6), 909–928. https://doi.org/10.1108/MEQ-03-2015-0028

Alhumid, H. A., Haider, H., Alsaleem, S. S., Alinizzi, M., Shafiquzaman, M., & Sadiq, R. (2019). Performance assessment model for municipal solid waste management systems: Development and implementation. Environments - MDPI, 6(2). https://doi.org/10.3390/environments 6020019

Brunner, P. H., Fellner, J., 2007. Setting priorities for waste management strategies in developing countries. Waste Management & Research, 25(3), 234–240.

Chen, Y. C., 2018. Effects of urbanization on municipal solid waste composition. Waste Management, 79, 828–836. https://doi.org/10.1016/j.wasman.2018.04.017

Chandrappa, R., Das, D. B., 2024. Waste quantities and characteristics. In Solid Waste Management: Principles and Practice (pp. 47–87). https://doi.org/10.1007/978-3-031-50442-6_2

Cai, H., Duan, Z., Lv, J., Zhou, L., Zhou, Q., Yan, Y., Long, J., & Du, J. (2025). Assessing solid waste management effectiveness using the "Zero-waste Index" — Exploration and practice in Chongqing. Circular Economy, 100167. https://doi.org/10.1016/j.cec.2025.100167

Colivicchi, I., Fabbri, T., & Lombardi, G. V. (2025). Performance indicators for measuring municipal and national targets: a comprehensive analysis in Waste Management. In Environment, Development and Sustainability. Springer Science and Business Media B.V. https://doi.org/10.1007/s10668-025-06423-w

Emmanouil, C., Roumeliotis, D., Kostas, A., & Vagiona, D. G. (2025). Assessment of Circular Economy Implementation in Municipal Waste Management Through Performance Indicators and Citizens' Opinion in a City in Western Greece. Sustainability (Switzerland), 17(5). https://doi.org/10.3390/su17052265

Government of Kerala. (n.d.). Haritha Keralam Mission. https://haritham.kerala.gov.in/

Greene, K. L., Tonjes, D. J., 2014. Quantitative assessments of municipal waste management systems: Using different indicators to compare and rank programs in New York State. Waste Management, 34, 825–836. http://dx.doi.org/10.1016/j.wasman.2013.12.020

Manipura, A., Drechsel, P., Jayawardhana, H. M. L. C., Shanmugapriya, R., Mannapperuma, N. R., & Jayatilleke, H. S. I. (2025). Improved performance indicators and institutional setup for more viable compost production from municipal solid waste in Sri Lanka. Waste Management Bulletin, 3(3). https://doi.org/10.1016/j.wmb.2025.100217

MSW_AnnualReport_2021-22 CPCB. (n.d.).

Suchitwa mission. (2021). Integrated waste management, Kerala.

Singh, D. J., Dikshit, A. K., Dangi, M. B., Tchobanoglous, G., & Kumar, S., 2025. Performance analysis of municipal solid waste management using technical indicators. Environmental and Sustainability Indicators, 26, 100693. https://doi.org/10.1016/j.indic.2025.100693

Singh, D., Dikshit, A. K., Kumar, S., 2024. Smart technological options in collection and transportation of municipal solid waste in urban areas: A mini review. Waste Management & Research, 42(1), 3–15.

Singh, D., Dikshit, A. K., Kumar, S., 2024a. A paradigm shift in municipal solid waste collection containers and vehicles in urban areas. In Invitation to National Level Conference on Sustainable Waste Management Practices pp. 129–142. Springer Nature Singapore. https://link.springer.com/chapter/10.1007/978-981-99-4186-5_10

Singh, D., Dikshit, A. K., Kumar, S., 2024b. Mitigating landfill gas emissions from open dumpsites using bio-cover. In Sustainable Management of Land, Water and Pollution of Built-Up Area pp. 189–198. https://doi.org/10.1007/978-3-031-56176-4_14

Singh, D., Tembhare, M., Machhirake, N., Kumar, S., 2023. Impact of municipal solid waste landfill leachate on biogas production rate. Journal of Environmental Management, 336, 117643. https://doi.org/10.1016/j.jenvman.2023.117643

Singh, D., Dikshit, A. K., Kumar, S., 2023a. A paradigm shift in municipal solid waste collection containers and vehicles in urban areas. In Invitation to National Level Conference on Sustainable Waste Management Practices. Springer Nature Singapore.

Singh, D., Dikshit, A. K., Kumar, S., 2023b. Quantifying the impact of municipal solid waste litter using environmental status index in urban areas. Journal of Cleaner Production, 430, 139653. https://doi.org/10.1016/j.jclepro.2023.139653

Singh, D., Chavan, D., Pandey, A. K., Periyaswami, L., Kumar, S., 2021. Determination of landfill gas generation potential from lignocellulose biomass contents of municipal solid waste. Science of the Total Environment, 785, 147243. https://doi.org/10.1016/j.scitotenv.2021.147243

Suryawan, I. W. K., & Lee, C. H. (2024). Achieving zero waste for landfills by employing adaptive municipal solid waste management services. Ecological Indicators, 165. https://doi.org/10.1016/j.ecolind.2024.112191

Trushnaa, T., Krishnan, K., Sonic, R., Singh, S., Kalyanasundaram, M., Annerstedt, K. S., Pathak, A., Purohit, M., Lundborg, C. S., Sabde, Y., Atkins, S., Sahoo, K. C., Rousta, K., Diwan, V., 2024. Interventions to promote household waste segregation: A systematic review. Heliyon, 10, e24332. https://doi.org/10.1016/j.heliyon.2024.e24332

Wu, H., Chen, R., Yuan, H., Yong, Q., Weng, X., Zuo, J., Zillante, G., 2024. An evaluation model for city-scale construction and demolition waste management effectiveness: A case study in China. Waste Management, 182, 284–298. https://doi.org/10.1016/j.wasman.2024.04.047

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MINING ALUMINUM FROM LANDFILLS: CHALLENGES, TECHNOLOGIES, AND THE CASE FOR PRE-LANDFILL INTERVENTION

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ABSTRACT
Aluminum, a critical mineral in modern industry and energy transition, presents substantial environmental and economic burdens when produced through primary methods due to its high energy consumption and associated emissions. Although secondary production through recycling offers significant energy and emissions savings, it is widely believed that a large volume of aluminum still ends up in Municipal Solid Waste Landfills, particularly in the form of used beverage cans (UBCs) and other post-consumer products. This paper explores the feasibility and challenges of recovering aluminum from landfills through landfill mining (LFM). It presents an in-depth review of the historical and current state of aluminum waste generation in the United States, highlighting the growing accumulation of aluminum in landfills and the limitations of existing recycling practices. The study evaluates the physiochemical conditions of landfills affecting aluminum stability and outlines key technologies for material recovery, including air classification, magnetic separation, eddy current separation, electrostatic separation, sink-float techniques, optical sorting and LIBS. Despite technological advances, the recovery of aluminum from landfills remains hindered by numerous barriers such as technical inefficiencies, contamination, oxidation, high operational costs, regulatory ambiguity, and public opposition. Crucially, a techno-economic assessment demonstrates that standalone aluminum recovery from landfill is profoundly uneconomic under any plausible combination of parameters. The findings emphasize the importance of pre-landfill aluminum recovery as a more viable and sustainable strategy. By diverting aluminum from disposal before landfilling, the environmental, economic, and logistical burdens of post-disposal recovery can be substantially mitigated. This review advocates for upstream integration of aluminum recovery into waste management systems as a pathway toward resource efficiency, landfill space conservation, and reduced reliance on energy-intensive primary production.

1. INTRODUCTION

Aluminum, crucial for its properties and the third most abundant element in the earth's crust, was first utilized in the 19th century (Ashkenazi, 2019). Its extraction, initially hindered by its high affinity for oxygen, became viable with the Hall-Héroult process in 1886 (Rabinovich, 2013). Today, aluminum production is dominated by the combined Bayer and Hall-Héroult processes, and is an important metal in many industries (Ashkenazi, 2019).

1.1 Aluminum as a critical mineral and its relevance in the energy transition

Although aluminum is abundant in the Earth's crust, it is classified as a critical mineral under the U.S. Energy Act of 2020 because it is essential to economic and national security and has a vulnerable supply chain (Fortier et al., 2018). Aluminum requires a considerable amount of energy to be converted to its metallic state from its oxide (Al_2O_3), making availability in the metallic form more important than availability of the ore. Most of this energy must be provided in the form of electricity, not just heat. The energy-intensive nature of aluminum production is a key factor in its classification as a critical mineral (Energy, 2023). Due to high energy costs, especially in the United States and Europe, multiple smelters have been forced to shut down or reduce their production output of primary aluminum, leading to a reduced number of operational smelters in some regions. In response, the industry is pursuing significant innovations, including the development of inert

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anode technology and other process improvements aimed at drastically lowering energy consumption and achieving near-zero direct emissions from the smelting process (Ahmad et al., 2025; Munshi). Concurrently, there is a trend for some primary production facilities to integrate or transition towards secondary aluminum production to improve economic and environmental sustainability.

1.2 Aluminum Production

Aluminum production consists of primary production (from bauxite) and secondary production (recycling scrap) (Padamata et al., 2021). Primary aluminum is produced through the Bayer process, which refines bauxite into alumina, followed by the Hall–Héroult electrolytic process, which reduces alumina to metallic aluminum (Buffington, 2012). These processes are highly energy intensive, requiring approximately 23–24 kWh per kilogram of aluminum, largely in the form of electricity for electrolysis (Choate & Green, 2003). According to literature, for every 4kg bauxite, 2 kg of aluminum oxide can be obtained and 1kg of aluminum metal produced (David & Kopac, 2013). Approximately, 23.8 kWh (comprising heat energy for Bayer refining and electrical energy for Hall-Heroult smelting) is required to produce a kilogram of aluminum from bauxite under the conventional method of aluminum production (Choate & Green, 2003). As a result, primary aluminum production contributes roughly 1% of global CO_2 emissions, with life-cycle emissions ranging from 14.98 to 23.96 tons CO_2-eq per ton of aluminum, depending on the energy source (Ali Hasanbeigi, November 21, 2017; Shen & Zhang, 2024). Research is actively pursuing decarbonization, notably inert anodes that could eliminate direct CO_2 emissions from electrolysis if powered by renewables (Saevarsdottir & Kvande, 2025). While not yet commercially widespread, this technology represents a critical pathway for reducing the future carbon footprint of primary aluminum (Chen et al., 2025). Waste generation, particularly red mud from the Bayer process, is another major issue: 1-1.5 tons of highly alkaline residue per ton of alumina, with ~117 million tons generated annually (Archambo & Kawatra, 2021; Wen et al., 2024).

The challenges associated with primary aluminum production has shifted the world's focus on secondary aluminum production (Tabereaux & Peterson, 2024). Secondary aluminum production is basically collecting and recycling aluminum scrap into usable aluminum (Capuzzi & Timelli, 2018). This type of aluminum production is 95.0% more energy-efficient than primary aluminum production (Green, 2007). This is because in secondary aluminum production, the energy intensive processes (Bayer and Hall-Heroult) in primary aluminum production are eliminated. Modern recycling achieves material yields >95.0%, with oxidation losses as low as 1-5% (Al-Alimi et al., 2024b; Capuzzi & Timelli, 2018). Producing one ton of secondary aluminum requires only 750-1500 kWh (directory, 2025), with CO_2-eq emissions of ~0.32 metric tons, compared to 14.98-23.96 tons for primary aluminum (Ali Hasanbeigi, November 21, 2017; Shen & Zhang, 2024).

Secondary aluminum production can be effective tool to reduce aluminum importation in countries that depend heavily on aluminum imports, such as the United States of America. Currently, the production of primary aluminum has decreased significantly in the U.S., and and the U.S. imported 5.54 million tons in 2023 (Chen & Graedel, 2012). Data from the EPA suggest that 3.9 million tons of aluminum waste was generated in 2018, of which about 2.7 million tons was landfilled (EPA, 2023). This suggests that improving the recycling industry in the USA can significantly reduce the amount of aluminum waste landfilled and consequently reduce the USA's dependency on other countries for raw aluminum.

1.3 Aluminum recycling as a share of total aluminum consumed is declining, even in states with container deposit laws

Despite the long-standing environmental and economic advantages associated with aluminum recycling, recent data indicate a concerning downward trend in the share of recycled aluminum relative to total aluminum consumption in the United States. This decline persists even in states that have adopted policies specifically designed to promote recycling behaviors, such as container deposit laws.

1.3.1 Trends and Data on Aluminum Recycling

According to the U.S. Geological Survey, the United States recovered approximately 3.2 million tons of aluminum from purchased scrap in 2021. This amount includes both new (manufacturing) scrap and old (post-consumer) scrap, with new scrap accounting for 53.0% and old scrap making up 47.0%. Notably, aluminum recovered from old scrap was equivalent to just about 30.0% of the nation's apparent consumption. This data reveals a critical gap that a substantial portion of aluminum scrap from post-consumer products, is not being recycled (Center-USGS, 2022).

Aluminum recycling in the United States is assessed using multiple metrics, each targeting different components of the recycling system. End-of-life (EOL) indicators such as the Collection Rate (CR) and Recycling Rate (RR) measure how much post-consumer aluminum is recovered and recycled. The Domestic Recycling Rate (DRR) refines this by excluding exports, offering a clearer view of materials retained and processed within the U.S. system. Recycling Input Rates (RIRs) evaluate the proportion of recycled aluminum in manufacturing inputs but can be inflated by the inclusion of pre-consumer (new) scrap, which does not reflect actual recovery from waste. The Old Scrap Ratio (OSR) addresses this limitation by quantifying only post-consumer recycled content (Association, 2024).

More recently, the Consumer Recycling Rate, which stood at 43.0% in 2023, has been emphasized for its accuracy in capturing domestic recycling behavior, as it excludes imported scrap. In contrast, the Industry Recycling Rate, reported at 57.0%, includes imported used beverage cans (UBCs) and exported material recycled abroad. The Aluminum Association includes imported UBCs to reflect total scrap processed by U.S. producers; however, this approach may overstate domestic recycling effectiveness. For evaluating material circularity and national policy outcomes, DRR, OSR, and the Consumer Recycling Rate offer more relevant and transparent indicators than industry-aggregated rates (Association, 2024; Institute, 2024).

1.3.2 Aluminum Beverage Cans: A Case Study in Declining Recovery

Aluminum Used Beverage Cans (UBCs) have historically played a central role in aluminum recycling due to their ubiquity and ease of collection. Recycling of UBCs in the U.S. peaked in 1992 with a rate of approximately 65.0%. According to the Aluminum Association and the Can Manufacturers Institute (CMI), the national recycling rate for aluminum cans fell to 43.0% in 2023, marking one of the lowest levels in recent decades. This rate represents a significant decline from the long-term average recycling rate of approximately 52.0% recorded since 1990. The reported statistics are derived from a comprehensive survey of aluminum can manufacturers and recyclers, complemented by governmental and other relevant data sources (Association, 2024). This downward trend coincides with a decrease in the inflation-adjusted market value of used aluminum cans, which has reduced the economic incentive (deposit) for collection and recycling.

A major issue lies in the recycled content of new aluminum cans, which typically ranges between 33.0% and 50.0%. Although it is technically possible to manufacture a can from 100.0% recycled material, doing so is considered inefficient within the current supply chain structure. This limitation arises from the inherently labor-intensive and disaggregated structure of the post-consumer collection system for used beverage cans (UBCs), which hinders efficient aggregation and processing at scale (Buffington, 2012). While aluminum cans are recycled at far higher rates than glass or plastic, these rates have fallen below 50% in recent years.

Despite aluminum's inherent recyclability and high material value, the U.S. continues to discard a vast quantity of used aluminum cans. In 2023, although approximately 46 billion aluminum cans were recycled, over 61 billion were discarded in landfills, resulting in an estimated loss of $1.2 billion worth of recyclable material (Association, 2024). This loss is equivalent to each person in the U.S. throwing away nearly 15 twelve-packs of aluminum cans annually, a striking indicator of the inefficiencies in the nation's recycling system.

Although aluminum still maintains a higher recycling rate than glass 39.6% and PET plastic bottles 20.0%, the recent decline represents a significant missed opportunity given aluminum's economic and environmental advantages (Association, 2024). Enhancing the collection, sorting, and processing of aluminum cans could yield substantial benefits in terms of resource efficiency, greenhouse gas reduction, and domestic material supply security (Gitlitz, 2002).

1.4 Scope of the Review

Given the persistent landfilling of large quantities of highly recyclable aluminum, this paper critically evaluates landfill mining (LFM) as a potential solution. We synthesize historical waste data, review separation technologies, and present a techno-economic assessment that quantifies the profound economic infeasibility of standalone aluminum recovery from landfills. We then analyze the environmental, technical, regulatory, and social barriers that render LFM impractical at scale. The central argument of this paper is that pre-landfill intervention – recovering aluminum before it is buried – is not merely preferable but necessary. Upstream recovery preserves material quality, avoids oxidation and contamination, eliminates the need for costly excavation, and aligns with circular economy principles. We conclude by outlining policy pathways and research priorities to shift the paradigm from post-hoc mining to proactive diversion.

1.5 Review Methodology

This review was conducted following a systematic approach to ensure comprehensive coverage and analytical thoroughness. Literature was identified through searches in the Scopus, Web of Science, and Google Scholar databases, using keyword combinations including "aluminum recycling," "landfill mining," "municipal solid waste aluminum," "aluminum recovery," "secondary aluminum," "used beverage cans," and "separation technologies." The search was not restricted by start date, with a cutoff in December 2024, to capture both foundational and recent developments.

Inclusion criteria prioritized peer-reviewed journal articles, authoritative industry reports and seminal books. Governmental and institutional data were included for quantitative analysis of waste generation and recycling trends (e.g., from the Aluminum Association, U.S. EPA, USGS). Exclusion criteria removed non-English publications, articles focusing solely on primary production without a waste/recovery link, and very short conference abstracts lacking substantive data. After screening titles, abstracts, and full texts, 114 sources were selected for in-depth analysis and synthesis. Evidence was synthesized thematically to address the core research questions: the scale of aluminum in landfills, the physicochemical fate of landfilled aluminum, the technological pathways for its recovery, and the associated economic and regulatory barriers. Quantitative data from U.S. agencies were harmonized and, where necessary, interpolated to construct cumulative time-series analyses. Findings from technological and case studies were compared to evaluate efficiency, readiness, and feasibility, leading to the critical assessment and advocacy for pre-landfill intervention presented in this review.

While previous reviews have provided broad overviews of landfill mining and material recovery challenges, often from global or European perspectives (Jain et al., 2023; Krook et al., 2012), this review offers a targeted and original contribution focused on the United States context. First, it presents a data-driven quantification of the historical accumulation of aluminum in U.S. landfills from 1960 to 2018, integrating fragmented national datasets to demonstrate the scale of aluminum resource loss. Second, the study bridges aluminum corrosion science with landfill biogeochemistry, using Pourbaix-based frameworks to assess aluminum survivability under realistic landfill conditions and to move beyond generic recovery potential toward a material-specific evaluation of recoverability. Third, the review provides a dedicated assessment of the economic and regulatory viability of aluminum landfill mining in the United States, explicitly contrasting U.S. policy and market conditions with the more established European experience. Collectively, these contributions culminate in a focused argument that upstream, pre-landfill aluminum recovery represents the most sustainable and practical pathway for improving resource efficiency, reducing environmental risk, and strengthening domestic material security within U.S. waste management systems.

2. STATE OF ALUMINUM WASTE MANAGEMENT IN THE U.S.

The U.S. has witnessed a substantial increase in aluminum usage over the past six decades, leading to a corresponding rise in aluminum waste (Chen & Graedel, 2012; Kulas et al.). While the U.S. Environmental Protection Agency (EPA) reports annual aluminum generation data, cumulative generation data over multiple decades are not directly provided (Agency, 2018a, 2018b, 2018c; Agency, 2020; Recovery, 2014). Therefore, this study calculated the cumulative aluminum generation from 1960 to 2018 by first using interpolation to estimate the amount of aluminum generated in the missing years and then summing them with the annual generation data reported by the EPA. This approach provides a more comprehensive perspective on the total aluminum generated over the decades.

Data from the U.S. EPA show that the total cumulative amount of aluminum generated from the early 1960s to 2018 exceeded 130 million U.S tons as shown in Figure 1. Annual generation rose from 340,000 U.S tons in the 1960s to nearly 4 million U.S tons by 2018. This growth is largely driven by increased consumption in packaging (especially beverage cans), durable goods (including appliances and electronics). The U.S. EPA did not provide specific data on the amount of aluminum from durable goods that was generated in their 2018 report.

To estimate this quantity, the annual aluminum generation from containers and packaging is subtracted from the total annual aluminum generation, with the remaining quantity attributed to durable goods. A graphical representation of this trend (Figure 1) shows exponential growth across all major aluminum generation categories, with containers and packaging leading the increase, reaching a cumulative 83 million U.S tons in 2018. While aluminum generation in the U.S. has increased exponentially, the amount of aluminum that ends up in landfills has also grown over the decades. This clearly highlights a gap in the aluminum recycling culture in the U.S.

The U.S. EPA did not provide specific data on the amount of aluminum from durable goods that ends up in landfills in their 2018 report. To estimate this quantity, the annual amount of aluminum landfilled from containers and packaging is subtracted from the total annual aluminum landfilled, and the remaining amount is attributed to durable goods. A similar methodology used to calculate cumulative aluminum generation from 1960 to 2018 was applied to estimate the cumulative amount of aluminum entering landfills, based on data reported by the EPA. The cumulative landfill data reveal a concerning buildup of aluminum waste, with over 90 million tons discarded by 2018. As shown in Figure 2, containers and packaging account for more than half of this accumulated waste, while durable goods represent a growing portion due to their increasing presence in consumer products (Agency, 2018a, 2018b, 2018c; Agency, 2020; Recovery, 2014).

2.1 Aluminum Waste by Type

Containers and packaging (beverage cans, food cans, food containers and foil) and durable goods (products that last three years or more) constitute the largest sources of aluminum in municipal solid waste. As shown in Figure 3, approximately 1 million U.S tons of containers and packaging ended up in landfills 2018, while the cumulative amount

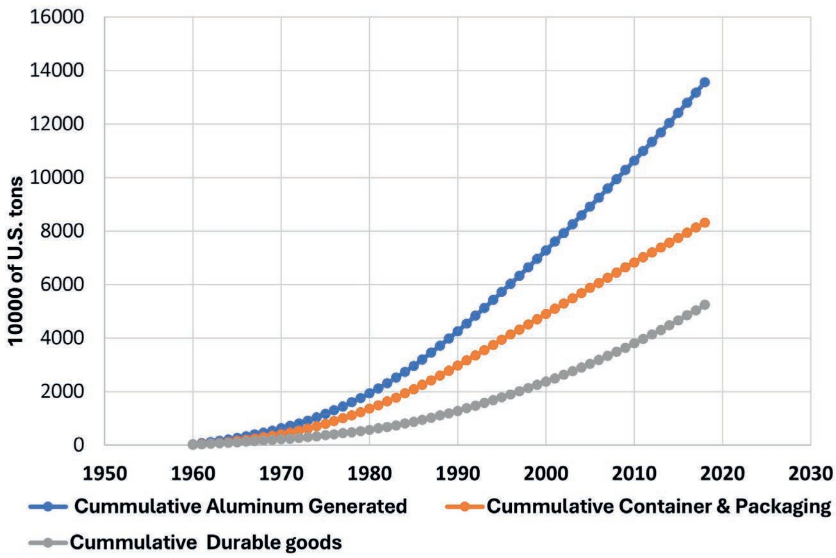

FIGURE 1: Cumulative aluminum waste generation in the United States from 1960 to 2018. Data are sourced from the EPA's 2018 report, Materials in the Municipal Waste Stream, with missing years interpolated using available data provided in the report (EPA, 2023).

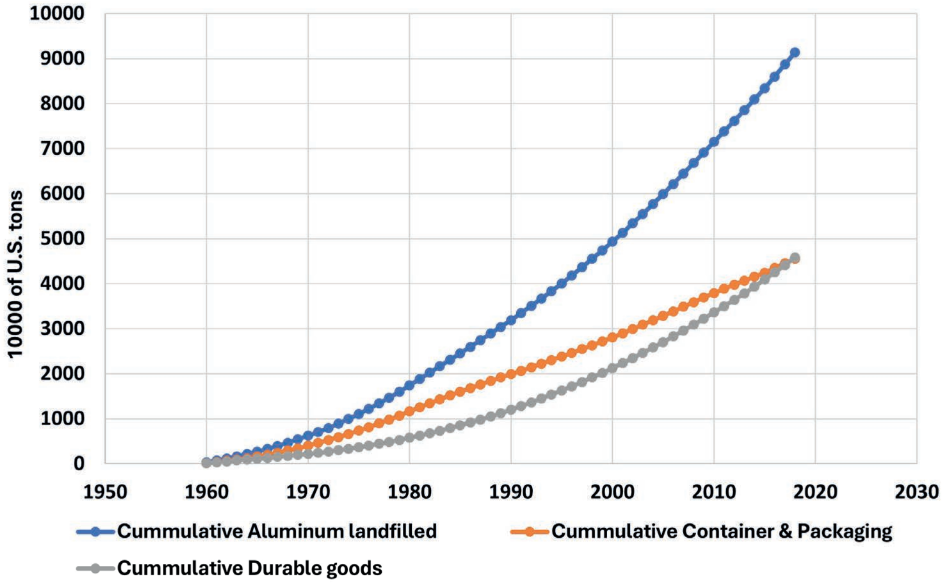

FIGURE 2: Cumulative aluminum landfilled in the United States from 1960 to 2018. Data are sourced from the EPA's 2018 report, Materials in the Municipal Waste Stream, with missing years interpolated using available data provided in the report (EPA, 2023).

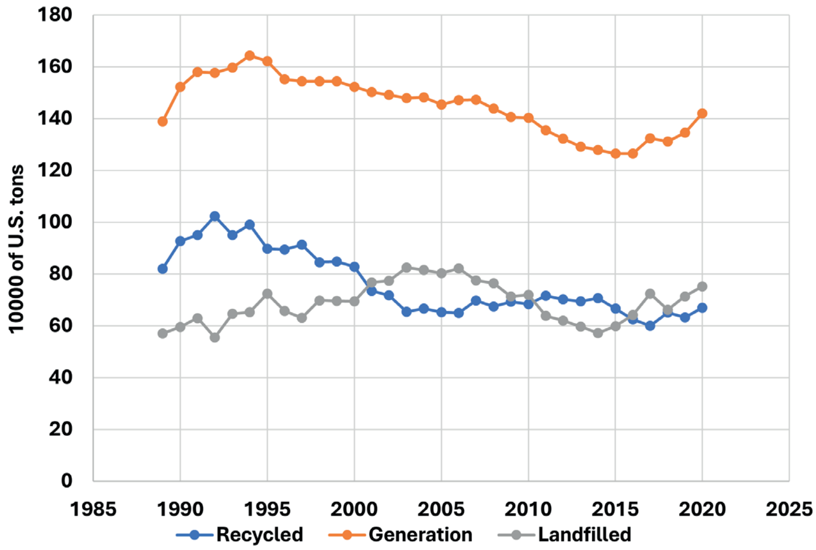

FIGURE 3: Generation, recycling, and landfilling of used beverage cans (UBCs) in the United States. Data exclude imported UBCs and are sourced from the Aluminum Association (Agency, 2020).

is 45.5 million U.S. tons, with beverage cans contributing the most to this total (Agency, 2020). As illustrated in Figure 2, the cumulative quantity of aluminum from durable goods disposed of in landfills has increased substantially over the decades, reaching approximately 45.8 million U.S. tons by 2018. In that same year, the annual landfill contribution from durable goods was estimated at 1.6 million U.S. tons.

Despite the high recyclability and economic value of aluminum, a substantial portion of aluminum waste, particularly in the form of Used Beverage Cans (UBCs), continues to be disposed of in landfills annually in the United States. Data from the Aluminum Association over the past three decades indicates a consistent increase in the volume of aluminum UBCs being landfilled since the early 2000s. In 2018 alone, approximately 660,000 U.S. tons of aluminum UBCs were landfilled, representing nearly half of the total generated as shown in Figure 3 (Agency, 2020). Between 1995 and 2015, UBCs waste generation declined from over 1.6 million to around 1.3 million U.S. tons. This may be due to lightweighting of aluminum cans by manufacturers and innovations in beverage cans design incorporating other materials such plastics.

The persistent landfilling of nearly half of all generated aluminum UBCs, despite their high economic value and recyclability, points to systemic failures in the U.S. recycling system. Several interconnected factors likely contribute to this gap. Economically, the fluctuating market price for scrap aluminum and the relatively low (and inflation-eroded) deposit value in bottle bill states can weaken financial incentives for consumer recycling and for municipalities to operate robust collection programs. Logistically, the reliance on single-stream recycling, while convenient, often leads to contamination of material streams, reducing the quality and value of collected aluminum (Morawski, 2010). Furthermore, the absence of a federal container deposit law (only 10 states have such laws) means a majority of the U.S. population lacks a direct economic incentive to return cans. Public awareness and convenience also play a role; recycling rates are often lower in multi-family housing and public spaces where access to recycling bins is limited (Buffington, 2012; Gitlitz, 2002). Addressing this issue requires a multi-pronged strategy involving policy (e.g., extended producer responsibility, modernized deposit laws), investment in sorting technology, and public education.

By 2021, the gap between generation, recycling, and landfilling had further narrowed, with landfilling and recycling rates approaching equivalence. This trend signifies a significant missed opportunity for material recovery. Aluminum cans are among the most energy-intensive materials to produce from virgin resources, yet they can be recycled indefinitely without any degradation in quality. The landfilling of this material not only contributes to environmental degradation but also results in the loss of substantial embodied energy and economic value (Al-Alimi et al., 2024a; Gaines, 1981; Green, 2007; Miteva & Hodjaoglu).

3. EFFECTS OF LANDFILL PHYSIOCHEMICAL PROPERTIES ON ALUMINUM

The substantial accumulation of aluminum in municipal solid waste landfills represents a potentially significant secondary resource; however, its recovery potential depends not on total buried mass but on the fraction that retains its metallic form. The long-term preservation of metallic aluminum is governed by the dynamic and heterogeneous physicochemical environment within landfill systems, where evolving pH, redox potential (Eh), moisture availability, and waste composition control corrosion processes and material stability (Stark et al., 2012).

Landfill conditions evolve as organic waste undergoes biodegradation, producing spatial and temporal variations in temperature, moisture, oxygen availability, and chemical composition (Igwegbe et al., 2024). Depth strongly influences these conditions: shallow zones experience greater oxygen intrusion and moisture fluctuations, while deeper layers are typically compacted, saturated, and anaerobic. These gradients regulate pH and Eh, which together determine aluminum stability.

The Pourbaix diagram for aluminum provides a thermodynamic framework for interpreting these processes (Pourbaix, 1974). Higher Eh values indicate oxidizing environments associated with aerobic activity, whereas lower Eh values reflect reducing anaerobic conditions. Aluminum remains corrosion-resistant within a pH range of approximately 4-9 due to formation of a protective oxide layer. Outside this range, the oxide film destabilizes. Acidic environments (pH < 4), generated during acidogenic decomposition, and highly alkaline conditions (pH > 9), often associated with ash residues or concrete degradation, both promote aluminum dissolution and oxidation (Pourbaix, 1974).

Moisture further accelerates degradation by enabling electrochemical reactions and microbial activity. Organic matter decomposition produces carbon dioxide and organic acids that promote pitting corrosion. Under alkaline anaerobic conditions, aluminum may react directly with water:

$$2Al + 6H_2O \rightarrow 2Al(OH)_3 + 3H_2\uparrow$$

forming aluminum hydroxides and hydrogen gas (Calder & Stark, 2010; Stark et al., 2012). Field observations of hydrogen generation and localized heating confirm that aluminum oxidation can persist for decades after burial. These reactions progressively convert metallic aluminum (Al^0) into hydroxides and aluminosilicate phases that are unrecoverable through conventional separation or remelting (Calder & Stark, 2010).

Beyond oxidation, Al^{3+} released during corrosion can complex with humic and fulvic substances in landfill environments, forming stable organo-aluminum species that persist in leachate or become incorporated into fine fractions (Lee et al., 2023). Because this aluminum is no longer metallic or conductive, it cannot be recovered through conventional mechanical separation technologies (Parrodi et al., 2018). Recovery would require energy- and chemical-intensive hydrometallurgical treatment, rendering it economically unattractive (David & Kopac, 2013). This pathway represents an additional irreversible loss mechanism, further reducing practical recoverability and strengthening the case for pre-landfill recovery.

Material characteristics also influence degradation rates. Thicker, coated used beverage cans (UBCs) corrode relatively slowly, whereas thin, uncoated aluminum foils rapidly degrade and fragment into fine particles that become incorporated into soil-like fractions. Because aluminum is recyclable only when its metallic structure is preserved, such transformations directly reduce recoverable resource potential (Alicia Vallejo-Olivares, October 7th,2020; Guide, 2025)

The cumulative effect of these degradation pathways is quantifiable in actual landfill mining operations. A Belgian landfill mining study reported recovery of approximately 70% of aluminum as metallic concentrate, while about 30% was lost to fine residues due to corrosion and particle size reduction (Lucas et al., 2019; Wen et al., 2025). This loss reflects irreversible chemical transformation rather than processing inefficiency. Recovery rates are highly site-dependent: landfill age and hydrogeological conditions control moisture exposure and long-term chemical evolution (Wen et al., 2025). Younger landfills generally retain more metallic aluminum, whereas older sites expose materials to decades of chemically aggressive conditions, progressively reducing recoverable fractions.

Consequently, aluminum recovery potential varies widely across landfill environments and cannot be inferred from disposal quantities alone. Evidence suggests that in mature landfills, up to roughly 30% of aluminum mass may already be irreversibly lost through corrosion, with the remaining fraction representing an upper bound for recovery.

In wetter or more chemically aggressive settings, losses may be greater (Lucas et al., 2019).

These findings shift landfill mining evaluation from theoretical resource estimation toward conditional recoverability. Although metallic aluminum persists in landfills, progressive degradation imposes an inherent reduction in recoverable material over time, reinforcing the importance of upstream recovery and pre-landfill intervention to preserve aluminum in its recyclable metallic state.

4. LANDFILL MINING

Landfill mining (LFM) is an emerging and innovative waste management strategy that focuses on the extraction and recovery of resouces from landfills (Krook et al., 2012). With the tremendous amount of UBCs and other aluminum waste in landfills, mining landfills to recover these aluminum waste can serve as a way to mitigate conventional aluminum mining and greatly reduce the U.S. dependence on imports. Primarily there are two types of landfill mining, which are in-situ landfill mining and ex-situ landfill mining (Jones et al., 2013).

In-situ landfill mining involves the recovery of resources from landfills without physical excavation (Burlakovs et al., 2017). While historically focused on methane extraction, recent technological advances have expanded the focus to include the extraction of dissolved metals and nutrients from leachate. Although significant research and pilot-scale innovation over the past decade demonstrate the technical feasibility of recovering resources like ammonia, phosphorus, and metals, these processes have not yet achieved widespread commercial adoption (Wang et al., 2024). However, this method is not suitable for aluminum recovery intended for recycling. Aluminum present in leachate exists in ionic or oxidized forms and cannot be directly recycled. Instead, it would require electrolysis to be converted back into its metallic state, a process akin to primary aluminum production, which is highly energy-intensive and costly. Therefore, in-situ mining is not a practical approach for recovering recyclable aluminum from landfills.

Ex-situ landfill mining, however, involves material recovery from landfills by first excavating the landfill and processing them to recover valuable materials for recycling. Aluminum waste recovered through this type of landfill mining is suitable for recycling, provided the material has not undergone oxidation (Pecorini & Iannelli, 2020).

4.1 Theoretical Framework: Landfill Mining in the Context of Circular Economy

The concept of landfill mining must be understood within the broader framework of the circular economy, which seeks to decouple economic growth from resource consumption through closed-loop systems (Kirchherr et al., 2017). Traditional linear economic models follow a 'take-make-dispose' pattern, while circular economy principles emphasize waste prevention, resource efficiency, and material recovery (Sariatli, 2017). Landfill mining represents a 'retroactive' circular strategy, attempting to recover value from already discarded materials. However, this approach conflicts with the circular economy's emphasis on upstream design and prevention. This tension creates a critical paradox: while landfill mining recovers resources, its necessity signals failure in upstream circular systems. Evaluating landfill mining through circular economy metrics, such as material circularity indicators, resource productivity, and system-level impacts, reveals that pre-landfill interventions align more closely with circular principles by preventing waste generation rather than remediating it (Goh et al., 2025; Jones et al., 2013; Kirchherr et al., 2017).

The comparative analysis in Table 1 reveals significant international variation in approaches to aluminum recovery and landfill mining. European countries, particularly Sweden , have pioneered landfill mining primarily as an environmental remediation strategy, with aluminum recovery as a secondary benefit (Johansson et al., 2017);(Krook & Baas, 2013; Vollprecht et al., 2021). In contrast, developing economies often achieve higher aluminum collection rates through extensive informal sectors (waste pickers, itinerant waste buyers and small-scale scrap shops), though with significant environmental and social costs (Ferronato & Torretta, 2019). Japan's approach demonstrates that high recycling rates (exceeding 90% for aluminum cans) can be achieved through integrated policy frameworks, advanced technology, and strong public participation, reducing the need for landfill mining (Pereira, 2025). The United States occupies a middle ground, with advanced technological capacity but fragmented policies that result in suboptimal recovery rates. Meanwhile, China's state-driven approach – under the National Sword Policy and Circular Economy Promotion Law – emphasizes "urban mining" through government-backed demonstration projects, with recycling rates estimated at 50-60% and rising, though constrained by quality control and energy-intensity challenges (Li et al., 2020; Lindstrom, 2024). These international comparisons highlight that successful aluminum recovery requires context-specific integration of technological, economic, and policy factors (Cadavid, 2025).

4.2 Techniques employed in recovery of aluminum waste

Following landfill excavation, the next phase is the separation stage, which involves removing ferrous metals, nonferrous metals (such as copper and aluminum), plastics, and other materials from the waste stream using various separation technologies. Given the heterogeneous nature of landfill waste, a preliminary shredding process is typically employed to reduce the size of waste materials followed by a screening process, thereby enhancing the efficiency and effectiveness of the separation operations (Quaghebeur et al., 2013). Typically, a combination of two or more separation technologies is employed to selectively recover aluminum from the waste stream.

4.2.1 Air Separation

Air separation is primarily applied as a pre-concentration step to enhance downstream aluminum recovery by removing low-density contaminants such as fines, paper, textiles, and light plastics from excavated landfill material. By reducing the mass of organics and soil-like fines going to subsequent separation stages, air classification improves the efficiency and recovery of aluminum (Brooks

TABLE 1: International policy frameworks for aluminum recovery and landfill mining.

Country/Region	Policy Framework	Aluminum Recycling Rate	Landfill Mining Status	Key Innovations	Major Challenges
United States	Market-driven approach; state-level regulations; 10 states with container deposit laws	47% for aluminum cans (2024);	Limited to remediation projects; few states have regulations; not yet adopted for resource recovery	Advanced MRF technologies; private sector R&D; efficient eddy current separation	Fragmented policy; contamination in single stream; low economic incentives
European Union	Circular Economy Action Plan; Landfill Directive (1999/31/EC); Extended Producer Responsibility (EPR)	60-75% (varies by member state); EU target: 50% by 2025	Most advanced; multiple pilot & operational projects; viewed as resource recovery within CE framework	Integrated circular economy approach; landfill mining pilot projects (e.g., BELGIUM, SWEDEN); standardized metrics	Regulatory ambiguity; high operational costs; public acceptance issues
Japan	Sound Material-Cycle Society; Home Appliance Recycling Law; Container & Packaging Recycling Law	>90% for aluminum cans; ~98% for automotive aluminum	Limited activity; focus on "urban mining" from active waste streams rather than historic landfills	Sophisticated collection & sorting systems; high public participation; advanced urban mining	Limited landfill space; high operational costs; focus on prevention over remediation
Developing Countries (e.g., Brazil, India)	Informal sector dominant; evolving EPR frameworks; limited formal regulations	60-95% (highly variable; often high due to informal collection)	Rare as formal engineering projects; extensive informal scavenging at active dumpsites	Low-cost, labor-intensive networks achieve high collection rates; adaptive informal technologies	Significant environmental & health costs; low quality of scrap; lack of worker protection
China	National Sword Policy; Circular Economy Promotion Law; "Urban Mining (demonstration bases)	50-60% (estimated, improving rapidly)	Emerging focus; government-backed pilot projects; emphasis on "urban mining" from industrial waste	Large-scale implementation potential; state investment; integration with manufacturing supply chains	Quality control; environmental compliance in processing; energy-intensive recovery

et al., 2019; Gesing & Wolanski, 2001). Air separators are capable of handling particles as small as 50-1000 µm, making them suitable for preparing landfill-derived waste for aluminum-focused recovery circuits (Shapiro & Galperin, 2005).

4.2.2 Magnetic Separation

Magnetic separation is used to remove ferrous metals from the waste stream before targeting non-ferrous aluminum recovery (Brooks et al., 2019). A permanent magnet positioned beneath or above a conveyor belt attracts iron and other ferrous metals, which are diverted into a separate collection stream, as depicted in Figure 4 (Kelly & Apelian, 2016). Although magnetic separators do not recover aluminum directly, they play a critical supporting role by eliminating ferrous materials that would otherwise interfere with eddy current separation of aluminum. Additionally, recovered ferrous metals represent a valuable co-product that can partially offset landfill mining operational costs.

4.2.3 Eddy current separator

Eddy current separators (ECS) constitute the core technology for recovering aluminum from mixed waste streams. High-speed rotating magnetic rotors induce eddy currents in electrically conductive, non-ferrous metals such as aluminum, generating repulsive forces that eject these materials away from non-conductive waste (Smith et al., 2019) as shown in Figure 5. As a result, aluminum packaging, beverage cans, and foil are selectively recovered into a concentrated non-ferrous metal fraction. ECS performance strongly influences aluminum recovery efficiency in landfill mining and material recovery facilities, making it the most critical unit operation for aluminum separation (Padamata et al., 2021; Smith et al., 2019).

After non-ferrous metals have been separated from the waste stream, additional processing steps are required to selectively recover aluminum from the mixed nonferrous fraction. The following are some of the technologies commonly employed to achieve selective aluminum recovery, this normally takes place in a material recovery facility.

4.2.4 Electrostatic separation

Following ECS, the recovered non-ferrous fraction often contains a mixture of aluminum and copper. Electrostatic separation exploits differences in electrical conductivity to further refine aluminum purity as shown in Figure 6 (Dascalescu et al., 2016). When mixed metal is exposed to a high-voltage field, highly conductive metals such as copper exhibit greater deflection than aluminum, enabling their separation into distinct streams (Veit et al., 2005). This step is particularly important for producing aluminum fractions that meet smelter-grade purity requirements. Effective operation requires fine particle sizing and low moisture content, which may increase preprocessing costs. The main disadvantages of this separator are that the materials must be finely sized and be completely dry for effective separation.

4.2.5 Sink float method

SSink-float separation can be applied as a secondary refining step to separate aluminum from heavier non-ferrous metals and residual inorganics based on density differences as illustrated in Figure 7. Aluminum, with a density of ~2.7 g/cm³, preferentially floats in suitable media, while copper and other heavy metals sink (Padamata et al., 2021). Although effective, the use of heavy liquids and the need for post-separation drying increase operational complexity and cost, limiting widespread application in landfill mining unless high-purity aluminum recovery is required.

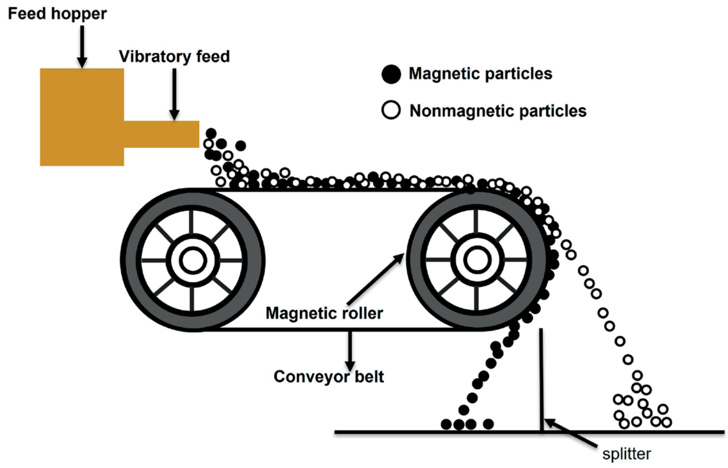

FIGURE 4: Schematic diagram of a magnetic separator comprising a vibratory feed hopper, conveyor belt, magnetic roller, and splitter for the separation of ferrous materials from non-magnetic particles.

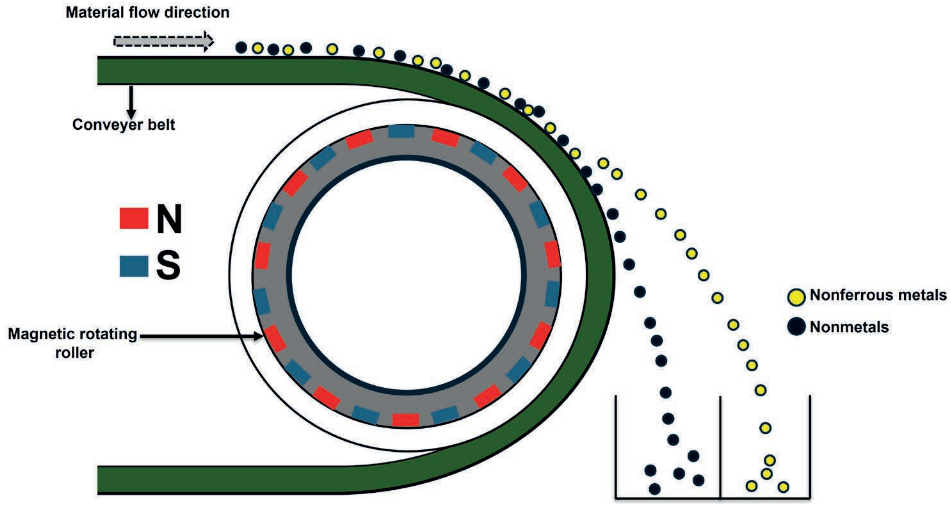

FIGURE 5: Schematic diagram of an eddy current separator comprising a conveyor belt and magnetic rotating roller consisting of alternating north and south poles.

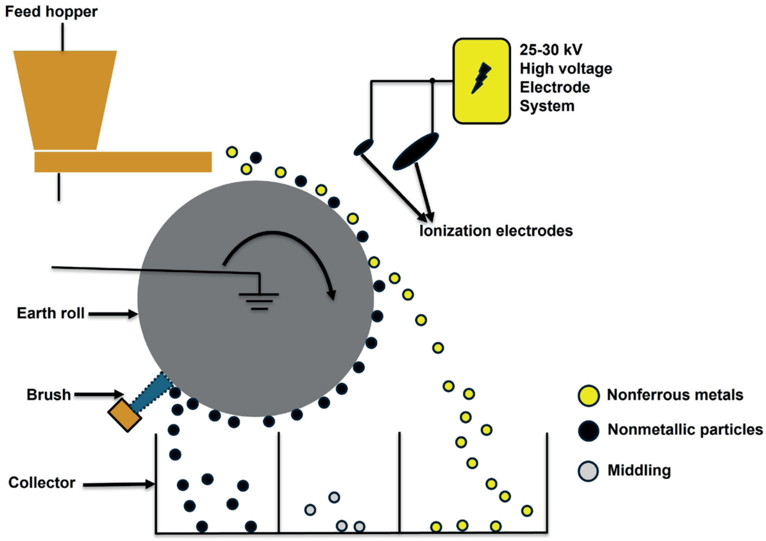

FIGURE 6: Schematic diagram of an electrostatic separator consisting of a vibratory feed hopper, high-voltage electrode system, earth roll, and collection chamber for product separation.

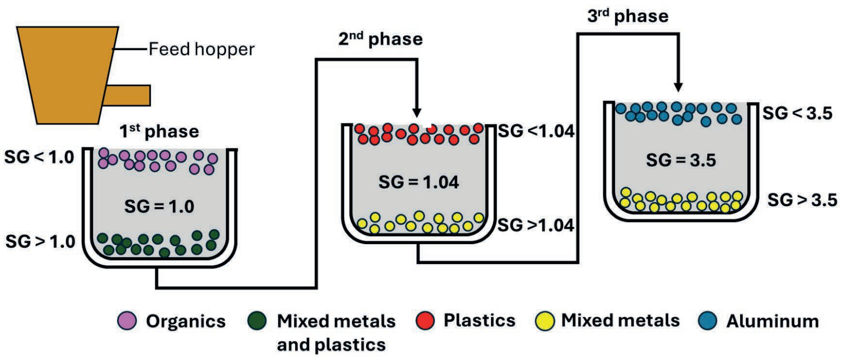

FIGURE 7: Schematic diagram illustrating the sink-float separation technique, highlighting the different phases involved in isolating aluminum from the waste stream.

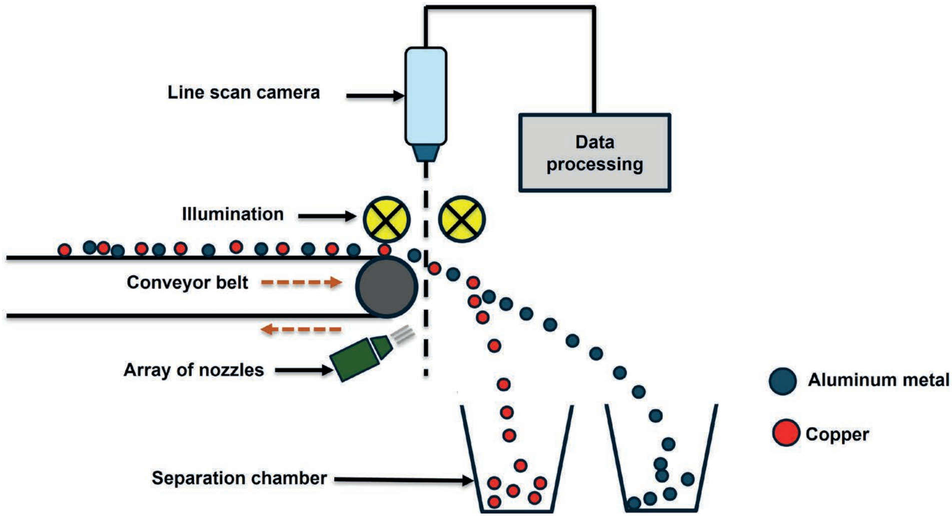

FIGURE 8: A schematic diagram of an optical separator consisting of camera, conveyor belt, nozzles and separation chamber, demonstrating the separation of aluminum from copper.

4.2.6 Optical based sorting

Optical based sorting systems enable targeted identification of aluminum based on surface color, spectral response, and inductive signatures (Mutz et al., 2005).These systems can selectively eject aluminum particles from mixed metal streams (Figure 8) using compressed-air nozzles and are increasingly applied where higher aluminum purity or alloy-specific sorting is desired. When integrated after ECS sensor-based sorting improves aluminum product quality and market value. This separator is suitable for separating materials with particle size above 10mm. Although optical separators are highly effective, they are associated with high operational costs.

4.2.7 Laser-Induced Breakdown Spectroscopy (LIBS)

Laser-Induced Breakdown Spectroscopy (LIBS) is an emerging technique for real-time, elemental identification of metals in mixed waste streams. By focusing a high-energy laser pulse on a particle, LIBS generates plasma whose emitted light spectrum reveals the elemental composition of the material (Cremers & Radziemski, 2006). When integrated downstream of ECS, LIBS can provide rapid, in-line detection and sorting of aluminum from residual copper, zinc, or other non-ferrous metals, enabling precise aluminum alloy-specific separation and recovery (Gurell et al., 2012). The key advantages of LIBS include high-speed analysis, minimal sample preparation, and the ability to detect trace contaminants that may affect smelter-grade purity. Limitations include higher equipment costs and the need for extensive calibration to handle heterogeneous landfill-derived materials effectively.

Each separation technology presents distinct trade-offs between recovery efficiency, operational requirements, and cost as shown in table 2. While individual technologies can achieve high recovery rates in a controlled settings, their effectiveness diminishes significantly in landfill mining applications due to material degradation, high moisture content, contamination, and heterogeneity (Xia et al., 2017). For instance, eddy current separation, typically achieving 85-95% recovery with clean industrial scrap, may achieve low recovery rates with landfill-aged aluminum due to oxidation and surface contamination (Huang et al., 2021). Similarly, electrostatic separation requires completely dry feed material, a condition that is rarely achievable for excavated landfill waste without extensive preprocessing. Excavated waste typically exhibits high moisture content and elevated relative humidity, necessitating energy-intensive drying steps that substantially increase operational costs (Silveira et al., 2018). This gap highlights that landfill mining cannot simply replicate MRF workflows but requires adapted or novel solutions. Emerging technologies such as artificial intelligence (AI)-driven optical sorters, robotic picking, and multi-sensor fusion systems offer potential pathways to better handle complex, degraded waste streams by making more nuanced material identifications. However, their high capital costs and unproven efficacy at landfill mining scale represent significant barriers (Gundupalli et al., 2017). Therefore, the central technological question for landfill mining shifts from what works in an MRF to what can be adapted or invented to work cost-effectively on low-grade, historical waste. These limitations explain why most successful landfill mining operations employ integrated, multi-stage separation systems, despite their higher capital requirements.

5. TECHNO-ECONOMIC ASSESSMENT OF ALUMINUM RECOVERY FROM LANDFILL: IDENTIFYING BREAKEVEN THRESHOLDS AND ECONOMIC CONSTRAINTS

This study presents a techno-economic assessment (TEA) of a commercial-scale aluminum recovery stream. By utilizing a base-case model derived from literature and realistic operational parameters, such as an annual project scale of 100,000 tonnes and a separation efficiency of 70%, we evaluate the net economic balance of standalone aluminum recovery. Furthermore, this analysis establishes critical breakeven thresholds for aluminum content and operational expenditures (OPEX), providing a rigorous framework to determine whether single-stream recovery is a feasible industrial strategy or if a transition toward multi-stream revenue models is required for financial sustainability.

5.1 Techno-economic assessment methodology

The net economic balance N for an aluminum recovery stream is calculated as:

$$R_{Al} = M_{waste} \times Y_{Al} \times \eta \times f_{cont} \times P_{Al}$$
$$C_{total} = M_{waste} \times C \ per\ tonne$$
$$N = R_{Al} - C_{total}$$

R_{Al} - Revenue generated from selling recovered aluminum
Operational Cost (OPEX) - cost of excavation, transportation and separation and recovery of metal.
Capital Expenditure (CAPEX) - equipment and machinery investment

Base-case parameters are summarized in Table 3 Values are drawn from peer-reviewed literature (Hogland et al., 2011; Ratcliffe et al., 2012; van Vossen & Prent, 2013; Wagner & Raymond, 2015) and industry reports, with conservative assumptions favoring economic viability.

5.2 Base-case results

$$R_{Al} = 100{,}000 \times 0.008 \times 0.70 \times 0.50 \times 2{,}200 = USD\ 616{,}000$$
$$C_{total} = 100{,}000 \times 40 = USD\ 4{,}000{,}000$$
$$N = 616{,}000 - 4{,}000{,}000 = USD\text{-}3{,}384{,}000$$

Net loss: USD -3.38 million. This loss equates to USD -33.84 per tonne of waste processed.This implies that standalone aluminum recovery is not economically feasible.

TABLE 2: Comparative analysis of aluminum separation technologies.

Technology	Recovery Efficiency for Aluminum	Key Advantages	Key Limitations for landfill waste	Operation Cost
Air Separation	60-75% for bulk density separation	Low capital cost, simple operation, high throughput, effective for lightweight material removal	Ineffective on wet waste; limited to dry, pre-shredded material; misses small or embedded Al	Low
Magnetic Separation	Not for aluminum (95%+ for ferrous removal)	Highly efficient for ferrous metals, continuous operation, low maintenance, improves downstream Al quality	Only removes ferrous metals; requires additional processes for aluminum recovery; ineffective for non-magnetic materials	Low-Medium
Eddy Current Separation	70-85% for clean Al scrap; 50-70% for landfill-aged Al	Effective for aluminum, copper, brass; handles varied particle sizes; non-contact method	Efficiency decreases with oxidation, small particles (<5mm), moisture; high energy consumption; sensitive to material conductivity	Medium-High
Electrostatic Separation	75-90% for conductive metals	High purity achievable; separates based on electrical conductivity; effective for aluminum and copper separation	Requires completely dry, finely sized materials (<5mm); sensitive to surface contamination; high voltage requirements	Medium
Sink-Float Separation	80-90% based on density differences	Can separate multiple materials simultaneously; high purity achievable; effective for mixed metal streams	Environmental concerns with heavy liquids; requires drying; high operational costs; media management needed	High
Optical Sorting	85-95% for identifiable aluminum pieces	High accuracy; effectively remove aluminum from waste stream; non-contact process; real-time processing	Requires >10mm particle size; sensitive to surface contamination and coatings; high capital investment; requires clean material presentation	High
Laser-Induced Breakdown Spectroscopy (LIBS)	85–95% (when integrated with pre-separation technologies)	Real-time elemental identification; enables alloy-specific sorting; minimal sample preparation; detects trace contaminants; high-speed, in-line analysis	High equipment cost; requires calibration for heterogeneous landfill-derived materials; performance depends on consistent particle presentation	High

TABLE 3: Base-case model parameters for techno-economic assessment.

Parameter	Symbol	Base-Case Value	Rationale & Source
Project Scale	M_waste	100,000 ton	Represents a realistic, commercial-scale operation
Total OPEX & CAPEX	C_Total	USD 40 / Mt	Conservative synthesis of literature
Al Content	Y - Al	0.8 wt.% (0.008)	Mid-point of reported range (0.3–1.5%) in landfill
Separation Efficiency	η	70%	Realistic recovery rate for mechanical separation
Contamination Discount	f_cont	0.50	Reflects a 50% price reduction for low-grade, oxidized landfill scrap
Aluminum Price	P_Al	USD 2,200 / tonne	Approximate 5-year average LME price

5.3 Breakeven thresholds

To determine under what conditions the operation could break even (N≥0), we solved for the required value of each key parameter. Table 4 presents these thresholds.

Every single breakeven threshold lies far outside the realm of economic or technical feasibility. No combination of plausible parameter adjustments can render standalone aluminum recovery profitable.

5.4 Comparison with successful multi-stream models

The uneconomic outcome of single-stream recovery contrasts sharply with profitable multi-stream operations. Table 5 compares the base-case model with the Ashfill project in Maine, USA, illustrating the advantages of recovering multiple materials (Wagner & Raymond, 2015). European pilot projects, including EURELCO and the Horizon 2020 NEW-MINE initiative, further confirm that standalone recovery is rarely viable. NEW-MINE demonstrated that profitability required an Integrated Value Recovery approach combining (1) material recycling, (2) waste-to-energy of high-calorific fractions, and (3) landfill reclamation and remediation. This holistic strategy captures additional value from avoided remediation costs and reusable land, emphasizing that the feasibility of Enhanced Landfill Mining often depends on supportive regulatory frameworks and policy incentives when raw material markets alone are insufficient (Blengini et al., 2019; Hernandez Parrodi et al., 2019).

5.5 Implication: Single-stream aluminum mining is economically irrational

The TEA demonstrates that standalone aluminum recovery from landfill is profoundly uneconomic, regardless of project scale. The required aluminum grade (3.6 wt.%) is more than quadruple the realistic maximum; the required aluminum price (USD 7,143/tonne) is without historical precedent; and the required OPEX (USD 6.16/tonne) is an order of magnitude below current best practice. These findings lead to an unequivocal conclusion: any economically viable landfill mining operation must adopt a multi-stream revenue model, recovering ferrous metals, copper, plastics, and potentially realizing land value or airspace recovery. A general form of such a model is:

$$R_{total} = R_{Al} + R_{Fe} + R_{Cu} + R_{plastics} + V_{Landspace}$$

where $V_{Landspace}$ represents the avoided cost of new landfill capacity or the value of reclaimed land. This multi-stream approach is the only pathway – demonstrated by the Ashfill case – toward positive economics.

6. BARRIERS TO LANDFILL MINING

Landfill mining (LFM) has been explored as a strategy for sustainable waste management and resource recovery and has been implemented in the U.S. and other countries for land redevelopment, airspace recovery, and environmental remediation (Jain et al., 2023). However, its broader adoption remains limited by interconnected environmental, economic, societal, technological, and regulatory barriers.

TABLE 4: Breakeven thresholds for standalone aluminum recovery (N ≥ 0).

Parameter Adjusted for Breakeven	Required Threshold Value	Comment on Practical Attainability
Minimum Aluminum Price (P_Al)	USD 7,143 / tonne	Over 3x the historical average; economically implausible
Minimum Aluminum Content (Y_Al)	3.6 wt.%	Far exceeds typical landfill composition (<1.5%)
Maximum Allowable OPEX (C_total)	USD 6.16 / tonne	technologically impossible
Minimum Scrap Quality (f_cont)	1.0	Unattainable for contaminated landfill scrap

TABLE 5: Contrast with the case study's successful model.

Parameter	Case study 1 (Raw MSW)	Case Study 2 (Ashfill)
Feedstock	Raw MSW (0.8% Al)	Incinerator Ash (7% metal)
OPEX	$40/Mt	$158/Mt (but higher revenue)
Revenue	Only Al	Ferrous + Non-ferrous + Stainless + Zorba
Outcome	Net Loss ($-3.38M)	Profitable ($7.42M)
Net gain/loss per tonne	$ -33.84	$16.96

Environmental risks arise from the disturbance of stabilized waste, which can release methane and volatile organic compounds and increase leachate generation, potentially contaminating soil and groundwater, particularly in older landfills without engineered containment systems (Jain et al., 2023). Although hazardous materials typically constitute less than 1% of landfill content, accidental release remains a concern, and reuse of untreated soil-like fractions may cause secondary contamination (Burlakovs et al., 2017; Gurusamy & Thangam, 2023).

Economic feasibility remains a major constraint, as high costs associated with excavation, separation, transportation, and residual waste management often exceed revenues from recovered materials. While metals provide some value, most waste fractions lack sufficient market demand, making project viability highly dependent on policy incentives, infrastructure availability, and commodity prices (Frändegård et al., 2015; Krook & Baas, 2013; Söderholm & Ekvall, 2020; Van Passel et al., 2013). Without subsidies or favorable regulatory conditions, most projects remain economically marginal, with unrecovered residues returned to landfills, increasing operational costs (Frändegård et al., 2015; Lawal, 2024).

Societal opposition further limits implementation due to concerns about health risks, odors, noise, air pollution, and traffic, compounded by the imbalance between localized short-term impacts and delayed benefits (Einhäupl et al., 2021; Johansson et al., 2017; Johansson & Pettersson, 2022).

Regulatory frameworks in both the U.S. and EU were designed for waste containment rather than resource recovery, creating uncertainty regarding permitting, liability, and cost allocation among stakeholders (Taelman et al., 2018; Velenturf & Purnell, 2017; Wilts et al., 2016). The absence of comprehensive regulations, combined with limited municipal capacity and the lack of standardized resource assessment frameworks such as United Nations Framework Classification for Resources (UNFC), further constrains large-scale implementation.

7. THE PARADIGM SHIFT TOWARD PRE-LANDFILL INTERVENTION

Pre-landfill recovery refers to the diversion and capture of recyclable materials before disposal in municipal solid waste landfills (Cimpan et al., 2015). Unlike landfill mining, which attempts recovery after materials have been burried, pre-landfill intervention targets materials within collection systems, source separation programs, and material recovery facilities (MRFs), where quality and economic value remain largely intact (Cimpan et al., 2015). For aluminum, upstream recovery preserves metallic aluminum (Al^0), which is essential for efficient recycling. Pre-landfill recovery therefore shifts the focus from attempting to recover burried materials to preventing degradation altogether.

7.1 Comparative Advantages of Pre-Landfill Intervention

Upstream recovery maintains aluminum purity and metallic continuity, enabling efficient remelting with minimal losses. In contrast, landfill exposure leads to oxidation, contamination, and fragmentation that reduce recyclability (Hull et al., 2005; Martin et al., 2013). Pre-landfill recovery eliminates major cost drivers associated with landfill mining, including excavation, waste handling, and intensive preprocessing (Zhou et al., 2015). Materials intercepted earlier in the waste stream require significantly less energy and processing (Raabe, 2023). Early recovery reduces landfill volume, prevents long-term corrosion reactions, and maximizes displacement of energy-intensive primary aluminum production. Because secondary aluminum production requires only a fraction of the energy of primary production, upstream diversion yields substantial emissions reductions (Anna et al., 2026).

7.2 Applicability of Existing Technologies Upstream

A critical insight emerging from landfill mining research is that most technologies proposed for post-disposal recovery are not inherently unsuitable; rather, they are applied too late in the material lifecycle (Krook & Baas, 2013). Technologies such as eddy current separation, air classification, optical sorting, magnetic separation, and density-based separation are already capable of efficiently recovering aluminum when feedstocks remain relatively clean and size-consistent (Cui & Forssberg, 2003; Vermeșan et al., 2019).

When deployed upstream in MRFs or dedicated recycling facilities, these technologies operate under far more favorable conditions. Waste streams are less compacted, moisture content is lower and contamination levels are much lower. Consequently, separation efficiencies increase while energy consumption and equipment wear decrease (Cimpan et al., 2015).

Adapting landfill-mining technologies for pre-landfill recovery therefore represents a technological reallocation rather than technological innovation. The challenge is not the absence of recovery technologies but the timing and system integration of their deployment within waste management infrastructure.

7.3 Policy and Infrastructure Pathways

Achieving large-scale pre-landfill aluminum recovery requires coordinated policy and infrastructure interventions that address systemic inefficiencies in collection and sorting systems (Hariyani et al., 2025). Evidence consistently shows that container deposit programs significantly increase recovery rates for aluminum beverage cans by providing direct economic incentives for consumer participation; modernizing deposit values to reflect inflation and expanding geographic coverage could substantially reduce aluminum leakage into landfills (Buffington, 2012; Gitlitz, 2005; Wen et al., 2025). Extended Producer Responsibility (EPR) frameworks further support recovery by shifting partial responsibility for end-of-life material management to producers, thereby incentivizing product designs that facilitate recyclability while funding improved collection infrastructure. For aluminum packaging, EPR mechanisms can stabilize recycling financing and reduce dependence on volatile commodity markets (Quinn; Thakur & Kalra, 2026).

Infrastructure modernization also plays a critical role. Upgrading Material Recovery Facility (MRF) systems with advanced sorting technologies, including improved eddy current separators and optical and artificial intelligence (AI) sorting systems, can significantly enhance aluminum capture rates, particularly within single-stream recycling systems where contamination currently limits recovery efficiency (Kannan et al., 2025; Staub, 2020). Expanding recycling access in multi-family housing, public spaces, and commercial environments addresses logistical barriers that contribute to aluminum disposal, while consistent labeling standards and public education initiatives are necessary to reduce contamination and improve participation (Kostakis et al., 2025; Trushna et al., 2024).

Collectively, these pathways shift waste management from reactive disposal toward proactive resource management.

7.4 The Circular Economy Connection

Pre-landfill intervention aligns directly with circular economy principles, which prioritize waste prevention, material retention, and resource efficiency over end-of-pipe recovery strategies (Ghisellini et al., 2016). While landfill mining represents a retroactive attempt to reintroduce materials into economic circulation, it inherently reflects earlier failures in product design, collection systems, and recycling infrastructure.

Circular economy frameworks emphasize maintaining materials at their highest possible value for as long as possible (Blomsma & Tennant, 2020). Recovering aluminum before disposal preserves both material quality and embodied energy, enabling true closed-loop recycling. Unlike the circular economy goal of maintaining material integrity, landfilling aluminum may lead to the irreversible loss of metallic properties due to corrosion, effectively removing it from the recyclable material loop. From a system's perspective, landfill mining should therefore be viewed not as a primary circular strategy but as a corrective measure for historical inefficiencies. Pre-landfill recovery, by comparison, operationalizes circular economy objectives by preventing resource loss at the outset. Transitioning toward upstream intervention transforms aluminum waste management from a remediation challenge into a resource optimization strategy, strengthening domestic material security while reducing environmental burdens associated with primary production.

8. CONCLUSIONS

Recovering aluminum from landfills via landfill mining is technically, economically, and environmentally challenging. Landfilled aluminum suffers from severe contamination, oxidation, and corrosion, with up to 30% potentially unrecoverable (Lucas et al., 2019). Excavation and sorting costs often exceed the market value of the recovered material, while waste heterogeneity creates significant operational limitations. Economic and environmental assessments are inconsistent, reflecting differing assumptions and highlighting the need for standardized methodologies in the field (Frändegård et al., 2015; Krook & Baas, 2013).

Pre-landfill intervention offers a far more effective pathway. Recovering aluminum upstream preserves material quality, maximizes yields, reduces waste volumes, and avoids the resource-intensive processes associated with landfill mining. Strengthening U.S. recycling systems through expanded container deposit laws, extended producer responsibility programs, modernization of Material Recovery Facilities with advanced optical and AI-based sorting technologies, and enhanced public engagement promoting design-for-recycling and consumer education could divert an additional 815,000 tons of aluminum beverage cans annually, increasing the national used beverage can (UBC) recycling rate to approximately 85% (Institute, 2025).

Integrating aluminum recovery upstream within a circular economy framework mitigates the technical, financial, and environmental burdens of post-disposal recovery. The imperative is clear: the industry must shift from reactive landfill mining to proactive diversion to enhance resource efficiency, conserve landfill space, and strengthen domestic material security.

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DECLARATION OF THE USE OF AI

ChatGPT was employed to support language refinement, grammar correction, and clarity improvement. All intellectual contributions, data analysis, interpretations, and conclusions are the author's own responsibility. The use of AI did not involve generating or altering scientific results or research findings.

REFERENCES

Agency, U. S. E. P. (2018a). Aluminum: Material-Specific Data

Agency, U. S. E. P. (2018b). Guide to the Facts and Figures Report about Materials, Waste and Recycling

Agency, U. S. E. P. (2018c). Studies, Summary Tables, and Data Related to the Advancing Sustainable Materials Management Report

Agency, U. S. E. P. (2020). Sustainable Materials Management (SMM) - Materials and Waste Management in the United States Key Facts and Figures

Ahmad, S., Steen, J., Zajac, T., Azadi, M., & Ali, S. H. (2025). Carbon emissions in metal manufacturing productivity: A global analysis of aluminium smelting. Energy Research & Social Science, 129, 104343.

Al-Alimi, S., Yusuf, N. K., Ghaleb, A. M., Lajis, M. A., Shamsudin, S., Zhou, W., Altharan, Y. M., Abdulwahab, H. S., Saif, Y., & Didane, D. H. (2024a). Recycling aluminium for sustainable development: A review of different processing technologies in green manufacturing. Results in engineering, 102566.

Al-Alimi, S., Yusuf, N. K., Ghaleb, A. M., Lajis, M. A., Shamsudin, S., Zhou, W., Altharan, Y. M., Abdulwahab, H. S., Saif, Y., & Didane, D. H. (2024b). Recycling aluminium for sustainable development: A review of different processing technologies in green manufacturing. Results in engineering, 23, 102566.

Ali Hasanbeigi, P. D. (November 21, 2017). Aluminum Industry: 10 Emerging Technologies for Energy-efficiency and GHG Emissions Reduction. Global Efficiency Interligence. Retrieved June 19, 2025 from

Alicia Vallejo-Olivares, H. P., Mertol Gökelma, Hans J. Roven, Trond Furu, Anne Kvithyld, Gabriella Tranell. (October 7th,2020). Light Metals 2021: Compaction of aluminium foil and its effect on oxidation and recycling yield (L. Perander, Ed.). Springer Nature. https://doi.org/http://dx.doi.org/10.1007/978-3-030-65396-5_96

Anna, N., Camil, T., Constantin, I., Doina, R., & Violeta, M. L. (2026). Recycled Versus Primary Aluminum in European Automotive Industry: Trends, Challenges, and Opportunities. Recycling, 11(1), 19.
Archambo, M., & Kawatra, S. (2021). Red mud: fundamentals and new avenues for utilization. Mineral processing and extractive metallurgy review, 42(7), 427–450.
Ashkenazi, D. (2019). How aluminum changed the world: A metallurgical revolution through technological and cultural perspectives. Technological Forecasting and Social Change, 143, 101–113.
Association, T. A. (2024, 12/05/2024). Amid Recycling Rate Decline, Aluminum Beverage Can Remains Most Recycled Drinks Package. https://www.aluminum.org/news/amid-recycling-rate-decline-aluminum-beverage-can-remains-most-recycled-drinks-package
Blengini, G., Mathieux, F., Mancini, L., Nyberg, M., & Viegas, H. (2019). Recovery of critical and other raw materials from mining waste and landfills. State of Play on Existing Practices, 1–125.
Blomsma, F., & Tennant, M. (2020). Circular economy: Preserving materials or products? Introducing the Resource States framework. Resources, Conservation and Recycling, 156, 104698.
Brooks, L., Gaustad, G., Gesing, A., Mortvedt, T., & Freire, F. (2019). Ferrous and non-ferrous recycling: challenges and potential technology solutions. Waste management, 85, 519–528.
Buffington, J. (2012). The beverage can in the United States: achieving a 100% recycled aluminum can through supply chain innovation. Jom, 64(8), 923–932.
Burlakovs, J., Kriipsalu, M., Klavins, M., Bhatnagar, A., Vincevica-Gaile, Z., Stenis, J., Jani, Y., Mykhaylenko, V., Denafas, G., & Turkadze, T. (2017). Paradigms on landfill mining: From dump site scavenging to ecosystem services revitalization. Resources, Conservation and Recycling, 123, 73–84.
Cadavid, J. Q. (2025). The UBC dilemma: Why the US is falling behind in recycling aluminium beverage can [Research].
Calder, G. V., & Stark, T. D. (2010). Aluminum reactions and problems in municipal solid waste landfills. Practice Periodical of Hazardous, Toxic, and Radioactive Waste Management, 14(4), 258–265.
Capuzzi, S., & Timelli, G. (2018). Preparation and melting of scrap in aluminum recycling: A review. Metals, 8(4), 249.
Center-USGS, N. M. I. (2022). Aluminum Statistics and Information. https://www.usgs.gov/centers/national-minerals-information-center/aluminum-statistics-and-information
Chen, K., Bao, S., Li, J., Li, J., Shi, M., Shi, X., & Ma, J. (2025). Research Progress on Inert Anode Materials for Aluminum Electrolysis. TMS Annual Meeting & Exhibition,
Chen, W.-Q., & Graedel, T. E. (2012). Dynamic analysis of aluminum stocks and flows in the United States: 1900–2009. Ecological Economics, 81, 92–102.
Choate, W. T., & Green, J. A. (2003). US energy requirements for aluminum production: historical perspective, theoretical limits and new opportunities. US Department of Energy, Energy Efficiency and Renewable Energy.
Cimpan, C., Maul, A., Jansen, M., Pretz, T., & Wenzel, H. (2015). Central sorting and recovery of MSW recyclable materials: A review of technological state-of-the-art, cases, practice and implications for materials recycling. Journal of Environmental Management, 156, 181–199.
Cremers, D. A., & Radziemski, L. J. (2006). History and fundamentals of LIBS. Laser Induced Breakdown Spectroscopy: Fundamentals and Applications, 1, 1–39.
Cui, J., & Forssberg, E. (2003). Mechanical recycling of waste electric and electronic equipment: a review. Journal of hazardous materials, 99(3), 243–263.
Dascalescu, L., Zeghloul, T., & Iuga, A. (2016). Electrostatic separation of metals and plastics from waste electrical and electronic equipment. In WEEE recycling (pp. 75–106). Elsevier.
David, E., & Kopac, J. (2013). Aluminum recovery as a product with high added value using aluminum hazardous waste. Journal of hazardous materials, 261, 316–324.
directory, S. (2025). Why is aluminum recycling so effective?
Einhäupl, P., Van Acker, K., Peremans, H., & Van Passel, S. (2021). The conceptualization of societal impacts of landfill mining–A system dynamics approach. Journal of Cleaner Production, 296, 126351.
Energy, U. S. D. o. (2023). Critical Minerals & Materials Program
EPA, U. S. (2023). Facts and Figures about Materials, Waste and Recycling
Ferronato, N., & Torretta, V. (2019). Waste mismanagement in developing countries: A review of global issues. International journal of environmental research and public health, 16(6), 1060.
Fortier, S. M., Nassar, N. T., Lederer, G. W., Brainard, J., Gambogi, J., & McCullough, E. A. (2018). Draft critical mineral list—Summary of methodology and background information—US Geological Survey technical input document in response to Secretarial Order No. 3359 (2331-1258).
Frändegård, P., Krook, J., & Svensson, N. (2015). Integrating remediation and resource recovery: On the economic conditions of landfill mining. Waste Management, 42, 137–147.
Gaines, L. L. (1981). Energy and materials use in the production and recycling of consumer-goods packaging.
Gesing, A., & Wolanski, R. (2001). Recycling light metals from end-of-life vehicle. Jom, 53(11), 21–23.
Ghisellini, P., Cialani, C., & Ulgiati, S. (2016). A review on circular economy: the expected transition to a balanced interplay of environmental and economic systems. Journal of Cleaner Production, 114, 11–32.
Gitlitz, J. (2005). Drink and deposit. Waste Management World.
Gitlitz, J. S. (2002). Trashed Cans: The Global Environmental Impacts of Aluminum Can Wasting in America. Container Recycling Institute.
Goh, K. C., Kurniawan, T. A., Othman, M. H. D., Jumaniyozov, K., Aziz, F., Anouzla, A., Ali, I., Khan, M. I., Alsultan, G. A., & Zhang, D. (2025). Promoting landfill mining for sustainable resource recovery, circular economy, and climate mitigation: a review. Frontiers in Environmental Science, 13, 1643195.
Green, J. A. (2007). Aluminum recycling and processing for energy conservation and sustainability. ASM International.
Guide, A. (2025). An aluminium beverage can. Aluminum Guide. Retrieved 08/05/2025 from
Gundupalli, S. P., Hait, S., & Thakur, A. (2017). A review on automated sorting of source-separated municipal solid waste for recycling. Waste Management, 60, 56–74.
Gurell, J., Bengtson, A., Falkenström, M., & Hansson, B. (2012). Laser induced breakdown spectroscopy for fast elemental analysis and sorting of metallic scrap pieces using certified reference materials. Spectrochimica Acta Part B: Atomic Spectroscopy, 74, 46–50.
Gurusamy, S., & Thangam, R. S. P. (2023). Potential health risk assessment of contaminants in soil-like material recovered from landfill mining. Environmental Monitoring and Assessment, 195(2), 330.
Hariyani, D., Hariyani, P., Mishra, S., & Sharma, M. K. (2025). A literature review on waste management treatment and control techniques. Sustainable Futures, 9, 100728.
Hernandez Parrodi, J. C., Lucas, H., Gigantino, M., Sauve, G., Esguerra, J. L., Einhaupl, P., Vollprecht, D., Pomberger, R., Friedrich, B., & Van Acker, K. (2019). Integration of resource recovery into current waste management through (enhanced) landfill mining. Detritus, 8, 141–156.
Hogland, W., Hogland, M., & Marques, M. (2011). Material recovery, energy utilization and economics of landfill mining in the EU (directive) perspective. Proceedings of "Sustainable Solid Waste Management In Eastern Europe-Prospects For The Future.
Huang, Z., Zhu, J., Wu, X., Qiu, R., Xu, Z., & Ruan, J. (2021). Eddy current separation can be used in separation of non-ferrous particles from crushed waste printed circuit boards. Journal of Cleaner Production, 312, 127755.
Hull, R. M., Krogmann, U., & Strom, P. F. (2005). Composition and characteristics of excavated materials from a New Jersey landfill. Journal of environmental engineering, 131(3), 478–490.
Igwegbe, C. A., López-Maldonado, E. A., Landázuri, A. C., Ovuoraye, P. E., Ogbu, A. I., Vela-García, N., & Białowiec, A. (2024). Sustainable municipal landfill leachate management: Current practices, challenges, and future directions. Desalination and Water Treatment, 100709.
Institute, C. R. (2025). Improving Efficiency and Sustainability in Aluminum Beverage Can Recycling, January 2025
Institute, T. A. A. a. C. M. (2024). The Aluminum Can Advantage:
Sustainability Key Performance
Indicators
Jain, M., Kumar, A., & Kumar, A. (2023). Landfill mining: A review on material recovery and its utilization challenges. Process Safety and Environmental Protection, 169, 948–958.
Johansson, N., Krook, J., & Eklund, M. (2017). The institutional capacity for a resource transition—A critical review of Swedish governmental commissions on landfill mining. Environmental Science & Policy, 70, 46–53.

Johansson, O., & Pettersson, M. (2022). Environmental law issues in connection with landfill mining. Detritus, 18, 77–84.
Jones, P. T., Geysen, D., Tielemans, Y., Van Passel, S., Pontikes, Y., Blanpain, B., Quaghebeur, M., & Hoekstra, N. (2013). Enhanced Landfill Mining in view of multiple resource recovery: a critical review. Journal of Cleaner Production, 55, 45–55.
Kannan, E., Yuvarani, B., Sanjaykumar, S., Tamilvanan, T., Vasanth, J., & Manikandan, S. (2025). A High Efficient and Cost Effective Waste Sorting Solution using Advanced Sensors and AI. 2025 5th International Conference on Trends in Material Science and Inventive Materials (ICTMIM),
Kelly, S., & Apelian, D. (2016). Automotive aluminum recycling at end of life: a grave-to-gate analysis. Worcester Polytechnic Institute.
Kirchherr, J., Reike, D., & Hekkert, M. (2017). Conceptualizing the circular economy: An analysis of 114 definitions. Resources, Conservation and Recycling, 127, 221–232.
Kostakis, I., Halkos, G., & Sardianou, E. (2025). Sustainability in municipal waste recycling: An empirical investigation of economic and institutional determinants. Global Economics Research, 100018.
Krook, J., & Baas, L. (2013). Getting serious about mining the technosphere: a review of recent landfill mining and urban mining research. Journal of Cleaner Production, 55, 1–9.
Krook, J., Svensson, N., & Eklund, M. (2012). Landfill mining: A critical review of two decades of research. Waste Management, 32(3), 513–520.
Kulas, D., Amponsah-Berko, D., Eisele, T., & Handler, R. Landfill Mining for Aluminum and Copper Recovery: An Economic and Environmental Feasibility Study. 2025 AIChE Annual Meeting,
Lawal, S. O. (2024). The economics of recycling: A review compiled with tax and subsidiary, implication for government, decision-makers, enterprises, community, and analysis cost/benefit and market. ASEAN Journal of Economic and Economic Education, 3(2), 165–188.
Lee, H., Coulon, F., & Wagland, S. (2023). The influence of humic acid on metal (loid) s leaching in landfill leachate for enhancing landfill mining. Science of the Total Environment, 896, 165250.
Li, Y., Yue, Q., He, J., Zhao, F., & Wang, H. (2020). When will the arrival of China's secondary aluminum era? Resources Policy, 65, 101573.
Lindstrom, S. (2024). Transitioning to a Circular Economy: Evaluating National Waste Management Policies. Future-Environmental Policy and Sustainable Development, 1(2).
Lucas, H. I., López, C. G., Parrodi, J. C. H., Vollprecht, D., Raulf, K., Pomberger, R., Pretz, T., & Friedrich, B. (2019). Quality assessment of nonferrous metals recovered by means of landfill mining: A case study in Belgium. Detritus, 8(December), 79–90.
Martin, J. W., Stark, T. D., Thalhamer, T., Gerbasi-Graf, G. T., & Gortner, R. E. (2013). Detection of aluminum waste reactions and waste fires. Journal of Hazardous, Toxic, and Radioactive Waste, 17(3), 164–174.
Miteva, A., & Hodjaoglu, G. Applications of Recycled Aluminum in the Modern Food Industry. Resources, Conservation and Recycling, 114, 18–31.
Morawski, C. (2010). Single stream uncovered. Resource Recycling, 21–25.
Munshi, B. ELYSIS™ Inert-Anode Aluminium Smelting: Electrochemistry, Scale-Up, and the India Adoption Pathway.
Mutz, S., Chatziavgoustis, T., & Pretz, T. (2005). Sorting of aluminum and its alloys within the European automobile recycling. Proceedings of 5th International Exhibition & Conference on Environmental Technology, Athens Greece, 3-6 February, C81,
Padamata, S. K., Yasinskiy, A., & Polyakov, P. (2021). A review of secondary aluminum production and its byproducts. Jom, 73(9), 2603–2614.
Parrodi, J. C. H., Höllen, D., & Pomberger, R. (2018). Potential and main technological challenges for material and energy recovery from fine fractions of landfill mining: A critical review. Detritus, 3(september 2018), 19–29.
Pecorini, I., & Iannelli, R. (2020). Characterization of excavated waste of different ages in view of multiple resource recovery in landfill mining. Sustainability, 12(5), 1780.
Pereira, A. C. (2025). From can to sheet: A critical review of aluminum can recycling processes, technologies, and sustainability dimensions (2020–2025). Revista FT, 10(2), 1026–1019.
Pourbaix, M. (1974). Atlas of Electrochemical Equilibria in Aqueoes Solution (N. A. o. C. Engineers, Ed. 2 ed.). National Association of Corrosion Engineers.
Quaghebeur, M., Laenen, B., Geysen, D., Nielsen, P., Pontikes, Y., Van Gerven, T., & Spooren, J. (2013). Characterization of landfilled materials: screening of the enhanced landfill mining potential. Journal of Cleaner Production, 55, 72–83.
Quinn, E. Circular economy and the potential for extended producer responsibility policies.
Raabe, D. (2023). The materials science behind sustainable metals and alloys. Chemical reviews, 123(5), 2436–2608.
Rabinovich, D. (2013). The allure of aluminium. Nature Chemistry, 5(1), 76–76.
Ratcliffe, A., Prent, O., & van Vossen, W. (2012). Feasibility of material recovery from landfills (MFL) in the European Union. The ISWA World Waste Congress,
Recovery, U. S. E. P. A. O. o. R. C. a. (2014). Municipal Solid Waste Generation,Recycling, and Disposal in the United States: Facts and Figures. A methodology Document
Saevarsdottir, G., & Kvande, H. (2025). Development of the Carbon Footprint of Primary Aluminum Production. TMS Annual Meeting & Exhibition,
Sariatli, F. (2017). Linear economy versus circular economy: a comparative and analyzer study for optimization of economy for sustainability. Visegrad Journal on Bioeconomy and Sustainable Development, 6(1), 31–34.
Shapiro, M., & Galperin, V. (2005). Air classification of solid particles: a review. Chemical Engineering and Processing: Process Intensification, 44(2), 279–285.
Shen, A., & Zhang, J. (2024). Technologies for CO2 emission reduction and low-carbon development in primary aluminum industry in China: A review. Renewable and Sustainable Energy Reviews, 189, 113965.
Silveira, A., Cella, M., Tanabe, E., & Bertuol, D. (2018). Application of tribo-electrostatic separation in the recycling of plastic wastes. Process Safety and Environmental Protection, 114, 219–228.
Smith, Y. R., Nagel, J. R., & Rajamani, R. K. (2019). Eddy current separation for recovery of non-ferrous metallic particles: A comprehensive review. Minerals Engineering, 133, 149–159.
Söderholm, P., & Ekvall, T. (2020). Metal markets and recycling policies: impacts and challenges. Mineral Economics, 33(1), 257–272.
Stark, T. D., Martin, J. W., Gerbasi, G. T., Thalhamer, T., & Gortner, R. E. (2012). Aluminum waste reaction indicators in a municipal solid waste landfill. Journal of geotechnical and geoenvironmental engineering, 138(3), 252–261.
Staub, C. (2020). Equipment upgrades drive greater UBC recovery
Tabereaux, A. T., & Peterson, R. D. (2024). Aluminum production. In Treatise on process metallurgy (pp. 625–676). Elsevier.
Taelman, S. E., Tonini, D., Wandl, A., & Dewulf, J. (2018). A holistic sustainability framework for waste management in European cities: Concept development. Sustainability, 10(7), 2184.
Thakur, V., & Kalra, S. (2026). Approaches to Extended Producer Responsibility: Fostering Sustainable Consumption and Circular Economy. In Green Horizons: Advancing Climate Solutions Through Green Finance, Circular Economy, and ESG (pp. 291–316). Springer.
Trushna, T., Krishnan, K., Soni, R., Singh, S., Kalyanasundaram, M., Annerstedt, K. S., Pathak, A., Purohit, M., Lundbog, C. S., & Sabde, Y. (2024). Interventions to promote household waste segregation: A systematic review. Heliyon, 10(2).
Van Passel, S., Dubois, M., Eyckmans, J., De Gheldere, S., Ang, F., Jones, P. T., & Van Acker, K. (2013). The economics of enhanced landfill mining: private and societal performance drivers. Journal of Cleaner Production, 55, 92–102.
van Vossen, W., & Prent, O. (2013). Feasibility study sustainable material and energy recovery from landfills in Europe. ISTANBUL3WCONGRESS 2013, 347.
Veit, H. M., Diehl, T. R., Salami, A. P., Rodrigues, J. d. S., Bernardes, A. M., & Tenório, J. A. S. (2005). Utilization of magnetic and electrostatic separation in the recycling of printed circuit boards scrap. Waste management, 25(1), 67–74.
Velenturf, A. P., & Purnell, P. (2017). Resource recovery from waste: Restoring the balance between resource scarcity and waste overload. Sustainability, 9(9), 1603.
Vermeșan, H., Tiuc, A.-E., & Purcar, M. (2019). Advanced recovery techniques for waste materials from IT and telecommunication equipment printed circuit boards. Sustainability, 12(1), 74.
Vollprecht, D., Machiels, L., & Jones, P. T. (2021). The EU training network for resource recovery through enhanced landfill mining—A review. Processes, 9(2), 394.

Wagner, T. P., & Raymond, T. (2015). Landfill mining: Case study of a successful metals recovery project. Waste management, 45, 448–457.
Wang, J., Shi, A., Yue, D., Wang, C., Liu, Y., & Shi, Y. (2024). Resources Recovery from Landfill Leachate: Current Status, Challenges, and Prospects. A Review of Landfill Leachate: Characterization Leachate Environment Impacts and Sustainable Treatment Methods, 215–238.
Wen, J., Yu, W., Yuan, H., Chen, H., Xiong, T., & Yang, F. (2024). A new strategy for extracting aluminum and iron from red mud via vacuum thermal reduction, alkali-leaching and magnetic separation. Journal of Cleaner Production, 485, 144409.
Wen, P., Li, X., Shi, G., Wang, Z., & Ma, Z. (2025). Age and hydrogeological conditions of landfill site control groundwater pollution in landfill site. Waste management, 208, 115171.
Wilts, H., Von Gries, N., & Bahn-Walkowiak, B. (2016). From waste management to resource efficiency—The need for policy mixes. Sustainability, 8(7), 622.
Xia, Y., He, P., Shao, L., & Zhang, H. (2017). Metal distribution characteristic of MSWI bottom ash in view of metal recovery. Journal of Environmental Sciences, 52, 178–189.
Zhou, C., Gong, Z., Hu, J., Cao, A., & Liang, H. (2015). A cost-benefit analysis of landfill mining and material recycling in China. Waste Management, 35, 191–198.

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TEMPORARY MATERIAL HUBS TO ENHANCE CIRCULAR ECONOMY: A CONCEPTUAL FRAMEWORK

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ABSTRACT

This study introduces the integrative concept of Temporary Material Hubs (TMHs) as a newly adapted approach to enhance long-term improvement in circularity by storing prospectively valuable waste, under current conditions not feasible or possible to recover, for future recycling capabilities. TMHs aim to optimize resource recovery by prolonging the lifespan of materials in anthropogenic cycles and avoiding premature disposal. Unlike landfilling with subsequent landfill mining (disposal and later excavation), TMHs proactively store such materials in controlled "hubs" to preserve value and enable future high-quality recovery. The conceptual framework is complemented by the known final sink concept to maintain clean material cycles. Given the lack of a clear definition of recyclability, this paper further proposes recycling pillars as guiding principles in the context of TMHs: environmental and health protection, availability of adequate recycling technologies, and economic feasibility including the availability of markets. Exemplary candidate materials for TMHs currently envisage, e.g., end-of-life wind blades and incineration residues. A SWOT analysis was used to discuss the strengths of TMHs in promoting resource optimization through postponed recycling, while identifying weaknesses such as uncertain costs and current lack of accurate technical implementation concepts. Opportunities lie in supporting European circularity goals and reducing primary material extraction, whereas threats include future regulatory uncertainties and inaccurate estimations of future waste recyclability. This study prepares the ground for future research and risk-assessment on technical, economical, and societal factors necessary for implementing TMHs on an industrial scale to ensure better functioning of the circular economy and a sustainable future.

1. INTRODUCTION

The waste sector is adapting to societal and environmental needs while navigating through restricting policies and remaining tightly connected to the economy. According to the European waste hierarchy, the highest priority is waste prevention and minimization (European Parliament (EP) and the Council, 2024b). However, modern society inevitably produces waste, and globally, the current waste management system still relies mainly on disposal (landfilling) (Kaza et al., 2018). New EU policies and strategies such as the European Green Deal strive for sustainable future based on the concept of circular economy, which conserves resources and keeps them in anthropogenetic cycles (European Council, 2024). Anthropogenic resources and waste are emerging as future sources of materials (Ghisellini et al., 2022).

The overall aim to foster a circular economy is compromised by the decrease in the share of secondary materials entering the global economy, from 9.1% in 2018 to 7.2% in 2023 (Fraser et al., 2024). The European circular material use rate was 11.8% in 2023 (European Environment Agency (EEA), 2025). In addition, resource consumption has increased significantly; society consumed a similar volume of resources from 2018 to 2024 as during the entire 20th century (582 versus 740 billion tonnes, respectively (Fraser et al., 2024)).

To counteract this trend, the European Commission (EC) sets clear targets for reuse and recycling of waste, e.g., 70% for construction and demolition waste by 2020 and a 65% target for municipal solid waste by 2035 (EP and the Council, 2024b). To further minimize environmental impact, the EU Landfill Directive restricts landfilling of recyclable waste from 2030 if suitable for material and energy recovery and caps landfilling of municipal solid waste at 10% by 2035 (EP and the Council, 2024a). Despite these efforts, 22.6% of municipal solid waste was

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still landfilled in the EU in 2019 as the primary treatment step (EC, 2020c).

Although recycling (followed by recovery) is at the bottom of the "10R" hierarchy (Vermeulen et al., 2019), it remains a cornerstone of a circular economy, keeping materials in anthropogenic cycles and maximizing their value. However, suitable markets for secondary materials are often not available. Some secondary raw material (SRM) markets, e.g., aluminium, glass and paper, are well-functioning with significant market shares of recyclates compared to raw materials (EEA, 2022, 2023). For instance, aluminium is highly circular and recyclable, maintaining its quality through repeated processing. Its primary production is very energy-intensive, but recycling saves up to 95% of energy and CO_2 emissions (European Aluminium, 2017, 2020).

In contrast, SRM markets for biowaste, construction and demolition waste, plastics, textiles and wood are generally less functional. The EEA attributes this to smaller market size, weaker demand, and inconsistent material specifications, hindering industrial applications (EEA, 2023). For instance, only a small percentage of construction and demolition waste is economically valuable, leading to low-quality recycling and a global recovery rate below one-third of the total amount of this waste (EEA, 2022; Ghisellini et al., 2022). While technology is available, market challenges, rather than technical limitations, appear to hinder recycling rates. The market is predominantly local (EEA, 2022).

To ensure sustainability, product designers must consider ecodesign rules embedded in the Ecodesign for Sustainable Products Regulation (EP and the Council, 2024c). Meanwhile, the waste management sector must handle already produced waste and products not yet targeted for recycling. Some materials are currently not feasibly recyclable or lack SRM markets but could become recyclable in future. However, the current waste management system prioritizes immediate processing, often through energy recovery or disposal, for materials/waste lacking immediate value, due to legal or financial issues. Storing them in specifically designed temporary deposits for postponed recycling could enhance circularity and material recovery. Already in the 1990s, ideas of prolonged storage for recyclable materials were explored. For example, a patent proposed temporarily storing plastic waste in bales within landfills (Herhof Umwelttechnik GmbH, 1992). This method aimed to reduce fire risks during storage by mixing plastic waste with low calorific value additives, such as processed construction waste or other inert materials, and bundling it into bales. The goal was to make the stored waste reusable in the future, expecting advancements in plastic recycling technology within 5 to 10 years (Herhof Umwelttechnik GmbH, 1992). However, this concept has not yet become widely established. Current concepts of temporary storage of waste are usually short-term and for specific waste streams (e.g., e-waste, hazardous waste, waste from disasters, demolition waste from military actions), which require proper disposal or recycling at a postponed date.

Conversely, legislative efforts to reduce disposal and energy recovery in favour of maximized material recovery may increase contaminated materials in anthropogenic cycles. Contaminated waste poses risks to the environment, human health, clean cycles, and high-quality secondary materials, requiring safe disposal (Kral et al., 2013). The EEA emphasizes the importance of sustaining clean material cycles for a functioning circular economy, both from a safety and an economic perspective (EEA, 2017). The Regulation on the Registration, Evaluation, Authorization and Restriction of Chemicals (REACH) promotes "safe-by-design" chemicals, substituting hazardous chemicals by safer ones. However, hazardous chemicals and so called persistent pollutants can persist in older products and recovered materials (EC, 2020b), especially within open-loop recycling systems (Pivnenko, 2016). Pivnenko (2016) proposed a hierarchy for contaminant mitigation, recommending incineration or adequate final sink (controlled landfill) when contaminant removal technologies or organizational approaches are unavailable.

This study proposes a conceptual framework to enhance circularity in the mid- to long term by introducing Temporary Materials Hubs (TMHs) for controlled, prolonged interim storage of materials/waste, targeting for the postponement of recycling until technological advancements or favourable economic conditions make it feasible and environmentally safe. By formalizing TMHs as a distinct strategy, introducing "recycling pillars" as a novel decision-making framework for evaluating recyclability, and integrating final sinks to maintain clean material cycles, this approach addresses a gap in current waste management strategies and meaningfully advances circular economy efforts.

Currently, the considerations focus on the EU, proposing a conceptual framework based on the region's waste management priorities and EU-waste hierarchy; however, with adaptations to meet legal, technical and economic conditions of other regions, the framework could reach global applicability.

2. MATERIALS AND METHODS

The concept is grounded in an integrative literature review, a methodological approach designed to systematically generate new concepts or ideas from existing literature (Torraco, 2016). Findings were used to create a conceptual framework and a renewed organizational approach in waste management, differentiated from existing interim storage strategies.

The integrative literature research was crucial for developing TMHs, as the explorative nature of this research required a structured yet flexible approach, studying mostly mature topics. During the literature research, we identified six key themes, which are visualized in a concept map in Figure 1, along with associated key words demonstrating the systematic process behind the analysis. Storage of radioactive waste was excluded, ensuring the focus on relevant studies. Five materials/types of waste with limited recyclability were chosen to highlight the need for this concept. Two purposes are explored in detail, and a SWOT analysis assesses the theoretical concept's performance. The TMHs concept is complemented by final sinks to ensure safe disposal of non-recyclable (e.g., asbestos, wooden railway sleepers), contaminated (e.g., textiles containing perfluoroalkyl and polyfluoroalkyl substances (PFAS)) or degraded materials in order to sustain clean material cycles and protect the environment. Detailed considerations of the final sink concept are excluded, as they are covered by existing publications (Cossu, 2016; Kral et al., 2013, 2019; Stanisavljevic & Brunner, 2021).

European Directives and national legislations were studied to examine the current legal state of waste storage and circularity, while EC documents provided data on specific waste streams. In total, 134 references are cited in our study, including peer-reviewed studies as a majority, followed by EU policy documents and research reports.

Discussions with waste management experts and scientists, mainly from Austria and Germany (e.g., concept presentation and discussion at young scientist congress hosted by DGAW), and peer groups (e.g., academic mentors, faculty members and fellows of the interdisciplinary BOKU-Doctoral School "Transitions to Sustainability" (T2S)), served as a valuable complement and provided additional input also from other disciplines. Experts individually evaluated the general concept of TMHs, no structured interviews were conducted at this stage.

3. CURRENT RECYCLING LIMITATIONS AND CHALLENGES

Recyclability determines whether materials/waste remain within the anthropogenic cycles. The Waste Framework Directive very generally describes recycling as "any recovery operation by which waste materials are reprocessed into products, materials or substances whether for the original or other purposes" (EP and the Council, 2024b), providing no criteria to assess "recyclability". In general, a single definition of recyclability is missing on the EU level. Currently only the Proposal for a Regulation on packaging and packaging waste briefly addresses this issue (EC, 2022). Similarly, the EU Landfill Directive sets restrictions for landfilling waste suitable for recycling or other material or energy recovery without defining such waste properties (EP and the Council, 2024a). Geist & Balle (2025) currently also emphasized the critical need for a conceptual basis to improve recyclability within the framework of the circular economy.

Sustainable and circular practices can be encouraged through incentives, while penalties can deter undesirable behaviours; however, enforcement of such legislation can be challenging (Fraser et al., 2024). An example of a regulatory incentive is the minimum content requirement for recycled material in packaging (EP, 2024a). The Single-Use Plastics Directive mandates that by 2025, beverage PET bottles must contain at least 25% recycled plastic, based on an average for all PET bottles sold in a Member State (EP and the Council, 2019). This legislation is expected to drive up the price of recycled PET, potentially surpassing that of virgin PET. This creates the need for high-quality recycled plastics, and the market for recycled and virgin material could be separated (EEA, 2022; Larrain et al., 2021).

Environmental and health protection is pivotal and should be the primary determinant of waste recyclability. Adequate recycling technologies often lag behind the development of new materials. For instance, while Bakelite, the first synthetic mass-produced plastic, was synthetized in 1907 and polyethylene in 1935 (Andrady & Neal, 2009; Plastics Europe, 2021), plastic recycling only began in the 1970s (Hopewell et al., 2009), and even after more than 50 years, it is still not established comprehensively. The widespread application of recycling technologies is tightly connected to economic feasibility. Even in the absence of contamination and with available recycling technologies, economic factors can hinder recycling. The main

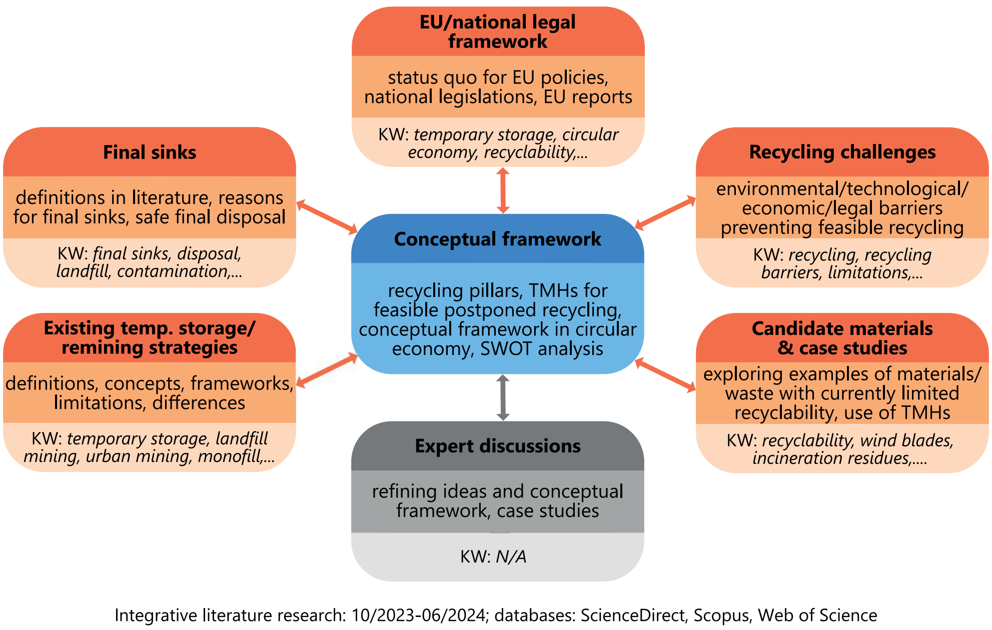

FIGURE 1: A concept map visualizing the six key themes and the methodological approach of the integrative literature review and the development of conceptual framework.

TABLE 1: Exemplary limitations and challenges affecting recycling efforts divided into four categories, then in alphabetical order.

Category	Limitations and challenges
Environmental issues	• Increasing number of substances identified as harmful (European Chemicals Agency, 2024) • Waste contamination by hazardous substances, despite labelled as "absolute non-hazardous" (European Environmental Bureau, 2017)
Technical issues	• Compromised quality of recycling outputs (EEA, 2022; European Environmental Bureau, 2017; Johansson et al., 2020; Kusenberg et al., 2022; Larrain et al., 2021) • Lack of capacity of recycling facilities (EP, 2024b; Xie et al., 2024) • Lack of end-of-life considerations from product designers (missing "design for recycling") (Sommer et al., 2020) • Potential contamination of waste streams and presence of hazardous substances – affecting e.g., food contact product safety (EC, 2024; EEA, 2022) • Quality of collected waste (EEA, 2022)
Economic issues	• Competition with energy recovery (EEA, 2022) • Difficult assessment of potential environmental benefits of recycling in monetary terms (Larrain et al., 2021) • Higher value of virgin material compared to recycled feedstock (EEA, 2022; Hestin et al., 2015) • Lack of economic incentives to prioritize recyclability in product design (EEA, 2022) • Prices of recycled plastic products are coupled to oil prices (Larrain et al., 2021) • Volatility of primary raw market and consequently impaired stability of SRM market development (EEA, 2022)
Legislative issues	• Insufficient end-of-waste legislation and lack of EU-wide quality standards (EEA, 2022, 2023) • Lack of legislation regarding the provision of information about the presence and nature of hazardous substances in "end-of-life-products" (European Environmental Bureau, 2017) • Lack of regulatory incentives to prioritize recyclability in product design (EEA, 2022) • Lack of taxes to strengthen competitiveness of recycled materials (EEA, 2023) • Limitations in waste classification (European Environmental Bureau, 2017)

limitations and challenges influencing recycling efforts are summarized in Table 1.

Given the current lack of a universal definition of recyclability at the EU level and as guiding principles in the context of the proposed conceptual framework, this study defines the key factors determining recyclability to be: (1) contribution to environmental and health protection, (2) availability of adequate recycling technologies and capacities, and (3) economic feasibility, including the existence of markets. The combination of all these pillars makes materials/waste theoretically recyclable (Figure 2). Legislation plays a crucial role in supporting or hindering these pillars and consequently recycling efforts by setting targets and restrictions.

Figure 2 shows, for example, that if a SRM market is unavailable or recycling is generally not economically feasible, waste is prone to be disposed of in landfills or subjected to thermal treatment (upper left grey triangle in Figure 2). This applies to, e.g., some types of plastic waste (Lim et al., 2023). Similarly, some materials/waste lack adequate recycling technologies (upper right grey triangle in Figure 2), e.g., the recycling outputs do not reach sufficient quality – such as in the case of composites (EC, 2023c). The lack of feasible recycling options can be addressed by TMHs (represented by the light green triangles in Figure 2). Vice versa, if recycling technologies are available and SRM markets exist but the recycling outputs do not comply with existing environmental standards, final sinks should be employed to keep the material cycles clean.

To improve the broader applicability of the concept of recycling pillars, it is necessary on the mid-term to define specific quantitative indicators. Such indicators might include, for example, the technology readiness level (to assess the availability of adequate recycling technologies), carbon footprint reductions or toxicity risk score (to evaluate environmental and health impacts) or a range of mar-

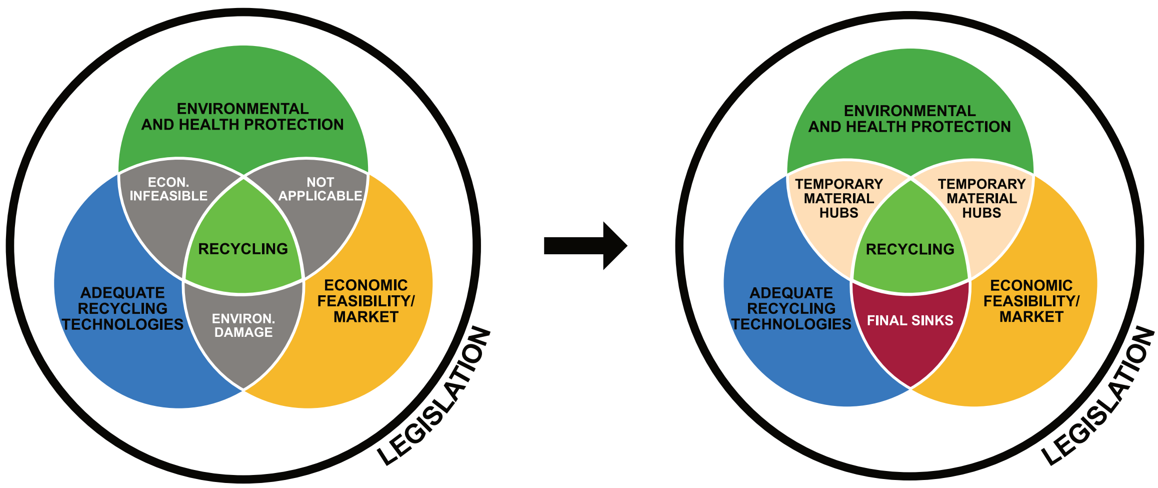

FIGURE 2: Visualisation of "recycling pillars" and gaps (grey areas) as well as Temporary Material Hubs and final sinks as a proposal to fill in these gaps in an improved circular economy.

ket indicators such as break-even virgin material price (to determine economic feasibility). However, to develop new, adapt or apply existing indicators is quite use-case specific and must be done individually for the considered material/waste. Van Nielen et al. (2022) developed a framework assessing recyclability of minor metals and defined a set of specifically related factors/indicators for this use case. The following chapter explains the concept of TMHs complemented by final sinks in a bigger detail.

4. THE CONCEPT OF TEMPORARY MATERIAL HUBS

4.1 Conceptual framework

The scheme of a circular economy desired by the EU (Figure 3 a) is a semi-closed loop where most materials circulate, some raw materials enter the cycle and a portion is disposed of as residual waste (EP, 2023). This model effectively utilizes currently valuable waste but directs materials with potential future value (examples listed in 4.2), to disposal or energy recovery, which is problematic as primary resources depletion accelerates (Jensen et al., 2012).

A new enhanced material recovery loop (Figure 3 b) replaces residual waste proposed for disposal with TMHs, designed to temporarily store materials/waste unsuitable for immediate recycling within the present waste management infrastructure and market situation. These materials are intended for targeted re-mining followed by further processing (recycling), meaning remining is not just an option, it is the focused obligation. Diverting materials/waste from disposal (or thermal treatment) to storage for future recycling is logical given scarcity of natural resources and advancing recycling technologies. This concept demands storage periods exceeding current legislation, such as the 3-year limit in EU (EP and the Council, 2024a).

TMHs support the ambitious goals outlined in the Waste Framework Directive and the Landfill Directive to increase the amount of waste recycled and reduce the amount of (municipal) waste sent to landfills (EP and the Council, 2024a, 2024b). They align with the new Circular Economy Action Plan by aiming to lower the demand for extraction of primary resources and increase the circular material use (EC, 2020a).

TMHs can be categorized into two types based on the properties of the stored material/waste and the processes and operations taking place during the storage period: static and dynamic. In static TMHs, materials/waste are stored in an intact state with optional pre-treatment and remain unchanged until remining and recycling. These TMHs are adequate for inert materials/waste. On the other hand, materials/waste in dynamic TMHs undergo desirable and engineered transformations before re-entering recycling options and material cycles and might require additional monitoring and regulation.

A comprehensive database and mapping of stored materials/waste ensures targeted excavation/remining with the intention of postponed recycling, making TMHs more cost-effective than landfill mining due to upfront planning allowing feasible and environmentally safe extraction of stored materials/waste. Examples of candidate materials/waste for both types of TMHs are listed in Table 2, with detailed purposes in subsections 4.2 and 4.3.

To complement TMHs and the enhanced material recovery loop, waste unsuitable for both immediate and postponed recycling is directed to a safe disposal – final sinks (Figure 3 b). While a harmonized definition of final sinks is lacking, Kral et al. (2013) defined them as "a sink that either destroys a substance completely, or that holds a substance for a very long time period". Stanisavljevic & Brunner (2021) perceive (final) sinks as a crucial part of the circular economy as they accommodate hazardous materials/waste without economic value. Final sinks serve to safely remove materials or waste unsuitable for recycling from the material cycles, even as efforts focus on reducing landfilling and increasing material recovery. For instance, this regards paper products that remain contaminated by polychlorinated biphenyls even after chemical cleaning, as well as contaminated polymers or paperboard recyclates (EEA, 2017; Mofokeng et al., 2024; Pivnenko, 2016; Zhong et al., 2025). Besides contamination, there are also limitations caused by material degradation during the repeating recycling processes (e.g., cellulose, polymer or carbon fibres after several recycling cycles) (Isa et al., 2022; Yang &

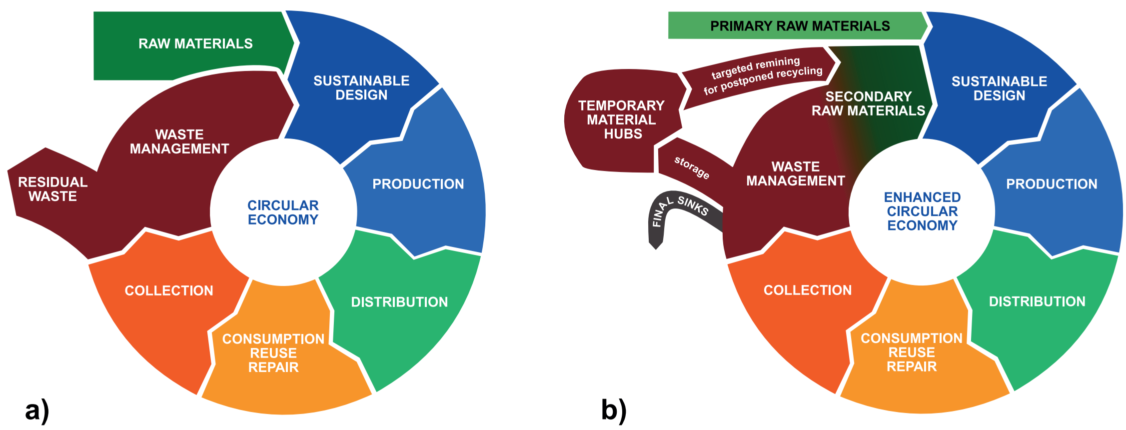

FIGURE 3: An innovative concept of a circular economy; a) currently desired circular economy loop (adapted from EP, 2023); b) circular economy loop including TMHs and final sinks.

Berglund, 2020). Kral et al. (2013) emphasize the need for safe final sinks for all unwanted substances, mandating a reassessment of material design strategies when suitable sinks are unavailable.

Final sinks can be both, natural or anthropogenic (Stanisavljevic & Brunner, 2021). An appropriate anthropogenic final sink for most organic pollutants, such as polybrominated diphenyl ethers, is state-of-the-art waste incineration; as it is for many other hazardous organic substances (Stanisavljevic & Brunner, 2021; Vyzinkarova & Brunner, 2013). According to Stanisavljevic & Brunner (2021), even inorganic substances can use incineration plants as their final sinks. Furthermore, controlled landfills operating based on the concept of a multi-barrier system (Stief, 1989) serve as today's final sinks. To maintain clean cycles and functional industry, society should focus on delivering the optimal amount of secondary resources rather than the maximal amount (Kral et al., 2013).

4.2 Exemplary candidate materials/waste for Temporary Material Hubs

In order to assess the need for TMHs and to roughly estimate their future demand, a screening of the current situation regarding the generation of "new or future waste streams" within the EU was carried out. Candidate materials/waste for TMHs are summarized in Table 2, together with their waste code, the amount currently generated and disposed in the EU, and also the related waste management and recycling issues. With respect to the proposed concept shown in Figure 2, these respective streams fall within the grey zones between recycling pillars.

Two examples (end-of-life wind blades and incineration residues) are then used as exemplary purposes in subsections 4.3 and 4.4.

4.3 Exemplary purpose for static Temporary Material Hubs

Diverse complex materials previously used in specific applications are becoming widely present in waste management – e.g., carbon fibre-reinforced polymers. Being used in wind blades, they are expanding due to the boom of renewable energy sources aimed at achieving net-zero emissions of greenhouse gases by 2050 (EP, 2021). Wind blades are also made of glass-reinforced polymers (the majority). Altogether, these materials are known as composites; they are resistant towards severe weather conditions with a usual life-span of 20-25 years (National Grid, 2023; Spini & Bettini, 2024). Estimates for wind blade waste generation in the EU between 2020 and 2030 vary widely. Sommer et al. (2020) project 570,000,000 tonnes of

TABLE 2: Currently relevant candidate materials/waste for TMHs; alphabetical order (DTMHs – dynamic TMHs; STMHs – static TMHs).

Material/waste/ product	Waste code	Amount generated in the EU	Disposal rate in the EU	Waste management and recycling issues	Type of TMHs
Composites/ wind blades	17 02 03[1] or 17 09 04[2]	Differs: 185,000 tonnes by 2030[3]; 570 million tonnes by 2030[4]; 350,000 tonnes by 2030[5]	• Lack of reliable data[1,6–10] • Landfilling or incineration without energy recovery of all composites: approximately 40-70% (160-280,000 tonnes/year)[6]	• The majority is co-processed in cement kilns, incinerated or landfilled[1,7–10] • Material degrades during recycling[11] • Virgin material is more valuable[11]	STMHs
Incineration residues	19 01 11*, 19 01 12[12]	19-31.8 million tonnes/ year[13,14]	• Landfilling: 11.3-16 million tonnes/year[14]	• Few environmentally safe recycling methods[15] • Environmentally challenging use of recyclates, often resulting in additional landfill waste[16]	DTMHs
PVC	02 01 04, 20 01 39[12]	2.5-4.1 million tonnes/ year[17]	• Incineration with energy recovery: 46% (1.5-1.9 million tonnes/ year)[17] • Landfilling or incineration without energy recovery: 19% (0.5-0.8 million tonnes/year)[17]	• Primarily incinerated rather than recycled; landfilling is restricted[18,19] • Incineration: danger of dioxin production[19] • Recycling technologies: limited by costs, high energy consumption, environmental impact, problematic quality of produced feedstock, low yield and the need for further research[20]	STMHs
Textiles/ clothing	20 01 10[12]	5.2 million tonnes/year[21]	• Incineration or landfilling: 78% (4.1 million tonnes/year)[21]	• The majority is incinerated, landfilled or exported to low-income countries[22] • Costly sorting and separation; low-quality secondary feedstock[23] • Contamination, e.g., by per- and polyfluoroalkyl substances[24]	STMHs
Tires	16 01 03[12]	300 million pieces/year [25,26]; over 3.5 million tonnes/ year[26]	• Landfilling is prohibited[27] • Incineration for energy production: 40% (1.4 million tonnes/year)[28]	• The majority is incinerated or recycled for low-value applications[26] • Recycling technologies: high investments costs, low efficiencies or being a threat to the environment[29] • Ongoing landfilling despite its unsuitability (banned in the EU)[27,30,31]	STMHs

[1] WindEurope (2020), [2] LAGA (2019), [3] Lichtenegger et al. (2020), [4] Sommer et al. (2020), [5] WindEurope (2021), [6] EuCIA (2023), [7] EC (2023c), [8] Gast et al. (2024), [9] Spini & Bettini (2024), [10] Volk et al. (2021), [11] Composites UK (2016), [12] EC (2014), [13] Lamers (2015), [14] Zero Waste Europe (2022), [15] Cao et al. (2024), [16] Fellner et al. (2015), [17] EC & Ramboll (2022), [18] Lahl & Zeschmar-Lahl (2024), [19] Miliute-Plepiene et al. (2021), [20] Ait-Touchente et al. (2024), [21] EC (2023a), [22] Stipanovic et al. (2023), [23] Andini et al. (2024), [24] Umweltbundesamt (2024), [25] EC (2023b), 26 Roetman et al. (2024), [27] EP and the Council (2024a), [28] Allen (2024), [29] Wu et al. (2024), [30] Dobrotă et al. (2020), [31] Guo et al. (2024)

wind blade waste to be generated between 2020 and 2030. The number was also cited in an EU article (EC, 2023c); however, the respective study shows some inconsistencies in data provided and therefore, this high number should be considered with caution. Lichtenegger et al. (2020) concluded that there will be approximately 185,000 tonnes of wind blade waste by 2030. WindEurope (2021) expects this amount to be around 350,000 tonnes. The installed capacity is predicted to increase from 255 GW in 2022 to a total capacity of 384 GW in 2027 (Gast et al., 2024; Sommer et al., 2020; WindEurope, 2023). Nonetheless, wind blade composites shall account for only 10% of thermoset polymer composite waste in Europe by 2025 (WindEurope, 2020).

Recycling of wind blades (approx. 2-5% of the wind turbine by mass) poses industrial challenges that are not satisfactorily resolved (Alves Dias et al., 2020; Schmidt, 2024; WindEurope, 2020). The majority of wind blade waste is currently co-processed in cement kilns, incinerated or landfilled, although data on the final destination of dismantled blades is scarce (EC, 2023c; Gast et al., 2024; Spini & Bettini, 2024; Volk et al., 2021; WindEurope, 2020). While virgin material is more valuable due to the degradation of recycled fibres (Composites UK, 2016), recycled feedstock can be used in industries tolerating lower quality and length of carbon fibres, such as the manufacture of hobby products for sports and leisure time, as suggested by Limburg and Quicker (2016). The costs of carbon fibres obtained from pyrolysis are up to four times lower than the costs of virgin material (approximately 5,000 EUR per tonne compared to approximately 20,000 EUR per tonne, respectively) (LAGA, 2019). The precursor used to manufacture carbon fibres significantly influences the final costs (Nunna et al., 2019), contributing over 50% to the total costs of the final product (Ellringmann et al., 2016; Nunna et al., 2019). Producing virgin carbon fibres is highly energy-intensive, requiring approximately 1,150 MJ/kg, whereas fibre recycling needs only 10% of this energy (Salas et al., 2023). Wind blades were labelled as "future waste" – a waste stream that is usually created by long-lived products that have already entered anthropogenic cycles in significant quantities. These are expected to increase in the future together with an increase in waste generation, and there are no feasible recycling technologies for them yet (Pomberger & Ragossnig, 2014).

Materials as composites cannot be currently feasibly recycled because the outputs from mechanic recycling do not reach sufficient quality; while chemical methods come with high implementation costs and consume vast amounts of energy (EC, 2023c). The outputs of thermal recycling methods bear disadvantages of compromised fibre length together with low polymer recovery rate. Due to their rather stable properties, dismantled wind blades could be stored in static TMHs and extracted when favourable recycling conditions are reached.

4.4 Exemplary purpose for dynamic Temporary Material Hubs

Out of 229 million tonnes of municipal waste generated in the EU in 2022, 59 million tonnes (26%) were incinerated (Eurostat, 2024). Incineration residues are unavoidable mineral products of waste incineration and contain, e.g., metals and rare earth elements. One ton of waste produces approximately 0.2-0.25 tonnes of incineration residues (Chen et al., 2023; Lamers, 2015; Reig et al., 2023). The main management option remains landfilling (Margallo et al., 2015), the most practical reuse option appears to be in road or embankment constructions (Oehmig et al., 2015).

There are currently few environmentally safe recycling methods (Cao et al., 2024). Current technologies enable the extraction of valuable compounds from incineration residues or the removal of harmful substances – e.g., metals can be extracted by leaching in acids and further used in industry (Cao et al., 2024; Reig et al., 2023). While some European countries (Netherlands, Switzerland) have attempted to recycle incineration residues, they have encountered environmental challenges, often resulting in additional landfill waste (Fellner et al., 2015). Incineration residues could benefit from the concept of dynamic TMHs, although considering that a large mass of residues must be handled, sufficient storage space must be provided.

A similar concept has already been described by Sapsford et al. (2023). Their ASPIRE concept is inspired by natural lithospheric weathering and pedogenetic processes, aiming to mimic those with waste, as observed in mine tailings and smelter residues. Carbon dioxide might be mitigated through ash weathering and ash could be reused as cement mixture or in agri-/silviculture as source of nutrients given its content of critical elements and minerals. Stored material could be "cleaned" or concentrated during its time in a facility and after the defined time period, it can be remined (Sapsford et al., 2023).

Weathering significantly changes the properties of incineration residues (Chimenos et al., 2003). Chen et al. (2023) and Chimenos et al. (2003) state that reactive substances in waste residues react and form physically and chemically stable phases with reduced release of heavy metals. This is in harmony with findings made by Obernberger and Supancic (2024) who have described ageing of incineration ashes during storage periods, and Mostbauer et al. (2008) who analysed the chemical weathering of waste incineration residues and the sequestration of CO_2 through the so-called BABIU process (Bottom Ash for Biogas Upgrading). Processes taking place during ash ageing/weathering include reduction of pH, Ca solubility reduction and volume stabilization (Gori et al., 2013; Obernberger & Supancic, 2024). While natural weathering and pedogenetic processes last long time periods, the processes in TMHs have to be usable for anthropogenically relevant timescales. Although incineration residues show reduced carbonation efficiencies compared to reference materials (e.g., composite of spent refractories with uptake up to 438 kg of CO_2 per ton of material), they absorb CO_2 (Schinnerl et al., 2024). In the case of bottom ash, the uptake is approximately 60 kg of CO_2 per ton of material (ibid.).

5. DISCUSSION

The lack of a clear recyclability definition at the EU level complicates recycling efforts. Defining recyclability would simplify deciding whether a material shall be kept in anthropogenetic cycles (either to be recycled directly or to be kept in TMHs) or if it should find its final sink. Pomberger and Bezama (2024) discussed the need for redefinition and sharpening of the term recyclability and pointed out the differences between theoretical, technical and real recyclability. Esguerra et al. (2024) describe recyclability as the potential for sorting out and subsequent recycling in corresponding facilities, thus focussing on the technical issues. The non-profit initiative RecyClass names a definition stating that plastics considered recyclable must meet conditions similar to those in the Proposal for a Regulation on packaging and packaging waste (RecyClass, 2024).

Recycling options and efforts are tightly connected to regulative challenges. There is, for example, a need for improved end-of-waste legislation and inclusion of EU-wide technical standards assuring that recycling operators produce material of stable quality (EEA, 2022, 2023). The competitiveness of recycled materials needs to be enhanced by the implementation of taxes or subventions, and the implementation of reasonable quotes to include recyclates in new products (EEA, 2023).

5.1 Previous concepts on remining and temporary storage

Storage of waste described in previous literature and regulations is usually connected to subsequent utilization or disposal in a short time. According to the EU Waste Framework Directive, temporary storage means storing waste before transportation to a waste treatment facility (EP and the Council, 2024b). The EU Landfill Directive states that storage of waste prior to disposal for over 1 year, and prior to recovery or treatment for over 3 years, is legally considered a landfill (EP and the Council, 2024a). In that case, the site has to comply with landfill regulations (Lehmphul, 2024).

Such time periods might not be sufficient as certain emerging waste streams need longer times in storage before their feasible utilization. Thus, TMHs have a clear intention of postponed recycling of the material/waste, unlike the current concepts of storage that can in certain cases serve to postpone disposal.

The case studies in literature emphasize the importance of further research of storage solutions for postponed resource recovery, that highlight the potential application and necessity of TMHs. One notable example is sewage sludge incineration and the subsequent treatment of ashes. Research has shown that, especially in case of mono-incineration, the storage of ashes should be considered, since the recovery of phosphorus from sewage sludge ash is not feasible at the present phosphorus market price (Bagheri et al., 2024; Montag et al., 2015). However, it can become economically feasible if the phosphorus price rises. Companies such as EasyMining with its Ash2Phos technology are on the way towards the commercialization of phosphorus recovery, with their recovery plant to start operations in early 2027 (EasyMining, 2025).

A step towards partial implementation of this concept can already be seen in the city of Zurich, which operates a sewage sludge mono-incineration plant with a future dedication to store produced ashes and enable future phosphorus recovery and recycling (Eicher, 2025; Montag et al., 2015).

A similar approach can be observed in the wind energy sector. Several European companies involved in wind blade decommissioning are already postponing disposal while independently researching economically feasible solutions or anticipating their emergence in the near future. This strategy underscores the justification for the costs associated with storage or potential postponed disposal (Nagle et al., 2022).

The German federal state waste working group LAGA (2019) addresses the storage of carbon fibre-reinforced material until adequate recycling routes are available. The need for temporary material storage before recovery is supported by the concern that carbon-reinforced polymers' fibre dust might be carcinogenic and have negative effects on health similar to those of as asbestos. Therefore, further research is needed to assure appropriate handling (LAGA, 2019).

The Austrian ministry is planning to temporarily allow previously forbidden landfilling of carbon- and glass-reinforced polymers. This decision is based on the current lack of recycling capacity or its insufficient performance (BMK, 2024). This intent contradicts the wind industry's goal to have landfilling of decommissioned wind blades banned by 2025 (WindEurope, 2021).

The few temporary storage concepts previously described in international literature are usually related to specific cases and waste streams (Table 3). The concepts range from disaster management, short-term storage of demolition waste from military actions and hazardous waste on drilling sites to so-called monofills for future mining. However, they do not necessarily lead to subsequent recycling (Table 3).

The "dead storage" of products is a specific and special case of unintended and undesirable temporary storage. This practice has negative effects on the circular economy as valuable substances remain in drawers rather than in recycling facilities and cannot become secondary feedstock (Wilson et al., 2017). The products lose their economic value with the duration of storage (Inghels & Bahlmann, 2021); engineered TMHs should have the reverse effect – the value of material/waste is unlocked with age.

The concept of TMHs shares similarities with approaches such as urban mining, landfill mining and monofilling, but key distinctions set TMHs apart. Urban mining is primarily driven by the aim to recover materials directly from the urban environment (Ghisellini et al., 2022). TMHs complement urban mining by providing storage for materials from deconstruction or building mining that lack immediate market demand. The ultimate purpose of TMHs is to prevent disposal in a landfill altogether and make postponed use of materials already present in anthropogenic deposits.

Landfill mining, as a subcategory of urban mining (Johansson et al., 2013), addresses disposal as the least desired option in the waste hierarchy and aims to "reverse" its effects (Ghisellini et al, 2022). Enhanced landfill mining builds upon this concept by considering the valorisation of

TABLE 3: Previous concepts on remining and temporary storage; alphabetical order.

Concept	Description	Reference
Hibernation/dead storage of products	Retention of small end-of-life electronics (e.g., mobile phones) by their users to have the security of a substitution device in case the primary one breaks down or else	Inghels and Bahlmann (2021); Wilson et al. (2017)
Landfill mining	Extraction of minerals or other solid resources from previously (often uncontrolled) disposed waste	Krook et al. (2012)
Monofills	Separate disposal of single waste fractions on landfills (compartments); e.g., monofills of e-waste for future mining due to current limitations in availability of treatment/recycling technologies; incineration ash monofills	Kahhat & Kavazanjian (2010); Liu et al. (2022)
Temporary storage	Storage of mineral-rich waste materials for future re-mining of valuable materials	Sapsford et al. (2023)
Temporary storage device	Storage of hazardous waste on drilling sites	Liu et al. (2023)
Temporary storage of demolition waste from military actions	Storage of waste to protect the environment and assure the best possible treatment after the termination of martial law	Cabinet of Ministers of Ukraine (2022)
Temporary storage sites	Storage of waste during disasters before it can be adequately treated	Lontoc et al. (2023)
Urban mining	"Systematic reuse of materials from urban areas"	Brunner (2011)

mined waste for both material and energetic use (Jones et al., 2013; Parrodi et al., 2020). The key distinction between TMHs and landfill mining lies in TMHs' strict commitment to material recovery. In landfill mining, remining is an option, while TMHs make it an obligation. Moreover, the physical space of TMHs is not only limited to "landfill area", also further options like unused factories, storage halls, brownfields etc can be adapted and used.

Monofills, in contrast, are landfills or compartments of landfills designed to dispose a single type of waste. Examples include e-waste monofills proposed by Kahhat & Kavazanjian (2010) and also incineration residue monofills (Roessler et al., 2017). Mining a monofill to recover materials is similar to landfill mining, but the homogenous nature of waste in monofills may simplify the mining efforts compared to mixed waste landfills. TMHs differ in that remining is inherent to their design, and they are intended to be employed for restricted time periods, offering a flexible and potentially modular solution for efficient remining. Nevertheless, well-considered and specifically planned monofills, if designed for high-quality remining, can act as a transitional form of TMHs, bridging the gap between current and future recycling advancements.

5.2 Factors impacting the performance of the Temporary Material Hubs

Several factors must be considered, researched and clarified to allow a successful implementation and integration of TMHs into the circular economy in practice. As part of this exploratory study, a conceptual SWOT analysis was conducted to assess the theoretical potential performance of TMHs based on the data acquired from literature screening and discussions with experts and peer groups (Figure 4).

The biggest strength of the TMHs concept lies in the optimization of resource use and the reduction of waste going to (final) disposal, and due to the need of a comprehensive database of stored waste, material traceability should be enhanced.

However, a key weakness is the current difficulty in estimating the economic situation (prospective costs and revenues) due to the lack of detailed technical implementation concepts, suitable siting and adequate storage capacities.

The opportunity in the use of TMHs is the support of the European circularity goals, allowing feasible recycling of currently non-valuable materials/waste, thereby unlocking their value in the future. This would lead to a reduction in raw material extraction. TMHs may also help balancing the ratio of materials to be recycled and the available recycling capacities by retaining a certain amount of material. Extended Producer Responsibility (EPR) schemes for certain waste streams might support TMHs by ensuring the financing of the storage for postponed proper handling and recycling. Despite challenges in measuring their effectiveness, EPR systems generally yield positive outcomes (The Environmental Research & Education Foundation, 2025).

The biggest threats stem from uncertainties faced by the concept, such as uncertain material flows in the future, as well as market situation and future changes in regulations. Another threat lies in the estimation of storage duration, meaning if and when materials/waste will become recyclable, and if/when markets will emerge. These issues will need a thorough assessment. Moreover, similar to all waste management facilities, underperforming TMHs might pose a threat to the environment.

Future research must focus on the duration of storage, technical and siting properties, and the estimation of costs/revenues, market development and risk-assessment. An adaptation of the regulatory framework would be also needed to allow the implementation of such concept.

5.2.1 Duration of storage

The proposed conceptual framework supports the long-term sustainability goals, producing results that extend beyond short-term objectives, such as those tied to 5-year policy cycles (EEA, 2024). The duration of storage must be tailored to specific materials/waste and respective recycling technologies and infrastructure. For example, a study by the EC expects that the performance of solvolysis and pyrolysis for recycling wind-turbine blade waste will improve in the near future (EC, 2023c). Garcia-Gutierrez et

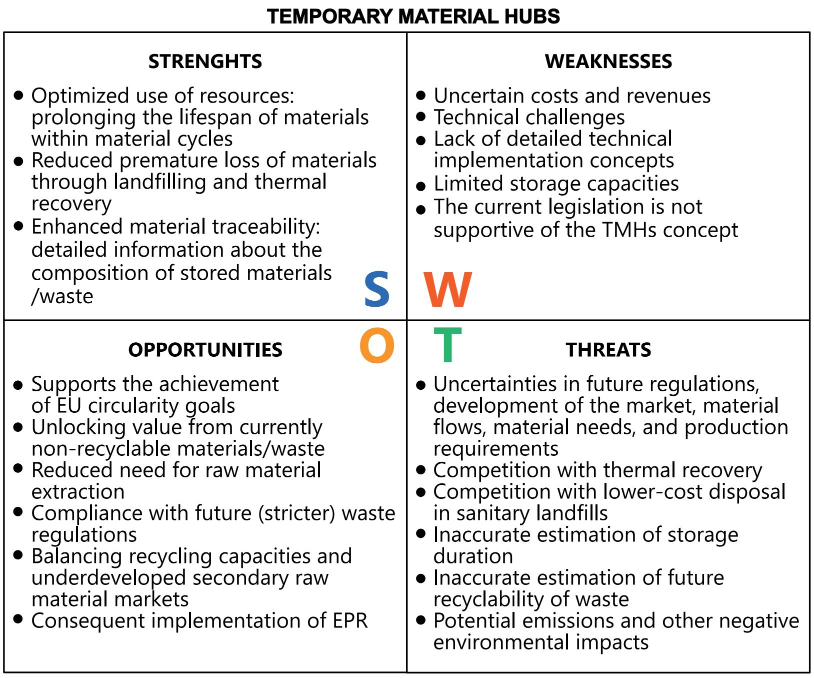

FIGURE 4: SWOT analysis of the potential performance of TMHs.

al. (2023) and Werner et al. (2022) estimate that most polymer chemical-recycling technologies will reach positive net earnings before 2040 (methanolysis in 2025, pyrolysis in 2033). The projections are based on expected cost reduction in recycling technologies (37.5% in the case of chemical recycling technologies, mechanical recycling costs are expected to remain the same) and increased costs of virgin material (increase by 71%). However, the authors acknowledge significant uncertainties in the estimates (Garcia-Gutierrez et al., 2023; Werner et al., 2022) and according to Islam et al. (2025), methanolysis is still in need of further research and optimisation. Based on the predictions, the storage time of certain plastic waste streams in TMHs may last up to ~ 15 years from 2025. Furthermore, conditions of potential premature remining (before the expiration of designated storage time) in case of favourable conditions must be determined.

The storage duration estimations might be based on the technology readiness level (TRL) of respective technologies. Rybicka et al. (2016) researched the TRL of composite management technologies, concluding that in 2016, incineration and landfilling were the most advanced, achieving the highest (maximal) TRL 9, pyrolysis (carbon fibres) and mechanical grinding of fibres (glass fibres) achieved TRL 8, while pyrolysis of glass fibres and mechanical grinding of carbon fibres were at TRL 6-7. Most composite recycling technologies, however, remained in the early stages, with TRLs of 3-4 (Rybicka et al., 2016). Within past years, none of the composite recycling technologies achieved a TRL allowing industrial-scale application (WindEurope, 2020), and there is still a lack of effective solution (Tortorici et al., 2025).

Generally, storage should last as long as necessary to achieve feasible recycling conditions but as short as possible to minimize the storage costs (Obernberger & Supancic, 2024). However, we propose that “temporary” in this concept may cover a time frame of about 5 to max. 20 years.

The duration of storage introduces a potential risk of unwanted alteration and degradation of the stored material/waste, which could jeopardize subsequent material recovery. To mitigate the risk, the facilities must be designed to withstand adverse impacting conditions, e.g., weather conditions, and prevent damage over time. One example of such degradation is corrosion. Even for very rigid and highly durable materials such as wind blades, this risk must be considered, especially if the blades are already degraded at the end of their life cycle (Muntenita et al., 2024).

The general framework for the decision-making under which conditions waste should be “remined” from the TMHs and reintroduced into material cycles is provided by the three pillars addressing “recyclability” (see Figure 2). However, we emphasize that each case (waste type) needs to be assessed individually, taking into account its specific conditions and pre-selected indicators, as mentioned in chapter 3.

5.2.2 Technical design and infrastructure

It is pivotal to know the composition of stored waste to ensure safety of storage and the effective option of future remining. This might require chemical analyses, assessment of potential contamination and thorough mapping and documentation, e.g., similar to the procedure for hazardous waste disposal in German underground landfills in Ordinance on Landfills and Long-Term Storage (Bundesministerium der Justiz, 2009). Based on the composition, static or dynamic TMHs are employed.

Static TMHs should be controlled and engineered to ensure that stored waste does not undergo any unwanted alteration or degradation during storage time. Certain abandoned facilities could be employed (such as old industrial or military buildings), inspiration could be found in underground disposal of hazardous waste as mentioned above.

For the preparation of Refused Derived Fuels (RDF) intended for energy recovery in diverse industries, storage facilities are already often needed and applied. Storage options for TMHs can be based on this, but require further (sometimes more complex) conditions, especially as they are intended to be stored for much longer periods of time. Thus, THMs would be more stringent than facilities for RDF storage. This is due to the clear intention of subsequent material recovery, which demands different, mainly higher quality standards compared to energy recovery. For instance, while Romaszewski and Fitas (2024) noted that open-air storage has no adverse effects on the thermal recovery of RDF, the same approach might not be suitable for TMHs, where material integrity is critical for future recycling processes.

Dynamic TMHs need to be engineered and monitored to allow stored waste to undergo certain desired processes. Those are, for example, the weathering processes of incineration residues for which controlled landfill sites with fading landfill-gas generation (with remaining high CO_2 concentrations; compare BABIU Process (Mostbauer et al., 2008) and state-of-the-art barriers could be used.

In certain cases, an Environmental Impact Assessment may be required to avoid potential adverse impacts on the environment.

It is important to note that TMHs may require a substantial area, which could lead to challenges such as land-use conflicts and competing interests (Montag et al., 2015). In theory, facilities such as mined (empty) landfills could be employed if environmental protection and properties allowing secure storage of materials/waste are assured. Consideration should be given to the option of "reusing" such facilities and adapting them to various types of materials/waste in parallel or subsequently rather than aiming for one-time employment. Additionally, case-specific pre-treatment would affect the design of the storage and the infrastructure setting. The quality and amount of the regarded waste stream and the status of the existing infrastructure will determine whether local or centralized TMHs should be created and if additional infrastructure needs to be established. The most significant transportation load is expected at the beginning of storage process and when remined. The extent of pre-treatment and transportation involved will also affect the overall costs of the concept.

5.2.3 Economic and legal factors

The costs of storage for postponed recycling might be higher than those of conventional waste management options. Competing with the TMHs are waste-to-energy facilities for thermal recovery of materials/waste, and landfills presenting a cheap yet environmentally/circularly unfavourable waste management alternative. Potential (mid- to long-term) environmental benefits of (postponed) recycling compared to energy recovery, landfilling and also primary raw material extraction must be considered. When considering the negative environmental externalities, the concept of TMHs may compete with the conventional waste management options in monetary terms. Furthermore, unlocking the value of currently non-valuable materials/waste may create revenues in the future that need to be estimated. Such estimations are currently a source of uncertainties and pose a potential weakness of TMHs. However, further research and modelling of cost scenarios shall reduce these economic risks.

A key priority for future research is conducting a comprehensive cost-benefit analysis, particularly to compare TMHs with thermal treatment and landfilling. This will require detailed modelling to account for variables such as market development and resource scarcity. Furthermore, an LCA analysis is recommended to evaluate and compare the environmental impacts of these addressed waste management strategies, providing a more holistic understanding of their long-term viability.

Platforms like digital marketplaces could facilitate the recycling of stored materials by enhancing their visibility and enabling trading. However, these platforms can only function if a market already exists, which is one of the challenges waste recycling might face, as explained in Figure 2. For this reason, TMHs should be implemented prior to the establishment of digital markets. Some platforms already enable the resale of dismantled wind blades for reuse, such as "Business in Wind" (2025).

To implement the TMHs concept, both national and European legislation would need to be adapted. One of the most critical changes would involve extending the duration of waste storage beyond the currently permitted periods outside a landfill regime (EP and the Council, 2024a). Moreover, implementing taxes or levies on primary raw materials could help improve the competitiveness of recycled materials (EEA, 2023), which would support the concept of TMHs.

6. CONCLUSIONS AND OUTLOOK

Optimizing resource use is a crucial step towards enhancing the circular economy. Unlike traditional waste management strategies, this study introduces a forward-looking approach for storing prospectively valuable waste streams for future recycling, rather than relying on immediate disposal or energy recovery. While it builds on existing concepts, it introduces several theoretical advancements: it (1) formalizes the concept of TMHs as a distinct strategy for bridging the temporal gap between current technological and economic capabilities and future advancements, (2) proposes the recycling pillars as a novel decision-making framework for evaluating recyclability, and (3) integrates the concept of final sinks to maintain clean material cycles. The comprehensive and integrative considerations of these diverse aspects in one central concept represent a meaningful progress, which has not been systematically addressed in previous literature.

This study provides a theoretical foundation for further use-case-specific research and practical implementation of the TMHs concept. Further details and application specifications, which may have a strong impact on the realisation of such a concept, have still to be researched and defined. However, the authors are of the opinion that it is important to disseminate the concept now in order to initiate a discussion within the scientific community and among diverse stakeholders in the waste and resource management sectors. Storing materials/waste for postponed recycling appears justifiable based on the current and prospective advancements in recycling technologies and secondary markets. A SWOT analysis, as outlined in this study, can be used to assess the principal feasibility of the TMHs concept for specific applications and cases – considering factors such as waste type, existing infrastructure, location, storage duration, and financial viability. Additionally, this analysis helps to identify uncertainties that may challenge implementation while also highlighting potential opportunities and benefits.

To build on this groundwork, future research should include case-based modelling and real-world pilot studies to quantitatively assess the concept's feasibility and consider cost- and risk-assessment approaches, and potential pre-treatment requirements for the stored waste. Furthermore, engaging with stakeholders – such as manufacturers, waste sector operators and policy makers – through interviews, would expand the study's perspective beyond academia, incorporating practical and industry-relevant insights. For successful implementation, both European and national legal frameworks must be adapted, particularly concerning the definition of recyclability, the extension of the permitted storage time, as well as the creation of conditions that foster secondary raw material markets. In addition to these technical, economic and legal considerations, future research should address the social dimensions to better understand societal implications and acceptance.

This study serves as a blueprint for innovative approaches to enhance the circular economy and aims to initiate a debate on the need for both, temporary material/waste storage and safe final sinks.

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REFERENCES

Ait-Touchente, Z., Khellaf, M., Raffin, G., Lebaz, N., & Elaissari, A. (2024). Recent advances in polyvinyl chloride (PVC) recycling. Polymers for Advanced Technologies, 35(1), e6228. https://doi.org/10.1002/pat.6228

Allen, M. (2024). Recycling tyres and plastics with an ancient heating method | Horizon Magazine. https://projects.research-and-innovation.ec.europa.eu/en/horizon-magazine/recycling-tyres-and-plastics-ancient-heating-method

Alves Dias, P., Pavel, C., Plazzotta, B., & Carrara, S. (2020). Raw materials demand for wind and solar PV technologies in the transition towards a decarbonised energy system. Publications Office of the European Union. https://data.europa.eu/doi/10.2760/160859

Andini, E., Bhalode, P., Gantert, E., Sadula, S., & Vlachos, D. G. (2024). Chemical recycling of mixed textile waste. Science Advances, 10(27), eado6827. https://doi.org/10.1126/sciadv.ado6827

Andrady, A. L., & Neal, M. A. (2009). Applications and societal benefits of plastics. Philosophical Transactions of the Royal Society B: Biological Sciences, 364(1526), 1977–1984. https://doi.org/10.1098/rstb.2008.0304

Bagheri, M., Gómez-Sanabria, A., & Höglund-Isaksson, L. (2024). Economic feasibility and direct greenhouse gas emissions from different phosphorus recovery methods in Swedish wastewater treatment plants. Sustainable Production and Consumption, 49, 462–473. https://doi.org/10.1016/j.spc.2024.07.007

BMK. (2024). Novelle der Deponieverordnung CFK und GFK Abfälle. https://www.wko.at/oe/news/0119-tf-wfa-novelle-deponievo.pdf

Brunner, P. H. (2011). Urban Mining A Contribution to Reindustrializing the City. Journal of Industrial Ecology, 15(3), 339–341. https://doi.org/10.1111/j.1530-9290.2011.00345.x

Bund/Länder-Arbeitsgemeinschaft Abfall (LAGA). (2019). Entsorgung faserhaltiger Abfälle—Abschlussbericht. https://proenvi.de/recht/LAGA/Bericht-Laga-Entsorgung-faserhaltige-abfaelle_201907.pdf

Bundesministerium der Justiz. (2009). § 1 DepV - Einzelnorm. Bundesministerium der Justiz. https://www.gesetze-im-internet.de/depv_2009/DepV.pdf

Business in Wind. (2025). Business in Wind—Sharing commitment. Business in Wind. https://businessinwind.com/

Cabinet of Ministers of Ukraine. (2022). On Approval of the Procedure for Waste Management Generated in Connection with Damage (Destruction) of Buildings and Structures as a Result of Hostilities, Terrorist Acts, Sabotage or Works to Eliminate Their Consequences and Amendments to Certain Resolutions of the Cabinet of Ministers of Ukraine.

Cao, C., Yuan, Z., Liu, H., Fei, X., Esteban, J., & She, Q. (2024). Insights into the usage of biobased organic acids for treating municipal solid waste incineration bottom ash towards metal removal and material recycling. Separation and Purification Technology, 353, 128330. https://doi.org/10.1016/j.seppur.2024.128330

Chen, B., Perumal, P., Illikainen, M., & Ye, G. (2023). A review on the utilization of municipal solid waste incineration (MSWI) bottom ash as a mineral resource for construction materials. Journal of Building Engineering, 71, 106386. https://doi.org/10.1016/j.jobe.2023.106386

Chimenos, J. M., Fernández, A. I., Miralles, L., Segarra, M., & Espiell, F. (2003). Short-term natural weathering of MSWI bottom ash as a function of particle size. Waste Management, 23(10), 887–895. https://doi.org/10.1016/S0956-053X(03)00074-6

Composites UK. (2016). Composites Recycling – Where are we now? End of Life and Recycling | Composites UK. https://compositesuk.co.uk/industry-support/environmental/end-of-life-and-recycling/

Cossu. (2016). Back to Earth Sites: From "nasty and unsightly" landfilling to final sink and geological repository. Waste Management, 55, 1–2. https://doi.org/10.1016/j.wasman.2016.07.028

Dobrotă, D., Dobrotă, G., & Dobrescu, T. (2020). Improvement of waste tyre recycling technology based on a new tyre markings. Journal of Cleaner Production, 260, 121141. https://doi.org/10.1016/j.jclepro.2020.121141

EasyMining. (2025). The world's first Ash2Phos plant. https://www.easymining.com/projects/plant-projects/Ash2Phos-plant-schkopau/

Eicher, N. (2025). Klärschlammbehandlung. Kanton Zürich. https://www.zh.ch/de/umwelt-tiere/abfall-rohstoffe/abfaelle/abfallanlagen/klaerschlammbehandlung.html

Ellringmann, T., Wilms, C., Warnecke, M., Seide, G., & Gries, T. (2016). Carbon fiber production costing: A modular approach. Textile Research Journal, 86(2), 178–190. https://doi.org/10.1177/0040517514532161

Esguerra, J. L., Carlsson, A., Johansson, J., & Anderberg, S. (2024). Characterization, recyclability, and significance of plastic packaging in mixed municipal solid waste for achieving recycling targets in a Swedish city. Journal of Cleaner Production, 468, 143014. https://doi.org/10.1016/j.jclepro.2024.143014

EuCIA. (2023). EuCIA launches European Composites Waste & Recycling Survey—JEC. Https://Www.Jeccomposites.Com/. https://www.jeccomposites.com/news/spotted-by-jec/eucia-launches-european-composites-waste-recycling-survey/

European Aluminium. (2017). Activity Report 2016. https://european-aluminium.eu/wp-content/uploads/2022/10/activity-report-2016_web.pdf

European Aluminium. (2020). Circular Aluminium Action Plan—Executive summary. https://european-aluminium.eu/wp-content/uploads/2022/08/european-aluminium-circular-aluminium-action-plan.pdf
European Chemicals Agency. (2024). All news—ECHA. https://echa.europa.eu/news-and-events/news-alerts/all-news/-/asset_publisher/yhAseXkvBI2u/
European Commission. (2014). 2014/955/EU: Commission Decision of 18 December 2014 amending Decision 2000/532/EC on the list of waste pursuant to Directive 2008/98/EC of the European Parliament and of the Council Text with EEA relevance. http://data.europa.eu/eli/dec/2014/955/oj/eng
European Commission. (2020a). A new Circular Economy Action Plan. https://eur-lex.europa.eu/legal-content/EN/TXT/?qid=1583933814386&uri=COM:2020:98:FIN
European Commission. (2020b). Communication From The Commission To The European Parliament, The Council, The European Economic And Social Committee And The Committee Of The Regions A new Circular Economy Action Plan For a cleaner and more competitive Europe. https://eur-lex.europa.eu/legal-content/EN/TXT/?qid=1583933814386&uri=COM:2020:98:FIN
European Commission. (2020c). Strategic Plan 2020-2024. https://commission.europa.eu/system/files/2020-10/env_sp_2020_2024_en.pdf
European Commission. (2022). Proposal for a REGULATION OF THE EUROPEAN PARLIAMENT AND OF THE COUNCIL on packaging and packaging waste, amending Regulation (EU) 2019/1020 and Directive (EU) 2019/904, and repealing Directive 94/62/EC. https://eur-lex.europa.eu/legal-content/EN/TXT/?uri=CELEX-%3A52022PC0677
European Commission. (2023a). Circular economy for textiles [Text]. European Commission - European Commission. https://ec.europa.eu/commission/presscorner/detail/en/ip_23_3635
European Commission. (2023b). LIFE project aims to revolutionise Europe's approach to recycling end-of-life tyres. - European Commission. https://cinea.ec.europa.eu/news-events/news/life-project-aims-revolutionise-europes-approach-recycling-end-life-tyres-2023-12-14_en
European Commission. (2023c). More circular, less carbon: Chemical recycling holds promise for wind-turbine blade waste. https://environment.ec.europa.eu/news/more-circular-less-carbon-chemical-recycling-holds-promise-wind-turbine-blade-waste-2023-10-19_en
European Commission. (2024). Plastic Recycling—European Commission. https://food.ec.europa.eu/food-safety/chemical-safety/food-contact-materials/plastic-recycling_en
European Commission & Ramboll. (2022). The use of PVC (poly vinyl chloride) in the context of a non-toxic environment: Final report. Publications Office of the European Union. https://data.europa.eu/doi/10.2779/375357
European Council. (2024). European Green Deal. European Council. https://www.consilium.europa.eu/en/policies/green-deal/
European Environment Agency. (2017). Circular by design—Products in the circular economy—European Environment Agency (Publication 6/2017). https://www.eea.europa.eu/publications/circular-by-design
European Environment Agency. (2022). Investigating Europe's secondary raw material markets—European Environment Agency [Publication]. https://www.eea.europa.eu/publications/investigating-europes-secondary-raw-material
European Environment Agency. (2023). Markets for many commonly recycled materials struggle in the EU. European Environment Agency. https://www.eea.europa.eu/en/newsroom/news/markets-commonly-recycled-materials-struggle
European Environment Agency. (2024). Europe's sustainability transitions outlook (TH-AL-24-011-EN-N; p. 70). European Environment Agency. https://www.eea.europa.eu/publications/europes-sustainability-transitions-outlook
European Environment Agency. (2025). Circular material use rate in Europe. European Environment Agency. https://www.eea.europa.eu/en/analysis/indicators/circular-material-use-rate-in-europe
European Environmental Bureau. (2017). Keeping it clean: How to protect the circular economy from hazardous substances - EEB - The European Environmental Bureau. https://eeb.org/library/keeping-it-clean-how-to-protect-the-circular-economy-from-hazardous-substances/
European Parliament. (2021). European Parliament resolution of 15 January 2020 on the European Green Deal (2019/2956(RSP)). https://eur-lex.europa.eu/legal-content/EN/TXT/?uri=CELEX-%3A52020IP0005
European Parliament. (2023). Circular economy: Definition, importance and benefits. https://www.europarl.europa.eu/news/en/headlines/economy/20151201STO05603/circular-economy-definition-importance-and-benefits
European Parliament. (2024a). Deal on new rules for more sustainable packaging in the EU. News - European Parliament. https://www.europarl.europa.eu/news/en/press-room/20240301IPR18595/deal-on-new-rules-for-more-sustainable-packaging-in-the-eu
European Parliament. (2024b). Plastic waste and recycling in the EU: Facts and figures. Topics | European Parliament. https://www.europarl.europa.eu/topics/en/article/20181212STO21610/plastic-waste-and-recycling-in-the-eu-facts-and-figures
European Parliament and the Council. (2019). Directive—2019/904—EN - SUP Directive—EUR-Lex. https://eur-lex.europa.eu/eli/dir/2019/904/oj
European Parliament and the Council. (2024a). Council Directive 1999/31/EC of 26 April 1999 on the landfill of waste. http://data.europa.eu/eli/dir/1999/31/2024-08-04/eng
European Parliament and the Council. (2024b). Directive 2008/98/EC of the European Parliament and of the Council of 19 November 2008 on waste and repealing certain Directives (Text with EEA relevance) Text with EEA relevance. http://data.europa.eu/eli/dir/2008/98/2024-02-18/eng
European Parliament and the Council. (2024c). Regulation (EU) 2024/1781 of the European Parliament and of the Council of 13 June 2024 establishing a framework for the setting of ecodesign requirements for sustainable products, amending Directive (EU) 2020/1828 and Regulation (EU) 2023/1542 and repealing Directive 2009/125/EC (Text with EEA relevance). http://data.europa.eu/eli/reg/2024/1781/oj/eng
Eurostat. (2024). Municipal waste statistics. https://ec.europa.eu/eurostat/statistics-explained/index.php?title=Municipal_waste_statistics
Fellner, J., Lederer, J., Purgar, A., Winterstetter, A., Rechberger, H., Winter, F., & Laner, D. (2015). Evaluation of resource recovery from waste incineration residues – The case of zinc. Waste Management, 37, 95–103. https://doi.org/10.1016/j.wasman.2014.10.010
Fraser, M., Conde, Á., & Laxmi, H. (2024). The Circularity Gap Report. https://www.circularity-gap.world/2024#download
Garcia-Gutierrez, P., Amadei, A. M., Klenert, D., Nessi, S., Tonini, D., Tosches, D., Ardente, F., & Saveyn, H. (2023). Environmental and economic assessment of plastic waste recycling. JRC Publications Repository. https://doi.org/10.2760/0472
Gast, L., Meng, F., & Morgan, D. (2024). Assessing the circularity of onshore wind turbines: Using material flow analysis for improving end-of-life resource management. Resources, Conservation and Recycling, 204, 107468. https://doi.org/10.1016/j.resconrec.2024.107468
Geist, H., & Balle, F. (2025). Recyclability: Redefining the concept for the circular economy. Journal of Industrial Ecology, 29(5), 1505–1522. https://doi.org/10.1111/jiec.70082
Ghisellini, P., Ncube, A., Casazza, M., & Passaro, R. (2022). Toward circular and socially just urban mining in global societies and cities: Present state and future perspectives. Frontiers in Sustainable Cities, 4. https://doi.org/10.3389/frsc.2022.930061
Gori, M., Bergfeldt, B., Reichelt, J., & Sirini, P. (2013). Effect of natural ageing on volume stability of MSW and wood waste incineration residues. Waste Management, 33(4), 850–857. https://doi.org/10.1016/j.wasman.2012.12.005
Guo, Z., Qiu, J., Kirichek, A., Zhou, H., Liu, C., & Yang, L. (2024). Recycling waste tyre polymer for production of fibre reinforced cemented tailings backfill in green mining. Science of The Total Environment, 908, 168320. https://doi.org/10.1016/j.scitotenv.2023.168320
Herhof Umwelttechnik GmbH. (1992). Verfahren zur Zwischenlagerung von Kunststoffabfällen (European Union Patent EP0500007A2). https://patents.google.com/patent/EP0500007A2/de
Hestin, M., Faninger, T., & Milios, L. (2015). Increased EU Plastics Recycling Targets: Environmental, Economic and Social Impact Assessment. Plastic Recyclers Europe. http://www.plasticsrecyclers.eu/wp-content/uploads/2022/10/increased-eu-plastics-recycling-targets.pdf

Hopewell, J., Dvorak, R., & Kosior, E. (2009). Plastics recycling: Challenges and opportunities. Philosophical Transactions of the Royal Society B: Biological Sciences, 364(1526), 2115–2126. https://doi.org/10.1098/rstb.2008.0311

Inghels, D., & Bahlmann, M. D. (2021). Hibernation of mobile phones in the Netherlands: The role of brands, perceived value, and incentive structures. Resources, Conservation and Recycling, 164, 105178. https://doi.org/10.1016/j.resconrec.2020.105178

Isa, A., Nosbi, N., Che Ismail, M., Md Akil, H., Wan Ali, W. F. F., & Omar, M. F. (2022). A Review on Recycling of Carbon Fibres: Methods to Reinforce and Expected Fibre Composite Degradations. Materials, 15(14), 4991. https://doi.org/10.3390/ma15144991

Islam, Md. M., Haque, N., Lau, D., Bhuiyan, M., & Pramanik, B. K. (2025). Comparative life cycle assessment of end–of–life strategies for post–consumer polylactic acid waste: Environmental trade–offs and uncertainty analysis. Chemical Engineering Journal, 522, 167057. https://doi.org/10.1016/j.cej.2025.167057

Jensen, D., Clover, J., Lonergan, S., Levy, M., Bowling, B., Bromwich, B., Halle, S., Barefoot, N., Hamro-Drotz, D., Mourad, B., & Sexton, R. (2012). Renewable Resources and Conflict. UN Interagency Framework Team for Preventive Action. https://www.un.org/en/land-natural-resources-conflict/renewable-resources.shtml

Johansson, N., Krook, J., Eklund, M., & Berglund, B. (2013). An integrated review of concepts and initiatives for mining the technosphere: Towards a new taxonomy. Journal of Cleaner Production, 55, 35–44. https://doi.org/10.1016/j.jclepro.2012.04.007

Johansson, N., Velis, C., & Corvellec, H. (2020). Towards clean material cycles: Is there a policy conflict between circular economy and non-toxic environment? Waste Management & Research: The Journal for a Sustainable Circular Economy, 38(7), 705–707. https://doi.org/10.1177/0734242X20934251

Jones, P. T., Geysen, D., Tielemans, Y., Van Passel, S., Pontikes, Y., Blanpain, B., Quaghebeur, M., & Hoekstra, N. (2013). Enhanced Landfill Mining in view of multiple resource recovery: A critical review. Journal of Cleaner Production, 55, 45–55. https://doi.org/10.1016/j.jclepro.2012.05.021

Kahhat, R. F., & Kavazanjian, E. (2010). Preliminary feasibility study on the use of mono-disposal landfills for e-waste as temporary storage for future mining: 2010 IEEE International Symposium on Sustainable Systems and Technology, ISSST 2010. Proceedings of the 2010 IEEE International Symposium on Sustainable Systems and Technology, ISSST 2010. https://doi.org/10.1109/ISSST.2010.5507740

Kaza, S., Yao, L. C., Bhada-Tata, P., & Van Woerden, F. (2018). What a Waste 2.0: A Global Snapshot of Solid Waste Management to 2050. Washington, DC: World Bank. https://doi.org/10.1596/978-1-4648-1329-0

Kral, U., Kellner, K., & Brunner, P. H. (2013). Sustainable resource use requires "clean cycles" and safe "final sinks." Science of The Total Environment, 461–462, 819–822. https://doi.org/10.1016/j.scitotenv.2012.08.094

Kral, U., Morf, L. S., Vyzinkarova, D., & Brunner, P. H. (2019). Cycles and sinks: Two key elements of a circular economy. Journal of Material Cycles and Waste Management, 21(1), 1–9. https://doi.org/10.1007/s10163-018-0786-6

Krook, J., Svensson, N., & Eklund, M. (2012). Landfill mining: A critical review of two decades of research. Waste Management, 32(3), 513–520. https://doi.org/10.1016/j.wasman.2011.10.015

Kusenberg, M., Eschenbacher, A., Djokic, M. R., Zayoud, A., Ragaert, K., De Meester, S., & Van Geem, K. M. (2022). Opportunities and challenges for the application of post-consumer plastic waste pyrolysis oils as steam cracker feedstocks: To decontaminate or not to decontaminate? Waste Management, 138, 83–115. https://doi.org/10.1016/j.wasman.2021.11.009

Lahl, U., & Zeschmar-Lahl, B. (2024). More than 30 Years of PVC Recycling in Europe—A Critical Inventory. Sustainability, 16(9), Article 9. https://doi.org/10.3390/su16093854

Lamers, F. (2015). Treatment of Bottom Ashes of Waste-to-Energy Installations – State of the Art. TK Verlag - Fachverlag Für Kreislaufwirtschaft. https://ask-eu.de/Artikel/27666/Treatment-of-Bottom-Ashes-of-Waste-to-Energy-Installations-State-of-the-Art-.htm

Larrain, M., Van Passel, S., Thomassen, G., Van Gorp, B., Nhu, T. T., Huysveld, S., Van Geem, K. M., De Meester, S., & Billen, P. (2021). Techno-economic assessment of mechanical recycling of challenging post-consumer plastic packaging waste. Resources, Conservation and Recycling, 170, 105607. https://doi.org/10.1016/j.resconrec.2021.105607

Lehmphul, K. (2024). Landfill. Umweltbundesamt; Umweltbundesamt. https://www.umweltbundesamt.de/en/topics/waste-resources/waste-disposal/landfill

Lichtenegger, G., Rentizelas, A. A., Trivyza, N., & Siegl, S. (2020). Offshore and onshore wind turbine blade waste material forecast at a regional level in Europe until 2050. Waste Management, 106, 120–131. https://doi.org/10.1016/j.wasman.2020.03.018

Lim, J., Ahn, Y., & Kim, J. (2023). Optimal sorting and recycling of plastic waste as a renewable energy resource considering economic feasibility and environmental pollution. Process Safety and Environmental Protection, 169, 685–696. https://doi.org/10.1016/j.psep.2022.11.027

Limburg, M., & Quicker, P. (2016). Entsorgung von Carbonfasern – Probleme des Recyclings und Auswirkungen auf die Abfallverbrennung – [Print, online]. Berliner Abfallwirtschafts- und Energiekonferenz 2016, Nietwerder. TK-Verlag.

Liu, Y., Mendoza-Perilla, P., Clavier, K. A., Tolaymat, T. M., Bowden, J. A., Solo-Gabriele, H. M., & Townsend, T. G. (2022). Municipal solid waste incineration (MSWI) ash co-disposal: Influence on per- and polyfluoroalkyl substances (PFAS) concentration in landfill leachate. Waste Management, 144, 49–56. https://doi.org/10.1016/j.wasman.2022.03.009

Lontoc, G. D., Diola, Ma. B. L. D., & Peralta, M. H. T. (2023). Multi-criteria evaluation of suitable locations for temporary disaster waste storage sites: The case of Cavite, Philippines. Journal of Material Cycles and Waste Management, 25(5), 2794–2808. https://doi.org/10.1007/s10163-023-01705-9

Margallo, M., Taddei, M. B. M., Hernández-Pellón, A., Aldaco, R., & Irabien, Á. (2015). Environmental sustainability assessment of the management of municipal solid waste incineration residues: A review of the current situation. Clean Technologies and Environmental Policy, 17(5), 1333–1353. https://doi.org/10.1007/s10098-015-0961-6

Miliute-Plepiene, J., Fråne, A., & Almasi, A. M. (2021). Overview of polyvinyl chloride (PVC) waste management practices in the Nordic countries. Cleaner Engineering and Technology, 4, 100246. https://doi.org/10.1016/j.clet.2021.100246

Mofokeng, N. N., Madikizela, L. M., Tiggelman, I., & Chimuka, L. (2024). Chemical profiling of paper recycling grades using GC-MS and LC-MS: An exploration of contaminants and their possible sources. Waste Management, 189, 148–158. https://doi.org/10.1016/j.wasman.2024.08.014

Montag, D., Everding, W., Malms, S., Pinnekamp, J., Reinhardt, J., Fehrenbach, H., Arnold, U., Trimborn, M., Goldbach, H., Klett, W., & Lammers, T. (2015). Bewertung konkreter Maßnahmen einer weitergehenden Phosphorrückgewinnung aus relevanten Stoffströmen sowie zum effizienten Phosphoreinsatz. Umweltbundesamt. https://www.umweltbundesamt.de/publikationen/bewertung-konkreter-massnahmen-einer-weitergehenden

Mostbauer, P., Lenz, S., & Lechner, P. (2008). MSWI Bottom Ash for Upgrading of Biogas and Landfill Gas. Environmental Technology, 29(7), 757–764. https://doi.org/10.1080/09593330801987061

Muntenita, C., Titire, L., Chivu, M., Podaru, G., & Marin, R. (2024). Wind Turbine Blade Material Behavior in Abrasive Wear Conditions. Polymers, 16(24), 3483. https://doi.org/10.3390/polym16243483

Nagle, A. J., Mullally, G., Leahy, P. G., & Dunphy, N. P. (2022). Life cycle assessment of the use of decommissioned wind blades in second life applications. Journal of Environmental Management, 302, 113994. https://doi.org/10.1016/j.jenvman.2021.113994

National Grid. (2023). Can wind turbine blades be recycled? | What happens to old wind turbine blades? | National Grid Group. https://www.nationalgrid.com/stories/energy-explained/can-wind-turbine-blades-be-recycled

Nunna, S., Blanchard, P., Buckmaster, D., Davis, S., & Naebe, M. (2019). Development of a cost model for the production of carbon fibres. Heliyon, 5, e02698. https://doi.org/10.1016/j.heliyon.2019.e02698

Obernberger, I., & Supancic, K. (2024). FACT-SHEET: Zwischenlagerung von Pflanzenaschen. https://www.forstholzpapier.at/images/FHP-Arbeitskreise_/AK_Energie/FACTSHEET_Forststra%C3%9Fenbau.pdf

Oehmig, W. N., Roessler, J. G., Blaisi, N. I., & Townsend, T. G. (2015). Contemporary practices and findings essential to the development of effective MSWI ash reuse policy in the United States. Environmental Science & Policy, 51, 304–312. https://doi.org/10.1016/j.envsci.2015.04.024

Parrodi, J. C. H., Vollprecht, D., & Pomberger, R. (2020). Case study on enhanced landfill mining at Mont-Saint-Guibert landfill in Belgium: Physico-chemical characterization and valorization potential of combustibles and inert fractions recovered from fine fractions. Detritus, 10, 44. https://doi.org/10.31025/2611-4135/2020.13941

Pivnenko, K. (2016). Waste material recycling: Assessment of contaminants limiting recycling. https://doi.org/10.13140/RG.2.1.2202.0722

Plastics Europe. (2021). History of plastics. Plastics Europe. https://plasticseurope.org/plastics-explained/history-of-plastics/

Pomberger, R., & Bezama, A. (2024). About theoretical, technical and real recyclability. Waste Management & Research, 42(9), 713–714. https://doi.org/10.1177/0734242X241267184

Pomberger, R., & Ragossnig, A. (2014). Future waste – waste future. Waste Management & Research, 32(2), 89–90. https://doi.org/10.1177/0734242X14521344

RecyClass. (2024). RecyClass Recyclability Methodology. https://recyclass.eu/wp-content/uploads/2024/03/RECYCLASS-RECYCLABILITY-METHODOLOGY_v.2.3.pdf

Reig, M., Vecino, X., Valderrama, C., Sirés, I., & Luis Cortina, J. (2023). Waste-to-energy bottom ash management: Copper recovery by electrowinning. Separation and Purification Technology, 311, 123256. https://doi.org/10.1016/j.seppur.2023.123256

Roessler, J. G., Townsend, T. G., & Kanneganti, A. (2017). Waste to energy ash monofill mining: An environmental characterization of recovered material. Journal of Hazardous Materials, 328, 63–69. https://doi.org/10.1016/j.jhazmat.2017.01.011

Roetman, E., Joustra, J., Heideman, G., & Balkenende, R. (2024). Does the Rubber Meet the Road? Assessing the Potential of Devulcanization Technologies for the Innovation of Tire Rubber Recycling. Sustainability, 16(7), Article 7. https://doi.org/10.3390/su16072900

Romaszewski, T., & Fitas, J. (2024). Properties of RDF after Prolonged Storage. Sustainability, 16(5), Article 5. https://doi.org/10.3390/su16052051

Rybicka, J., Tiwari, A., & Leeke, G. A. (2016). Technology readiness level assessment of composites recycling technologies. Journal of Cleaner Production, 112, 1001–1012. https://doi.org/10.1016/j.jclepro.2015.08.104

Salas, A., Berrio, M. E., Martel, S., Díaz-Gómez, A., Palacio, D. A., Tuninetti, V., Medina, C., & Meléndrez, M. F. (2023). Towards recycling of waste carbon fiber: Strength, morphology and structural features of recovered carbon fibers. Waste Management, 165, 59–69. https://doi.org/10.1016/j.wasman.2023.04.017

Sapsford, D. J., Stewart, D. I., Sinnett, D. E., Burke, I. T., Cleall, P. J., Harbottle, M. J., Mayes, W., Owen, N. E., Sardo, A. M., & Weightman, A. (2023). Circular economy landfills for temporary storage and treatment of mineral-rich wastes. Proceedings of the Institution of Civil Engineers - Waste and Resource Management. https://doi.org/10.1680/jwarm.22.00008

Schinnerl, F., Sattler, T., Noori-Khadjavi, G., & Lehner, M. (2024). Direct aqueous mineral carbonation of secondary materials for carbon dioxide storage. Journal of CO2 Utilization, 88, 102942. https://doi.org/10.1016/j.jcou.2024.102942

Schmidt, B. (2024, 11). Klimastrategie 2040. VOEB - Future Waste, Wien.

Sommer, V., Stockschläder, J., & Walther, G. (2020). Estimation of glass and carbon fiber reinforced plastic waste from end-of-life rotor blades of wind power plants within the European Union. Waste Management, 115, 83–94. https://doi.org/10.1016/j.wasman.2020.06.043

Spini, F., & Bettini, P. (2024). End-of-Life wind turbine blades: Review on recycling strategies. Composites Part B: Engineering, 275, 111290. https://doi.org/10.1016/j.compositesb.2024.111290

Stanisavljevic, N., & Brunner, P. H. (2021). Megacities need both: Circular economy and final sinks! Waste Management & Research, 39(12), 1437–1439.

Stief, K. (1989). Strategy in landfilling solid wastes. In P. Baccini (Ed.), The Landfill (pp. 275–291). Springer. https://doi.org/10.1007/BFb0011268

Stipanovic, H., Bäck, T., & Tischberger-Aldrian, A. (2023). Characterisation of post-consumer textiles using near-infrared spectrometers: NIR 2023.

The Environmental Research & Education Foundation. (2025). Extended Producer Responsibility Literature Review. www.erefdn.org

Torraco, R. J. (2016). Writing Integrative Literature Reviews: Using the Past and Present to Explore the Future. Human Resource Development Review, 15(4), 404–428. https://doi.org/10.1177/1534484316671606

Tortorici, D., Adriani, A., & Laurenzi, S. (2025). Development of a sustainable chemical recycling process of carbon fibers from epoxy-based composites. Composites Science and Technology, 270, 111295. https://doi.org/10.1016/j.compscitech.2025.111295

Umweltbundesamt. (2024). Ewigkeitschemikalien PFAS in Abfällen. Umweltbundesamt; Umweltbundesamt. https://www.umweltbundesamt.de/themen/ewigkeitschemikalien-pfas-in-abfaellen

van Nielen, S. S., Kleijn, R., Sprecher, B., Miranda Xicotencatl, B., & Tukker, A. (2022). Early-stage assessment of minor metal recyclability. Resources, Conservation and Recycling, 176, 105881. https://doi.org/10.1016/j.resconrec.2021.105881

Vermeulen, W., Reike, D., & Witjes, S. (2019). Circular Economy 3.0—Solving confusion around new conceptions of circularity by synthesising and re-organising the 3R's concept into a 10R hierarchy. 2019. https://www.researchgate.net/publication/335602859_Circular_Economy_30_-_Solving_confusion_around_new_conceptions_of_circularity_by_synthesising_and_re-organising_the_3R's_concept_into_a_10R_hierarchy

Volk, R., Stallkamp, C., Herbst, M., & Schultmann, F. (2021). Regional rotor blade waste quantification in Germany until 2040. Resources, Conservation and Recycling, 172, 105667. https://doi.org/10.1016/j.resconrec.2021.105667

Vyzinkarova, D., & Brunner, P. H. (2013). Substance Flow Analysis of Wastes Containing Polybrominated Diphenyl Ethers. Journal of Industrial Ecology, 17(6), 900–911. https://doi.org/10.1111/jiec.12054

Werner, M., Narayanan, A., Rabenschlag, O., Nagarajan, P., Saboo, R., Banatao, R., Moedritzer, S., Zilnik, D., Wong, T., & Meza, M. (2022). Closing the Plastics Circularity Gap—Full Report. Google. https://bbia.org.uk/wp-content/uploads/2022/04/closing-plastics-gap-full-report.pdf

Wilson, G. T., Smalley, G., Suckling, J. R., Lilley, D., Lee, J., & Mawle, R. (2017). The hibernating mobile phone: Dead storage as a barrier to efficient electronic waste recovery. Waste Management, 60, 521–533. https://doi.org/10.1016/j.wasman.2016.12.023

WindEurope. (2020). Accelerating Wind Turbine Blade Circularity. WindEurope. https://windeurope.org/intelligence-platform/product/accelerating-wind-turbine-blade-circularity/

WindEurope. (2021). How to build a circular economy for wind turbine blades through policy and partnerships. WindEurope. https://windeurope.org/policy/position-papers/how-to-build-a-circular-economy-for-wind-turbine-blades-through-policy-and-partnerships/

WindEurope. (2023). Wind energy in Europe: 2022 Statistics and the outlook for 2023-2027. WindEurope. https://windeurope.org/intelligence-platform/product/wind-energy-in-europe-2022-statistics-and-the-outlook-for-2023-2027/

Wu, Q., Zhang, Q., Chen, X., Song, G., & Xiao, J. (2024). Life cycle assessment of waste tire recycling: Upgraded pyrolytic products for new tire production. Sustainable Production and Consumption, 46, 294–309. https://doi.org/10.1016/j.spc.2024.02.029

Xie, S., Lim, Y. T., Wang, H., Yi, W., & Antwi-Afari, M. F. (2024). Location and Capacity Optimization of Waste Recycling Centers: Mathematical Models and Numerical Experiments. Applied Sciences, 14(16), Article 16. https://doi.org/10.3390/app14167039

Yang, X., & Berglund, L. A. (2020). Recycling without Fiber Degradation—Strong Paper Structures for 3D Forming Based on Nanostructurally Tailored Wood Holocellulose Fibers. ACS Sustainable Chemistry & Engineering, 8(2), 1146–1154. https://doi.org/10.1021/acssuschemeng.9b06176

Zero Waste Europe. (2022). Incineration Residues: The dust under the carpet - Zero Waste Europe. Zero Waste Europe. https://zerowasteeurope.eu/press-release/incineration-residues-the-dust-under-the-carpet/

Zhong, H.-N., Su, Q.-Z., Chen, S., Li, D., Sui, H., Zhu, L., Dong, B., Wu, S., & Wang, X. (2025). Revealing contaminants in China's recycled PET: Enabling safe food contact applications. Resources, Conservation and Recycling, 212, 107947. https://doi.org/10.1016/j.resconrec.2024.107947

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FROM RECLAIMER INTEGRATION TO GETTING INTEGRATED INTO THEIR RECYCLINGSCAPE: THREE STORIES FROM BRAZIL

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ABSTRACT

Catadores in Brazil, reclaimers in South Africa, diverters in Canada, recyclers (recicladores) in other South American countries, waste pickers in India, and all over the world. Who are informal workers acting at the core of the circular economy? Identifying and acknowledging their role in the economic, social, and political system is not merely a semantic exercise but a critical intervention in transforming waste governance, changing our understanding of waste and waste production, and doing justice to a population whose rights have historically been denied. Focusing on Brazil, where catadores are formally recognized as a professional category by law and in official job classifications and statistics, this study supports recognizing catadores' work as political work and the need to overcome a charity-driven, paternalistic perspective that reproduces marginalization rather than supporting integration. A research agenda is put forward to advance the debate on integration and reclaimers from a transformative (i.e., systemic change) perspective.

1. INTRODUCTION

Municipal solid waste (MSW) generation worldwide is projected to rise sharply, as major producers like the US fail to implement meaningful changes in consumption patterns, while lower and middle-income countries increase both production and consumption of goods (UNEP, 2024). Even more concerning is the projected growth of uncontrolled waste (UNEP, 2024), the fraction of discarded materials that escape formal collection and proper disposal, ending up in the environment or managed inappropriately, e.g., being burned. However, the boundaries between formal and informal waste management are often blurred, and "formal infrastructures", especially incineration, do not necessarily reduce socio-environmental harms. Practices such as excavating, collecting, sorting, and reusing discarded materials have existed since ancient times (Freestone, 2015; Healey, 1978) and remain deeply rooted in community-based organizations in contexts such as Brazil (Maiello, 2022), Colombia (Parra & Abizaid, 2021), and India. The lack of inclusion and recognition of this informal sector is increasingly acknowledged as a major barrier to effective waste management, where 40% of global informal workers operate in the waste sector (UNEP, 2024).

This research investigates municipal integration processes in Brazil and their relationship to the recovery of recyclable materials. The cities of Belo Horizonte, São Paulo, and Volta Redonda were selected for their high income per capita and associated waste generation (Downs & Medina, 2000). Using the extended case method and Samson's (2020) typology, the study's main contribution lies in applying this framework to Brazil's distinctive context of reclaimer institutionalization. Material recovery data were obtained from the Brazilian National Sanitation Information System (SNIS), which was subsequently replaced by the National System of Information on Basic Sanitation (SINISA).

2. WHO ARE RECLAIMERS AND WHO ARE CATADORES?

We use the terms "reclaimers" and "catadores" interchangeably to refer to workers engaged in the manual collection and sorting of recyclable materials in Brazil. Other terms are also used in English-language literature, such as recyclers, collectors, diverters, informal recyclers (Gutberlet & Carenzo, 2020), refuse sorters (Bouvier & Dias, 2021), independent street or dump pickers (Rutkowski & Rutkowski, 2015), and waste pickers. While the latter is the most common term (Ali & Yusuf, 2021; Gutberlet & Carenzo, 2020; Gutberlet, 2023; Gutberlet & de Carvalho Vallin, 2024; Kalina & Tilley, 2020; Rutkowski, 2020; Schamber & Bon, 2022; Yamane et al., 2023), it carries pejorative connotations that associate their work with rubbish and

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limit it to the act of picking (Samson, 2020, p.195), misrecognizing the tacit and experiential knowledge involved in identifying and revaluing different discarded materials (Ibid.) as well as the complex, embodied labor required to navigate scattered spaces where materials are disposed and organized.

Reclaimers are specialized service providers (Gutberlet & Carenzo, 2020; Gutberlet et al., 2017; Dias, 2006) who contribute to the extension of landfill lifespans (Kadyamadare & Samson, 2024) and the reduction of CO_2 equivalent emissions (Chintan, 2013; in Departments of Environment, Forestry and Fisheries & Science and Innovation of South Africa, 2020, p. 21; IPCC, 2023, pp. 103, 105). Despite these contributions, they face social stigma, institutional barriers, precarious work, and low pay in the billionaire waste management industry (UNEP, 2024), with negative physical and mental health impacts (Gutberlet & Carenzo, 2020; Sekhwela & Samson, 2020; Parizeau, 2015) (Figure 1).

In Brazil, catadores are not a homogeneous, monolithic category but a diverse group, ranging from freelance individuals to cooperative members, and from occasional actors supplementing their income to political activists in a national movement (Maiello, 2022), who negotiate concessions, resist exclusion, and construct group identity and dignity in diverse ways. Most operate informally, with women constituting the majority (Bouvier et al., 2022; Marques et al., 2020). Decades of practice, sometimes across generations, have enabled nationwide political organization since 1999 (Maiello, 2022) and inclusion in the National Registry of Professions (CBO) since 2002 (Ibid.), positioning Brazil as a leader in the formal recognition compared with similar economies, which is the result of political mobilization and social struggles of catadores (Ibid.).

3. INTEGRATION THEORY

Integration, for Samson (2020), does not mean opening institutional processes to include reclaimers' informal practices, but rather rethinking the economic and political systems surrounding reclaimers' organizations and the recyclingscape (Ibid.), in which the value of discarded materials and their circular potential are recognized, contrasting with the wastescape. Integration is a contested, power-laden process (Sekhwela & Samson, 2020) that can reinforce or challenge existing power relations, eliciting transformations.

Samson (2020) identifies four dominant conceptualizations: a) charity, b) (tokenistic) participation, c) multifaceted process, and d) social transformation, which form a dynamic continuum rather than isolated siloes, as real-world initiatives rarely fit into a single category. Reflecting on the political work of integration, Samson identifies five forms of neglection, referred to as erasures under the charity model: i) economic role and fair recognition of labor; ii) knowledge and agency (e.g. epistemological misrecognition); iii) recyclingscape (e.g. the informal sociospatial system created by reclaimers whose erasure entails displacement); iv) sphere of accumulation related to rescued materials, livelihood and economic neglect; v) and humanity. This theoretical framework is instrumental for exploring the different levels of integration and, for relevant institutions (e.g., municipalities) and researchers, for understanding the implications of a charity approach (Figure 2).

The Brazilian context is especially relevant for understanding "reclaiming" as political work (Samson, 2020) and catadores not as victims of marginalization, but as actors with agency and power. We applied Samson's typology to map how three Brazilian municipalities, within major industrial regions, relate to local communities of catadores.

FIGURE 1: Skilled catadores, sorting recyclables, enabling the circularity of various materials. Credits: @catakiapp. Photo: Júlia Nagle | @ju.nagle.

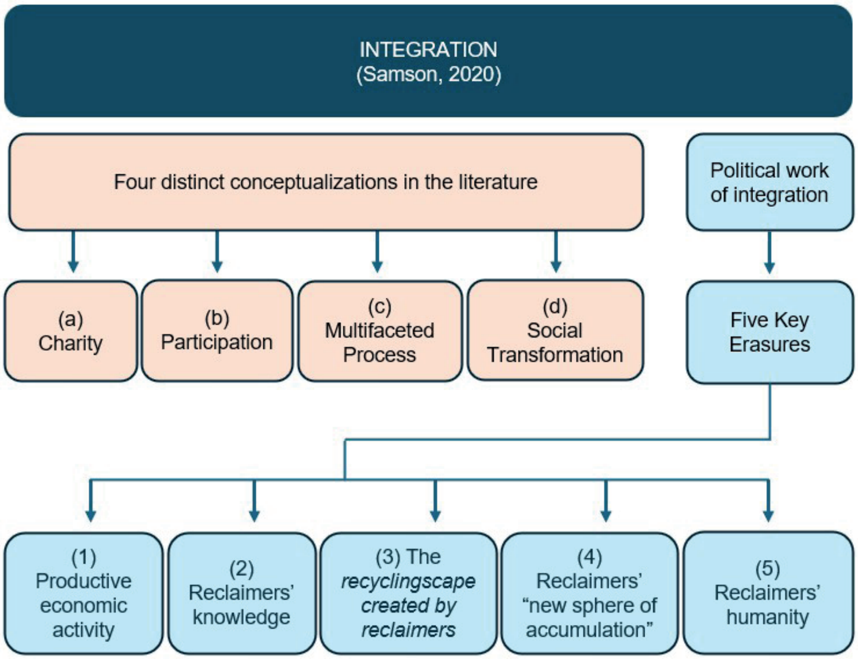

FIGURE 2: Visualization of the theoretical framework.

4. STORIES FROM THREE CITIES

This study adopts an extended case method (Burawoy, 2009) as an overarching framework. A transdisciplinary approach was used for data collection, incorporating insights from scholars, local activists, and grassroots initiatives, including catadores' associations in Rio de Janeiro (Maiello, 2022), virtual engagement with Pimp My Carroça, and an in-depth interview from a previous study with the Volta Redonda Secretariat of Environment.

In total, 1 595 pages from nine policy documents and 11 news articles (2019-2024) were analyzed, capturing legal frameworks and reclaimers' lived experiences. Three municipalities representing core industrialized metropolitan regions, differing in size and political culture, were selected (Table 1).

Our central guiding research question is: to what extent are different approaches to reclaiming integration associated with the performance of recyclable material recovery at the municipal level?

A codebook (Saldaña, 2013) based on Samson's (2020) framework was developed, including a summative reliability check (Drisko & Maschi, 2016). Quantitative data on recyclable recovery were sourced from SNIS database, version 2024.006 (http://app4.mdr.gov.br/serieHistorica/) filtered by category and municipality. From 290 indicators and components (2002-2022), nine key metrics were selected: CO116 (collected by public agent), CO117 (by private agents), CO142 (by other agents), CS009 (amount recovered), CS026 (by selective collection service), CS048 (collected by reclaimers), IN031 (recovery rate), IN032 (per capita recovered) and IN054 (per capita collected), supplemented by data from the Brazilian Institute of Geography and Statistics (IBGE). Data inconsistencies were identified and corrected.

TABLE 1: Profile of the selected municipalities.

Ref.	Indicator	Belo Horizonte, Minas Gerais	São Paulo, São Paulo	Volta Redonda, Rio de Janeiro
1	Urbanized territory (km^2)	274	914	43
2	Population	2 315 560	11 451 999	261 563
3	GDP (R$, billions)	105.8	828.9	19.6
4	PIB per capita (R$)	41 818.32	66 872.84	71 551.44
5	Total waste generated (kg/inhabitant/year)	315.6	359.7	262
6	Recyclables collected per capita (selective collection) (kg/inhabitant/year)	2.73	6.47	4.12
7	Number of cooperatives/ associations	6	26	3
8	Number of reclaimers affiliated with associations	197	1 050	34
9	Estimated number of informal reclaimers	Data not available	Data not available	Data not available

Sources: [1-2] 2022 IBGE; [3-4] 2021 IBGE; [5] SNIS-2023 (CO119); [6] SNIS-2023 (IN054); [7] SNIS-2023 (CA006); [8] SNIS-2023 (CA007); [9] SNIS-2023

5. INTEGRATING THE RECYCLINGSCAPE? WHAT DATA CAN(NOT) TELL US?

Our findings contribute empirical grounding to Samson's conceptualization. From a conceptual perspective, Samson also identifies Brazil as an advanced reality in the political recognition of catadores' work and activism, as evidenced by the notion of reciclagem inclusiva (inclusive recycling) developed by the catadores movement, which resonates with her conceptualization of integration. While, in theory, we agree with the author and acknowledge the political value of the Brazilian recyclingscape, our research reveals gaps that complicate the picture, pointing to avenues for a research agenda. The following presents the results for each of the investigated municipalities.

In Belo Horizonte, the 1999 Municipal Waste & Citizenship Forum laid the foundation for the national reclaimers' movement (Dias, 2006), raising awareness of integrated waste management (Ibid.). The Municipal Waste Management Law No. 10 534/2012 assigns the Urban Cleaning Authority (SLU) the duty to involve catadores' cooperatives and those forming cooperatives in formal collection (Art. 34). However, the law assigns to catadores only a complementary role (Ibid., para. 3), meaning they are not responsible for the service provision. While this is one of the first laws to establish reclaimers' involvement, it reflects a charity-oriented approach (Samson), focusing on their labor rather than their professionalism. Catadores have provided door-to-door sorting since 2019, improving the quality of discarded material and municipal recycling rates (Reclaimers' National Movement, 2019). Despite this, they face institutional neglect, as during the COVID-19 pandemic (Gonçalves et al., 2022), when 309 formal and roughly 3 000 informal catadores' livelihoods were disrupted (Dias, 2020) (Figure 3).

In São Paulo, the country's wealthiest city, urban vulnerabilities are addressed through Law No. 16 050/2014. Unlike Belo Horizonte, this is not a law on waste management, but a broader legislative instrument. On paper, it is a progressive policy, establishing the right to the city as a principle to ensure universalized access to urban services and amenities (para. 5). The law makes a step forward compared to Belo Horizonte by including catadores within cooperatives and as individuals. Moreover, under this law, the city commits to providing space and involvement, and "find a strategy" to ensure contractual stability. However, these are vague commitments, and, as in the case of Belo Horizonte, they address reclaimers as vulnerable people to be involved. We are far from the Samson's perspective on integration as the process of designing policy for the recyclingscape, and still closer to her definition of charity integration. Alongside public institutions, grassroots initiatives such as Pimp My Carroça report that most catadores continue to face discrimination, police violence, low remuneration, and high risk of accidents, with social stigma remaining a persistent barrier (Pimp My Carroça, 2024). Exclusion from decision-making persists (Gringos Podcast, 2024) as observed in the local sorted collection program (Decree No. 48 799/2007, Art. 4) and local initiatives, such as the Open Letter, supported by 36 institutions, denouncing the worsening socioeconomic conditions during the pandemic (Figure 4).

In Volta Redonda, Law No. 5 762/2020, driven by catadores and allies, legally integrates catadores, including those outside cooperatives, into the public sorted collection and decision-making via the deliberative body CACS-VR, where catadores are members. However, implementation challenges include governance tensions due to the hollowing out of public policies (Gripp & Valério, 2023) and warehouse irregularities which fall under the responsibilities of the municipality (Public Labor Prosecution Office in Rio de Janeiro, 2023), and threats to livelihoods from a proposed incineration plant, later halted by judi-

FIGURE 3: Specialized catadores in Belo Horizonte. Credits/Photo: ASMARE @asmaremg.

FIGURE 4: NGO Pimp My Carroça at COP 30, hosting a panel in the Green Zone, bringing the catadores' agenda to the forefront of climate discussions. Credits/Photo: Pimp My Carroça @pimpmycarroca.

cial intervention (Federal Public Defender's Office, 2022) (Figure 5).

Regarding content analysis, it revealed charity was the dominant framing in policy documents, followed by multifaceted process, participation, and social transformation. In gray literature, charity also prevails, followed by social transformation and multifaceted process, while participation is absent (Table 2).

Analysis of material recovery revealed a systematic gap in data reporting for component CS048 across all cities, with missing years such as 7 in Belo Horizonte, 17 in São Paulo, and 16 in Volta Redonda, reflecting the misrecognition of reclaimers' direct contributions and limiting the assessment of their economic and environmental impact. Recyclables recovery per capita was significantly low in all cities, roughly half the national. Municipal recovery rates were similarly low across all cities, well below the national rate. These low figures coincide with the charity-oriented integration approach observed across all cities, suggesting an association with framing catadores primarily as recipi-

FIGURE 5: Skilled catadores in Volta Redonda. Credits/Photo: Folha Verde Cooperative.

ents of support rather than as active participants in waste governance. However, given the missing data and the exploratory nature of this research, based on case studies, these findings should be considered a speculation to be validated in future studies (Table 3).

6. BEYOND CHARITY, TOKENISM, AND PATERNALISM IN RECLAIMERS' INTEGRATION: A RESEARCH AGENDA

Samson's theory enabled a critical view of the concept of "inclusão de catadores" (reclaimers' inclusion) in Brazil's legislative framework, illuminating a charity approach by defining it and unravelling its underpinning structures, namely the five forms of erasure. Each of the five erasures relates to a specific sphere of misrecognition: i) labor and skills; ii) epistemological; iii) spatial; iv) economic; v) human and moral.

Despite some progress in social inclusion through associational practices and formal policies, our analysis shows a prevailing charity approach to reclaimers' integration, raising questions about the implementation of many of the political achievements of catadores' social and political movements. Problematic structural discontinuities and gaps in data production limit local public decision-making (Maiello et al., 2015) and affect social inclusion. Samson's transformative view of integration, as changing the system around catadores' recyclingscape rather than involving them in the formal wastescape, helped us identify structural paternalistic (i.e., charity-like) patterns and imagine integration beyond tokenistic approaches. Integration is not a neutral policy goal, or a process of educating catadores to integrate them (Gutberlet & de Carvalho Vallin, 2024), but a discursive, contested process and political tool, exposing the hiatus between discourse, both in the policy and in the academic literature, and the practice in waste governance.

We acknowledge that the research has limitations. Its qualitative nature introduces multiple selection biases, from case identification to single-coder document interpretation. Remote data collection of empirical materials limited its breadth, as only half of municipalities have a waste management plan (Ministry of Cities, 2023, p.24). Incomplete national datasets, transitioning from SNIS to SINISA, and limited direct interaction with reclaimers constrained our ability to fully answer the research questions.

Building on this acknowledgement and aiming to advance Samson's transformative perspective on integration as systemic change, we propose a three-part research agenda using the five Samson erasures as entry points. First, we see multi-sited ethnographic research (Marcus, 1995) as the most suitable approach to explore both labor/skills and spatial erasures, allowing to observe, from multiple spatial foci and multiple scales (from the reclaimer's bodies to the region of waste transit, crossing through the organizational sites) the recyclingscape, its everyday functioning and the tensions with the formal waste management system. The ethnographic approach allows the building of necessary relations, breaking unintentional hierarchies and challenging power structures, all of which

TABLE 2: Results of the content analysis in descending order by code.

Codes	BH		SP		VR		Total Count	
	P	GL	P	GL	P	GL	P	GL
c assistance to work/ passive targets	22	9	43	8	27	6	92	23
m labor requires S-C-P-L-E actions	14	1	38	2	29	0	81	3
c exclusion of reclaimers' recycling system	8	3	4	6	5	4	17	13
t new understanding of the world & role	1	12	8	5	2	0	11	17
t social justice to forge citizenship	2	1	8	2	2	0	12	3
p attention to key aspects for success	4	0	6	0	4	0	14	0
p participatory approaches to work	4	0	5	0	1	0	10	0
m sociocultural actions for unhindered work	0	1	2	0	0	0	2	1

Legend: BH=Belo Horizonte; SP=São Paulo; VR=Volta Redonda; P=Policy; GL=Gray Literature

TABLE 3: Recovery of recyclables and the integration approach among municipalities.

Ref.	Data	BH	SP	VR
1	Recyclables recovery per capita (kg/inhabitant/year)	2.4	2.8	2.1
2	National estimate of recyclables recovery per capita (kg/ inhabitant/year)		5.5	
3	Municipal recovery rate of recyclable materials (%)	0.76	0.77	0.79
4	National recovered rate of recyclable material (%)		2.37	
5	Integration (mostly)	Charity-driven	Charity-driven	Charity-driven

Sources/ Legend: [1] SNIS, version 2024.006 (IN032); [2] Brazil recovered 1.12 million tons of dry recyclable waste from all 5 060 municipalities in 2022 (Ministry of Cities – Brazil, 2023, p.116), or 5.52 kg/ inhabitant/year per capita, considering a population of 203.1 million in that year (IBGE Census 2022); [3] SNIS, version 2024.006 (IN031); [4] Ministry of Cities – Brazil, 2023, p.120; [5] Results of content analysis of this present study based on the Samson's (2020) framework.

are constantly involved in the research process. Second, acknowledging epistemological injustices inherent in capitalist society and even in participatory research (Kimura & Kinchy, 2016), we recommend a feminist-inspired citizen science (CS) approach (Engelmann et al., 2022) to address the data gap identified here. This entails a new way of engaging with CS, coupled with multi-sited and extended ethnography (Burawoy, 2009), addressing both epistemological and human erasures, by involving multiple actors in the waste management process, both users and service providers, such as reclaimers and municipal utility operators, to create connections rather than reproducing hierarchies and siloed social spaces. We understand a feminist approach to citizen science as an approach informed by an ethics of care (in this case, for knowledge and common data collection) and shared responsibilities in the public service (Pioggia, 2024). Finally, to address the economic erasure, we recommend adopting a diverse economy (Gibson-Graham, 2008) perspective as an overarching theoretical framework that allows us to recognize the economic power of the recyclingscape, its transformative potential, and stimulate the imagination that is necessary to make new, inclusive, and circular alternatives possible (Roelvink et al., 2015).

REFERENCES

Ali, A. F., & Yusuf, F. I. (2021). Prevalence of injuries among waste pickers. A case study in Nigeria. Detritus, 17, 89–96. https://doi.org/10.31025/2611-4135/2021.15144

Azevedo, A. M. M. D., Gutberlet, J., Araújo, S. D. D., & Duarte, F. H. (2022). Impactos da Covid-19 sobre catadores de materiais recicláveis organizados no Estado de São Paulo [Impacts of COVID-19 on Organized Recyclable Waste Pickers in the State of São Paulo]. Ambiente & Sociedade, 25, e00881. https://doi.org/10.1590/1809-4422asoc20210088r1vu2022l2ao

Bouvier, M., & Dias, S. M. (2021). Waste Pickers in Brazil: A Statistical Profile (No. 29; WIEGO Statistical Briefs). Women in Informal Employment: Globalizing and Organizing (WIEGO). https://www.wiego.org/wp-content/uploads/2021/12/WIEGO_Statistical_Brief_N29_Brazil_WPs.pdf

Bouvier, M., Vanek, J., & Roubaud, F. (2022). Informal Workers in Brazil: A Statistical Profile (No. 33; WIEGO Statistical Briefs). Women in Informal Employment: Globalizing and Organizing (WIEGO). https://www.wiego.org/wp-content/uploads/2022/09/WIEGO_Statistical_Brief_N33_IWs_Brazil.pdf

Burawoy, M. (2009). The extended case method: Four countries, four decades, four great transformations, and one theoretical tradition. University of California Press. https://doi.org/10.1525/9780520943384

Department of Environment, Forestry and Fisheries & Department of Science and Innovation. (2020). Waste picker integration guideline for South Africa: Building the Recycling Economy and Improving Livelihoods through Integration of the Informal Sector. DEFF and DST: Pretoria. https://www.gtac.gov.za/resource/waste-picker-integration-guideline-for-south-africa/

Dias, R. (2020, March 30). Coronavírus: Catadores de recicláveis vivem drama em BH [Coronavirus: Recycling collectors experience drama in BH]. Estado de Minas. https://www.em.com.br/app/noticia/gerais/2020/03/30/interna_gerais,1133935/coronavirus-catadores-de-reciclaveis-vivem-drama-em-bh.shtml

Dias, S. M. (2006). Waste & Citizenship Forum – Achievements and limitations (Paper No. 11). Solid waste, health and the Millennium Development Goals, Kolkata, India. https://www.wiego.org/wp-content/uploads/2019/09/Dias_WASTE_AND_CITIZENSHIP_FORUMS.pdf

Dias, S. M. (2011). Recycling in Belo Horizonte, Brazil – An Overview of Inclusive Programming (No. 3; WIEGO Policy Brief - Urban Policies). Women in Informal Employment: Globalizing and Organizing (WIEGO). https://www.wiego.org/wp-content/uploads/2019/09/Dias_WIEGO_PB3.pdf

Dias, S. M. (2016). Waste pickers and cities. Environment and Urbanization, 28(2), 375–390. https://doi.org/10.1177/0956247816657302

Downs, M., & Medina, M. (2000). A Short History of Scavenging. Comparative Civilizations Review, 42(42), 4. https://scholarsarchive.byu.edu/ccr/vol42/iss42/4

Drisko, J., & Maschi, T. (2015). Content analysis (1st ed.). Oxford University Press, New York. https://doi.org/10.1093/acprof:oso/9780190215491.001.0001

Engelmann, S., Dyer, S., Malcolm, L., & Powers, D. (2022). Open-weather: Speculative-feminist propositions for planetary images in an era of climate crisis. Geoforum, 137, 237–247. https://doi.org/10.1016/j.geoforum.2022.09.004

Estúdio Folha. (2023, August). Parceria melhora renda para catadores de material reciclável em São Paulo [Partnership improves income for reclaimers in São Paulo]. https://estudio.folha.uol.com.br/prefeitura-de-saopaulo/2023/08/parceria-melhora-renda-para-catadores-de-material-reciclavel-em-sao-paulo.shtml

Federal Public Defender's Office. (2022). Audiência Pública debate situação de catadores em Volta Redonda/RJ [Public hearing debates the situation of reclaimers in Volta Redonda/RJ]. https://www.dpu.def.br/noticias-rio-de-janeiro/67821-audiencia-publica-debate-situacao-de-catadores-em-volta-redonda-rj

Felicio, A. B., Dias, G., & Giancola, C. (2024, January). Aniversário de SP: iniciativa promove melhorias em condições de trabalho de catadores de materiais recicláveis [SP's anniversary: initiative promotes improvements in working conditions of reclaimers]. TV Globo and G1 SP. https://g1.globo.com/sp/sao-paulo/noticia/2024/01/23/aniversario-de-sp-iniciativa-promove-melhorias-em-condicoes-de-trabalho-de-catadores-de-materiais-reciclaveis.ghtml

Freestone, I. C. (2015). The Recycling and Reuse of Roman Glass: Analytical Approaches. Journal of Glass Studies, 57, 29–40. http://www.jstor.org/stable/24726946

Gibson-Graham, J. K. (2008). Diverse economies: performative practices for `other worlds'. Progress in Human Geography, 32(5), 613–632. https://doi.org/10.1177/0309132508090821

Gonçalves, J. T., Lima, F., Lana, B. S., & Manzanares, R. D. (Eds.). (2022, January 24). Situação dos catadores de materiais recicláveis em Belo Horizonte (MG) durante a pandemia de COVID-19 [Situation of reclaimers in Belo Horizonte (MG) during the COVID-19 pandemic]. Even3. https://www.even3.com.br/anais/x_enanppas2021/395465-situacao-dos-catadores-de-materiais-reciclaveis-em-belo-horizonte-(mg)-durante-a-pandemia-de-covid-19/

Gringos Podcast. (2024, January 29). Mundano (grafiteiro, artivista) – Gringos Podcast #298 [Mundano (graffiti artist, artivist) – Gringos Podcast #298]. [Vídeo]. YouTube. https://www.youtube.com/watch?v=CI1ZKvV97QU

Gripp, D. L. R., & Valério, M. P. (2023, November 20). Articulação das cooperativas de materiais recicláveis de Volta Redonda com as entidades de apoio e de defesa de direitos humanos para garantir a implementação da política municipal de coleta seletiva com priorização dos catadores [Coordination of recyclable materials cooperatives in Volta Redonda with support and human rights organizations to ensure the implementation of the municipal selective collection policy, with prioritization of reclaimers]. Anais do 1º Simpósio brasileiro de ensino, pesquisa e extensão em tecnologia social (SEPETS). https://www.even3.com.br/anais/1_sepets/706404-articulacao-das-cooperativas-de-catadores-de-materiais-reciclaveis-de-volta-redonda-com-as-entid%E2%80%A6/

Gutberlet, J. (2023). Grassroots eco-social innovations driving an inclusive circular economy. Detritus, 22, 3–12. https://doi.org/10.31025/2611-4135/2023.17252

Gutberlet, J., & Carenzo, S. (2020). Waste pickers at the heart of the circular economy: A perspective of inclusive recycling from the global south. Worldwide Waste, 3(1), 6. https://doi.org/10.5334/wwwj.50

Gutberlet, J., Carenzo, S., Kain, J.-H., & Mantovani Martiniano De Azevedo, A. (2017). Waste picker organizations and their contribution to the circular economy: Two case studies from a global south perspective. Resources, 6(4), 52. https://doi.org/10.3390/resources6040052

Gutberlet, J., & De Carvalho Vallin, I. (2024). Waste picker rights and social inclusion: The creation of a university with knowledge democracy. Detritus, 26, 114–122. https://doi.org/10.31025/2611-4135/2024.18358

Gutberlet, J., Besen, G.R., & Morais, L. (2020). Participatory solid waste governance and the role of social and solidarity economy: Experiences from São Paulo, Brazil. Detritus, 13, 167–180. https://doi.org/10.31025/2611-4135/2020.14024

Healy, J. F. (1978). Mining and Metallurgy in the Greek and Roman World (316 p.). London: Thames and Hudson.
Kadyamadare, G., & Samson, M. (2024). Thinking households—How resident conceptualizations of waste, reclaimers and separation at source shape recycling practices. Urban Forum, 35(1), 25–46. https://doi.org/10.1007/s12132-023-09497-9
Kalina, M., & Tilley, E. (2020). 'Bad' trash: Problematizing waste in Blantyre, Malawi. Detritus, 12, 187–200. https://doi.org/10.31025/2611-4135/2020.13994
Kimura, A. H., & Kinchy, A. (2016). Citizen science: Probing the virtues and contexts of participatory research. Engaging Science, Technology, and Society, 2, 331–361. https://doi.org/10.17351/ests2016.99
Maiello, A., De Paiva Britto, A. L. N., Mello, Y. R., & De Oliveira Barbosa, P. S. (2015). (Un)Used and (Un)usable? The role of indicators in local decision-making. A Brazilian case study. Futures, 74, 80–92. https://doi.org/10.1016/j.futures.2014.11.002
Maiello, A., Britto, A. L. N. D. P., & Valle, T. F. (2018). Implementação da Política Nacional de Resíduos Sólidos [Implementation of the National Solid Waste Policy]. Revista de Administração Pública, 52(1), 24–51. https://doi.org/10.1590/0034-7612155117
Maiello, A. (2022). In, out, or beyond? Waste pickers and policy networks: a story from Jardim Gramacho (Rio de Janeiro). Sustainability, 14(24), 16977. https://doi.org/10.3390/su142416977
Mallmann, D. (2024, February). 30 toneladas de materiais recicláveis são coletadas durante o Carnaval de BH [30 tons of recyclable materials are collected during the BH Carnival]. CNN Belo Horizonte. https://www.cnnbrasil.com.br/nacional/30-toneladas-de-materiais-reciclaveis-sao-coletadas-durante-o-carnaval-de-bh/
Marcus, G. E. (1995). Ethnography in/of the World System: The Emergence of Multi-Sited Ethnography. Annual Review of Anthropology, 24, 95–117. https://doi.org/10.1146/annurev.an.24.100195.000523
Marques, C. P., Zolnikov, T. R., Noronha, J. M. de, Angulo-Tuesta, A., Bashashi, M., & Cruvinel, V. R. N. (2021). Social vulnerabilities of female waste pickers in Brasília, Brazil. Archives of Environmental & Occupational Health, 76(3), 173–180. https://doi.org/10.1080/19338244.2020.1787315
Medina, M. (2008, October). The informal recycling sector in developing countries: Organizing waste pickers to enhance their impact. GRIDLINES, No. 44. https://documents1.worldbank.org/curated/en/227581468156575228/pdf/472210BRI0Box31ing1sectors01PUBLIC1.pdf
Ministry of Cities. (2023). Diagnóstico Temático: Manejo de Resíduos Sólidos Urbanos – Visão Geral [Thematic Diagnosis: Management of Urban Solid Waste - Overview]. Secretariat for Environmental Sanitation, National Sanitation Information System (SNIS). https://antigo.mdr.gov.br/images/stories/ArquivosSNSA/Arquivos_PDF/Snis/RESIDUOS_SOLIDOS/DIAGNOSTICO_TEMATICO_VISAO_GERAL_RS_SNIS_2023_ATUALIZADO.pdf
Ministry of the Environment. (2022). Plano Nacional de Resíduos Sólidos - Planares [National Solid Waste Plan – Planares]. https://www.gov.br/mma/pt-br/acesso-a-informacao/acoes-e-programas/programa-projetos-acoes-obras-atividades/agendaambientalurbana/lixao-zero/plano_nacional_de_residuos_solidos-1.pdf
Municipality of Belo Horizonte. (2017). Plano Municipal de Gestão Integrada de Resíduos Sólidos de Belo Horizonte (PMGIRS-BH) [Municipal Integrated Solid Waste Management Plan of Belo Horizonte (PMGIRS-BH)]. https://prefeitura.pbh.gov.br/sites/default/files/estrutura-de-governo/slu/2018/documentos/versao_final_pmgirs-bh_mma.pdf
Municipality of Belo Horizonte. (2021). Isopor passa a ser aceito na coleta seletiva de Belo Horizonte [Styrofoam is now accepted in the selective collection in Belo Horizonte]. https://prefeitura.pbh.gov.br/noticias/isopor-passa-ser-aceito-na-coleta-seletiva-de-belo-horizonte-0?__goc_wbp__=238483002W-BoT9z9-Ej17-SUaVm7zBHPrs10
Municipality of São Paulo. (2014). Comitê Intersecretarial para a Política Municipal de Resíduos Sólidos – Plano de Gestão Integrada de Resíduos Sólidos da Cidade de São Paulo [Intersecretarial Committee for Municipal Solid Waste Policy: Integrated Solid Waste Management Plan of São Paulo]. https://www.prefeitura.sp.gov.br/cidade/secretarias/spregula/residuos_solidos/noticias/?p=169047
Municipality of Volta Redonda. (2015). Documento 3: Plano Municipal de Gestão Integrada de Resíduos Sólidos – PMGIRS [Document 3: Municipal Plan for Integrated Solid Waste Management – PMGIRS]. https://www2.voltaredonda.rj.gov.br/projetos/saneamento/mod/consulta_publica_2015/pdf/DOC3_PMGIRS.pdf
Parizeau, K. (2015). Urban political ecologies of informal recyclers health in Buenos Aires, Argentina. Health & Place, 33, 67–74. https://doi.org/10.1016/j.healthplace.2015.02.007
Parra, F., & Abizaid, O. (2021). La formalización de la población recicladora en Colombia como prestadora del servicio público de reciclaje. Logros, oportunidades, restricciones y amenazas [The formalization of the recycling population in Colombia as providers of the public recycling service: achievements, opportunities, constraints, and threats]. (No. 12; WIEGO Technical note). Women in Informal Employment: Globalizing and Organizing (WIEGO). https://www.wiego.org/wp-content/uploads/2022/08/nota-tecnica-wiego-12-parra-abizaid.pdf
Pioggia, A. (2024). Cura e pubblica amministrazione: Come il pensiero femminista può cambiare in meglio le nostre amministrazioni [Care and Public Administration: How Feminist Thinking Can Improve Our Administrations]. Bologna: Il Mulino
Pimp My Carroça. (2024, March). Pesquisa Cataki 2022 traz dados inéditos sobre catadores e catadoras autônomos [Cataki 2022 Survey brings unprecedented data on self-employed reclaimers]. https://pimpmycarroca.com/pesquisa-cataki-2022-traz-dados-ineditos-sobre-catadores-e-catadoras-autonomos/
Public Labor Prosecution Office in Rio de Janeiro. (2022). MPT-RJ, MPE, DPU e DPE ajuízam ação para que Município de Volta Redonda assegure trabalho digno a catadoras e catadores de recicláveis das cooperativas da cidade [MPT-RJ, MPE, DPU and DPE file a lawsuit so that the Municipality of Volta Redonda ensures decent work for reclaimers from the city's cooperatives]. 1st Regional Office. https://www.prt1.mpt.mp.br/informe-se/noticias-do-mpt-rj/1619-mpt-rj-ajuiza-acao-para-que-municipio-de-volta-redonda-assegure-trabalho-digno-a-catadoras-e-catadores-de-reciclaveis-das-cooperativas-da-cidade
Public Labor Prosecution Office in Rio de Janeiro. (2023). Justiça do Trabalho determina que Prefeitura de Volta Redonda efetue diversas adequações no meio ambiente de trabalho das cooperativas de catadores da cidade [Labor Court orders that Volta Redonda City Hall make several adjustments to the working conditions of reclaimers' cooperatives]. https://www.prt1.mpt.mp.br/procuradorias/ptm-vr/1935-justica-do-trabalho-determina-que-prefeitura-de-volta-redonda-efetue-diversas-adequacoes-no-meio-ambiente-de-trabalho-das-cooperativas-de-catadores-da-cidade
Reclaimers' National Movement. (2019, September). Coleta seletiva em Belo Horizonte agora é com catadores [Sorted collection in Belo Horizonte is now with reclaimers!]. https://www.mncr.org.br/noticias/blog-sudeste/coleta-seletiva-em-belo-hozironte-agora-e-com-catadores
Reclaimers' National Movement. (2020, September). Catadores conquistam ampliação do auxílio emergencial da cidade de SP [Catadores achieve expansion of emergency aid in the city of São Paulo]. https://www.mncr.org.br/noticias/blog-sudeste/catadores-conquistam-ampliacao-de-auxilio-emergencial-da-cidade-de-sp
Reclaimers' National Movement. (2023, October). Coopamare resiste a mais uma tentativa de despejo em São Paulo. E vai ficar! [Coopamare resists another eviction attempt in São Paulo. Moreover, it will stay!] https://www.mncr.org.br/noticias/blog-sudeste/coopamare-resiste-a-mais-uma-tentativa-de-despejo-em-sao-paulo-e-vai-ficar
Roelvink, G., St. Martin, K., & Gibson-Graham, J. K. (Eds.). (2015). Making Other Worlds Possible: Performing Diverse Economies. University of Minnesota Press. https://doi.org/10.5749/j.ctt130jtq1
Rutkowski, J. E. (2020). Inclusive packaging recycling systems: Improving sustainable waste management for a circular economy. Detritus, 13, 29–46. https://doi.org/10.31025/2611-4135/2020.14037
Rutkowski, J. E., & Rutkowski, E. W. (2015). Expanding worldwide urban solid waste recycling: The Brazilian social technology in waste pickers' inclusion. Waste Management & Research: The Journal for a Sustainable Circular Economy, 33(12), 1084–1093. https://doi.org/10.1177/0734242X15607424
Saldaña, J. (2013). The coding manual for qualitative researchers (2nd ed). SAGE.
Samson, M. (2020). The political work of waste picker integration. In M. Chen & F. Carré, The Informal Economy Revisited (1st ed., pp. 195–200). Routledge. https://doi.org/10.4324/9780429200724-34
Santiago, C. D. (2024). A Política Nacional de Resíduos Sólidos: Um olhar sobre a governança [The National Solid Waste Policy: A Perspective on Governance]. Boletim Regional, Urbano e Ambiental (BRUA): N. 31, 31, 117–125. https://doi.org/10.38116/brua31art9

Schamber, P., & Bon, S. (2022). Extended producer responsibility (EPR) and packaging regulations in Argentina: Reflections on the aspects associated with the blocking of the draft legislation initiatives. Detritus, 19, 18–27. https://doi.org/10.31025/2611-4135/2022.15195

Sekhwela, M. M., & Samson, M. (2020). Contested understandings of reclaimer integration—Insights from a failed Johannesburg pilot project. Urban Forum, 31(1), 21–39. https://doi.org/10.1007/s12132-019-09377-1

Sousa, M. (2023, June). Catadores de recicláveis realizam ato em prol de direitos em SP [Reclaimers carry out an act in favor of rights in SP]. CicloVivo. https://ciclovivo.com.br/fique-ligado/eventos/catadores-de-reciclaveis-realizam-ato-em-prol-de-direitos-em-sp/

United Nations Environment Programme. (2024). Global Waste Management Outlook 2024: Beyond an age of waste – Turning rubbish into a resource. Nairobi. https://wedocs.unep.org/20.500.11822/44939

Wadham, H., & Warren, R. C. (2014). Telling organizational tales: The extended case method in practice. Organizational Research Methods, 17(1), 5–22. https://doi.org/10.1177/1094428113513619

Yamane, L. H., Dutra, R. M. D. S., & Siman, R. R. (2023). Assessment and perception of occupational risks in waste picker organizations: A portrait of waste pickers' situation after formal integration. Detritus, 22, 13–26. https://doi.org/10.31025/2611-4135/2023.17253

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CISA

CHARACTERISTICS OF RECYCLED AGGREGATES FROM CONSTRUCTION AND DEMOLITION WASTE ACCORDING TO THE ITALIAN EoW REGULATION

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ABSTRACT

Construction and Demolition Waste (CDW) accounts for approximately one-third of the industrial waste generated in the European Union, posing significant environmental challenges. The building sector may significantly increase sustainability through effective CDW management and the production of good quality recycled aggregates (RAs). The construction sector is one of the main sectors involved in achieving environmental sustainability through the application of circular economy principles. In Europe, CDW recovery rates generally exceed regulatory target; only few countries don't reach 70% recovery by weight outlined in Directive 2008/98/EC. ISPRA (Italian Institute for Environmental Protection and Research) data shows that the CDW recovery rate in Italy is over 81%. However, diverging perspectives exist; reports shows illegal CDW dumping in some Regions, compromising environmental and public health efforts. Despite progress, RAs still present challenges, mostly because of its heterogeneity. To safeguard the recovery, the End-of-Waste regulation for inert waste was introduced in Italy: the requirements introduced by Ministerial Decree 152/2022, replaced by the Ministerial Decree n.127/2024, could involve several issues to promote the CDW recovery. The aim of this study is to illustrate the characteristics of RAs with respect to the requirements set by the EoW regulation, in order to evaluate the effect of its implementation on the recovery feasibility. About 2500 laboratory certificates (chemical analysis and leaching tests) are collected; they referred to CDW and RAs quality monitoring in 2020-2022 carried out by 28 Italian recycling plants. These data were analyzed in order to statistically verify the compliance of RAs with the EoW requirements.

1. INTRODUCTION

The "linear" economy, also known as the "take-make-dispose" model, has been the dominant industrial system for the past 150 years. It involves extracting raw materials, mass consumption, and producing waste once the product's life cycle has ended. This approach is recognized as unsustainable and it is necessary to adopt a new "circular" economy model that reduces and eliminates waste, diversifies material sources, and extends the life of products (Papadaki at al., 2022).

The Community regulatory framework focuses on the recovery and use of waste materials to preserve natural resources through the Waste Framework Directive 2008/98/EC and its amendment EU Directive 2018/851. The European Commission's action plan prioritizes seven crucial sectors for a circular economy, including buildings, plastics, textiles, electronic waste, food and water, packaging, batteries and vehicles.

The building sector is a major contributor to the depletion of natural resources due to its intensive use. Construction and demolition waste (CDW), listed in Chapter 17 of the European Waste List (EWC), is one of the largest waste streams produced in Italy and Europe. In 2023, Italy produced over 61 million tonnes of CDW (ISPRA, 2025), excluding excavated soil and rock, and dredging spoil. It represents the fourth largest producer in Europe after Germany, France, and the United Kingdom (Eurostat, 2022).

The Italian CDW recovery rate in 2023 was 81% (ISPRA, 2025), exceeding the 70% goal established by European Directive 2008/98/EC. Countries with high recovery rates are the ones with the best practices in terms of waste treatment. For example, in the Netherlands, Luxembourg and Belgium, characterized by densely populated areas,

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great efforts have been made to reduce the disposal of mineral waste from construction and demolition. Many of the Member States with higher recovery rates have a good range of CDW management structures, such as Denmark, Italy, Ireland, Germany and Luxembourg (European Commission, 2017).

To do this, it is very important to know their composition, that is influenced by several factors, including the variability of raw materials, the construction practices, the demolition techniques (usually non-selective mode is adopted), and the lack of advanced treatment processes in recycling plants (Juenger and Siddique, 2015; Silva et al., 2017; Gálvez-Martos et al., 2018; Ruggeri et al., 2019).

The heterogeneous mixture of CDW components (Bianchini et al., 2020; Diotti et al., 2021) makes it difficult to recycle and market recycled aggregates (RAs) from their treatment. In fact, according to Caro et al. (2024), the average CDW in the EU has a high mixed fraction composition (59.2%). The individual fractions composing CDW are: concrete (24%), bricks (5%), metals (4.3%), wood (2.3%), hazardous components (1.8%), gypsum (1.4%), ceramics and tiles (1.2%), insulation materials (0.3%), plastic (0.2%), glass (0.2%), paper and cardboard (0.2%).

The chemical composition of RAs depends on the characteristics of the CDW from which they originate (Butera et al., 2014). Because CDW undergoes simple mechanical treatment, its elements often remain in RAs. Literature shows chromium as one of the most critical compounds. Total chromium, a major concrete component, can appear in high concentrations due to raw materials (Sinyoung et al., 2011; Estokova et al., 2018), but it also derives from ceramic and asphalt fractions. Zinc, vanadium, and nickel are also found, especially in cementitious and bituminous materials, with vanadium common in bitumen. Ceramic, clay, and brick fractions show high sulphate and fluoride levels, particularly in pure gypsum, with very high concentrations (Cilli et al., 2017; Imteaz et al., 2020). Bituminous mixtures contain significant hydrocarbons and polycyclic aromatic hydrocarbons (PAHs), mainly from traffic-related activities (Sadler et al., 1999).

In Italy, the recycled aggregates (RAs) from CDW treatment are mainly used for infrastructure construction (roads, railways, cycle paths, etc.) (81.2%), followed by fillings and other uses (11.6%). The using of RAs for non-structural concrete (6.8%), and structural concrete (0.4%) production is still very low. This is due to the low in-situ generation of RAs derived from concrete demolition only (EWC 170101), that can be used, according to regulations, for new concrete production (Fondazione per lo sviluppo sostenibile and FISE UNICIRCULAR, 2021).

To promote the shifting to a circular economy, the EU has established that waste status ends when a material is subjected to recovery processes in accordance with specific requirements. Currently, End-of-Waste (EoW) criteria have been defined, at European level, for the following categories of waste: scrap iron, steel and aluminium, glass cullet, and copper scrap. In Italy, according to the requirements reported in Legislative Decree 152/2006 (Italian Government, 2006) and subsequent amendments (article 184-ter), EoW regulations are available for the following waste categories: bituminous conglomerate, absorbent hygiene personal care products, SRF (Solid Recovered Fuel), End-of-Life Tires (ELT) derived rubber, paper and board, CDW. Additionally, EoW regulations are being drafted for road sweeping waste, textile waste, bituminous membrane, and gypsum-based waste.

As concerns CDW, EoW criteria have not yet been defined at European level, although some projects are available and a preliminary list of pre-selected CDW and by-product streams is investigated (Reetsch et al., 2024). The JRC has published a draft technical proposal on EoW criteria for mineral CDW, which will then be used as a basis by the European Commission for defining the relevant Regulation (Egle et al., 2025). Similar to Austria, France, the Netherlands, Belgium, the United Kingdom and Ireland, recently the EoW regulation for inert waste has been also published in Italy. Other countries (Denmark, Finland, Czech Republic, Germany, Spain and Sweden) that don't have the EoW criteria have nevertheless adopted leaching criteria following European standards (Velzeboer and van Zomeren, 2017).

An analysis of End of Waste (EoW) criteria across European countries shows differences in accepted input waste, though CDW—cement/concrete, bricks, tiles, and ceramics—is universally considered a basic material for recovery and RAs production. The Netherlands applies EoW rules for "stony waste" from construction, renovation, and demolition, limiting codes to EWC 17 01 01 (concrete), 17 01 02 (bricks), and 17 01 07 (mixtures). Belgium (Wallonia) only accepts chapter 17 codes, unlike Italy, which also allows others; Ireland permits a broad range, while France is very selective.

For RAs composition, Dutch limits for parameters like sum of PAHs and PCBs are less strict than Italy's. France allows concrete and mixed aggregates under limits for hydrocarbons (C10-C21), PAHs, TOC, and BTEX, generally higher than Italian criteria. Regarding pollutant release, leaching tests are different: e.g. Austria uses EN 12457-4 (10 L/kg liquid-to-solid ratio, 10 mm max size vs. Italy's 4 mm), while the Netherlands adopts EN 14405 up-flow percolation. Detected parameters also vary; Austria sets fewer limits. Overall, Italian values are among the most restrictive (Velzeboer and van Zomeren, 2017).

The first version of Italian EoW regulation published in 2022 (Ministerial Decree 152/2022, Italian Ministry of ecological transition, 2022) has raised some concerns in the construction and recycling sector, with the risk to increase the amount of CDW sent to landfills. It has been noted that it is challenging to comply with the EoW criteria, particularly those pertaining to the content of pollutants in the solid matrix that set (in 2022 release) the same (and very low) limit values for all applications of RAs derived from CDW treatment.

In 2024 the Ministerial Decree n. 127/2024 (Italian Ministry of environment and energy security, 2024) replaced the first EoW version, and some changes were implemented, in order to make its application more effective to move towards a circular economy approach. The addition of various threshold limits for the content of pollutants dependent on the types of RA use was the primary change with regard to the 2022 version.

This work aims to highlight the chemical and environmental characteristics of CDW and RAs obtained from their treatment and to compare them with the EoW criteria outlined in the recent Italian regulation. The statistical results presented are based on the elaboration of 2500 certificates of analysis obtained from 28 Italian CDW recycling plants.

2. MATERIALS AND METHODS

2.1 Data collected

Thanks to the collaboration with the National Association of Building Constructors (ANCE), the Italian CDW treatment facilities were contacted to carry out a survey using a questionnaire. This survey's objectives are to examine the kinds of treatments used in recycling facilities, the quantity and kind of CDW that are treated, the production of RAs, the internal quality control procedures for incoming waste and outgoing aggregates, and to gather historical information regarding the qualitative features of the treated CDW and the RA generated.

28 CDW recycling plants took part to the survey and provided the required data. The location of these plants is reported in Figure 1. It can be noted that the medium- and high-capacity plants (with CDW amount higher than 100 kt/year) account for about 40%.

The plants considered have been also classified by EWC codes of the waste they can receive. As shown in Figure 2, the most commonly treated waste is mixed fraction (EWC 17 09 04), followed by soil and rock (EWC 17 05 04), and concrete (EWC 17 01 01).

These plants provided 2500 certificates of analysis both on CDW and RAs referred to the period 2020-2022. Table 1 shows the number of available certificates divided according to the type of treated waste. The analyses include the determination of chemical composition (to measure the content of pollutants in the solid matrix) and leaching test in water carried out with a liquid-to-solid ratio of 10 L/kg, according to UNI EN 12457-2 (the leaching test with CO_2 saturated water is applied only to the samples of the plants located in Bolzano Province).

It is evident that around 70% of CDW certificates, including those pertaining to leaching tests and chemical composition, make reference to mixed waste (EWC 17 09 04). Regarding RAs, it is frequently impossible to know the type of CDW from which they were obtained (in 60% of cases for chemical composition and 74% for leaching test).

Although the number of tests available for CDW (1155) and RAs (1337) is similar, there are more chemical composition measurements available for CDW (77%) and more leaching test results for RAs (71%).

2.2 Elaboration criteria

As concerns graphical representation, the box-plot methodology was applied. This methodology allows evaluating the entirety of the data distribution in relation to

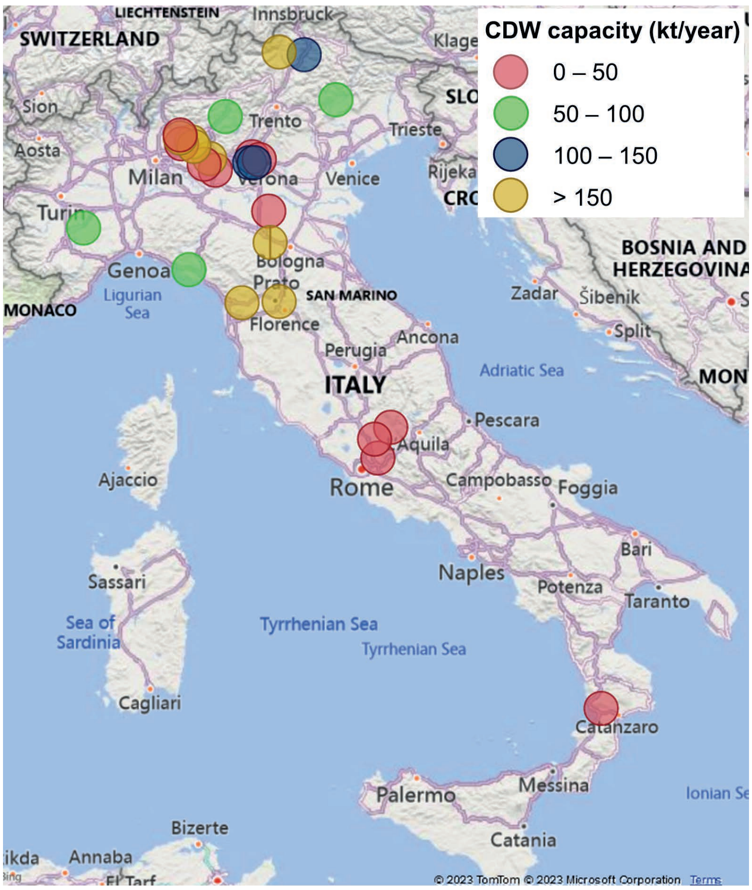

FIGURE 1: CDW treatment plants that took part in the survey: location and size category.

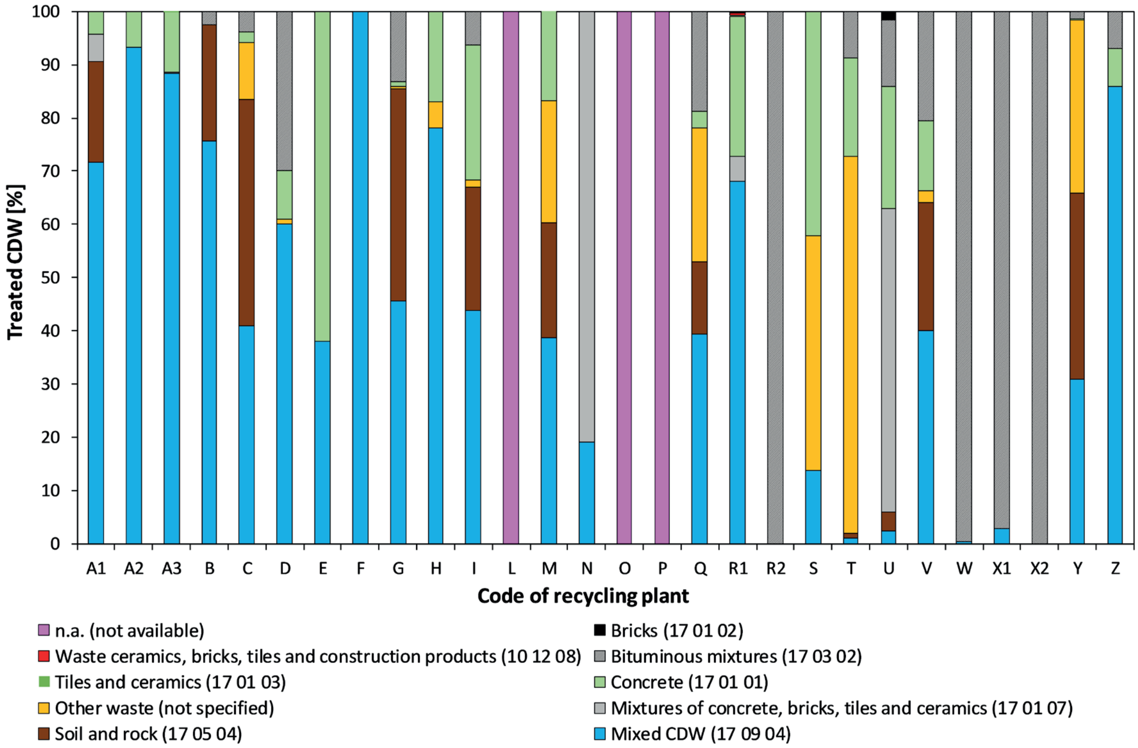

FIGURE 2: Distribution of treated CDW by EWC code (reference year: 2022).

specific statistical values including the 25th, 50th, and 75th percentiles. The basic idea is to identify the central observations with a "box" the central observations, with "tails" coming out of the box the values that are closest to the box, and with dots and stars the anomalous observations. A box plot represents the distribution of data and consists of a box from the first quartile (Q1) to the third quartile (Q3). The median of the data is represented by a horizontal line inside the box (Q2). Approximately 50% of the data falls within the box (IQR, calculated as the difference between Q3 and Q1).

The tails of the distribution, which are the extreme values, are represented by continuous lines starting from the edges of the box (Q1 and Q3) and ending with two other horizontal lines, the whiskers. The tails delimit the intervals (1.5 IQR) in which the lower values of Q1 and the higher values of Q3 are located. The limit values for the whiskers are defined as Q1 - 1.5 * IQR for the lower whisker and Q3 + 1.5 * IQR for the upper whisker. The values outside the tails are considered anomalous values and significantly deviate from the other data points in the same population. Outliers (between 1.5 and 3 times IQR) are considered normal while extreme values (over 3 IQR) are generally rare and far from the central data distribution represented by the box.

The results are also summarized in tables. For each parameter reported in the laboratory certificates, the minimum, maximum, and average values were given. The results obtained were compared with the following limit values:

- For the content of pollutants in RAs (chemical composition), the limit values set by the EoW regulation (Ministerial Decree n. 127/2024 - Annex 1, Table 2), which provides two different columns:
  - the column 1 - more restrictive limits - for the use of RAs in environmental reclamation and filling;
  - the column 2 - less restrictive limits - for other application of RAs such as bituminous mixtures, foun-

TABLE 1: Number of certificates of analysis available for CDW and RAs.

Type of waste (EWC)	Construction and demolition waste (CDW)		Recycled aggregates (RAs)	
	Chemical composition	Leaching test	Chemical composition	Leaching test
Concrete (17 01 01)	8	21	0	38
Mixtures of concrete, bricks, tiles and ceramics (17 01 07)	31	11	0	0
Bituminous mixtures (17 03 02)	102	47	72	232
Soil and rock (17 05 04)	74	34	10	18
Mixed CDW (17 09 04)	461	331	0	7
Other waste / not available	17	18	119	841
Total	693	462	201	1136

dation layer and sub-bases for roads, and concrete production.
- For the release of pollutants from RAs (results of leaching test):
  - the limit values established by EoW regulation (Ministerial Decree n. 127/2024 - Annex 1, Table 3);
  - the limit values set by Ministerial Decree n. 186/2006 for the waste recovery in simplified procedure (the limits are equal to the ones imposed by EoW regulation, with the exception of sulphates and chlorides).

In these tables, the results obtained are also reported in terms of percentages of exceeding (or compliance). In many cases, especially for the chemical composition, the instrumental determination value (detection limit - D.L.) reported in analytical certificates was higher than the threshold value (especially for the more restrictive limits). Therefore, the following percentages were calculated:

- certain exceedance: number of samples that certainly exceed the limit (e.g. CrVI = 2.47 mg/kg greater than the limit of 2 mg/kg), in relation to the total number of data available;
- possible exceedance: number of samples that could exceed the limit, taking into account only the values reported as < D.L., when D.L. is greater than the limit itself (example CrVI < 5 mg/kg compared to the limit of 2 mg/kg), in relation to the total number of data available;
- certain and possible exceedance: number of samples

TABLE 2A: Chemical composition of RAs and compliance/non-compliance with the more restrictive requirements provided by Italian EoW regulation.

Parameter		Number of data	Content			(A) Tab.2, Col. 1, Ministerial Decree 127/2024				
			Minimum	Average	Maximum	Limit values	Non-compliance			Certain compliance
							Certain	Possible	Certain and possible	
Chromium VI	$[mg/kg_{DM}]$	131	<0.1	0.27	<5	2	0.8%	1,5%	2.3%	97.7%
Benzene	$[mg/kg_{DM}]$	31	<0.002	0.20	<1	0.1	0.0%	32.3%	32.3%	67.7%
Ethylbenzene	$[mg/kg_{DM}]$	31	<0.002	0.22	<1	0.5	0.0%	6.5%	6.5%	93.5%
Styrene	$[mg/kg_{DM}]$	23	<0.002	0.12	<1	0.5	0.0%	8.7%	8.7%	91.3%
Toluene	$[mg/kg_{DM}]$	31	<0.002	0.22	<1	0.5	0.0%	6.5%	6.5%	93.5%
Xylene	$[mg/kg_{DM}]$	31	<0.002	0.22	<1	0.5	0.0%	6.5%	6.5%	93.5%
BTEXS	$[mg/kg_{DM}]$	25	<0.05	0.20	<0.5	1	0.0%	0.0%	0.0%	100.0%
Benzo(a)anthracene	$[mg/kg_{DM}]$	98	<0.005	4.44	<10	0.5	2.0%	42.9%	44.9%	55.1%
Benzo(a)pyrene	$[mg/kg_{DM}]$	98	<0.005	4.44	<10	0.1	5.1%	78.6%	83.7%	16.3%
Benzo(b)fluoranthene	$[mg/kg_{DM}]$	98	<0.005	4.51	<10	0.5	0.0%	42.9%	42.9%	57.1%
Benzo(k)fluoranthene	$[mg/kg_{DM}]$	73	<0.003	4.47	<10	0.5	0.0%	46.6%	46.6%	53.4%
Benzo(g.h.i)perylene	$[mg/kg_{DM}]$	98	<0.005	4.44	<10	0.1	5.1%	77.6%	82.7%	17.3%
Chrysene	$[mg/kg_{DM}]$	98	<0.005	4.44	<10	5	0.0%	41.8%	41.8%	58.2%
Dibenzo(a.e)pyrene	$[mg/kg_{DM}]$	96	<0.002	4.50	<10	0.1	0.0%	78.1%	78.1%	21.9%
Dibenzo(a.l)pyrene	$[mg/kg_{DM}]$	95	<0.002	4.54	<10	0.1	0.0%	77.9%	77.9%	22.1%
Dibenzo(a.i)pyrene	$[mg/kg_{DM}]$	98	<0.002	4.43	<10	0.1	0.0%	78.6%	78.6%	21.4%
Dibenzo(a.h)pyrene	$[mg/kg_{DM}]$	98	<0.002	4.43	<10	0.1	1.0%	78.6%	79.6%	20.4%
Dibenzo(a.h)anthracene	$[mg/kg_{DM}]$	88	<0.002	4.88	<10	0.1	1.1%	76.1%	77.3%	22.7%
Indenopyrene	$[mg/kg_{DM}]$	87	<0.004	4.97	<10	0.1	0.0%	74.7%	74.7%	25.3%
Pyrene	$[mg/kg_{DM}]$	86	<0.005	4.98	<10	5	0.0%	47.7%	47.7%	52.3%
Sum of PAH	$[mg/kg_{DM}]$	88	<0.005	12.85	<70	10	0.0%	11.4%	11.4%	88.6%
Phenol	$[mg/kg_{DM}]$	19	<0.0025	0.12	<1	1	0.0%	0.0%	0.0%	100.0%
PCB	$[mg/kg_{DM}]$	19	<0.001	0.11	<1	0.06	5.3%	10.5%	15.8%	84.2%
Heavy hydrocarbons (C>12)	$[mg/kg_{DM}]$	81	<20	361	9464	50	22.2%	0.0%	22.2%	77.8%
Asbestos	$[mg/kg_{DM}]$	197	Absent	63	<1000	100	0.0%	2.0%	2.0%	98.0%
Floating materials	$[cm^3/kg]$	6	0.01	0.01	0.01	5	0.0%	0.0%	0.0%	100.0%
Extraneous material	[%]	6	<0.01	0.34	<1	1	0.0%	0.0%	0.0%	100.0%

(A): use of RAs for environmental reclamation and filling

PAH: polycyclic aromatic hydrocarbons; PCB: polychlorinated biphenyls; BTEXS: sum of benzene, toluene, ethylbenzene, xylene, and styrene

TABLE 2B: Chemical composition of RAs and compliance/non-compliance with the less restrictive requirements provided by Italian EoW regulation.

Parameter		Number of data	Content			(B) Tab.2, Col. 2, Ministerial Decree 127/2024				
			Minimum	Average	Maxi-mum	Limit values	Non-compliance			Certain compli-ance
							Certain	Possible	Certain and possible	
Chromium VI	[mg/kg$_{DM}$]	131	<0.1	0.27	<5	15	0.0%	0.0%	0.0%	100.0%
Benzene	[mg/kg$_{DM}$]	31	<0.002	0.20	<1	2	0.0%	0.0%	0.0%	100.0%
Ethylbenzene	[mg/kg$_{DM}$]	31	<0.002	0.22	<1	50	0.0%	0.0%	0.0%	100.0%
Styrene	[mg/kg$_{DM}$]	23	<0.002	0.12	<1	50	0.0%	0.0%	0.0%	100.0%
Toluene	[mg/kg$_{DM}$]	31	<0.002	0.22	<1	50	0.0%	0.0%	0.0%	100.0%
Xylene	[mg/kg$_{DM}$]	31	<0.002	0.22	<1	50	0.0%	0.0%	0.0%	100.0%
BTEXS	[mg/kg$_{DM}$]	25	<0.05	0.20	<0.5	100	0.0%	0.0%	0.0%	100.0%
Benzo(a)anthracene	[mg/kg$_{DM}$]	98	<0.005	4.44	<10	10	0.0%	0.0%	0.0%	100.0%
Benzo(a)pyrene	[mg/kg$_{DM}$]	98	<0.005	4.44	<10	10	0.0%	0.0%	0.0%	100.0%
Benzo(b)fluoranthene	[mg/kg$_{DM}$]	98	<0.005	4.51	<10	10	0.0%	0.0%	0.0%	100.0%
Benzo(k)fluoranthene	[mg/kg$_{DM}$]	73	<0.003	4.47	<10	10	0.0%	0.0%	0.0%	100.0%
Benzo(g.h.i)perylene	[mg/kg$_{DM}$]	98	<0.005	4.44	<10	10	0.0%	0.0%	0.0%	100.0%
Chrysene	[mg/kg$_{DM}$]	98	<0.005	4.44	<10	50	0.0%	0.0%	0.0%	100.0%
Dibenzo(a.e)pyrene	[mg/kg$_{DM}$]	96	<0.002	4.50	<10	10	0.0%	0.0%	0.0%	100.0%
Dibenzo(a.l)pyrene	[mg/kg$_{DM}$]	95	<0.002	4.54	<10	10	0.0%	0.0%	0.0%	100.0%
Dibenzo(a.i)pyrene	[mg/kg$_{DM}$]	98	<0.002	4.43	<10	10	0.0%	0.0%	0.0%	100.0%
Dibenzo(a.h)pyrene	[mg/kg$_{DM}$]	98	<0.002	4.43	<10	10	0.0%	0.0%	0.0%	100.0%
Dibenzo(a.h)anthracene	[mg/kg$_{DM}$]	88	<0.002	4.88	<10	10	0.0%	0.0%	0.0%	100.0%
Indenopyrene	[mg/kg$_{DM}$]	87	<0.004	4.97	<10	5	0.0%	47.1%	47.1%	52.9%
Pyrene	[mg/kg$_{DM}$]	86	<0.005	4.98	<10	50	0.0%	0.0%	0.0%	100.0%
Sum of PAH	[mg/kg$_{DM}$]	88	<0.005	12.85	<70	100	0.0%	0.0%	0.0%	100.0%
Phenol	[mg/kg$_{DM}$]	19	<0.0025	0.12	<1	60	0.0%	0.0%	0.0%	100.0%
PCB	[mg/kg$_{DM}$]	19	<0.001	0.11	<1	5	0.0%	0.0%	0.0%	100.0%
Heavy hydrocarbons (C>12)	[mg/kg$_{DM}$]	81	<20	361	9464	750	11.1%	0.0%	11.1%	88.9%
Asbestos	[mg/kg$_{DM}$]	197	Absent	63	<1000	100	0.0%	2.0%	2.0%	98.0%
Floating materials	[cm^3/kg]	6	0.01	0.01	0.01	5	0.0%	0.0%	0.0%	100.0%
Extraneous material	[%]	6	<0.01	0.34	<1	1	0.0%	0.0%	0.0%	100%

(B): use of RAs for bituminous mixtures, foundation layer and sub-bases for roads, railways, airports and civil/industrial yards, concrete production.

PAH: polycyclic aromatic hydrocarbons; PCB: polychlorinated biphenyls; BTEXS: sum of benzene, toluene, ethylbenzene, xylene, and styrene ; DM: dry matter

that could exceed the limit, considering not only the certain exceedances (e.g. CrVI = 2.47 mg/kg greater than the limit of 2 mg/kg), but also values reported as < D.L., where D.L. exceeds the limit itself (example CrVI < 5 mg/kg compared to the limit of 2 mg/kg), in relation to the total number of data available;

- certain compliance: the number of samples that with certainty comply with the regulatory limit, in relation to the total number of available data.

3. RESULTS

3.1 Chemical composition of recycled aggregates

In this paragraph, the results about RAs chemical composition are shown. In Tables 2A and 2B, the non-compliance values are highlighted in grey cell. Moreover, the miniminum, average, and maximum contents of pollutants are reported (the average values are calculated considered the values lower than detection limit (D.L.) equal to D.L.).

Heavy hydrocarbons (C>12) are the most critical pollutants: it can be note (Table 2A) that 22% of the analyzed samples don't respect (certainly) the more restrictive limit (reported on EoW regulation – Ministerial Decree 127/2024, Table 2, column 1 – that regulates the use of RAs for environmental reclamation and filling). Benzo(a)pyrene, benzo(g,h,i)perylene, and PCB (polychlorinated biphenyls) show certain exceedances of about 5% with respect the more restrictive limit values. Despite this percentage be quite low, the possible non-compliance is high (also up to 79%) for several PAHs and benzene. Based on

TABLE 3: Results of leaching tests on RAs and non-compliance with requirements provided by the Italian EoW regulation (M.D. 127/2024) and the Italian waste recovery legislation (M.D. 186/2006).

Parameter		Number of data*	Minimum	Average	Median	Maximum	Ministerial Decree 127/2024, Tab. 3		Ministerial Decree 186/2006, Annex 3	
							Limit values	Non-compliance	Limit values	Non-compliance
pH	[-]	998	6.4	10.1	10.0	12.6	5.5-12	0.4%	5.5-12	0.4%
Nitrates	[mg/L]	1009	<0.1	11.4	5.0	155	50	0.4%	50	0.4%
Fluorides	[mg/L]	1009	<0.025	0.41	0.25	33.3	1.5	0.3%	1.5	0.3%
Sulphates	[mg/L]	1009	<1	55.5	29.2	816	750	0.2%	250	0.8%
Chlorides	[mg/L]	1009	0.55	11.1	10.0	89.0	750	0.0%	100	0.0%
Cyanide	[µg/L]	1007	<0.01	8.78	10.0	<50	50	0.0%	50	0.0%
Barium	[mg/L]	1009	<0.0001	0.07	0.09	<5	1	0.2%	1	0.2%
Copper	[mg/L]	1009	<0.0001	0.01	0.01	0.08	0.05	0.2%	0.05	0.2%
Zinc	[mg/L]	1009	<0.0001	0.07	0.05	4	3	0.1%	3	0.1%
Beryllium	[µg/L]	1009	<0.001	1.17	1.00	9.10	10	0.0%	10	0.0%
Cobalt	[µg/L]	1009	<0.005	3.91	5.00	28.8	250	0.0%	250	0.0%
Nickel	[µg/L]	1009	<0.0001	2.34	1.00	15.9	10	0.2%	10	0.2%
Vanadium	[µg/L]	1009	<0.01	15.3	10.7	211	250	0.0%	250	0.0%
Arsenic	[µg/L]	1007	<0.001	3.91	3.60	33.1	50	0.0%	50	0.0%
Cadmium	[µg/L]	1008	<0.001	0.57	0.50	<5	5	0.0%	5	0.0%
Total Chromium	[µg/L]	1008	<0.001	17.2	10.0	282	50	0.3%	50	0.3%
Lead	[µg/L]	1008	<0.005	4.15	5.00	45.0	50	0.0%	50	0.0%
Selenium	[µg/L]	1008	<0.005	1.73	1.00	14.2	10	0.1%	10	0.1%
Mercury	[µg/L]	1009	<0.0001	0.37	0.50	1.50	1	0.2%	1	0.2%
Asbestos	[mg/L]	554	0.00005	4.55	1.00	<30	n.p.	-	30	0.0%
COD	[mg/L]	1009	1	16.6	15.0	43.0	30	0.4%	30	0.4%

** The samples related to the leaching tests performed with CO_2 saturated water (not in according with UNI EN 12457-2) were not considered.*
n.p.: not provided

the available data, we are not able to clear identify the RA compliance or not, because the D.L. is not enough lower than the respective threshold.

The only parameter that certainly exceeds the less restrictive limits for RAs application in bituminous mixtures, sub-bases for roads and concrete production, etc.) (Table 2, column 2, M.D. 127/2024) are heavy hydrocarbons (C>12), with a percentage of exceedance of 11%. Indenopyrene and asbestos show possible non-compliance (Table 2B).

A comparison between the composition of CDW and RAs is reported in Figure 3, for the most critical parameters. It is important to note that the certificates of analysis related to the produced RAs do not correspond to those of the treated CDW to obtain such RAs; furthermore, for the same parameters, the instrumental detection limits are different. In fact, CDW are analyzed with the aim to verify the non-hazardous classification, while RAs must comply the requirements for their recovery.

It can be observed that the content of PCBs and heavy hydrocarbons (C>12) decreases from treated CDWs to produced RAs. Therefore, the treatments adopted in the CDW recycling plants are able to slightly reduce the content of pollutants. Instead, PAHs, such as benzo(a)pyrene and benzo(g,h,i)perylene, show opposite behaviour: this is caused by the presence of many analytical certificates in which the instrumental detection limit for RA is higher than CDW ones.

Figure 4 shows the effect of CDW type (based on EWC) used to produce the RAs on the content of most critical pollutants in RAs. Regarding heavy hydrocarbons (C>12) and PHAs, such as benzo(a)pyrene and benzo(g,h,i)perylene, the concentrations measured in RAs produced from bituminous mixtures (EWC 17 03 02) are significantly higher and much more scattered than RAs produced from other kind of waste. It is important to note that it is impossible to know the origin of the waste for a considerable number of RAs (reported in Figure 3 as "EWC n.a."). Based on the data analyzed (Figure 2 and Table 1), most of them are probably obtained from the mixed CDW (EWC 17 09 04), therefore with heterogenous quality.

As concerns heavy hydrocarbons, almost all the RAs obtained from bituminous mixtures don't comply the less restrictive limit - 750 mg/kg_{DM} - set by the EoW regulation (Ministerial Decree 127/2024). It should be noted that for this kind of RA are already available EoW criteria provided by Ministerial Decree 69/2018, which does not require the measurement of heavy hydrocarbons in the final product (bituminous conglomerate granulate), but rather requires monitoring the sum of PAHs (with a threshold limit of 100 mg/kg_{DM}) and asbestos (1000 mg/kg_{DM}).

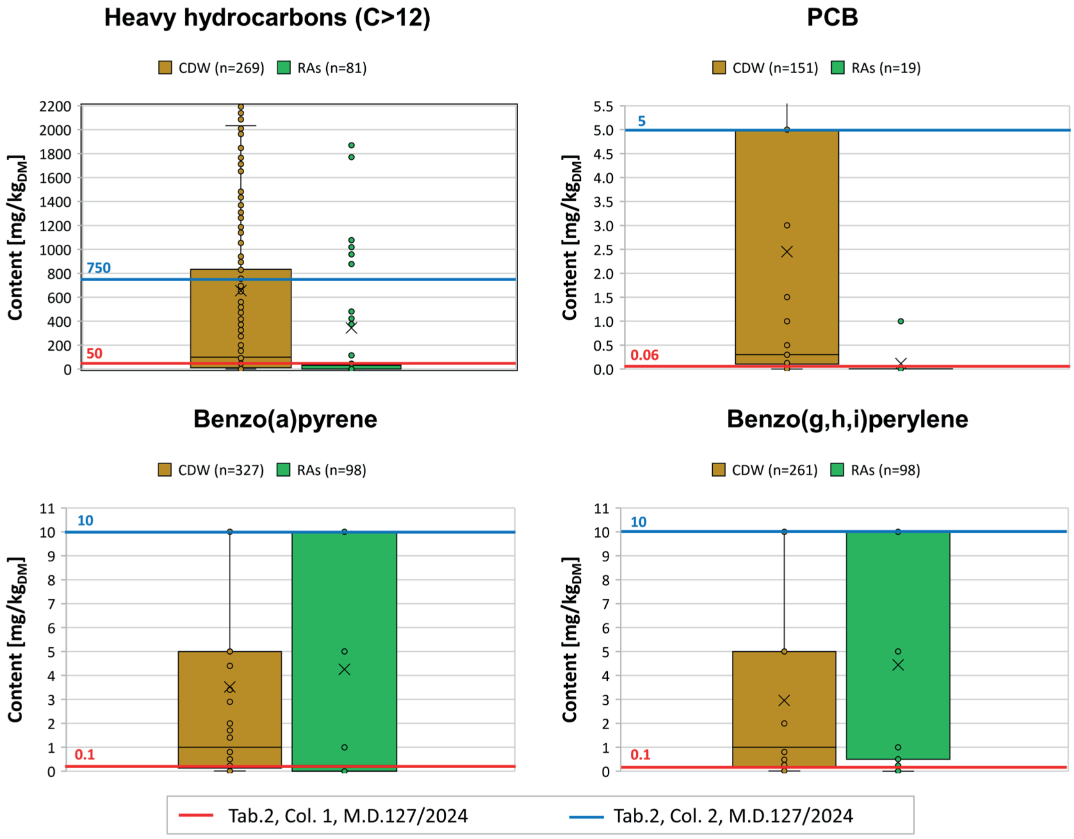

FIGURE 3: Comparison between CDW and RAs chemical composition – most critical pollutants (n: number of analyses).

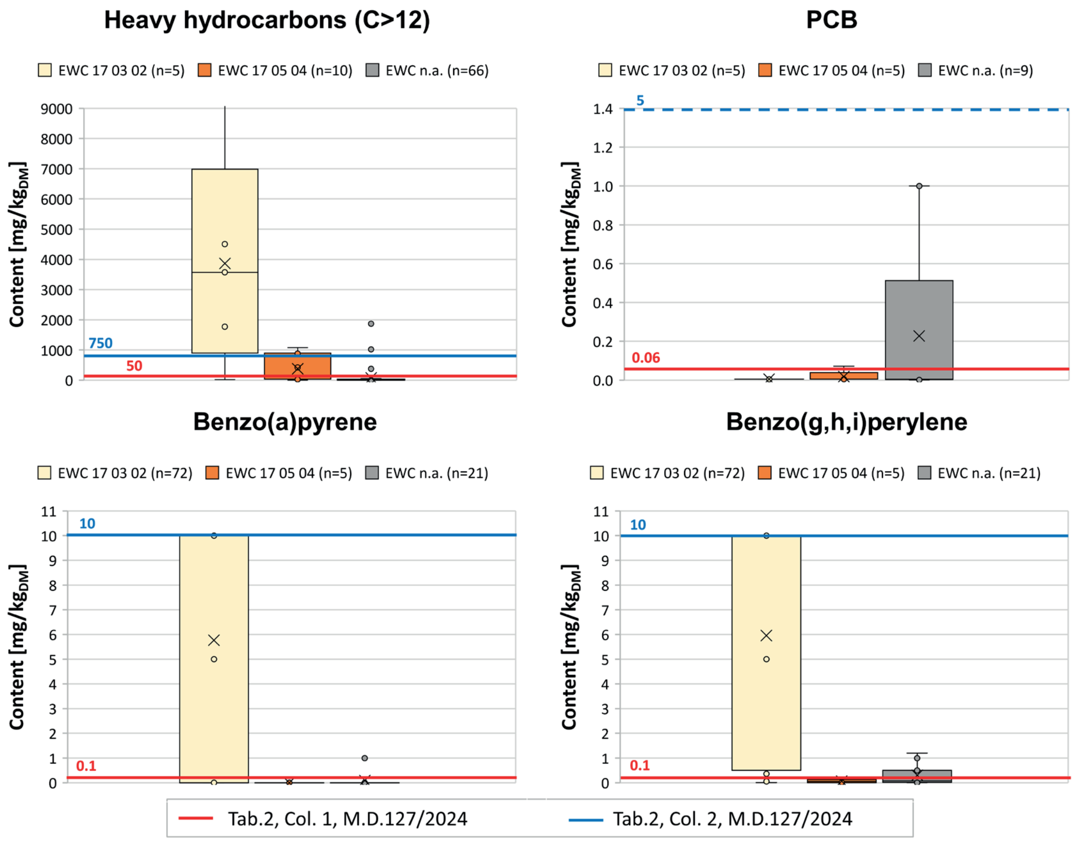

FIGURE 4: Effect of CDW type on the chemical composition of RAs – most critical pollutants (n: number of analyses; EWC n.a.: no information about CDW treated to produce RA).

The RAs obtained from the treatment of soil and rocks (EWC 17 05 04) often show high contents of heavy hydrocarbon that exceed the more restrictive limit (50 mg/kg$_{DM}$) and, occasionally, even the less restrictive limit. This could be due to the fact that during the excavation phase of this CDW, some fractions of bituminous mixtures are also collected and subsequently processed to produce the recycled aggregates.

As concerns PCBs, high content is observed for the RAs obtained by unknown CDW (probably most of them with significant concrete fraction).

3.2 Release of pollutants from recycled aggregates

As reported for the chemical composition, the results of the leaching test on RAs (according to UNI EN 12457-2) were elaborated (Table 3) in order to show the non-compliance both with the Italian EoW criteria (M.D. 127/2024) and the national regulation for waste recovery in simplified procedure (M.D. 186/2006). It can be noted that the limit values are the same, with the exception of sulphate and chloride, and the asbestos (this pollutant is not provided in EoW regulation). For each parameter in the database, minimum, maximum, average and median concentrations are also reported. Also in this case, the average and the median values are calculated considered the values lower than detection limit (D.L.) equal to D.L.. The non-compliance values are highlighted in grey cell.

Unlike the chemical composition, it was not necessary to calculate the possible non-compliance, because, for each parameter, the analysis performed is able to reach concentrations lower than the limit threshold.

It can be noted that there are no particular critical issues. Over 99% of the samples meet the EoW regulation's requirements. Few non-compliances (that do not exceed 1%) are referred to pH, nitrates, COD, fluorides, total chromium, sulphates, barium, nickel, copper, mercury, zinc, and selenium.

Figure 5 shows the results of leaching tests, only for the most critical parameters, using box-plot methodology, in order to see the variability of the pollutants release. It can be observed that:

- concerning pH, despite in most cases it complies with the regulatory limit (in the range 5.5 - 12), the measured values are generally between 9.5 and 11; the higher values are probably due to the presence of cementitious fraction; it is pointed out that pH is one of the most important parameters that affect the release especially of the heavy metals;
- COD is generally between 14 and 22 mg/L; also in this case, although the non-compliance is occasionally, sometimes the measured concentrations are quite close to the limit threshold (30 mg/L); this aspect could be due to the presence of soil fractions;
- total chromium and nickel show scattered concentra-

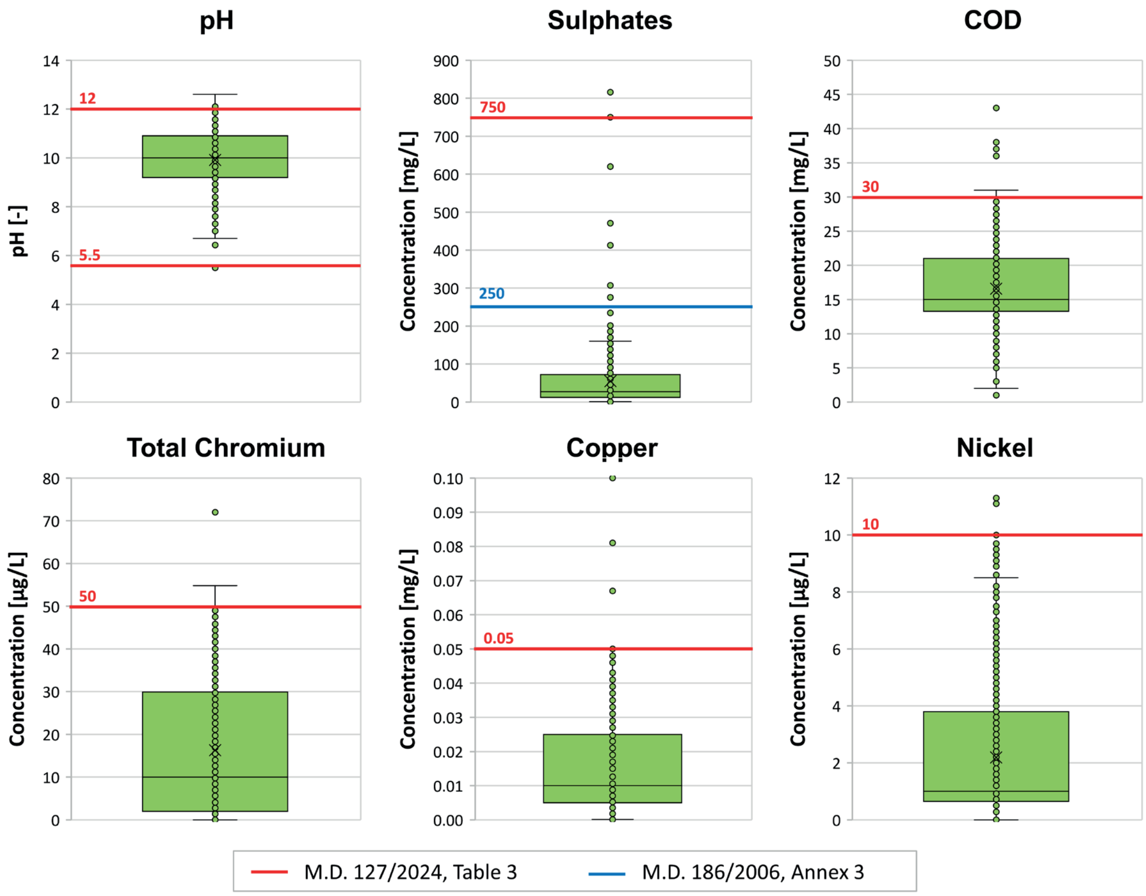

FIGURE 5: Results of leaching tests on RAs – most critical pollutants.

tions; however, the non-compliance with the limits set by EoW criteria (50 μg/L for total chromium and 10 μg/L for nickel) are very low;

- regarding copper, even if only three values exceed the limit values (0.05 mg/L), there are several concentrations close to this threshold;
- as concerns sulphates, most values are below both limit values (250 imposed by the waste recovery regulation and 750 mg/L provided by the EoW criteria), although some values exceed 250 mg/L (threshold for waste recovery in simplified procedure).

As mentioned above, the higher releases of COD (Figure 6) – in few cases also over than EoW limit values - are observed for RAs obtained from the treatment of soils and rocks (EWC 17 05 04) and from unknown waste (probably most of them are RAs derived from mixed CDW). This aspect could be due to the presence of residual soils (derived from the demolition operation) in the mixed CDW sent to the recycling plants.

As might be expected, high releases of total chromium are noted in RAs obtained from cement CDW (EWC 17 01 01). Moreover, RAs derived from unknown waste (n.a. in Figure 6) showed very scattered concentrations ang the highest values, also above the threshold; their quality is higly variable and occasionally cementitious and concrete fractions may be found.

3.3 Environmental and economic implications of non-compliant recycled aggregates

The use of RAs from CDW that do not comply with EoW regulation can have serious environmental and economic consequences. From an environmental standpoint, one major concern is the potential release of contaminants. Heavy metals such as chromium, lead, copper, and zinc can leach into soil and groundwater, posing risks to ecosystems and human health (Diotti et al., 2020; Vu Quoc Hung et al., 2022). Cement residues present in RAs tend to produce highly alkaline leachates, which can disrupt soil chemistry and harm microbial communities (Diotti et al., 2020). Elevated sulphate concentrations are another issue, as they may accelerate chemical degradation processes and contribute to groundwater contamination (Diotti et al., 2020). Furthermore, RAs that include asphalt can release PAHs, compounds known for their toxicity and persistence in the environment (Jandová et al., 2025). Studies also highlight that contaminant loads vary widely among recycled materials, and standard leaching tests may underestimate their ecotoxic potential, making risk assessment more complex (Maia et al., 2018; Vu Quoc Hung et al., 2022).

On the economic point of view, non-compliant RAs create several challenges. To meet environmental safety standards, additional treatments, such as advanced washing, are often required, which significantly increase production costs (Peiris et al., 2025a). If these materials cannot achieve End-of-Waste status, they remain classified as waste, leading to higher disposal costs and reducing their market value (García et al., 2024). This situation undermines the competitiveness of recycled aggregates compared to virgin materials and can discourage investment in recycling technologies (Dias et al., 2022). Moreover, non-compliance may result in exclusion from public procurement processes and green-certified projects, limiting market opportunities (García et al., 2024). Producers also face increased administrative burdens, including more frequent testing and certification, which further raises operational costs (García et al., 2024; Diotti et al., 2020). Ultimately, these factors can lead to greater volumes of waste being sent to landfills, counteracting circular economy goals and diminishing the environmental benefits of recycling (Peiris et al., 2025a).

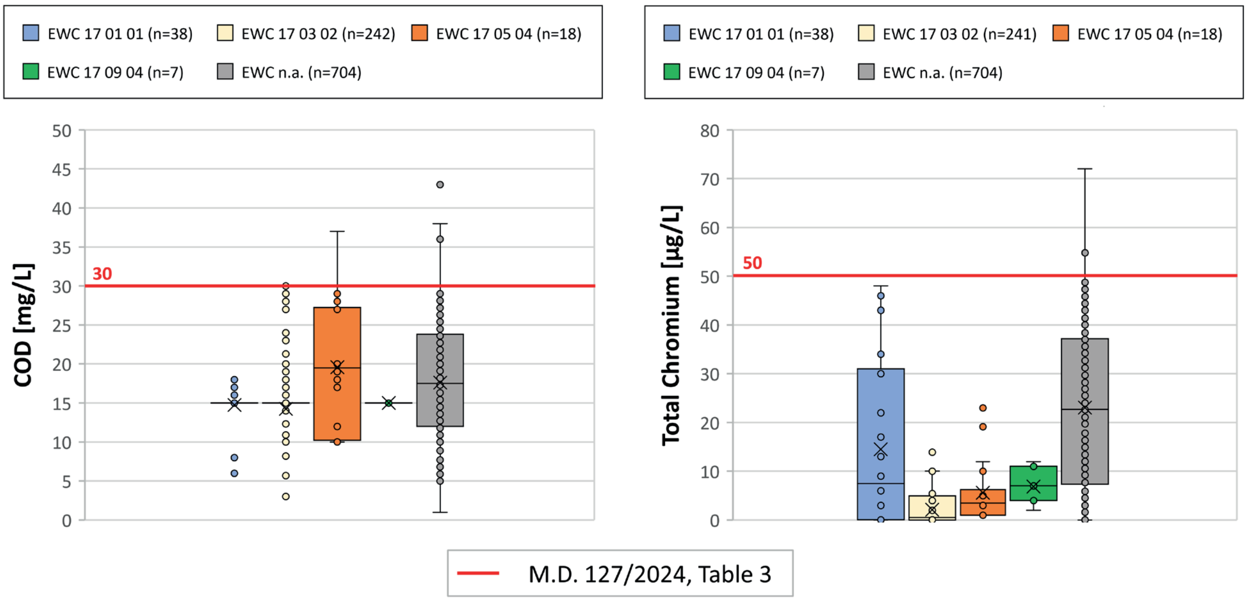

FIGURE 6: Effect of CDW type on COD and total Chromium release from RAs (n: number of analyses; EWC n.a.: no information about CDW treated to produce RA).

4. CONCLUSIONS

To promote waste reuse in the building sector, the Italian EoW requirements for inert waste were published in September 2024. There has been some news, mostly about the addition of different limit values for the pollutants' content depending on the kind of RA application.

This study aims to assess the impact of the EoW regulation's implementation on the viability of recovery by illustrating the quality of RAs in relation to the requirements established by the recent regulation. Therefore, a survey was carried out on 28 Italian recycling plants. The data collected (in 2020 - 2022 period) are referred both to the quality of waste treated by recycling plants (CDW) and to recycled aggregates produced (RAs) ones. About 2500 laboratory certifications (both for chemical analysis and leaching tests) are acquired.

Regarding the content of pollutants in RAs, the most critical parameters are heavy hydrocarbons (C>12), PAHs such as benzo(a)pyrene and benzo(g,h,i)perylene, and PCB. It is important to underline that the use of the collected data does not always allow to assess with certainty the non-compliances with the strictest limits established for the application of RAs in environmental reclamation and filling. In fact, the analysis results' detection limits are often greater than the more stringent threshold values.

The highest concentrations of heavy hydrocarbons and PAHs were detected in RAs from the treatment of bituminous mixtures (EWC 17 03 02). Moreover, for these wastes, there is already an EoW regulation (M.D. 69/2018) that has no requirement for heavy hydrocarbons in the final product (bituminous conglomerate granulate).

Even RAs obtained from soils and rocks (EWC 17 05 04) often show content of heavy hydrocarbons above the most restrictive limit (50 mg/kg_{DM}) and sometimes even above the least restrictive limit (750 mg/kg_{DM}), probably due to the presence of residues of bituminous mixtures during the soil excavation phase.

As concerns the release of pollutants from RAs, any critical issues are shown. For many parameters (pH, nitrates, COD, fluorides, total chromium, sulphates, barium, nickel, copper, mercury, zinc, and selenium) very few non-compliance (always below 1% of total samples) with EoW requirements.

RAs obtained from soils and rocks (EWC 17 05 04) show higher COD release, and RAs from cement waste (EWC 17 01 01) show higher release of total chromium, due to the presence of cementitious and concrete fractions.

Non-compliant recycled aggregates from CDW may leach heavy metals, sulphates, and PAHs, contaminating soil and groundwater. Economically, extra treatments and certifications increase costs, and failure to meet End-of-Waste standards results in landfill disposal and market exclusion. Overall, non-compliance weakens circular economy objectives and limits competitiveness in sustainable construction.

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REFERENCES

Bianchini, G., Ristovski, I., Milcov, I., Zupac, A., Natali, C., Salani, G.M., Marchina, C., Brombin, V., Ferraboschi, A., 2020. Chemical Characterisation of Construction and Demolition Waste in Skopje City and Its Surroundings (Republic of Macedonia). Sustainability, 12(5), 2055. https://doi.org/10.3390/su12052055

Butera, S., Christensen, T.H., Astrup, T.F., 2014. Composition and Leaching of Construction and Demolition Waste: Inorganic Elements and Organic Compounds. J Hazard Mater 276, 302-311. https://doi.org/10.1016/j.jhazmat.2014.05.033

Caro, D., Lodato, C., Damgaard, A., Cristobal, J., Foster, G., Flachenecker, F., Tonini, D., 2024. Environmental and socio-economic effects of construction and demolition waste recycling in the European Union. Sci Total Environ 168295. https://doi.org/10.1016/j.scitotenv.2023.168295

Cilli, E., Bruneaux, M.A., Chateau, L., Lucatelli, J.M., Peyratout, C., Smith, A., 2017. Cleanliness of Mixed Fired Clay Bricks Coming from Construction and Demolition Waste. Waste Biomass Valorization, 8, 2177-2185. https://doi.org/10.1007/s12649-016-9770-3

Dias, A., Nezami, S., Silvestre, J.S., Kurda, R., Silva, R., Martins, I., de Brito, J., 2022. Environmental and Economic Comparison of Natural and Recycled Aggregates Using LCA. Recycling, 7(4), 43. https://doi.org/10.3390/recycling7040043

Diotti, A., Pérez Galvin, A., Piccinali, A., Plizzari, G., Sorlini, S., 2020. Chemical and Leaching Behavior of Construction and Demolition Wastes and Recycled Aggregates. Sustainability, 12(24), 10326. https://doi.org/10.3390/su122410326

Diotti, A., Plizzari, G., Sorlini, S., 2021. Technical analysis of full-scale construction and demolition waste treatment plants: case studies of the Lombardy region, Italy. Detritus 15, 51-66. https://doi.org/10.31025/2611-4135/2021.14087

Egle, L., Milios, L., Cristóbal García, J., Saveyn, H., 2025. EU-wide End-of-Waste criteria for mineral construction and demolition waste. JRC draft technical proposal. Available online: https://www.gzs.si/Portals/Panoga-Gradbenistvo/Vsebine/novice-priponke/JRC_CDW-EOW_Technical%20proposal_draft_June2025.pdf (accessed on October 4, 2025)

European Commission: BRE, Deloitte, Directorate-General for Environment, FCT, ICEDD, RPS and VTT, 2017. Resource efficient use of mixed wastes improving management of construction and demolition waste – Final report, Publications Office. Available on line: https://data.europa.eu/doi/10.2779/99903 (accessed on October 4, 2025)

European Parliament, 2008. Directive 2008/98/EC of the European Parliament and of the Council of 19 November 2008 on waste and repealing certain Directives

European Parliament, 2018. Directive (EU) 2018/851 of the European Parliament and of the Council of 30 May 2018 amending Directive 2008/98/EC on waste

Eurostat, 2022. Generation of Waste by Economic Activity. Available online: https://ec.europa.eu/eurostat/databrowser/view/ten00106/default/table?lang=en (accessed on October 4, 2025).

Estokova, A., Palascakova, L., Kanuchova, M., 2018. Study on Cr(VI) Leaching from Cement and Cement Composites. Int J Environ Res Public Health 15, 824. https://doi.org/10.3390/ijerph15040824

Fondazione per lo sviluppo sostenibile, FISE UNICIRCULAR, 2021. The Italy of recycling, 2021. Available online: https://www.fondazionesvilupposostenibile.org/italia-del-riciclo-2021/ (accessed on September 19, 2025) (in Italian)

Gálvez-Martos, J.L., Styles, D., Schoenberger, H., Zeschmar-Lahl, B., 2018. Construction and Demolition Waste Best Management Practice in Europe. Resour Conserv Recycl 136, 166-178. https://doi.org/10.1016/j.resconrec.2018.04.016

García, C., Caro, D., Foster, G., Pristera, G., Gallo, F., Tonini, D., 2024. Techno-economic and environmental assessment of construction and demolition waste management in the European Union. JRC Technical Report Eur 31784 EN. https://doi.org/10.2760/721895

Government of Italy, 2006. Legislative Decree n. 152, 2006. Environmental Regulations (in Italian)

Imteaz, M.A., Arulrajah, A., Maghool, F., 2020. Environmental and Geotechnical Suitability of Recycling Waste Materials from Plasterboard Manufacturing. Waste Manag Res 38, 383-391. https://doi.org/10.1177/0734242X19881213

ISPRA, 2025. Report on Waste from Economic Activities - 2025, Reports 416/2025. ISBN: 978-88-448-1274-4. Available online: https://www.isprambiente.gov.it/it/pubblicazioni/rapporti/rapporto-rifiuti-speciali-edizione-2025 (accessed on September 19, 2025) (in Italian)

Italian Ministry of the environment and land protection, 2006. Ministerial Decree n. 186/2006. Regulation amending the Ministerial Decree of February 5, 1998 "Identification of non-hazardous waste subject to simplified recovery procedures, pursuant to articles 31 and 33 of the Legislative Decree n. 22 of February 5, 1997" (in Italian)

Italian Ministry of ecological transition, 2022. Ministerial Decree n. 152/2022. Regulation governing the end-of-waste status of inert waste from construction and demolition and other inert waste of mineral origin, pursuant to article 184-ter, paragraph 2, of Legislative Decree n. 152 of April 3, 2006 (in Italian)

Italian Ministry of environment and energy security, 2024. Ministerial Decree n. 127/2024 Regulation containing provisions for the cessation of the waste classification of inert waste from construction and demolition, other inert waste of mineral origin, pursuant to article 184-ter, paragraph 2, of Legislative Decree n. 152 of April, 2006 /2006 (in Italian)

Jandová, V., Bucková, M., Huzlík, J., Valentin, J., Effenberger, K., Svoboda, J., 2025. Release of PAHs from reclaimed asphalt mixtures into the water environment after passivation by cold in-place recycling technology. Environ Sci Pollution Res, 32, 2455-2466. https://doi.org/10.1007/s11356-024-35875-2

Juenger, M.C.G., Siddique, R., 2015. Recent advances in understanding the role of supplementary cementitious materials in concrete. Cement Concrete Res. 78, 71-80. http://doi.org/10.1016/j.cemconres.2015.03.018

Maia, M.B., de Brito, J., Martins, I.M., Silvestre, J.D., 2018. Toxicity of Recycled Concrete Aggregates: Review on Leaching Tests. Open Constr Build Technol J, 12, 187-196. https://doi.org/10.2174/1874836801812010187

Papadaki, D.; Nikolaou, D.A.; Assimakopoulos, M.N., 2022. Circular Environmental Impact of Recycled Building Materials and Residential Renewable Energy. Sustainability 14, 4039. https://doi.org/10.3390/su14074039

Peiris, D., Gunasekara, C., Law, D.W., Patrisia, Y., Tam, V.W.Y., Setunge, S., 2025a. Impact of treatment methods on recycled concrete aggregate performance: a comprehensive review. Environ Sci Pollution Res, 32, 14405-14438. https://doi.org/10.1007/s11356-025-36497-y

Peiris, D., Patrisia, Y., Gunasekara, C., Law, D.W., Tam, V.W.Y., Setunge, S., 2025b. Environmental impact of recycled aggregate treatment methods using life-cycle assessment and cost analysis. Int J Life Cycle Asses. https://doi.org/10.1007/s11367-025-02565-1

Reetsch, A., Tessien, J., Schröder, N., van 't Zelfde, J., Ooms, J., van Bruggen, R., Lundberg P., Zamparutti, T., 2024. Background Data Collection for Future EU EoW Criteria of Construction and Demolition Waste – 'Final Report. Available online: https://data.europa.eu/doi/10.2873/213625 (accessed on September 19, 2025)

Ruggeri, M., Pantini, S., Rigamonti, L., 2019. Assessing the impact of selective demolition techniques on C&D waste management. IOP Conference Series Earth and Environmental Science. https://doi.org/10.1088/1755-1315/296/1/012005

Sadler, R., Delamont, C., White, P., Connell, D., 1999. Contaminants in Soil as a Result of Leaching from Asphalt. Toxicol Environ Chem, 68, 71-81. http://doi.org/10.1080/02772249909358646

Silva, R.V., de Brito, J., Dhir, R.K., 2017. Availability and processing of recycled aggregates within the construction and demolition supply chain: A review. J. Cleaner Prod 143, 598-614. https://doi.org/10.1016/j.jclepro.2016.12.070

Sinyoung, S., Songsiriritthigul, P., Asavapisit, S., Kajitvichyanukul, P., 2011. Chromium Behavior during Cement-Production Processes: A Clinkerization, Hydration, and Leaching Study. J Hazard Mater 191, 296-305. http://doi.org/10.1016/j.jhazmat.2011.04.077

UNI EN 12457-2, 20024. Characterisation of waste – Leaching - Compliance test for leaching of granular waste materials and sludges - Part 2: One stage batch test at a liquid to solid ratio of 10 l/kg for materials with particle size below 4 mm (without or with size reduction) (in Italian)

Velzeboer, I., van Zomeren, A., 2017. End of Waste criteria for inert aggregates in member states. ECN Report. Available online: https://repository.tno.nl/SingleDoc?find=UID%20c7d4333a-45a8-4d57-bef9-f6ff84730dc1 (accessed on April 16, 2025).

Vu Quoc Hung, P., Egodawatta, P., Gallage, C., Dawes, L., 2022. Investigating Potential Environmental Issues of using Recycled Materials in Road Construction. Presented at EIRAI Conference.

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CRITICAL ANALYSIS OF SCOTTISH LOCAL AUTHORITIES BULKY WASTE SERVICES: IMPLICATIONS FOR LEAKS IN THE CIRCULAR ECONOMY MODEL

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ABSTRACT
Fly-tipping is the illegal disposal of waste onto land and can range from a bin bag of household waste to copious quantities of domestic, commercial or construction waste. A decade after the Paris Agreement 2015 on climate change, there is increased urgency to implement inclusive policies that can contribute to sustainable waste management services and reduce the environmental impacts of fly-tipping. This study explores the characteristics of bulky waste kerbside collection services through analysis of website disclosures from local authorities across Scotland. A database was constructed to identify potential barriers to service uptake and provide insights for policy makers and waste managers. It is crucial to ensure that materials from this discrete waste stream do not leak out of the circular economy approach being pursued by the Scottish Government. The findings highlight underlying factors that may influence resident engagement with bulky uplift services and emphasise the importance of effective communication and inclusive policies for local authorities and waste managers. Our findings reveal critical gaps in accessibility, affordability, and operational design that may inhibit service uptake and contribute to fly-tipping. Recommendations include the adoption of inclusive booking and payment methods, progressive pricing policies, and coordinated digital solutions such as a unified smartphone app. These measures could support legitimate disposal behaviours, reduce environmental harm caused by fly-tipping, and strengthen Scotland's circular economy outcomes. Mitigating fly-tipping and improving bulky item disposal routes can reduce greenhouse gas emissions from landfill and support Scotland's climate targets by retaining products and materials in circulation.

1. INTRODUCTION

Scotland is positioned as a forerunner in implementing the circular economy model, (Zero Waste Scotland, 2024a). This leadership has been reinforced by recent legislation with the introduction of the Circular Economy (Scotland) Bill 2024, (NetRegs, 2024). Waste management services in the UK, including Scotland, have been identified as the 26th best managed worldwide, (Statista, 2023). Despite this strong position, illegal disposal methods such as fly-tipping continue to represent significant 'leaks' within otherwise controlled waste systems. Globally, the circular economy model faces systemic leaks when materials escape controlled collection pathways through illegal disposal, inadequate infrastructure, or informal trade routes, (Cook & Velis, 2021; Song et al., 2023; UNEP, 2024). These leaks commonly involve household waste, construction and demolition debris, which together account for a significant share of uncontrolled waste flows worldwide, (Song et al., 2023; UNEP, 2024). Such losses compromise resource efficiency, increase greenhouse gas emissions from landfill, and perpetuate environmental injustice in regions with limited waste management capacity, (Cook & Velis, 2021; UNEP, 2024). Figure 1 illustrates fly-tipping as a leak from the circular economy model, occurring at the nexus of consumption and collection.

The circular economy model aims to retain materials of value within circulation promoting economic growth while reducing environmental impacts, (Takahashi, 2020). However, waste management problems are widely recognised as 'wicked issues' due to their social and political complexity, (Salvia et al., 2021). In this context, fly-tipping is not merely a behavioural issue but a symptom of systemic barriers to legitimate disposal. Framing fly-tipping as a leak from the circular economy model allows targeted analysis of specific waste streams and their disposal routes with a goal of mitigating leakage through improved service

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FIGURE 1: Fly-tipping as a leak from the circular economy model.

design and communication. Fly-tipping exemplifies these systemic leaks with waste composition varying across contexts but typically dominated by household bulky items alongside mixed household waste and construction materials, (Comerford et al., 2018; Hodsman & Williams, 2011; Scottish Government, 2023b). Globally, these uncontrolled flows undermine circular economy objectives by diverting recoverable materials into landfill or informal processing, increasing emissions and eroding the integrity of waste systems, (Song et al., 2023; UNEP, 2024). Similar challenges have been documented internationally, where bulky waste management and fly-tipping persist despite formal service provision. Studies from Europe, Australia, and North America highlight recurring issues of access, affordability, and communication, reinforcing the need for comparative insights to inform local policy development, (Comerford et al., 2018; Keske et al., 2018; Larsen et al., 2012).

Household bulky items are the discrete waste stream most frequently encountered in fly-tipping incidents across Scotland, (Scottish Government, 2023b). Bulky items or waste are items of furniture, white goods, e-waste or any item you would take with you when moving house, (Hodsman and Williams, 2011). These items can be difficult to dispose of due to their material composition and size, (Alexander et al., 2009a; Curran et al., 2007; Keske et al., 2018). Our study analysed the information disclosed on local authority websites to provide a narrative reflection on the underlying factors that influence service uptake. Ensuring that bulky items are collected by dedicated services mitigates their potential for fly-tipping which reduces economic costs to governments and environmental impacts, (Liu et al., 2017). The insights generated are relevant to policymakers and waste managers in Scotland and beyond, as fly-tipping of bulky items is a global issue requiring effective and transparent communication strategies for disposal services.

1.1 Background

Fly-tipping is the illegal dumping of waste and can range from a bin bag of household waste to large quantities of domestic, commercial or construction waste, (Scottish Government, 2023a). It is a global phenomenon that manifests heterogeneously dependent on the availability of waste management services. Developed countries with waste management services experience fly-tipping as an uncontrolled leak from a controlled system whereas developing countries lack basic waste management services and experience fly-tipping as large, open dumpsites, (Song et al., 2023; UNEP, 2024). A decade after the Paris Agreement 2015 on climate change, the urgency to implement sustainable waste management policies has intensified, (Dixon et al., 2022). Waste management is a critical system for climate action as the sector contributes approximately 4% of Scotland's greenhouse gas emissions and up to 15-20% globally, (Scottish Government, 2024; UNEP, 2024). Aligning disposal routes with circular economy principles reduces reliance on landfill, which is a major source of methane emissions, and promotes reuse and recycling pathways that conserve resources and lower carbon intensity, (Curran et al., 2007; Takahashi, 2020). Sustainable waste management directly supports climate targets under the Paris Agreement by reducing emissions associated with extraction, production, and waste degradation, (UNEP, 2024). Furthermore, the leak from the circular economy model identified in our study primarily consist of household bulky items (furniture, white goods, and e-waste) that escape formal collection systems and enter uncontrolled disposal routes. While precise quantities are unknown, bulky items are consistently reported as the most common materials in fly-tipping incidents across Scotland, followed by mixed bags of household waste then construction and demolition debris, (Scottish Government, 2023b). National estimates suggest that fly-tipping accounts for approximately 26,000 tonnes of waste annually, highlighting the potential scale of resource loss when bulky items are diverted from reuse or recycling pathways, (Scottish Government, 2023a; Zero Waste Scotland, 2017). Once dumped, these items are typically damaged and sent to landfill, eliminating any opportunity for recovery and increasing greenhouse gas emissions from decomposition, (Curran et al., 2007; Liu et al., 2017). Beyond environmental harm, the leak from the circular economy model imposes substantial economic costs, with clearance alone estimated at £8.9–12.7 million annually, (Scottish Government, 2023a, 2023b; Zero Waste Scotland, 2017). Addressing this leak of household bulky items is therefore critical to maintaining circular economy integrity and achieving climate and resource efficiency goals.

Scotland has pursued ambitious zero waste plans for over a decade, including the rollout of segregated kerbside recycling in 2012, (Griffiths et al., 2013). Zero waste strategies aim to reduce, reuse, and recycle all municipal waste streams in alignment with circular economy principles, (Scottish Government, 2020b; Zero Waste Scotland, 2024a). The Circular Economy (Scotland) Bill 2024 strengthens this framework by mandating a national strategy and introducing enhanced enforcement measures, such as increased fines and vehicle seizure powers for fly-tipping offences, (NetRegs, 2024). These developments reflect Scotland's commitments to climate action and emphasise the importance of addressing disposal routes for household bulky items as a waste stream frequently found in fly-tipping incidents, (Scottish Government, 2023b). Bulky items, including furniture, white goods, and e-waste, pose unique challenges due to their size, material complexity, and incompatibility with standard collection systems, (Hodsman and Williams, 2011; Keske et al., 2018). Their improper disposal not only contributes to environmental degradation but also incurs significant economic costs (Liu et al., 2017). As Scotland advances its circular economy ambitions, ensuring that bulky items are diverted from illegal disposal and integrated into circular reuse or recycling pathways is essential to reducing waste and supporting climate goals. Diverting bulky items from illegal disposal and integrating them into reuse or recycling systems prevents the loss of recoverable products and materials, avoiding emissions from landfill, reinforcing the link between waste management and climate action, (Curran et al., 2007; Scottish Government, 2024).

1.2 Economic and environmental impacts of fly-tipping

Fly-tipping is prohibited under section 33 (1) of the Environmental Protection Act 1990, (Murdo, 2022). Despite this, Scotland records between 60,000 and 66,000 fly-tipping incidents annually with estimated clear up costs of between £8.9m to £12.7m, (Scottish Government, 2023a, 2023b; Zero Waste Scotland, 2017). These figures likely underestimate the true cost, as incidents on private land are not systematically recorded and data collection by local authorities can be inconsistent, (Dixon et al., 2022; Scottish Government, 2023a). When indirect costs are included, albeit aggregated with littering costs, the total economic burden rises to an estimated £280.8million annually, (Scottish Government, 2023b). This represents a significant strain on public finances and taxpayers, (Webb et al., 2006; McNeill et al., 2021), especially in the context of an ongoing cost-of-living crisis, (Scottish Affairs Committee, 2024). These financial pressures underscore the importance of understanding the factors that influence uptake of legitimate waste services and how improved facilitation could reduce both economic and environmental impacts.

Beyond its financial burden, fly-tipping poses serious environmental and public health risks. The waste management sector contributes 4% of Scotland's greenhouse gas (GHG) emissions, making sustainable resource use essential to meeting climate targets, (Scottish Government, 2024). Fly-tipping is a serious source of pollution as the materials can contain substances that are destructive to land, the habitats of wildlife and human health, (Liu et al., 2017; Purdy et al., 2022; Song et al., 2023). Pollutants leaching from fly-tipping sites can contaminate soil and freshwater sources damaging fragile ecosystems, (UNEP, 2024). When dumped in or near our waterways, this can contaminate water sources or create flood risks by blocking rivers, (Purdy et al., 2022). Even after removal, items damaged by the illegitimate dumping process will always be destined for landfill or recovery despite any pre-dumping potential for reuse or recycling, (Curran et al., 2007). These outcomes undermine Scotland's circular potential and highlight the need for scrutiny of disposal routes, (Cook and Velis, 2021). Accordingly, our study analyses website disclosures from Scottish Local Authorities (SLA) concerning kerbside collection services for household bulky items given their prevalence in fly-tipping incidents across Scotland.

1.3 Scottish local authorities waste management services for household bulky items

Waste management in Scotland is delivered by thirty-two local authorities, commonly known as councils, responsible for providing multiple public services to the communities they serve. Household bulky items have two local authority disposal routes. There are dedicated facilities called Household Waste Recycling Centres (HWRCs) in every region where residents can transport their items for free, and there are chargeable kerbside collection services where residents can have bulky items picked up in proximity to their property. There are also other informal bulky item disposal routes which include to sell or donate. However, this study focuses upon formal disposal routes, specifically the kerbside collection service due to logistical challenges many residents face transporting bulky items. Bulky items form a discrete waste stream with significant implications for circular economy outcomes, (Scottish Government, 2016). However, their importance is often obscured by broader recycling targets and limitations in waste composition data, (Chung et al., 2010). Our study is the first known analysis of SLA household bulky item kerbside collection services. By examining service design and accessibility, we aim to identify opportunities to reduce fly-tipping and offer insights for policymakers in Scotland and other regions facing similar challenges.

2. METHODS

2.1 Setting and study design

Scotland is one of four countries within the United Kingdom. It shares a border with England and has over eight hundred islands, with an approximate population of 5.48 million, (Statista, 2024a). Some of the largest areas have the smallest population densities, (Scottish Government, 2023b). The country's diverse geography and population density result in regional variations in waste management practises. To account for these differences, this study uses the Rural & Environment Science & Analytical Services (RESAS) classification system, which categorises regions as urban city, urban with significant rural, rural, and highland and island (Scottish Government, 2018). This framework supports context-specific analysis of service provision and resident needs. To complement this, we used Scotland's census data on property types, and multiple deprivation using the national index to explore how these factors affect the capacity to store bulky items and influence disposal behaviours.

2.2 Data extraction

Website disclosures on SLA kerbside bulky item collections, known as and referred to hereafter as bulky uplifts, were accessed and analysed to consider the resident-facing process for service provision. Data was extracted from the websites of all 32 SLA. Data collection occurred in two phases: February to March 2024 and April to May 2024, with accuracy checks in September 2024 and March 2025. This allowed for two consecutive financial years to be incorporated into the analysis exploring temporal variations. The database compiled includes service details such as booking methods, accepted items, charges, payment options, and waiting times, alongside data on population density, deprivation levels, and property types.

2.3 Standardised charge benchmarking

To compare service costs across SLA, a standardised charge (£/item) was calculated based on a 'maximum utility' assumption. Maximum utility entails residents storing items until they have the maximum number of items collected per uplift arranged, (Curran et al., 2007). Several SLA with non-comparable pricing structures were excluded. While this approach simplifies complex pricing models, it provides a useful benchmark for understanding cost variation.

2.4 Smartphone app availability

Smartphone app availability was assessed by searching Apple and Android app stores in April 2024, with follow-up checks in August 2024 and March 2025. The analysis considered whether SLA offered apps, their functionality, developer details disclosed, and whether they supported bulky item bookings or fly-tipping reporting features.

2.5 Literature review contextualisation

A literature review was conducted using Scopus and Google Scholar to identify research on household bulky waste and fly-tipping. Three search strings were used:

- "household" AND "bulky" AND ("items" OR "waste");
- ("kerbside" OR "curbside") AND "bulky" AND "collection";
- "illegal dumping" AND "household" AND ("bulky" AND "items" OR "waste").

The review included peer-reviewed and grey literature to ensure broad coverage with articles from several databases: Emerald, Elsevier, JSTOR, Sage, SciFlo, SciSpace, Taylor and Francis, and Wiley. While current literature is important to frame the issue within contemporary thinking, applying a date range arbitrarily was not considered appropriate for this study. A flow diagram provided in Figure 2 illustrates the screening and inclusion process.

The review conducted was systematic in approach however, it does not meet strict criteria for a formal Systematic Literature Review (SLR) due to the inclusion of non-academic sources. While not presented as a standalone section, insights from this review informed the framing of the research and are referenced throughout the results and discussion.

3. RESULTS AND DISCUSSION

3.1 Scottish local authority bulky uplifts

Bulky uplifts are a separate service from regular kerbside collections and must be booked by contacting the local authority. Despite online assertions about free uplifts available, (Zero Waste Scotland, 2021), this is a service which is charged for by all but one of the SLA. This could be quite provocative or frustrating for residents who access a local authority website and find that free bulky uplifts are not available in their region. Especially when we consider that charges are in addition to council tax that residents already pay for other waste services, (Curran et al., 2006; McNeill et al., 2021). Failure to capture reusable bulky items within controlled systems not only undermines circular economy goals but also increases emissions associated with landfill and replacement production, (Curran et al., 2007; UNEP, 2024). Evidence from other regions demonstrates that the barriers identified in Scotland are not unique. In England, research found that charges for bulky uplifts and limited booking options reduced service uptake and increased the risk of fly-tipping, (Curran et al., 2006; Hodsman & Williams, 2011). Studies in Australia and Canada reported similar patterns, with seasonal free collection schemes improving compliance and reducing illegal dumping, (Comerford et al., 2018; Keske et al., 2018). In Denmark, monthly kerbside collections in urban areas were associated with higher reuse potential, while rural regions continued to face logistical challenges, (Larsen et al., 2012). These findings align

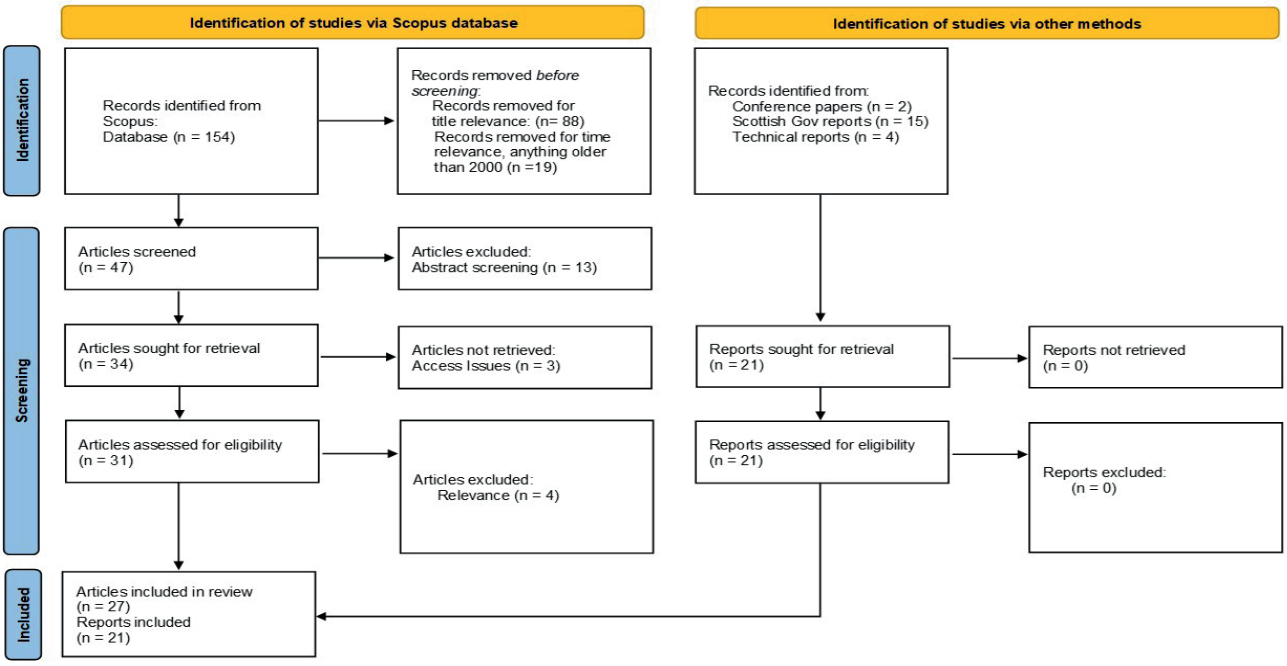

FIGURE 2: Systematised literature review flow diagram.

with our results, indicating that service design, affordability, and communication are critical determinants of legal disposal behaviour across diverse contexts. Addressing these factors through inclusive policy development is therefore essential to mitigate systemic leaks from the circular economy model. Figure 3 provides our resident facing service cycle for SLA bulky uplifts.

The service cycle provided has been used to systematically analyse SLA website disclosures for the points of contact identified. The analysis of SLA website disclosures leads to a better understanding of the resident-facing process for services and captures the underlying factors that require consideration for facilitation of bulky uplifts. Household bulky items are the most frequently encountered materials in fly-tipping incidents in Scotland, (Scottish Government, 2023b). The issue of household bulky items in fly-tipping incidents is not just a UK phenomenon, it has been observed as a source of fly-tipping or associated with

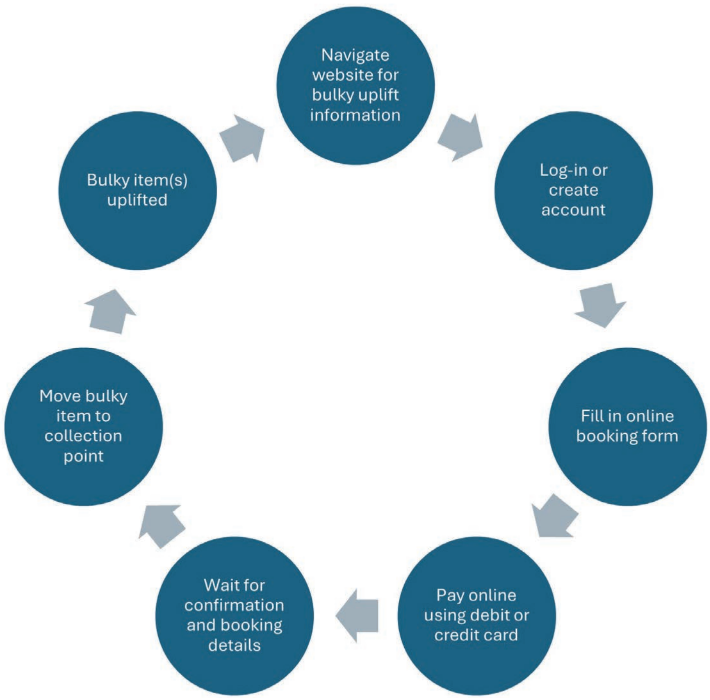

FIGURE 3: Service cycle for bulky uplifts provided by Scottish local authorities.

problematic disposal routes worldwide, (Comerford et al., 2018; Guérin et al., 2018; Hodsman and Williams, 2011; Ito and Colombo, 2019; Song et al., 2023; Vimpolšek et al., 2022). Our study analysed the resident facing process for bulky uplifts by examining SLA website disclosures. It must be acknowledged that the insights generated from our analysis of the SLA websites were critical assessment rather than formally derived opinions and perspectives of residents across Scotland.

3.2 Scottish local authority website navigation

Website navigation is a critical component of service accessibility. All SLA websites include a 'bins and recycling' tab on their homepage, yet locating specific information on bulky uplifts often requires navigation of multiple other pages. When considering bulky uplifts, charges, accepted items, and the process for both booking and receiving the service will be of key interest to residents. The layered structure of website pages could hinder service uptake, particularly for residents with limited digital fluency, (Purdy et al., 2022). Transparent communication is essential to facilitate informed decision-making around waste disposal, (Curran et al., 2006). The information provided on SLA websites should be transparent concerning service processes and provide residents further options for bulky items, such as reuse organisations and charities, (Curran and Williams, 2010; Cox et al., 2010; Zero Waste Scotland, 2024b). However, only 44% of SLA provide direct links to reuse organisations or charities, with a further 12% mentioning them without actionable guidance. This leaves a significant gap in the promotion of sustainable alternatives to end-of-life disposal for household bulky items. Given the diversity of resident preferences, SLA should consider expanding communication channels beyond websites. Options such as printed leaflets, social media, and smartphone apps could improve outreach and inclusivity, (Ito and Colombo, 2019).

3.3 Smartphone app availability across Scotland

Smartphone ownership in the UK reached 94% in 2024, (Statista, 2024b), positioning mobile platforms as a key channel for communicating public service delivery. Across Scotland, only 25% of SLA currently offer smartphone apps for public waste services information. These vary in functionality, with some supporting fly-tipping and other waste issue reporting, and others providing bins and recycling information. While no apps currently support bulky uplift bookings, this reflects the early stage of digital development in the sector and presents a timely opportunity for SLA to commit resources to a collaborative effort to define future app functionality. Our smartphone app search revealed that there are three apps which have been developed in-house by SLA, and one provided by an independent specialist company. In-house app development demonstrates initiative and SLA responsiveness to local needs. However, without coordination, this could result in fragmented systems for residents. A unified approach, led by a dedicated specialist, offers greater potential. Albion Environmental Ltd are a UK leading consultancy who specialise in waste management, environmental, health and safety solutions. As leading developers of a bins and recycling smartphone app, they are uniquely positioned to advance this effort. They provide a digital solution based on sector-specific expertise which is a foundation for scalable development. A single integrated platform could support user authentication, provide real-time updates when residents move between SLA regions, and offer consistent access to bin schedules, booking processes, and reuse options. Furthermore, such a system could be co-designed with SLA and residents to reflect local priorities and ensure inclusive service delivery nationwide.

Features such as image uploads, Apple/Android Pay integration, and notification features could enhance reuse potential while also generating valuable data to inform service improvements, (Wang et al., 2020; Salim et al., 2023). The booking process for bulky uplifts could be enabled with notifications and reminders of bookings. Features used in the gig-economy for estimated arrival times could be used by collection crews to provide further information to residents with bookings, (Purdy et al., 2022). Pro-active communications at key seasonal points, like Christmas and Spring sales, could facilitate reuse options for bulky items, (Alexander et al., 2009a; Keske et al., 2018), and notification features of smartphone apps could increase awareness of reuse opportunities for residents at these strategic moments. Furthermore, collaboration between SLA, reuse organisations, and charities could encourage diversion of reusable bulky items from SLA services simultaneously increasing environmental and social benefits, (Alexander and Smaje, 2008; Cox et al., 2010; Curran and Williams, 2010; Zero Waste Scotland, 2024b). Investment in a national digital solution would support Scotland's circular economy ambitions by improving accessibility, communication, transparency, and resident engagement. SLA should consider collaborative development of a smartphone app for public services using the sector-specific expertise available to ensure equitable service provision, avoid duplication of effort, and to conserve scare financial resources. However, reliance on digital solutions must also be balanced with other communication methods such as regular leaflets to account for the preferences of diverse populations, (Ito and Colombo, 2019). The key objective is to facilitate sustainable behaviour through provision of information using methods of communication that reflect the diverse needs of residents, designed in collaboration with experts in the industry and experts in lived experience.

3.4 Booking methods for bulky uplifts

Booking methods for bulky uplift services vary across SLA, with notable spatial and temporal differences. While online booking is the most common method, some SLA also offer phone, email, postal, or in-person options. Over the two financial periods analysed (2023-24 and 2024-25), several SLA expanded their booking channels, while others reduced them potentially limiting accessibility for residents. Figure 4 illustrates the booking methods available over the periods assessed.

The figure provides an overview of all SLA booking methods, however the spatial variations suggest that urban SLA predominantly rely on online-only booking, whereas rural, highland, and island regions tend to offer more

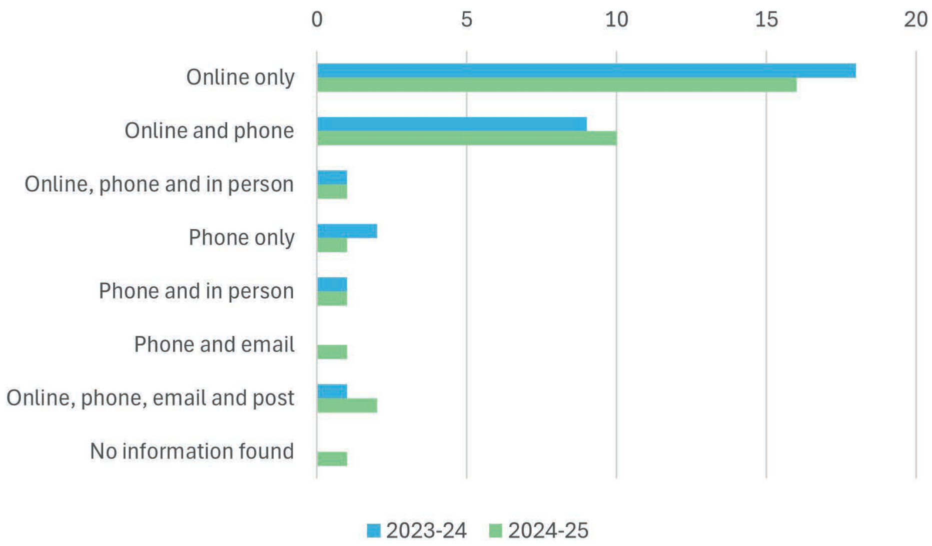

FIGURE 4: Overview of booking methods for bulky uplifts provided by Scottish local authorities over the periods 2023-24 and 2024-25.

varied options. However, one SLA removed in-person booking entirely, and another provided no booking information during the second period. These inconsistencies may create confusion and deter service uptake, particularly among residents who prefer or require non-digital communication methods (Ito and Colombo, 2019; Purdy et al., 2022). To improve accessibility and align with circular economy goals, SLA should offer a variety of booking options that reflect resident preferences. Integrating booking functionality into a unified smartphone app which could be co-designed with SLA and residents, would streamline the process, support authentication, and enable reuse assessment through image uploads (Wang et al., 2020). This would reduce barriers to service use and support legitimate disposal routes for bulky items.

3.5 Charges for bulky uplifts

Charges for bulky uplift services vary significantly across SLA, with most applying a minimum charge based on item volume; 75% increasing to 78% over the periods observed. While one SLA offers the service free of charge to all residents, the remaining authorities apply a range of pricing structures, including minimum charges, itemised lists, time-based charges, and blended models. These differences can be confusing for residents, especially when items like a three-piece suite are counted as multiple units for collection, (Purdy et al., 2022). Furthermore, residents may be unwilling to pay for services, (Alexander et al., 2009a). Resident perceptions in similar contexts have been that charges in addition to existing payments for regular waste management services are unreasonable, (Cox et al., 2010; Curran et al., 2006). The impact of charges is the decrease of demand for the service, not the decrease in need for the service, (Curran et al., 2006; Hodsman and Williams, 2011; Purdy et al., 2022). To enable comparison, a standardised charge (£/item) was calculated using a 'maximum utility' assumption, where residents will store items until the maximum number of items allowed per uplift is reached, (Curran et al., 2007). Table 1 presents the spatial and temporal variation in both minimum and standardised charges across four SLA classifications: urban cities, urban with significant rural, rural, and highland and island regions.

Table 1 shows that charges increased in most regions between 2023-24 and 2024-25, with rural and island communities consistently facing higher costs. This 'rural premium' reflects the logistical challenges of servicing large, sparsely populated areas, (Curran et al., 2006; Óskarsson et al., 2022). However, rising costs may discourage legal disposal and increase the risk of fly-tipping, particularly in areas with high deprivation, (Curran et al., 2006). SLA should exercise caution when adjusting charges and consider the broader implications for service uptake. Progressive pricing policies, such as free or discounted uplifts for low-income households, could reduce fly-tipping and support circular economy goals, (Tilley, 2020). This has been observed in countries where the fly-tipping of household waste has never been a major policy issue, (Ichinose, Yamamoto and Yoshida, 2013). To ensure equitable access to bulky uplift services, SLA should consider whether free bulky uplifts for all residents would mitigate fly-tipping

TABLE 1: Spatial and temporal variation of minimum charge and standardised charges (£/item) for Scottish local authority bulky uplifts for the periods 2023-24 and 2024-25.

	Urban cities		Urban with significant rural		Rural		Highland & island	
	2023-24	2024-25	2023-24	2024-25	2023-24	2024-25	2023-24	2024-25
Minimum charge	£17.25	£20.50	£33.40	£36.75	£30.90	£32.31	£39.50	£42.30
Standard charge	£5.58	£6.33	£5.21	£6.12	£7.89	£7.81	£7.57	£8.03

costs sufficiently to be viable. Even a free seasonal service can assist with waste reduction strategies and prove popular with residents, (Brosius et al., 2013; Comerford et al., 2018; Keske et al., 2018). If a universal free service cannot be justified, SLA should, at a minimum, ensure that pricing strategies account for both regional disparities and household income levels. This includes targeted discounts and benefits for residents facing financial hardship, particularly in areas with high deprivation. Such measures would support legitimate disposal behaviours and reduce the risk of fly-tipping, while aligning with Scotland's circular economy ambitions.

3.6 Discounts and free uplifts provided by Scottish local authorities

The availability of discounts or free bulky uplift services varies significantly across SLA, with notable spatial and temporal differences. Analysis of website disclosures from 2023-24 and 2024-25 revealed that most SLA do not offer financial support for residents on income-related benefits or in council housing. This is particularly concerning given the high levels of deprivation in many regions. According to the Scottish Index of Multiple Deprivation (SIMD), urban cities show deprivation levels ranging from 10.2% to 45.4%, while urban with significant rural areas range from 3.8% to 44.7%. Rural regions range from 2.6% to 31.3%, and highland and island communities from 0% to 10.4%, (Scottish Government, 2020a). These figures highlight the need for targeted support in areas where residents may face financial barriers to accessing legitimate disposal routes. Figure 5 illustrates the spatial and temporal variation of the availability of discounts or free bulky uplifts provided by SLA over the periods analysed.

As shown in Figure 5, urban SLA with higher deprivation levels consistently offered no discounts or free uplifts across both periods. In urban with significant rural regions, one SLA introduced a progressive policy offering free bulky uplifts to all residents, while another revoked a previously available set number of free uplifts. Rural SLA showed limited support, with only two offering targeted discounts or free uplifts. Highland and island SLA were similarly restrictive, with just one providing a discount for benefits recipients. These findings highlight a lack of consistency and equity in service provision. SLA serving areas with higher deprivation should prioritise inclusive discount and free uplift policies to ensure that financial barriers do not prevent residents from accessing legitimate disposal routes. Free or discounted bulky uplift services for low-income households could reduce fly-tipping, support circular economy goals, and promote social equity, (Curran et al., 2006; Hodsman and Williams, 2011; Purdy et al., 2022). To support transparency and uptake, SLA should clearly communicate eligibility criteria and integrate discount options into booking and payment systems. This offers further support for use of a smartphone app as a unified digital platform which could streamline these processes, ensuring that residents are aware of available support and can access services without unnecessary complexity.

3.7 Payment methods for bulky uplifts

Payment methods for bulky uplift services are inconsistently disclosed across SLA websites. Analysis from 2023-24 and 2024-25 shows that most SLA do not specify accepted payment options, with 22 out of 32 failing to provide this information in the latter period. This lack of transparency may deter residents from using the service, particularly those who rely on cash or alternative payment methods. In Scotland, cash remains more commonly used than in other parts of the UK, (UK Parliament, 2022), and digital exclusion continues to affect vulnerable groups. Restricting payment to debit or credit card alone may unintentionally exclude residents who do not have access to online banking or who prefer in-person transactions. This is especially concerning in the context of a cost-of-living crisis, where financial flexibility is crucial, (Scottish Affairs Committee, 2024). To improve accessibility and uptake, SLA should clearly communicate available payment options and consider offering a range of methods, including cash, PayPal, and mobile payment systems such as Android/Apple Pay and Google Pay. Integration of payment functionality into a unified smartphone app could further streamline the process, allowing residents to complete bookings and

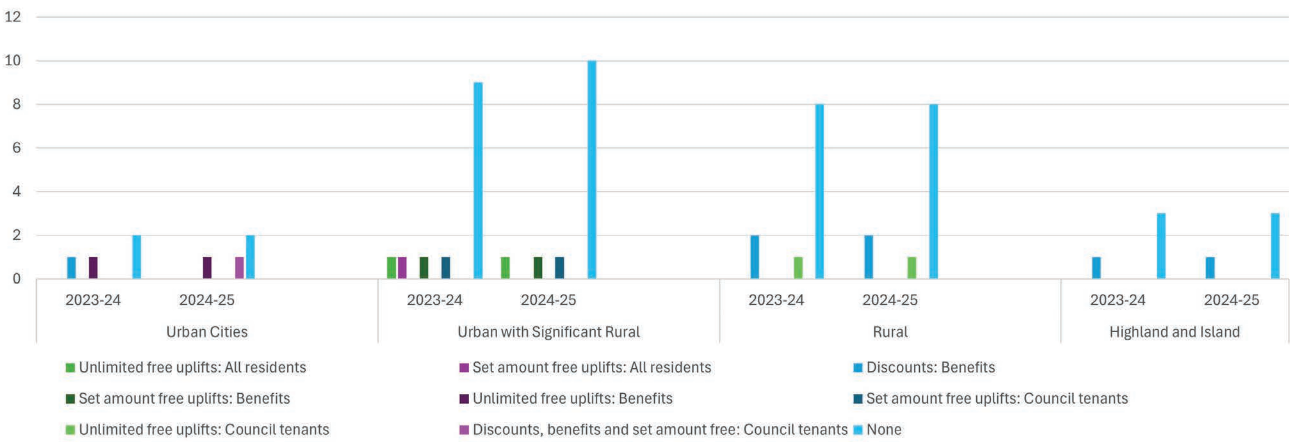

FIGURE 5: Spatial and temporal variation of the availability of discounts or free bulky uplifts provided by Scottish local authorities over the periods 2023-24 and 2024-25.

payments in one place. Inclusive payment design is essential to ensure that all residents can engage with bulky uplift services regardless of digital access or financial circumstances. Transparent and flexible payment systems would support equitable service provision and reduce barriers to legitimate disposal routes.

3.8 Waiting times for bulky uplifts

Waiting times for bulky uplift services are inconsistently disclosed across SLA websites. Analysis from 2023-24 and 2024-25 shows that 75% of SLA did not provide a clear timeframe for service delivery. Where timeframes were specified, they ranged from within 7 days to up to 28 days, with longer durations more common in highland and island regions as illustrated in Figure 6.

Waiting times can be a source of dissatisfaction for residents, (Curran et al., 2007; Hodsman and Williams, 2011). Uncertainty around waiting times may discourage residents from using legitimate disposal routes, particularly when alternative, informal options legitimate or otherwise offer faster collection, (Purdy et al., 2022). This is especially problematic for residents with limited storage space, such as those in flats or temporary accommodation, who may struggle to retain bulky items for extended periods (Alexander et al., 2009b). Clear communication of expected waiting times is essential to support service uptake and reduce the risk of fly-tipping. SLA should consider publishing standardised timeframes and offering real-time updates through digital platforms. Integration of booking and scheduling into a unified smartphone app could allow residents to receive notifications, estimated arrival times, and reminders which would mirror similar features used in the gig economy, (Purdy et al., 2022). Improved transparency around waiting times would enhance resident certainty, support planning, and contribute to more efficient service delivery. It would also help preserve the reuse potential of items by reducing the likelihood of damage during prolonged outdoor storage.

3.9 Collection point for bulky uplifts

All SLA require residents to place bulky items outside at their usual kerbside collection point. While this standardised approach simplifies logistics, it presents challenges for residents who may struggle to move large items safely, particularly older residents or those with mobility issues, (Curran et al., 2006; Hodsman and Williams, 2011; Purdy et al., 2022). Outdoor placement also risks damage to items from weather exposure or vandalism, which can compromise their reuse potential. Once damaged, items are typically diverted to landfill or recovery, even if they were originally suitable for reuse, (Alexander et al., 2009a; Curran et al., 2007). This undermines circular economy goals and increases the environmental impact. Charities and reuse organisations often collect items from inside homes, preserving their condition and extending their life cycle, (Hodsman and Williams, 2011; Wilkinson and Williams, 2020). SLA could adopt similar practises by training collection crews to safely retrieve items from within properties, particularly for vulnerable residents. This would improve accessibility, reduce health and safety risks, and support reuse outcomes.

In multiple household buildings, shared waste areas and limited storage space within individual properties further complicates bulky item disposal. Collaboration between SLA, housing associations, and landlords could help establish appropriate collection points and bespoke procedures for property types and resident needs, (Bernstad et al., 2012; Larsen et al., 2012; Ordoñez et al., 2015). Revisiting collection point policies with a focus on inclusivity and reuse potential could enhance service uptake and reduce fly-tipping. A flexible, resident-centred approach would better align with Scotland's circular economy ambitions and support equitable bulky waste management.

3.10 Spatial variation of property types across Scotland

Property type plays a significant role in shaping bulky waste disposal behaviours and service accessibility. Data

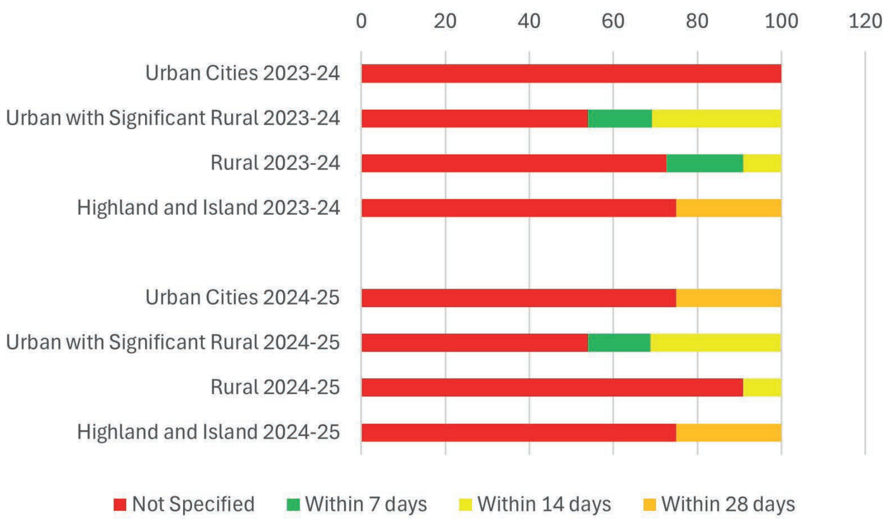

FIGURE 6: Spatial and temporal variation of waiting time disclosed for bulky uplifts by Scottish local authorities over the periods 2023-24 and 2024-25.

from Scotland's Census (2022) reveals distinct spatial patterns across SLA classifications. Urban cities and urban with significant rural regions have high population density and the greatest proportion of multiple household buildings, with deprivation levels ranging from 3.8% to 45.4%. In contrast, rural, highland and island regions are characterised by lower population densities, higher proportions of detached homes, and lower deprivation levels ranging from 0% to 31.3% (Scottish Government, 2020a). Our findings concerning area size, population density, and property type distribution are illustrated in Table 2.

Residents in multiple household properties, more commonly known as flats, often face constraints in storing bulky items, particularly when waiting for uplift services. Shared waste areas and limited indoor space can increase the likelihood of informal disposal, especially when waiting times are long or collection points are impractical, (Alexander et al., 2009b; Purdy et al., 2022). However, in high-density urban areas, placing bulky items on pavements poses a significant risk to public health and safety, (Alexander et al., 2009a). Residents in detached homes may have greater storage capacity, allowing them to use a 'maximum utility' approach however, outdoor storage can compromise reuse potential, (Curran et al., 2006). These spatial differences underscore the need for tailored service design. SLA should consider property type when developing collection policies, booking systems, and communication strategies. Collaboration with residents, housing associations and landlords could help establish appropriate collection points and support residents in multiple household buildings, (Bernstad et al., 2012; Larsen et al., 2012; Ordoñez et al., 2015). A resident-centred approach that accounts for property type and spatial context would improve service accessibility, reduce fly-tipping, and support Scotland's circular economy ambitions.

4. CONCLUSIONS

Our study on SLA website disclosures for bulky uplifts revealed critical gaps in accessibility, communication, and transparency that are presented for consideration of policy makers and waste managers. As these websites are the primary source of information for residents, it is crucial that they are easy to navigate and provide enough knowledge to assist in decision making, (Alexander et al., 2009a; Curran et al., 2006; Purdy et al., 2022). Smartphone apps offer promising opportunities to streamline booking, payment, and item classification, potentially enhancing reuse and reducing waste, (Purdy et al., 2022; Salim et al., 2023; Wang et al., 2020). Moreover, a national digital solution provides a unique opportunity for SLA to use sector-specific expertise and collaboration to ensure effective communication of public services. However, reliance on digital methods must be balanced with inclusive access for all residents. Offering a variety of communication, booking, and payment methods could facilitate uptake of bulky items kerbside collection services.

Charging structures vary across SLA, with most applying a minimum charge based on item volume. Rural, highland, and island communities often facing a rural premium with higher charges, (Curran et al., 2006; Óskarsson et al., 2022). These disparities may discourage legal disposal and increase the risk of fly-tipping, ultimately raising public clean-up costs. While some progressive policies do exist, with one region providing the bulky uplift service free of charge to all residents, these remain limited. Over the periods observed, nominal increases to charges were made that were reflected in the increases in standardised charges (£/item) over most regions. Charge increases should be exercised with caution as increasing costs for residents could negatively influence the demand for bulky uplifts, though not necessarily the need for them, (Curran et al., 2006; Hodsman and Williams, 2011; Purdy et al., 2022). Our findings suggest that free bulky uplift services should be considered by SLA, especially given the considerable clearance costs for fly-tipping. Furthermore, discounted bulky uplift services for low-income households should be a minimum standard, especially in regions with high levels of deprivation, (Curran et al., 2006; Hodsman and Williams, 2011; Purdy et al., 2022). Other opportunities for SLA are seasonal free uplift schemes for all residents which could prove popular mirroring successful models used elsewhere, (Brosius et al., 2013; Comerford et al., 2018; Keske et al., 2018).

Operational factors also influence service uptake. Uncertainty around waiting times, kerbside collection point requirements, and limited collection support from collection crews may deter residents, (Curran et al., 2007; Hodsman and Williams, 2011). Providing clear timeframes and allowing trained crews to collect items from within homes could improve accessibility, reduce health and safety risks, and preserve the reuse value of items. Additionally, property type affects storage and disposal options with residents in multiple household properties facing greater constraints than those in detached homes, (Alexander et al., 2009b; Purdy et al., 2022). Collaborative approaches between SLA, communities, housing associations and landlords could help develop inclusive policies for bulky uplift services which match local needs and priorities, (Bernstad et al., 2012; Larsen et al., 2012; Ordoñez et al., 2015). Further research is required to assess public willingness to engage in such efforts as this remains uncertain.

TABLE 2: Spatial variation of area size, population density, and property types, modified from (Scotland Census, 2022).

SLA region classification	Area m^2	Population density m^2	Whole houses %	Flats %	Mobile structures %
Urban cities	65	6,183	42.65	57.27	0.08
Urban with significant rural	260	1,082	71.74	28.12	0.14
Rural	1,992	210	80.22	19.53	0.25
Highland & island	1,215	38	87.27	12.44	0.29

This is the first study to consider analysis of local authority bulky uplift services from publicly available information across Scotland. Our future research will focus on exploring resident perspectives concerning fly-tipping and bulky item disposal practices to validate these findings and develop inclusive local policies to inform service provision. Mitigating fly-tipping as a leak from the circular economy model can contribute to waste reduction and climate targets providing safer, cleaner communities and ensuring a safer end of engineered life for household bulky items. A limitation to our approach is reliance on website disclosures, which may not fully capture operational nuances such as informal practises, seasonal variations, or resident experiences. While the analysis provides valuable insights into service design and accessibility that could facilitate understanding of service design in other regions, future research incorporating direct stakeholder perspectives would strengthen understanding of service barriers and opportunities for improvement. Our insights offer a foundation for evidence-based policy development, supporting more inclusive, efficient, and sustainable household bulky waste services across Scotland. By reducing uncontrolled disposal and promoting reuse and recycling, local authorities can contribute directly to Scotland's net-zero ambitions and global climate commitments, (Scottish Government, 2024; UNEP, 2024). Insights from international studies confirm that the systemic barriers observed in Scotland mirror those found globally, underscoring the need for locally tailored yet globally informed solutions to advance circular economy objectives, (Comerford et al., 2018; Curran et al., 2006; Hodsman & Williams, 2011; Keske et al., 2018; Larsen et al., 2012). Mitigating the leak of household bulky items from the circular economy model by improving service accessibility and promoting reuse pathways is essential to preserve product and material value, reduce landfill dependency, and strengthen circular economy outcomes.

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REFERENCES

Alexander, C., Curran, A., Smaje, C., Williams, I. 2009a. Evaluation of bulky waste and reuse schemes in England. Proceedings of Institution of Civil Engineers: Waste and Resource Management, 162(3), 141-150. https://doi.org/10.1680/warm.2009.162.3.141

Alexander, C., Smaje, C. 2008. Evaluating third sector reuse organisations in the UK: Case-studies and analysis of furniture reuse schemes. Resources, Conservation and Recycling, 52(5), 719-730. https://doi.org/10.1016/j.resconrec.2007.09.005

Alexander, C., Smaje, C., Timlett, R., Williams, I. 2009b. Improving social technologies for recycling. Proceedings of Institution of Civil Engineers: Waste and Resource Management, 162(1), 15-28. https://doi.org/10.1680/warm.2009.162.1.15

Bernstad, A., La Cour Jansen, J., Aspegren, H. 2012. Local strategies for efficient management of solid household waste-the full-scale Augustenborg experiment. Waste Management and Research, 30(2), 200-212. https://doi.org/10.1177/0734242X11410113

Brosius, N., Fernandez, K., Cherrier, H. 2013. Reacquiring consumer waste: Treasure in our trash? Journal of Public Policy & Marketing, 32(2), 286-301. https://www.jstor.org/stable/43305789

Chung, S., Lau, K., Zhang, C. 2010. Measuring bulky waste arisings in Hong Kong. Waste Management, 30(5), 737-743. https://doi.org/10.1016/j.wasman.2010.01.009

Comerford, E., Durante, J., Goldsworthy, R., Hall, V., Gooding, J., Quinn, B. 2018. Motivations for kerbside dumping: Evidence from Brisbane, Australia. Waste Management, 78, 490-496. https://doi.org/10.1016/j.wasman.2018.06.011

Cook, E., Velis, C. 2021. Global review on safer end of engineered life. https://doi.org/10.5518/100/58

Cox, J., Giorgi, S., Sharp, V., Strange, K., Wilson, D., Blakey, N. 2010. Household waste prevention - A review of evidence. Waste Management and Research, 28(3), 193-219. https://doi.org/10.1177/0734242X10361506

Curran, A., Heaven, S., Williams, I. 2006. An evaluation of council bulky waste collection services in England. CIWM Scientific & Technical Review, 2006, 12-25. https://core.ac.uk/download/pdf/23702.pdf

Curran, A., Williams, I., Heaven, S. 2007. Management of household bulky waste in England. Resources, Conservation and Recycling, 51(1), 78-92. https://doi.org/10.1016/j.resconrec.2006.08.003

Curran, A., Williams, I. 2010. The role of furniture and appliance re-use organisations in England and Wales. Resources, Conservation and Recycling, 54(10), 692-703. https://doi.org/10.1016/j.resconrec.2009.11.010

Dixon, A., Farrell, G., Tilley, N. 2022. Illegal waste fly-tipping in the Covid-19 pandemic: enhanced compliance, temporal displacement, and urban–rural variation. Crime Science, 11(1). https://doi.org/10.1186/s40163-022-00170-3

Griffiths, T., Coe, C., Fiori, F., Berry, I. 2013. Mattress recycling. Proceedings of Institution of Civil Engineers: Waste and Resource Management, 166(4), 158-166. https://doi.org/10.1680/warm.12.00011

Guérin, J., Paré, M., Lavoie, S., Bourgeois, N. 2018. The importance of characterizing residual household waste at the local level: A case study of Saguenay, Quebec (Canada). Waste Management, 77, 341-349. https://doi.org/10.1016/j.wasman.2018.04.019

Hodsman, C., Williams, I. 2011. Drivers for the fly-tipping of household bulky waste in England. Proceedings of the Institution of Civil Engineers: Municipal Engineer, 164(1), 33-44. https://doi.org/10.1680/muen.900027

Ichinose, D., Yamamoto, M., Yoshida, Y. 2013. Productive efficiency of public and private solid waste logistics and its implications for waste management policy. IATSS Research, 36(2), 98-105. https://doi.org/10.1016/j.iatssr.2013.01.002

Ito, M., Colombo, R. 2019. Bulky waste in the municipality of São Paulo: Management and valuation. Urbe, 11. https://doi.org/10.1590/2175-3369.011.e20180117

Keske, C., Mills, M., Godfrey, T., Tanguay, L., Dicker, J. 2018. Waste management in remote rural communities across Canadian North: Challenges and opportunities. Detritus, 2(2018), 63-77. https://doi.org/10.31025/2611-4135/2018.13641

Larsen, A., Petersen, C., Christensen, T. 2012. Bulky waste quantities and treatment methods in Denmark. Waste Management and Research, 30(2), 147-154. https://doi.org/10.1177/0734242X11417973

Liu, Y., Kong, F., Santibanez Gonzalez, E. 2017. Dumping, waste management and ecological security: Evidence from England. Journal of Cleaner Production, 167, 1425-1437. https://doi.org/10.1016/j.jclepro.2016.12.097

McNeill, P., Woodard, R., Williams, M. 2021. Waste duty of care regulations: Evaluation of enforcement strategies by local authorities in England. Journal of Cleaner Production, 312(2021), 127654. https://doi.org/10.1016/j.jclepro.2021.127654

Murdo, F. MSP. 2022. Proposed Fly Tipping (Scotland) Bill. https://www.parliament.scot/-/media/files/legislation/proposed-members-bills/murdo-fraser-consultation-document--final.pdf

NetRegs. 2024. Scottish environmental legislation: Waste. https://www.netregs.org.uk/legislation/scotland-environmental-legislation/current-legislation/waste/

Ordoñez, I., Harder, R., Nikitas, A., Rahe, U. 2015. Waste sorting in apartments: Integrating the perspective of the user. Journal of Cleaner Production, 106, 669-679. https://doi.org/10.1016/j.jclepro.2014.09.100

Óskarsson, G., Agnarsson, S., Davíðsdóttir, B. 2022. Waste management in Iceland: Challenges and costs related to achieving the EU municipal solid waste targets. Waste Management, 151, 131-141. https://doi.org/10.1016/j.wasman.2022.07.035

Purdy, R., Crocker, M., Borrion, H., Ekblom, P., Tompson, L., Galvin, J., Fouquet, R. 2022. Fly-tipping: Drivers, deterrents and impacts (Technical report for DEFRA). https://doi.org/10.13140/RG.2.2.19860.37769

Salim, H., Jackson, M., Stewart, R., Beal, C. 2023. Drivers-pressures-state-impact-response of solid waste management in remote communities: A systematic and critical review. Cleaner Waste Systems, 4. https://doi.org/10.1016/j.clwas.2023.100078

Salvia, G., Zimmermann, N., Willan, C., Hale, J., Gitau, H., Muindi, K., Gichana, E., Davies, M. 2021. The wicked problem of waste management: An attention-based analysis of stakeholder behaviours. Journal of Cleaner Production, 326. https://doi.org/10.1016/j.jclepro.2021.129200

Scotland's Census. 2022. 2022 Reports. https://www.scotlandscensus.gov.uk/2022-reports/

Scottish Affairs Committee. 2024. Cost of living: Impact on rural communities in Scotland first report of session 2023-24 Report, together with formal minutes relating to the report. https://publications.parliament.uk/pa/cm5804/cmselect/cmscotaf/153/report.html

Scottish Government. 2016. Making things last: a circular economy strategy for Scotland. https://www.gov.scot/publications/making-things-last-circular-economy-strategy-scotland/

Scottish Government. 2023a. National litter and flytipping strategy. https://www.gov.scot/publications/national-litter-flytipping-strategy/

Scottish Government. 2023b. Scale and cost of litter and flytipping Scotland. https://www.gov.scot/publications/scale-cost-litter-flytipping-scotland/pages/1/

Scottish Government. 2024. Scotland's circular economy and waste route map to 2030 consultation. https://www.gov.scot/publications/scotlands-circular-economy-waste-route-map-2030-consultation/

Scottish Government. 2020a. Scottish index of multiple deprivation 2020. https://www.gov.scot/collections/scottish-index-of-multiple-deprivation-2020/

Scottish Government. 2018. Understanding the Scottish rural economy. https://www.gov.scot/publications/understanding-scottish-rural-economy/

Scottish Government. 2020b. Update to the climate change plan 2018 - 2032: Securing a green recovery on a path to net zero. https://www.gov.scot/publications/securing-green-recovery-path-net-zero-update-climate-change-plan-20182032/

Statista. 2024a. Population of Scotland 1971-2023. https://www.statista.com/statistics/367788/population-of-scotland/

Statista. 2024b. Smartphone ownership penetration in the United Kingdom (UK) in 2012-2024, by age. https://www.statista.com/statistics/271851/smartphone-owners-in-the-united-kingdom-uk-by-age/

Statista. 2023. The state of global waste management. https://www.statista.com/chart/31449/waste-management-score-on-the-yale-environmental-performance-index-by-country/

Song, N., McLellan, I., Liu, W., Wang, Z., Hursthouse, A. 2023. The waste ban in China: What happened next? Assessing the impact of new policies on the waste management sector in China. Environmental Geochemistry and Health, 45(4), 1117-1131. https://doi.org/10.1007/s10653-021-01101-y

Takahashi, W. 2020. Economic rationalism or administrative rationalism? Curbside collection systems in Sweden and Japan. Journal of Cleaner Production, 242. https://doi.org/10.1016/j.jclepro.2019.118288

Tilley, N. 2020. UCL JDI Special Series on COVID-19: No. 1 Fly-tipping during a pandemic. https://www.ucl.ac.uk/jill-dando-institute/sites/jill-dando-institute/files/1_fly_tipping_final_.pdf

UK Parliament. 2022. Access to cash in Scotland: Government response to the committee's second report. https://publications.parliament.uk/pa/cm5803/cmselect/cmscotaf/695/report.html

UNEP. 2024. Beyond an age of waste: Turning rubbish into a resource. https://wedocs.unep.org/20.500.11822/44939

Vimpolšek, B., Leskovar, J., Lisec, A. 2022. Circularity of bulky waste: A case study of Krško in Slovenia. International Maritime Transport and Logistics Conference, 11(1), 180-188. https://doi.org/10.21622/MARLOG.2022.11.180

Wang, H., Li, Y., Dang, L., Ko, J., Han, D., Moon, H. 2020. Smartphone-based bulky waste classification using convolutional neural networks. Multimedia Tools and Applications, 79(39-40), 29411-29431. https://doi.org/10.1007/s11042-020-09571-5

Webb, B., Marshall, B., Czarnomski, S., Tilley, N. 2006. Fly-tipping: Causes, incentives and solutions. https://www.researchgate.net/publication/238523261_Fly-tipping_Causes_Incentives_and_Solutions

Wilkinson, A., Williams, I. 2020. Why do (W)eee hoard? The effect of consumer behaviour on the release of home entertainment products into the circular economy. Detritus, 12 (2020), 18-33. https://doi.org/10.31025/2611-4135/2020.14004

Zero Waste Scotland. 2017. Evidence review of flytipping behaviour. https://cdn.zerowastescotland.org.uk/managed-downloads/mf-qj9zglno-1676977132d

Zero Waste Scotland. 2024a. A decade of delivery: charting the impact of our activities from 2014 - 2024. https://www.zerowastescotland.org.uk/resources/zero-waste-scotland-decade-delivery

Zero Waste Scotland. 2024b. A feasibility study of regional materials hubs in Scotland. https://cdn.zerowastescotland.org.uk/managed-downloads/mf-qrrfa3ry-1679488452d

Zero Waste Scotland. 2021. What to do with household appliances and white goods? https://www.zerowastescotland.org.uk/resources/what-do-household-appliances-and-white-goods

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RECYCLED SISAL FIBER-REINFORCED POLYPROPYLENE DERIVED FROM INDUSTRIAL WASTE STREAM

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ABSTRACT
This paper presents the development and comprehensive characterization of recycled polypropylene (rPP) composites reinforced with recovered sisal fibers sourced from secondary agricultural waste streams. The rPP matrix was obtained from post-industrial waste generated during the manufacturing of household appliances at Hisense GORENJE company (Slovenia). To enhance the interfacial compatibility between the hydrophobic polymer matrix and the lignocellulosic reinforcement, maleic anhydride-grafted polypropylene (MAPP) was incorporated at a constant concentration of 3 wt.% as a compatibilizing agent. Composite formulations containing 10, 20, and 30 wt.% of sisal fibers were systematically investigated through mechanical testing (tensile strength, tensile modulus, and impact resistance), morphological analysis (SEM), spectroscopic characterization (FTIR), and thermal analyses (TGA–DTA and DSC). FTIR analysis suggested the interactions between the functional groups of MAPP and the lignocellulosic fibers; however, no distinct absorption band attributable to ester carbonyl formation was detected. SEM observations revealed improved fiber–matrix adhesion in the compatibilized systems, supporting the effectiveness of MAPP in promoting the interfacial bonding. The addition of sisal fibers significantly enhanced the mechanical performance of the rPP, with increases of up to 45% in tensile modulus and 92% in tensile strength at 30 wt.% fiber loading. Nevertheless, the composite containing 20 wt.% sisal fibers exhibited the most favorable balance between stiffness, strength and impact resistance. Overall, the results demonstrate the strong potential of rPP/sisal composites as lightweight, sustainable materials, enabling the dual valorization of industrial and agricultural waste streams and supporting circular-economy-oriented applications in the white-goods sector.

1. INTRODUCTION

The steady increase in global plastic production and consumption has resulted in a waste crisis of considerable magnitude, with annual volumes exceeding 400 million tons. Polyolefins, particularly polypropylene (PP), constitute approximately half of this waste stream. Due to their chemical inertness and extremely slow degradation, the management of post-consumer and post-industrial polyolefin waste presents a major environmental challenge, thereby underscoring the need for advanced valorization strategies and circular economy models (Singh et al, 2023).

The household appliance industry, characterized by high production volumes and a reliance on durable plastic components, is a significant contributor to this stream. At GORENJE, a leading European manufacturer, substantial quantities of post-industrial PP waste, primarily in the form of purges and production lumps, are generated. The INCIRCULAR project, co-funded by the European Union's I3 Programme, was initiated to transform this internally generated waste into high-performance composite materials, with the overarching objective of closing the material loop within the industrial supply chain. In contrast to many previous studies employing model-grade recycled polymers, the rPP used in this work originates from a controlled post-industrial waste stream with documented thermo-mechanical degradation, making the results directly transferable to industrial manufacturing conditions in the white-goods sector. In the pursuit of sustainable material solutions, natural fibers such as flax, hemp, jute, and sisal have emerged as promising alternatives for synthetic reinforcements, in-

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cluding glass fibers. These lignocellulosic reinforcements offer several environmental and functional advantages, including renewability, biodegradability, low density, and the ability to enhance the mechanical performance of polymer matrices (Ajayi et al., 2025). In this study, the sisal fibers (SIS250) were sourced from secondary agricultural waste streams, specifically disused coffee bean bags. This approach provides a dual environmental benefit: the revalorization of industrial waste (rPP) and the upcycling of agricultural waste (sisal fibers). Consequently, the resulting composites represent a high-value, circular-economy material solution (Mahmud et al., 2021). Although numerous studies have investigated natural fiber-reinforced polypropylene composites, the majority focus on virgin PP matrices or prioritize stiffness enhancement at high fiber loadings, often at the expense of impact resistance and processability. Studies specifically addressing recycled polypropylene reinforced with natural fibers frequently remain limited to laboratory-scale formulations or lack direct relevance to industrial waste streams with known processing histories. In this context, the present work advances the state of the art by moving beyond material substitution and focusing on formulation-driven performance optimization. Rather than maximizing fiber content alone, this study identifies an optimal reinforcement window that balances stiffness, strength, and impact resistance in a fully recycled composite system derived from real industrial and agricultural waste streams.

Polypropylene, although recyclable, undergoes considerable thermo-mechanical degradation during repeated reprocessing (Caicedo et. al., 2017). This degradation reduces its molecular weight and negatively affects its mechanical properties, including tensile strength and modulus. The accompanying decline in toughness, typically manifested as a reduction in impact strength of approximately 20%, renders pure recycled polypropylene (rPP) unsuitable for applications that require reliable resistance to dynamic or impact loading. To address this inherent brittleness, reinforcement with high-performance materials is essential (Sanadi et al., 2023; Zhao et al., 2022; Ai-Huei Chiou & Chia-Hua Lin, 2023). The incorporation of natural fibers into polyolefin matrices such as PP presents a fundamental materials challenge arising from the polarity incompatibility. Whereas PP is hydrophobic, sisal cellulose fibers are inherently hydrophilic, resulting in weak interfacial bonding and inefficient stress transfer across the fiber–matrix interface (Ajayi et al., 2025). To address this limitation, polypropylene grafted with maleic anhydride (MAPP) is commonly employed as an effective coupling agent. MAPP acts as a molecular bridge: the PP backbone ensures compatibility with the rPP matrix, while the maleic anhydride moieties chemically react with the hydroxyl (-OH) groups of the sisal fibers. This reaction forms covalent ester linkages that enhance interfacial adhesion and facilitate efficient load transfer within the composite (Ibrahim et al., 2017).

The main objective of this work is to rigorously evaluate and characterize rPP composites reinforced with recycled SIS250 sisal fiber. The effects of three fiber loadings (10, 20, and 30 wt.%) on the mechanical, thermal, and morphological properties of the material are investigated. The analysis aims to identify the optimal formulation that offers the best balance of strength, toughness, and processability, ensuring its suitability as a lightweight structural material for demanding applications in the white goods sector while supporting the waste-recovery model proposed under the INCIRCULAR project. The goal of this study is therefore not only to characterize the composites, but to identify a formulation window that reconciles mechanical performance, toughness, and processability, thereby enabling the practical substitution of virgin, glass-fiber-reinforced plastics in selected white-goods components.

2. MATERIALS AND METHODS

2.1 Matrix and Reinforcement Materials

2.1.1 Recycled Polymer Matrix (rPP)

The polymer matrix consisted of ground recycled polypropylene (rPP) obtained from a single post-industrial waste stream generated by GORENJE IPC Company during the manufacture of household appliances. The material originated from a controlled and homogeneous production process, and all experiments were conducted using material from the same production batch in order to eliminate batch-to-batch variability. It should be noted that the recycled polypropylene was not a neat polymer but contained approximately 30 wt.% of mineral fillers, primarily talc and calcium carbonate, as confirmed by compositional analysis. Thermal characterization by TGA under air further confirmed this filler content, showing a final residual mass of 21.46% at 800°C, corresponding to the thermally stable fraction of the mineral phase remaining after polymer decomposition and carbonate calcination. This inherent filler content contributes to the baseline stiffness and dimensional stability of the rPP, while also affecting its density and thermal behaviour. The presence of mineral fillers was taken into account when interpreting the mechanical and thermal performance of the developed composites.

2.1.2 Recycled Sisal Fiber Reinforcement

The reinforcement component selected were the recycled sisal fibers (SIS250), supplied by J. Rettenmaier & Söhne GmbH. Originating as secondary waste from coffee bean bags, these fibers highlight the circular nature of the material. Fiber loadings of 10, 20, and 30 wt.% were selected to establish a comprehensive performance profile and identify the most effective reinforcement concentration. The fibers were characterized to define their physicochemical properties. Particle size analysis indicated a median fiber length of approximately 300 µm, with the main fraction ranging between 100 and 300 µm. Elemental analysis revealed a carbon content of 38-45%, typical of lignocellulosic biomass. The moisture conFtent, determined according to ISO 18134-2, ranged between 5-8%, requiring drying at 70°C for 8h prior to compounding to prevent hydrolytic degradation during processing. Fiber purity was assessed by loss on ignition (LOI) at 950°C, yielding an ash content of 1.07% ± 0.19, confirming the effective removal of major inorganic impurities during the recycling process.

TABLE 1: Composition of rPP/Sisal (SIS250) Composite Formulations.

Formulation	Fiber load (wt.%)	rPP (wt.%)	SIS250 (wt.%)	MAPP (wt.%)
rPP	0	100	0	0
SIS250 10%	10	87	10	3
SIS250 20%	20	77	20	3
SIS250 30%	30	67	30	3

2.1.3 Coupling Agent (MAPP)

The commercial product Licocene® PP MA 7452 GR (Clariant), a maleic anhydride-grafted polypropylene (MAPP), was used as a compatibilizing agent. Its concentration was fixed at 3 wt.% in all formulations to improve the interfacial adhesion between the polar cellulosic fibers and the non-polar polymer matrix. This loading was selected based on our previous systematic study (Žepič Bogataj et al., 2019), in which 3 wt. % MAPP was identified as optimal for enhancing interfacial morphology and tensile performance in recycled polypropylene composites reinforced with cellulosic fibers, without compromising processability.

2.2 Formulation Design and Composite Preparation

Batches of composite materials were formulated and prepared with the rPP matrix and SIS250 sisal fibers in increasing concentrations as shown in Table 1.

2.3 Processing: Twin-Screw Extrusion Compounding and Injection Moulding

The preparation of the composite formulations involved a carefully controlled pre-mixing and compounding process, given the delicate nature of sisal fibers and the challenges associated with processing of rPP at high fiber loadings.

2.3.1 Pre-mixing and Matrix Preparation

The rPP matrix was first subjected to a classification process to ensure a uniform particle-size fraction, removing both coarse and excessively fine powder material—an essential step for maintaining a consistency during extrusion. The classified rPP, along with the SIS250 sisal fibers and MAPP pellets was then manually mixed to achieve initial homogenization prior to feeding.

2.3.2 Extrusion Compounding and Injection Moulding

Extrusion compounding process (Figure 1) was carried out using a USEON LAB-30 co-rotating twin-screw extruder with a length-to-diameter (L/D) ratio of 44. Processing parameters were carefully optimized to ensure uniform fiber dispersion and promote the MAPP reaction while avoiding thermal degradation of the sisal fibers. The barrel temperature was maintained between 30 and 60 °C in the Feed Zone and gradually increased to 190-195 °C at the Discharge Zone. Screw rotation speed was adjusted inversely with fiber content to minimize fiber damage, with 150 rpm employed for the 10 wt.% fiber formulation and 100 rpm for the 20 wt.% and 30 wt.% sisal composites.

After compounding, all composite formulations, including the reference rPP sample, were subjected to a drying process at 70 °C for 8 h before the injection moulding. Testing specimens were produced using a Fanuc Roboshot ALPHA S50IA/330/IT injection molding machine under controlled processing conditions. The melt temperature was set to 180°C and the mould temperature to 22°C. The injection pressure was 800 bar, with a holding pressure of 500 bar and a back pressure of 20 bar. The injection speed was maintained at 80 mm s^{-1}, and the screw rotation speed

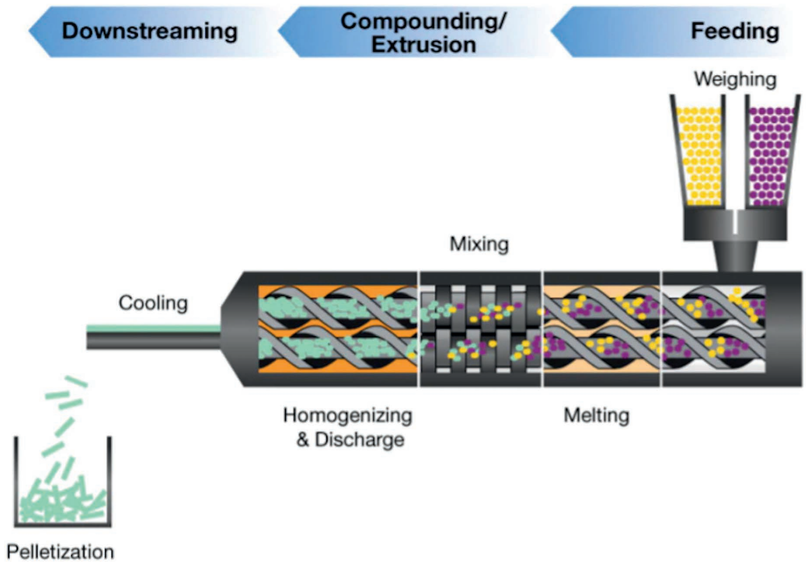

FIGURE 1: Twin extrusion compounding diagram.

was 50 rpm. The total injection cycle time was 40 s, including a cooling time of 20 s. A clamping force of 500 kN was applied during processing. All processing parameters were kept constant for all formulations to ensure comparability of the specimens.

2.4 Characterization Analyses

2.4.1 Mechanical Properties

Tensile strength, tensile modulus, and impact strength were determined in accordance with ISO 527-1/-2 and DIN EN ISO 179 standards. Uniaxial tensile tests were performed to evaluate tensile strength and modulus, providing insight into the load-bearing capacity and stiffness of the materials. Impact resistance was assessed using the unnotched Charpy method to quantify the energy absorption under dynamic loading conditions. For each material formulation, tensile parameters are reported as average values with standard deviation, calculated from measurements on ten specimens (n = 10). Impact strength results are presented as average values with the corresponding standard deviation, based on six specimens per formulation (n = 6). All specimens were conditioned and tested in accordance with the relevant standards to ensure reproducibility and comparability of the results. To evaluate the statistical significance of differences in tensile properties among the rPP and rPP/sisal formulations, one-way analysis of variance (ANOVA, single-factor analysis) was performed. The F-statistic and associated p-value were calculated using a significance level of 0.05.

2.4.2 Thermal Properties

Thermogravimetric analysis (TGA) was performed to assess the thermal stability and degradation behavior of the composites, while differential scanning calorimetry (DSC) was employed to investigate thermal transitions, including melting points and crystallinity, providing insights into the materials' structural characteristics and processability. Simultaneous TGA-DSC measurements were conducted using a TA Instruments SDT Q600 system, which records both heat flow and weight changes as a function of temperature under controlled atmospheres. The system operates from room temperature to 1450°C in either an oxidizing (air) or inert (N_2) atmosphere. Weight loss profiles were monitored as a function of temperature to characterize the thermal stability and decomposition behavior of the materials.

The relative change in onset degradation temperature (T_{onset}) was calculated with respect to neat rPP according to:

$$\Delta T_{onset}(\%) = \frac{T_{onset,ref} - T_{onset,sample}}{T_{onset,ref}} \times 100 \quad (1)$$

where:

$T_{onset,ref}$ is the onset degradation temperature of the reference material (e.g., neat rPP),

$T_{onset,sample}$ is the onset degradation temperature of the composite.

2.4.3 Chemical and Morphological Analysis

Fourier Transform Infrared Spectroscopy (FT-IR) was performed to identify the functional groups and molecular structure of the composite samples and the reference rPP. Spectra were recorded using a Spectrum 100 FT-IR spectrometer (PerkinElmer) equipped with a universal attenuated total reflectance (ATR) accessory. The fractured surfaces of mechanically tested specimens were examined using Scanning Electron Microscopy (SEM) to assess fiber dispersion and interfacial bonding. SEM observations were conducted on a Quanta 250 instrument (FEI Company, USA) under high vacuum at an acceleration voltage of 5 kV, a spot size of 2.0, and a working distance of approximately 9-10 mm.

3. RESULTS AND DISCUSSION

3.1 Mechanical Properties

3.1.1 Tensile strength and Tensile Modulus

Tensile testing confirmed the pronounced reinforcing effect of sisal fibers on recycled polypropylene matrix, demonstrating a progressive increase in both stiffness and strength with increasing fiber content (Table 2). The tensile modulus increased from 2414 MPa for neat rPP to 3492 MPa for the composite containing 30 wt.% SIS250 fibres, corresponding to an enhancement of approximately 45%. Similarly, the tensile strength increased from 21.8 MPa to 41.8 MPa, representing an improvement of about 92%. These results indicate highly efficient stress transfer from the polymer matrix to the fibrous reinforcement phase.

To statistically validate the observed trends, one-way ANOVA was performed on the tensile data. The statistical results are summarized in Table 2. The extremely low p-values ($p < 0.05$) confirm that the differences among formulations are statistically significant. Moreover, the high F-values indicate that the between-group variability (effect of fiber content) greatly exceeds the within-group experimental variability, confirming excellent reproducibility of the measurements. The magnitude of the mechanical improvements—particularly the 92% increase in tensile strength—further demonstrates that the observed enhancements are not only statistically significant but also

TABLE 2: Comparison of tensile parameters — tensile modulus (E_t), tensile strength (σ_M), and elongation at break (ε_t) (mean ± SD) — with corresponding one-way ANOVA statistical results (F-statistic and p–value).

Formulation	E_t [MPa]	σ_M [MPa]	ε_t [%]	F - value	p - value
rPP	2414 ± 58	21.8 ± 0.8	3.7 ± 1.0	11759.6	6.9×10^{-54}
SIS250 10%	2691 ± 34	30.8 ± 0.6	2.9 ± 0.4	36623.7	9.2×10^{-63}
SIS250 20%	3140 ± 42	38.2 ± 0.6	3.1 ± 0.1	37290.6	6.7×10^{-63}
SIS250 30%	3492 ± 58	41.8 ± 0.8	2.7 ± 0.3	25351.8	6.9×10^{-60}

physically meaningful, clearly exceeding typical experimental scatter associated with injection-molded composites.

These findings are consistent with literature reports. Joseph et al. (2003) observed comparable improvements in polypropylene-sisal composites, primarily attributed to the high intrinsic stiffness of fibers and improved interfacial adhesion when compatibilizers such as MAPP are employed. Similarly, Garkhail et al. 2000 reported simultaneous increases in modulus and strength with increasing sisal content, explaining this behavior by the fibers' ability to restrict matrix deformation and carry a signifciant portion of the applied load.

As commonly observed in natural fiber-reinforced composites, the increase in stiffness and strength was accompanied by a reduction in ductility (Table 2). The elongation at break decreased from 3.7% for neat rPP to 2.7% for the SIS250 30 wt.% composite, reflecting reduced polymer chain mobility and the more brittle mechanical response introduced by the fibrous phase. Overall, the results confirm that the incorporation of sisal fiber significantly enhances the stiffness and strength of recycled polypropylene composite, in agreement with well-established trends for lignocellulosic fiber-reinforced PP systems.

3.1.2 Impact strength

Charpy impact resistance (unnotched) indicates that the addition of sisal fiber enhances the toughness of rPP, but only up to an optimum reinforcement level. All fiber-reinforced formulations exhibit higher impact energy than neat rPP (10.6 kJ/m²), as shown in Figure 2. The SIS250 10% composite reaches 11.3 kJ/m² (approximately 7% increase), while SIS250 20% exhibits the highest value, 14.1 kJ/m² - an improvement of about 33% relative to the matrix. In contrast, the impact resistance of SIS250 30% decreases slightly to 12.9 kJ/m², although it remains above that of rPP. This trend indicates that, although tensile strength and modulus increase nearly monotonically up to 30 wt. % fiber loading, impact toughness exhibits an intermediate maximum at 20 wt.%, evidencing a stiffness-toughness trade-off typical of short-fiber composites. This behavior is consistent with the well-established inverse relationship between excessive stiffness and the ability of a composite to absorb impact energy.

At moderate fiber loadings, mechanisms such as fiber pull-out and crack deflection contribute to energy dissipation. However, at higher fiber contents, the formation of agglomerates and microstructural defects – also observed in the SEM analysis – becomes more pronounced. These defects act as stress concentration sites, limiting effective stress transfer across the interface and reducing the composite's ability to absorb dynamic energy. Similar trends have been reported for sisal-reinforced polypropylene composites, where impact resistance increases with fiber content only up to an optimal level and subsequently declines when the reinforcement becomes excessive, primarily due to a reduced effective matrix fraction and heightened stress concentration (Ferede & Atalie, 2022; Muralidhar, 2020). Overall, these findings are consistent with the tensile results: although maximum stiffness and strength are achieved at 30 wt.% of sisal fibres, the most favorable balance between toughness and rigidity occurs at 20 wt.%. This makes the SIS250 20% formulation the most suitable option for applications requiring enhanced impact absorption without compromising structural performance (Žepič Bogataj et al., 2019). In contrast, higher fiber loading (30 wt.%) may limit applicability in impact-sensitive components due to reduced energy dissipation despite increased stiffness.

3.2 Thermal properties

3.2.1 Thermogravimetric analysis (TGA)

TGA analysis performed up to 1000°C confirms the dual effect of sisal fibers on the thermal stability of the composite. All samples remain thermally stable up to approximately 280-300°C (Figure 3), however, the SIS250 formulations exhibit a slightly earlier onset of degradation compared with neat rPP. This behaviour is attributed to the lower intrinsic thermostability of the lignocellulosic constituents of sisal fibres - hemicellulose, cellulose, and lignin – which serve as activation sites for thermal degradation. These findings are consistent with previous studies, reporting that the incorporation of natural fibers decreases the onset temperature of polypropylene due to their higher reactivity and the presence of oxygen-containing functional groups (Patel et al., 2023; Arrakhiz et al., 2013). At high

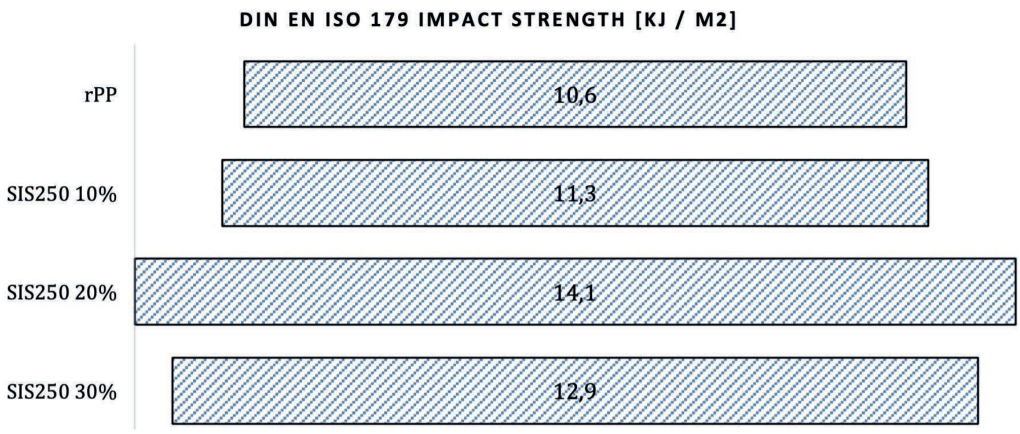

FIGURE 2: Impact Strength kJ/m^2.

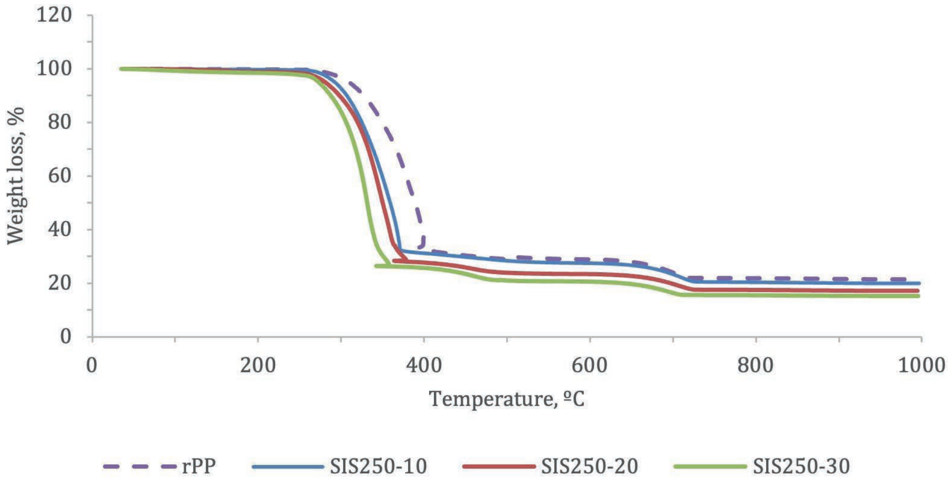

FIGURE 3: TGA of samples: rPP and SIS250 10, 20 and 30 wt.%.

temperatures (700-900°C), the composites show a greater carbonaceous residue than rPP, particularly at 30 wt.% fiber content, indicating enhanced char formation resulting from cellulose carbonization. This carbonaceous residue acts as a physical barrier that restricts heat and mass transfer between the flame and the polymer matrix, a mechanism well-documented to potentially delay flame spread in natural fiber composites (Fan & Naughton, 2016; Tajvidi et al., 2009). While this charring behavior is beneficial for fire-performance scenarios in white-goods components, the associated reduction in the degradation onset temperature (T_{onset}) imposes a limitation on the processing window. Consequently, strict temperature control (below 200°C) is required during extrusion and moulding to prevent the thermal deterioration while leveraging the flame-retardant potential of the high fiber loading. Overall, this thermal behaviour aligns with the morphological observations discussed below and the mechanical results presented earlier. Increasing the fiber content improves stiffness and load transfer but simultaneously introduces microstructural heterogeneities – evident in SEM images – that reduce toughness and increase the material susceptibility to early thermal degradation (Ordoñez Benavides, 2016).

Thus, the observed thermal behavior reinforces the overall interpretation of the materials performance: moderate fiber contents (20 wt.%) provide the best balance among mechanical properties, thermal stability, and microstructural homogeneity. In contrast, higher loadings (30 wt.%) promote increased formation of carbonaceous residue but at the cost of reduced initial thermal stability and diminished impact resistance. From the thermogravimetric analysis, the characteristic degradation temperatures and the final residual mass attributable to sisal can be determined. The corresponding values for each composite family are summarized in Table 3.

Where, T_{onset} (°C) is the initial degradation temperature, $T_{50\%}$ (°C) is the temperature at which 50% mass loss is reached, T_{max} (°C) indicates the temperature at which the maximum degradation rate occurs, and FR (%wt) means the final residue remaining ager degradation. The percentage decrease in onset degradation temperature (T_{onset}) was calculated relative to neat rPP using Equation 1. A reduction of 3.8%, 3.4% and 3.1% was observed for the composite containing 10, 20, and 30 wt. % of sisal fibers, respectively.

The results show that the fiber-reinforced samples exhibit a slightly lower degradation onset temperature (T_{onset}) than neat rPP, indicating that the incorporation of sisal fibers reduces the initial thermal stability of the material. This decrease is attributed to the lower intrinsic thermostability of the lignocellulosic constituents – primarily hemicellulose and cellulose – which degrade earlier than polypropylene and act as activation sites for thermal decomposition, particularly at higher fiber loadings. Consequently, the fibers do not inhibit degradation but rather promote its earlier initiation, a behavior widely reported for natural fiber-reinforced biocomposites (Krishnasamy et al. 2019; Munde et al. 2019; Joseph et al. 2003). Although the addition of sisal fibers reduced T_{onset} to approximately 263°C, this reduction does not compromise the manufacturing feasibility. The extrusion and injection moulding processes were optimized with a maximum temperature profile of 190-195°C

TABLE 3: Relevant temperatures vs. sisal fiber percentage.

Formulation	T_{onset} [°C]	$T_{50\%}$ [°C]	T_{max} [°C]	$FR_{\%}$ [%]	ΔT_{onset} [%]
rPP	274.31	377.55	722.72	21.46	
SIS250 10%	263.90	350.00	730.46	19.96	-3.79
SIS250 20%	265.03	343.14	729.50	17.22	-3.38
SIS250 30%	265.83	327.15	717.07	15.24	-3.09

for extrusion and 180°C for injection, respectively, ensuring a thermal safety margin greater than 60°C relative to the degradation onset. Under these conditions, the residence time in the processing equipment does not induce significant thermal deterioration of the fibers, thereby preserving their reinforcing efficiency. From a service-life perspective, natural fibers may act as thermal catalysts by locally weakening the crystalline structure of polypropylene and promoting earlier degradation. However, for the intended application in white-goods components (e.g., interior parts), where operating temperatures typically remain below 100°C, the composite functions well within its thermal stability range. Therefore, while the reduction in T_{onset} limits its potential use in high-temperature environments (>200°C), it does not adversely affect the durability under standard household appliance operating conditions. This behavior is consistent with the overall thermal and mechanical trends observed: increasing fiber content enhances composite stiffness but simultaneously introduces microstructural heterogeneities that facilitate earlier thermal activation.

3.3 Chemical and Morphological Characterization

3.3.1 Interfacial Interaction using FTIR Spectroscopy

The FTIR spectra of the composites (Figure 4) show that the characteristic absorption bands of polypropylene remain essentially unchanged after the incorporation of sisal fibers, indicating that the crystalline and chemical structure of the rPP matrix is preserved. Only slight variations in band intensity and baseline are observed, which are attributed to the presence of lignocellulosic components in the fibres - principally hydroxyl (–OH) groups and C–O–C vibrations. Regarding chemical interactions, the spectra did not reveal a distinct new peak corresponding to the ester linkage (expected near 1700-1740 cm^{-1}). This absence is attributed to the low concentration of MAPP coupling agent (3 wt. %) and the overlap of the carbonyl signal with the dominant spectral envelope of the polypropylene matrix. Such limitations are consistent with previous reports on MAPP-reinforced composites, where the esterification bands were either masked or below the detection limit of standard FTIR-ATR measurements (Bassyouni et al 2018; Bermello et al 2008).

While complementary techniques such as XPS could theoretically isolate surface chemical changes, the effectiveness of compatibilization in this study is strongly evidenced by the macroscopic performance. The significant increase in tensile strength (up to 92% relative to neat rPP) and the fiber wetting observed in SEM micrographs demonstrate efficient stress transfer across the interface. Overall, the FTIR analysis supports the interpretation that the composite behavior is governed by adequate interfacial compatibility and physical adhesion, which is corroborated by the mechanical results rather than by the isolated detection of new chemical bonds (Paredes García, 2024).

3.3.2 Fracture morphology by SEM

Scanning electron microscopy (SEM) analysis confirms the close relationship between microstructural features and the mechanical and thermal behavior of the rPP-sisal composites. Neat rPP exhibits a homogeneous and ductile fracture surface, whereas the formulations with moderate fiber contents, particularly SIS250 20%, display well-dispersed fibers with excellent wetting and strong interfacial adhesion (Figure 5). This is evidenced by matrix residues adhering to the fiber surfaces and the absence of significant voids. Such morphology promotes efficient

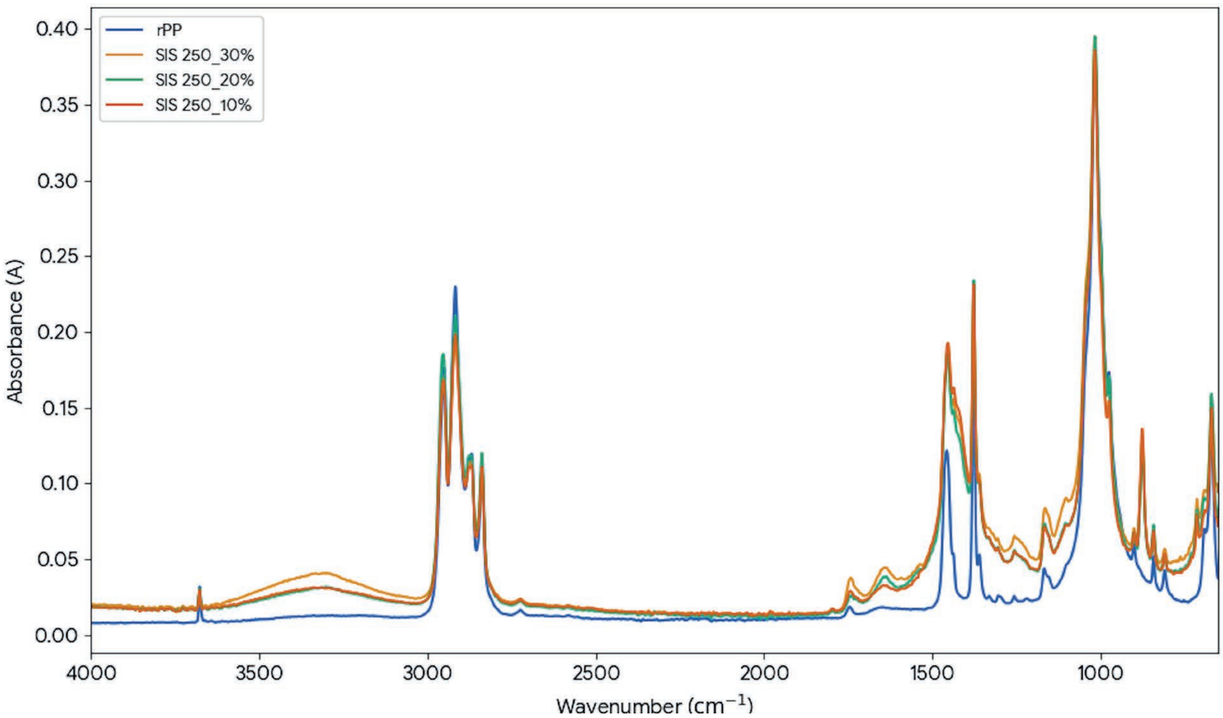

FIGURE 4: FTIR Spectra of a) rPP, and b) composite samples.

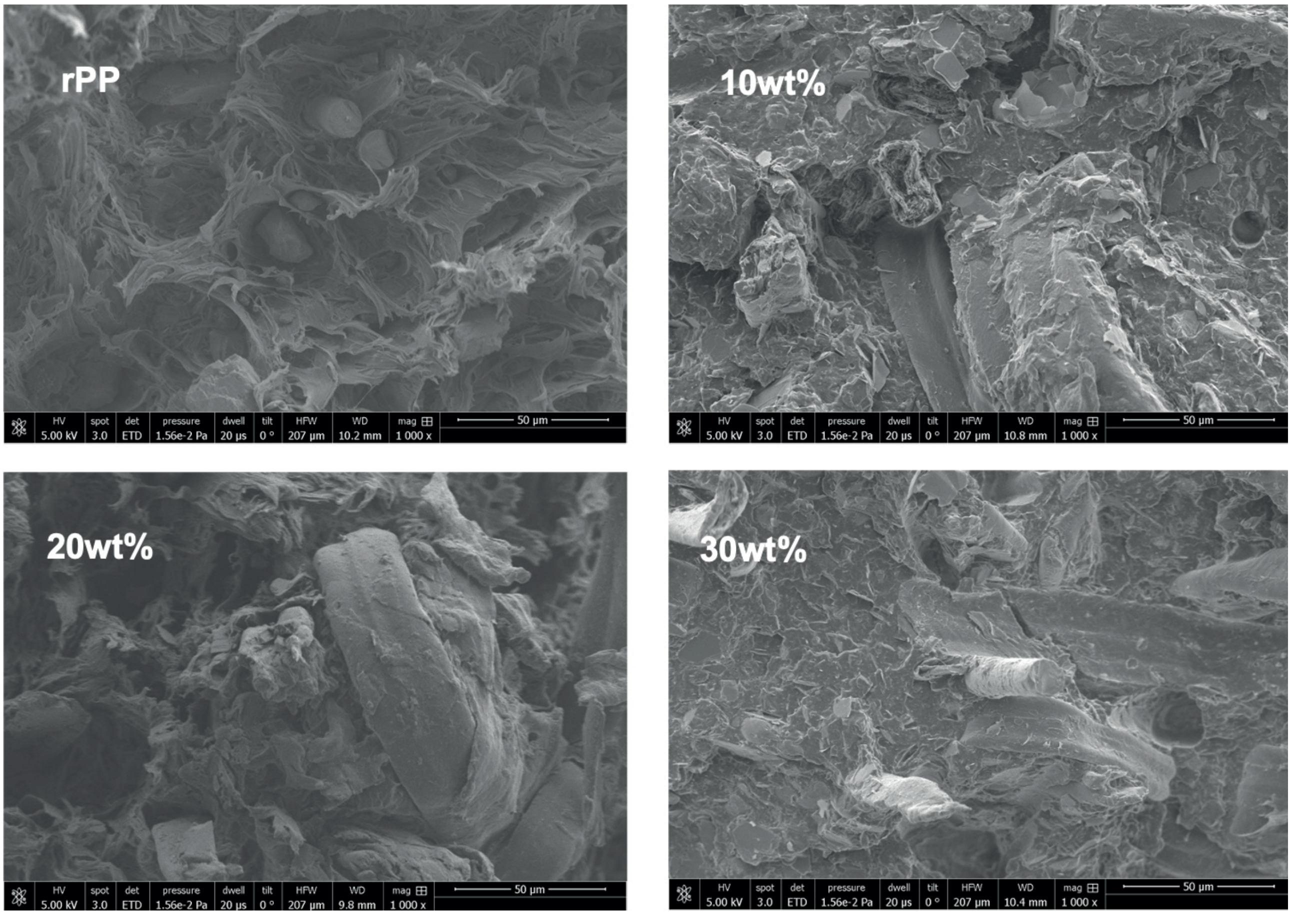

FIGURE 5: SEM images for rPP composites reinforced with 10 wt. %, 20 wt.% and 30 wt.% of sisal fibers.

load transfer through the MAPP-compatibilized interface and enables energy-dissipating mechanisms such as controlled fiber pull-out, in which fibers slide from the matrix during fracture. These mechanisms explain why this formulation achieves the most favorable balance among stiffness, strength, and toughness. In contrast, the composites containg 30 wt.% of sisal fibres, show the presence fo fiber agglomerates, interfacial voids, and regions of incomplete impregnation. These microstructural defects act as stress concentrators and reduce the materials ability to absorb energy, consistent with the observed decrease in impact resistance and elongation at break for this formulation. This behavior coincides with reports in the literature on PP composites reinforced with lignocellulosic fibers, where excessive fiber loading weakens interfacial efficiency and generates defects that diminish toughness despite increasing stiffness (Faruk et al., 2014; Essabir et al., 2013). Overall, the SEM observations corroborate the mechanical trends and are consistent with thermal analysis, which likewise indicates increased structural heterogeneity and higher initial reactivity in composites with elevated fiber content.

3.4 Industrial Feasibility, Resource Considerations and Potential Applications

The developed rPP/sisal composites demonstrate favorable potential for industrial implementation. Based on the present study, the 20 wt.% fiber formulation provides the optimal balance between stiffness, tensile strength, and impact resistance, while remaining compatible with realistic material availability. Post-industrial polypropylene waste is generated in substantially larger volumes than suitable lignocellulosic streams, suggesting that partial substitution, rather than high-fiber loadings, represents a more scalable strategy. Although natural fibers may be cost-sensitive in certain markets, sisal fibers are accessible not only through agricultural supply chains but also via urban post-consumer streams, such as discarded coffee sacks, ropes, carpets, and packaging textiles, providing additional opportunities for localized material recovery. From an economic perspective, the use of post-industrial polypropylene reduces reliance on virgin feedstock, while moderate fiber incorporation minimizes processing complications and avoids excessive material costs associated with high natural-fiber contents. As the composites were produced using conventional compounding and injection molding without specialized modifications, the approach is fully compatible with existing manufacturing infrastructure, supporting industrial scalability and transferability to regions with comparable waste streams and processing capabilities. Building on the mechanical and thermal characterization presented in this study, the 20 wt.% rPP/sisal formulation exhibits properties suitable for lightweight structural and semi-structural applications. Potential uses include white-goods components, such as clips, panels, and housings, as well as automotive interior parts requiring moderate stiffness and impact resistance. The materials are compatible with conventional injection molding, facilitating integration into existing manufacturing lines. Furthermore, the concept can be extended to hybrid composites or alternative lignocellulosic waste streams, enabling broader implementation within regional circular value chains and further supporting the objectives of the INCIRCULAR project.

4. CONCLUSIONS

This study demonstrates the technical feasibility and high potential of recycled polypropylene (rPP) composites reinforced with recycled sisal fiber (SIS250) for structural applications in the white goods sector. The proposed material system embodies the principles of the circular economy by simultaneously valorizing post-industrial polymer waste (rPP from Gorenje) and secondary agricultural waste (sisal fibers recovered from coffee bags). The most significant outcomes of this research are summarized as follows:

- Effective Mechanical Reinforcement: The addition of sisal fibers in combination with MAPP compatibilizer, produced highly efficient load transfer. The composites exhibited substantial improvements in stiffness (tensile modulus up to +45%) and tensile strength (up to +92%) at 30 wt.% sisal relative to neat rPP.
- Optimal Formulation: Although the 30 wt.% formulation provided the highest stiffness, the SIS250 20% composite offered the most balanced mechanical profile. It achieved the maximum Charpy impact strength (+33% compared with rPP) while maintaining significant gains in stiffness and strength, making it the most mechanically robust and energy-absorbing formulation.
- Morphological Validation: SEM observations confirmed excellent fiber-matrix adhesion and uniform dispersion at 20 wt.% of sisal fibres added, enabling efficient stress transfer. At 30 wt.%, however, fiber agglomerates and interfacial voids appeared, acting as stress concentrators and reducing toughness, fully consistent with the mechanical trends.
- Thermal Stability: The addition of sisal fibers slightly lowered the degradation onset temperature due to the lower intrinsic thermal stability of lignocellulosic components. However, composites with higher fiber contents exhibited increased carbonaceous residue at elevated temperatures. While this char formation suggests a positive tendency towards improved fire reaction behavior, specific flammability testing (e.g., UL-94) would be required to certify this performance.
- Industrial Viability: The SIS250 20% composite emerges as a promising candidate for semi-structural components in the white-goods sector. While long-term durability tests (fatigue, creep) are recommended for critical load-bearing parts, the current static mechanical profile meets the requirements for various rigid casing and internal support applications, demonstrating that the dual valorization of polymer and natural-fiber waste streams can yield high-performance materials. Nevertheless, future studies should evaluate the potential long-term effects, including moisture uptake, biodegradation, and UV exposure, to confirm the composite's durability under industrial service conditions.

Overall, this study supports the circular-economy strategy advanced within the INCIRCULAR project by demonstrating the technical feasibility of transforming post-industrial polypropylene waste into high-performance bio-composites through conventional processing technologies. The findings provide material-level validation for scalable circular value chains integrating recycled plastics and lignocellulosic reinforcements, supporting further pilot-scale implementation and end-of-life circularity assessments. While the environmental benefits of this approach are discussed qualitatively, a comprehensive life-cycle assessment (LCA) would be required to quantitatively substantiate reductions in environmental impact, representing an important direction for future work.

ACKNOWLEDGEMENTS

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REFERENCES

Ajayi, N. E., Rusnakova, S., Ajayi, A. E., Ogunleye, R. O., Agu, S. O., & Amenaghawon, A. N. (2025). A comprehensive review of natural fiber reinforced polymer composites as emerging materials for sustainable applications. Applied Materials Today, 43, 102666. https://doi.org/10.1016/j.apmt.2025.102666

Arrakhiz, F. Z., El Achaby, M., Malha, M., Bensalah, M. O., Fassi-Fehri, O., Bouhfid, R., Benmoussa, K., & Qaiss, A. (2013). Mechanical and thermal properties of natural fibers reinforced polymer composites: Doum/low density polyethylene. Materials & Design, 43, 200–205. https://doi.org/10.1016/j.matdes.2012.06.056

Bassyouni, M. (2018). Dynamic mechanical properties and characterization of chemically treated sisal fiber-reinforced polypropylene biocomposites. Journal of Reinforced Plastics and Composites, 37(23), 1402-1417. https://doi.org/10.1177/0731684418798049

Bermello, A., Díaz, D., Martínez, R., Quintana, G., Mieres, G., & Leal, J. A. (2008). Estudio de un material compuesto de partículas de bagazo de caña y matriz de polietileno mediante espectrometría FTIR. ICIDCA. Sobre los Derivados de la Caña de Azúcar, 42(1-3), 106-111. https://www.redalyc.org/pdf/2231/223120667015.pdf

Caicedo, C., Crespo-Delgado, L. M., De La Cruz-Rodríguez, H. y Álvarez-Jaramillo, N. A. (2017). Propiedades termo-mecánicas del Polipropileno: Efectos durante el reprocesamiento. Ingeniería Investigación y Tecnología, 18(3), 245-252. https://doi.org/10.22201/fi.25940732e.2017.18n3.022

Chiou, A.-H., & Lin, C.-H. (2023). Material and mechanical characterization of recycled polypropylene reinforced with different weight percentages of short glass fiber developed by injection molding. Heliyon, 9(9), e19403. https://doi.org/10.1016/j.heliyon.2023.e19403

Essabir, H., Hilali, E. M., Elgharad, A., Minor, H. E., Imad, A., Elamraoui, A., & Gaoudi, O. A. (2013). Mechanical and thermal properties of bio-composites based on polypropylene reinforced with Nut-shells of Argan particles. Materials & Design, 49, 442-448. https://doi.org/10.1016/j.matdes.2013.01.025

Fan, M., & Naughton, A. (2016). Mechanisms of thermal decomposition of natural fibre composites. Composites Part B: Engineering, 84, 290-302. https://doi.org/10.1016/j.compositesb.2015.10.038

Faruk, O., Bledzki, A. K., Fink, H.-P., & Sain, M. (2014). Progress Report on Natural Fiber Reinforced Composites. Macromolecular Materials and Engineering, 299(1), 9-26. https://doi.org/10.1002/mame.201300008

Ferede, E., & Atalie, D. (2022). Mechanical and water absorption characteristics of sisal fiber reinforced polypropylene composite. Journal of Natural Fibers, 19(16), 14825–14838. https://doi.org/10.1080/15440478.2022.2069188

Garkhail, S. K., Heijenrath, R. W. H., & Peijs, T. (2000). Mechanical Properties of Natural-Fibre-Mat- Reinforced Thermoplastics based on Flax Fibres and Polypropylene. Applied Composite Materials, 7, 351–372. https://doi.org/10.1023/A:1026590124038

Ibrahim, I. D., Jamiru, T., Sadiku, R. E., Agwuncha, S. C., & Munda, S. L. (2017). Dependency of the Mechanical Properties of Sisal Fiber Reinforced Recycled Polypropylene Composites on Fiber Surface Treatment, Fiber Content and Nanoclay. Journal of Polymers and the Environment, 25, 427–434. https://doi.org/10.1007/s10924-016-0823-2

International Organization for Standardization. (2023). ISO 179-1: Plastics—Determination of Charpy impact properties—Part 1: Non-instrumented impact test. https://www.iso.org/standard/84393.html

International Organization for Standardization. (2025). ISO 527-2: Plastics—Determination of tensile properties—Part 2: Test conditions for moulding and extrusion plastics. https://www.iso.org/es/norma/527-2

International Organization for Standardization. (2019). ISO 527-1: Plastics—Determination of tensile properties—Part 1: General principles. https://www.iso.org/standard/527-1

Joseph, K., Thomas, S., & Pavithran, C. (1999). Effect of chemical treatment on the tensile properties of short sisal-fibre-reinforced polyethylene composites. Composites Science and Technology, 59, 1625–1640. https://doi.org/10.1016/S0266-3538(99)00024-X

Joseph, P. V., Joseph, K., Thomas, S., Pillai, C. K. S., Prasad, V. S., Groeninckx, G., & Sarkissova, M. (2003). The thermal and crystallisation studies of short sisal fibre reinforced polypropylene composites. Composites Part A: Applied Science and Manufacturing, 34(3), 253-264. https://doi.org/10.1016/S1359-835X(02)00185-9

Joseph, P. V., Mathew, G., Joseph, K., Thomas, S., & Pradeep, P. (2003). Mechanical properties of short sisal fiber-reinforced polypropylene composites: Comparison of experimental data with theoretical predictions. Journal of Applied Polymer Science, 88(3), 602–611. https://doi.org/10.1002/app.11498

Krishnasamy, S., Thiagamani, S. M. K., Kumar, C. M., Nagarajan, R., Shahroze, R. M., Siengchin, S., Ismail, S. O., & Devi, I. M. P. (2019). Recent advances in thermal properties of hybrid cellulosic fiber reinforced polymer composites. International Journal of Biological Macromolecules. https://doi.org/10.1016/j.ijbiomac.2019.08.231

Mahmud, S., Hasan, K. M. F., Jahid, M. A., Li, M., Shi, S. Q., & Zhu, J. (2021). Comprehensive review on plant-fiber reinforced polymeric biocomposites. Journal of Materials Science. https://doi.org/10.1007/s10853-021-05774-9

Monteiro, S. N., Calado, V., Rodriguez, R. J. S., & Margem, F. M. (2012). Thermogravimetric behavior of natural fibers reinforced polymer composites—An overview. Materials Science and Engineering: A, 557, 17-28. https://doi.org/10.1016/j.msea.2012.05.109

Munde, Y. S., Ingle, R. B., & Siva, I. (2019). Effect of sisal fiber loading on mechanical, morphological and thermal properties of extruded polypropylene composites. Materials Research Express, 6(8), 085307. https://doi.org/10.1088/2053-1591/ab1dd1

Muralidhar, B. A. (2020). Characterization of sisal/polypropylene composites treated with plasma. Tekstilna Industrija, 3(4), 202–212. https://doi.org/10.31881/TLR.2020.15

Ordoñez Benavides, J. (2016). Evaluación de las propiedades térmicas y mecánicas del polipropileno reforzado con zeolitas tipo ZSM-5. Institución Universitaria ITM. Disponible en: https://hdl.handle.net/20.500.12622/502

Paredes García, F. F. (2024). Caracterización mecánica y por FTIR de bio-composites reforzados con residuos lignocelulósicos: Caracterización mecánica a tracción y caracterización por FTIR de bio-composites reforzados con residuos lignocelulósicos [Tesis de Maestría/Doctorado, Escuela Politécnica Nacional, 45 pp. In: EPN, 2024. http://bibdigital.epn.edu.ec/handle/15000/25311

Patel, R. V., Yadav, A., & Winczek, J. (2023). Physical, Mechanical, and Thermal Properties of Natural Fiber-Reinforced Epoxy Composites for Construction and Automotive Applications. Applied Sciences, 13(8), 5126. https://doi.org/10.3390/app13085126

Sanadi, A. R., & Stelte, W. (2023). Effect of the characteristics of maleic anhydride-grafted polypropylene (MAPP) compatibilizer on the properties of highly filled (85%) kenaf-polypropylene composites. Materials Research, 26, e20220428. https://doi.org/10.1590/1980-5373-MR-2022-0428

Singh, M., Mohanty, A., & Misra, M. (2023). Upcycling of waste polyolefins in natural fiber and sustainable filler-based biocomposites: A study on recent developments and future perspectives. Composites Part B: Engineering, 263. https://doi.org/10.1016/j.compositesb.2023.110852

Tajvidi, M., & Takemura, A. (2009). Thermal Degradation of Natural Fiber-reinforced Polypropylene Composites. Journal of Thermoplastic Composite Materials, 23(3), 281-298. https://doi.org/10.1177/0892705709347063

Zhao, X., Copenhaver, K., Wang, L., Korey, M., Gardner, D. J., Li, K., Lamm, M. E., Kishore, V., Bhagia, S., Tajvidi, M., Tekinalp, H., Oyedeji, O., Wasti, S., Webb, E., Ragauskas, A. J., Zhu, H., Peter, W. H., & Ozcan, S. (2022). Recycling of natural fiber composites: Challenges and opportunities. Resources, Conservation and Recycling, 177, 105962. https://doi.org/10.1016/j.resconrec.2021.105962

Zhao, Y., & Xianhui, H. (2022). Degradation mechanisms in recycled polypropylene and strategies for property recovery. Waste Management, 135, 12–25. https://doi.org/10.1016/j.wasman.2021.08.032

Žepič Bogataj, V., Fajs, P., Peñalva, C., Omahen, M., Čop, M., & Henttonen, A. (2019). Utilization of recycled polypropylene, cellulose and newsprint fibres for production of green composites. Detritus: Multidisc. J. for Waste Res. & Residues, 7, 36-43. 10.31025/2611-4135/2019.13857

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COMPOSTING WITH MEALWORMS FOR PLASTICS DEGRADATION

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ABSTRACT
Recycling plastics is an energy intensive process, but entomoremediation (the use of insects to transform environmental contaminants), and insect-driven plastics degradation (plastivory) may offer a natural, low energy, cost effective approach to plastics waste reduction. Polystyrene and some polyethylene materials are amenable to degradation by darkling ground beetle (Tenebrionidae) adults and larvae, and a wide range of polyethylene can be degraded by waxworm moth larvae. Mandibular activity of the insects chewing on plastics serves to increase the surface area of the polymers that can be attacked by insect digestive enzymes and gut bacteria that can degrade the polymers into individual monomers and other low molecular weight compounds. The objective of this work was to determine the feasibility of an innovative solution to the plastics waste problem using a holistic, low technology, low cost, and low maintenance technique. Plastivory by ground beetle mealworm larvae was tested using small batch, mixed food and plastics waste compost bins. Different combinations of food waste and plastics were assessed and ultimately modeled through a continuum model of mealworm population dynamics. Our idealized, coarse-grained continuum population model suggests that a modestly-sized compost bin can support a steady population of ~1100 mealworms that can degrade plastic at a rate of ~2 g of polystyrene foam per week. It is anticipated that this model will allow for assessing the extrapolation of empirically derived plastics degradation rates at various scales such as individuals and households, to industrial scales that would comply with municipal regulations.

1. INTRODUCTION

A common need on DoD/Army installations and throughout the world is to find efficient and less harmful means for waste reduction and novel sources of manufacturing feedstocks. Polyethylene (PE) or polystyrene (PS) are ubiquitous, recalcitrant plastic materials with breakdown times estimated at 400 years for many plastics (Parker 2019). PE and PS (Styrofoam) consist of long, linear, and/or branched chains of ethylene or styrene molecules, respectively. PS constitutes the most common class of plastics, least sustainable (Brandon et al.,, 2020), and the most difficult to transport and eliminate. Many recycling centers do not accept PS due to its bulky nature, so it is generally placed in landfills where it persists (Brandon et al., 2020).

Recycling plastics is both energy intensive and often cost prohibitive. An interesting and still very embryonic means for plastics waste reduction is to use insect-driven plastivory, or entomoremediation. Many forms of PS and some PE materials are amenable to breakdown by either darkling ground beetles in their adult and larval (mealworm) forms (Coleoptera: Tenebrionidae: *Tenebrio molitor* and Zophobas morio), or waxworm moth larvae (Lepidoptera: Pyralidae: *Achroia grisella* and *Galleria mellonella*) (Brandon et al., 2018; Kundungal et al., 2019; Urbanek et al., 2020; Yang et al., 2014; Yang et al., 2018; Yang et al., 2020; Yang et al., 2015a; Yang et al., 2015b). The surface area of the plastic polymers is increased by the insects chewing (mandibular activity) on plastics which allows for access by insect digestive enzymes and gut bacteria that can degrade the polymers into low molecular weight compounds or completely mineralize them into CO_2 (Brandon et al., 2018; Yang et al., 2015a).

Mealworms have been shown to consume numerous types of styrene polymers, but limited types of PE materials (as reviewed by (Pham et al., 2023)). In one feeding experiment, simple mono-compound PE and PS were introduced to mealworms separately and in combination (Brandon et al., 2018). Feeding and degradation of both plastics occurred at a high level separately but there was a preference for the specific type of PE used when the plastics were introduced together. This study observed shifts in the gut microbiome community structure when insects were feeding on different plastics as food sourc-

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es, but the dominant bacterial phyla were unchanged. It was observed that minor microbial phyla in the gut were more variable between PE and PS diets indicating that numerous bacteria may work together to degrade plastics in a nonspecific manner (Brandon et al., 2018). Isolated cultures of gut bacteria have been shown to degrade different plastics but the feasibility of commercialization of these processes is currently confounded by practical limitations. Among these obstacles, implementation of a plastics degrading bioreactor would have to be maintained in a liquid environment with strict oversight of the plastics being introduced and constant monitoring to assess the health of the system. Although this could be an exciting technology, it would require intensive engineering and extensive capital investment. The use of mealworms as a bioreactor could overcome many of these limitations. For example, an insect system could be regarded as a semi-autonomous, self-perpetuating reactor constantly introducing new plastics into the gut microbiome system, while reproducing to create more bioreactors. It is envisioned that this system would be largely self-sustaining as the mealworms and adults are mobile and capable of finding new sources of plastic in a mixed reactor setting or moving away from less desirable conditions. Furthermore, darkling beetles have a wide temperature tolerance as they are found in most ecosystems, typically scurrying along the ground, amongst leaves or detritus, or under rotting trees and logs throughout the year in more temperate environments. As a model system and something that could be used by anyone, the beetles are cosmopolitan throughout the world (non-invasive), flightless, non-biting, extremely low maintenance, highly reproductive, and inexpensive. Although the beetles can eat plastics exclusively, their health and eating vigor is vastly improved when offered a variety of other food sources such as grains, fruits, and vegetables. This suggests that the plastivory of these insects might best be leveraged in mixed waste, plastics-composting at a small, contained scale such as for households or small installation composting bins.

The goal of this effort is to provide a simplistic solution to the food and plastics waste problem by demonstrating the use of beetles/mealworms to facilitate plastics degradation using small-batch, food compost bins for either individual households or food service facilities. Extrapolation of previous experimental benchtop-scale rates of degradation of low-density polyethylene (LDPE), polystyrene (PS), and high-density polyethylene (HDPE) in a pliable form like shopping bags (~0.5 mg, 0.25 mg, and 0.05 mg plastics degraded/mealworm each day, respectively) will inform the end-user regarding the number of mealworms needed, effective ratios of food to plastics waste, and expected speed of plastics degradation.

As referenced in the Army Installations Strategy (https://apps.dtic.mil/sti/citations/AD1118920), Army installations are often dependent on infrastructure and services in surrounding communities, making them vulnerable to adversarial threats as well as subjecting them to regulations and standards of the specific municipalities. Ever-changing, municipality-specific regulations lead to inevitable gaps in Army-wide Facility Investment Plans for waste disposal, including food waste and recyclable materials. For example, the state of California signed SB54 (https://calrecycle.co.gov.gov/laws/rulemaking/sb54regulations/) into law in 2023 mandating that 30% of single use plastic packaging and food-ware must be recycled, increasing to 40% by 2028, 65% by 2030, and 100% of all packaging must be recyclable or compostable by 2032. Furthermore, the California Composting Law SB1383 (https://leginfo.legislature.ca.gov/faces/billNavClient.xhtml?bill_id=201520160SB1383) went into effect January 2022 requiring households and organizations to separate organic, compostable material from other garbage with the goal of a 75% reduction in statewide disposal of organic waste into landfills. Other municipalities may have less stringent or no projections for waste reduction, but they too will likely follow California's lead eventually. Fines for non-compliance may be high and it is still unclear how this will be monitored. An innovative, inexpensive method of reducing compostable food waste and select plastics material, while at the same time reducing disposal costs and potential fines and improving public perception of Army installations would be an easily achievable goal with a high return on investment.

2. MATERIAL AND METHODS

2.1 Supplies

Mealworms (*Tenebrio molitor*) were originally ordered from Carolina Biological Supply Company (Burlington, North Carolina) and sustained on a mixture of PS #6 foam (Uline Catalog No. s-23554, Pleasant Prairie, Wisconsin), HDPE #2 and LDPE #4 (Uline Catalog No. s-19363), and organic rolled oats (Bob's Red Mill, Milwaukie, OR). Mealworms were maintained over successive generations as reported in Jung, et al., 2024 at 25°C, in the dark, and at approximately 50% humidity.

2.2 First compost trial

Food wastes and plastics were added at a 50:1 ratio to a 3.8 L (23 × 23 × 11 cm) bin containing 100 g of oats and 5 g cardboard. Approximately 274 mealworms of various sizes weighing 13.6 g were added to each of three experimental and three control (no food waste) bins. All food and plastics wastes were itemized and weighed (Table 1). This trial was terminated after 36 days.

2.3 Second compost trial

Mealworms of various sizes were collected over a 2-week period and stored in 4°C. Healthy worms (n=1000; 48 g collective wieght) of various sizes were collected into three separate containers and weighed. These mealworms were added to 15 L (42.5 × 30.2 × 17.8 cm) bins with both food and non-food waste (food:non-food = 2.5:1); no bran or oats were added (Table 2). This trial was terminated after 30 days.

2.4 Large compost trial

Food and plastics waste from a cafeteria trash bin was sorted and recorded as type, form, and weight and distributed evenly to three experimental (mealworm-containing)

TABLE 1: Itemized food and plastics waste added to first compost trial. Exp=experimental trial.

	Exp A	Exp B	Exp C	Control A	Control B	Control C
Egg carton - foam (PS)	0.8	0.7	0.7	0.6	0.6	0.7
Biohazard bag (HDPE)	0.3	0.2	0.3	0.2	0.2	0.3
Storage bag (LDPE)	0.6	0.5	0.7	0.7	0.6	0.5
Foam Packing (LDPE)	0.4	0.3	0.3	0.4	0.4	0.4
Styrofoam blocks (PS)	0.9	0.9	0.9	0.9	0.9	0.9
Cardboard	4.8	5.1	4.8	4.2	4.5	4.9
Non-food waste (g)	**7.8**	**7.7**	**7.7**	**7.0**	**7.2**	**7.7**
Grapes	76.90	72.00	67.30			
Strawberry	71.10	71.20	69.70			
Apple	138.9	152.9	152.9			
Bread	23.60	23.70	26.30			
Raspberry	43.30	45.50	45.20			
Blueberry	30.10	30.30	30.80			
Total food (g)	383.9	395.6	392.2			
Ratio food:non-food	**49.22**	**51.38**	**50.94**			

and one control bin without mealworms. Roughly 1500 mealworms (83 g collective wieght of each batch) were added to each of the three bins which were 49 L (56 × 43 × 30.5 cm) and contained both food and non-food waste (food:non-food = 0.4:1) (Table 3) and 36 g oats. This trial was terminated after 100 days.

TABLE 2: Itemized food and plastics waste added to second compost trial. Exp=experimental trial.

	Exp A	Exp B	Exp C	Control (no worms)
Packing foam (PS)	13.8	15.8	16.9	14.9
Egg carton (PS)	6.20	5.90	6.30	6.80
Clamshell for berries (PET)	10.1	10.2	9.50	9.80
Small bubble wrap (PE)	2.70	2.80	2.90	2.80
Small crouton bag (PE)	0.43	0.43	0.43	0.43
Small storage bag (PE)	0.25	0.25	0.25	0.25
Large plastic bag (PE)	12.7	14.3	12.0	10.5
Cardboard	35.2	35.4	35.6	35.6
Non-food waste (g)	**46.2**	**49.7**	**48.3**	**45.5**
Croutons	17.5	17.6	17.7	13.3
Apple	30.2	37.5	38.0	36.6
Lettuce	4.90	5.50	5.20	5.30
Avocado shell	4.30	4.70	4.30	5.70
Strawberry	58.1	58.8	67.0	58.4
Grapes	43.1	45.6	42.5	43.6
Raspberry	13.8	12.9	13.7	12.6
Eggshell	2.90	3.30	3.10	4.00
Blueberry	16.2	13.4	15.9	19.2
Red onion	10.9	9.20	10.1	7.90
Total food (g)	**202**	**209**	**218**	**207**
Ratio food:non-food	**2.48**	**2.45**	**2.59**	**2.55**

2.5 Modeling to extrapolate to large-scale composting

Accurately tracking the number of mealworms in a population over time is a daunting task. An adult female darkling beetle can lay hundreds of eggs over her lifetime, and these eggs are very small and adhere to surfaces, making them difficult to even see, much less separate from a compost bin and count. During the early instars of their larval lifecycle, mealworms are likewise almost invisible to the naked eye. As the insect population grows over time, these counting difficulties are only exacerbated. To predict the long-term size of the mealworm population supported by differently sized compost bins, and thereby estimate the expected rate of plastic consumption, we devised an ODE (ordinary differential equation)-based population dynamics model. This model tracks the subpopulations of *T. molitor* in five different lifecycle stages: egg, larva, pupa, reproducing beetle, and non-reproducing beetle. The adult population is divided into two subpopulations since female darkling beetles do most of their egg laying within the first two months of their adult lives, even though their overall lifespan lasts ~6 months.

Assuming a roughly equal sex ratio across the total population, the temporal subpopulation of eggs, denoted as $N_E(t)$, will grow at a rate proportional to half the size of the young adult (reproducing) beetle subpopulation, $N_b(t)$. The constant of proportionality, which we term the fecundity, f, estimates the average number of eggs each young female lays in a week. The number of eggs is depleted as the eggs hatch into larvae. The rate of this process will be proportional to $N_E(t)$, with a constant of proportionality, η_e, which we dub the egg hatching rate. These considerations lead to the following differential equation for $N_E(t)$:

$$\frac{dN_E(t)}{dt} = \frac{1}{2} f N_b(t) - \eta_e N_E(t) \quad (1)$$

The subpopulation of larvae, which we denote $N_L(t)$, will grow at a rate $\eta_e N_E(t)$, since all eggs are assumed to even-

TABLE 3: Itemized food and plastics waste added to third compost trial. Exp=experimental trial.

	Exp A	Exp B	Exp C	Control (no worms)
Clamshell plate (PS)	55.7	34.0	48.9	35.0
Large drink cup (PS)	25.4	20.8	25.5	26.8
Small food bowl (PS)	9.50	7.60	11.4	8.40
Soft drink bottle sliced (PE)	9.90	8.10	13.3	12.6
Clear pie tray clamshell (PS)	8.90	9.80	8.60	9.60
Small fry tray	6.20	6.20	7.40	8.70
Coffee cup	23.3	20.7	14.9	16.1
Napkins	70.2	53.1	106.4	53.9
Non-food waste (g)	209	160	236	171
Food waste (g) (bananas, cornbread, collard greens, pie crust, broccoli, rice, hull peas)	71.2	77.4	74	78.1
Ratio food:non-food	0.340507	0.482845	0.313029	0.456458

tually hatch. This is an approximation, of course, but not an unreasonable one, so long as we assume the system is maintained at optimal breeding conditions. Larvae are lost to two processes: pupation and death, both of which should occur at rates proportional to $N_L(t)$. We approximate pupation as occurring at a constant rate, ρ, but we assume that the death rate should also depend upon the total population of larvae and adults, $N_T(t)$. In particular, we assume that the system has an optimal carrying capacity, K, determined by the external conditions and the available surface area available to the organisms, and the death rate should increase the more the total population of feeding organisms (i.e., excluding eggs and pupae) exceeds this capacity. These considerations lead to the following differential equation:

$$\frac{dN_L(t)}{dt} = \eta_e N_E(t) - \rho N_L(t) - \frac{\mu_L}{K} N_L(t) N_T(t) \tag{2}$$

wherein μ_L is the fixed mortality rate of larvae at capacity (N_T=K).

The population of pupae, $N_P(t)$, should grow at a rate $\rho N_L(t)$ and decline at a rate $\eta_p N_P(t)$, wherein η_p is the pupal hatching rate. This leads to the following differential equation for $N_P(t)$:

$$\frac{dN_P(t)}{dt} = \rho N_L(t) - \eta_p N_P(t) \tag{3}$$

Unlike darkling beetle eggs, which hatch at very high rates under optimal conditions, pupae have a much lower successful hatch rate, so we will assume that the young beetle population, $N_b(t)$, grows as $x_P \eta_p N_P(t)$, wherein $x_P \in [0,1]$ estimates the fraction of pupae that successfully emerge as adult beetles. The young beetle population will decline as a function of both their aging out of the reproductively active stage (parametrized at a rate α) and their death, modeled using a logistic term similar to that in Equation 2 (with the same capacity, K, but a different mortality rate, μ_B). These considerations lead to the following differential equation:

$$\frac{dN_b(t)}{dt} = \chi_P \eta_p N_P(t) - \alpha N_b(t) - \frac{\mu_B}{K} N_b(t) N_T(t) \tag{4}$$

Finally, the nonreproductive beetle population, $N_B(t)$, will grow as younger beetles age into the class and decline exclusively through death (with the same mortality rate, μ_B):

$$\frac{dN_B(t)}{dt} = \alpha N_b(t) - \frac{\mu_B}{K} N_B(t) N_T(t) \tag{5}$$

The total population of feeding organisms, $N_T(t)$ will always equal the sum total of the larval and adult populations:

$$N_T(t) = N_L(t) + N_b(t) + N_B(t) \tag{6}$$

Together, Equations 1-6 constitute a closed, nonlinear dynamical system that can be solved numerically. We will be principally interested in the long-time behavior of solutions, which, as we will discuss in the proceeding section, can be characterized analytically.

3. RESULTS AND DISCUSSION

3.1 First compost trial

The food:non-food ratio for the first trial was 50:1, using primarily fruit and bread, and the plastics were largely the plastics they had been exposed to previously (Figure 1). Approximately 274 mealworms (13.6 g) of various sizes were added to each bin and they were housed in the dark at 25°C. At 36 days the mealworms were counted (Figure 2) and the plastics were itemized, weighed, and the percent consumption by mealworms was determined (Figure 3). Interestingly, the experimental group experienced 71% attrition which was likely due primarily to the presence of fungi which had overtaken the experimental groups. In the absence of decomposing fruit, the control group only exhibited 35% attrition. The control group performed better due to numbers and health of survivors.

3.2 Second compost trial

The food:non-food ratio for the second trial was 2.5:1, using primarily fruits and vegetables, and the plastics were collected from household waste (Figure 4). Approximately 1000 mealworms (48 g) of various sizes were added to each bin and they were housed in the dark at 25°C. A larger container was used to provide enough space for the plastics. At 30 days the trial was terminated due to massive fungal growth. Numerous mealworms were floating in the decaying liquid of the food and few, if any, survivors were

FIGURE 1: First compost feeding trial showing mealworms added to an (A) experimental bin and a (B) control bin without food.

seen. This was surprising since there was more space available for the mealworms to climb away from the liquid and there was much less food available for decay than the first trial. However, it is possible that the plastic may have prevented natural evaporation of the decaying food, and this may have been remedied with the addition of oats as dry bedding and food that could have absorbed some of the moisture.

3.3 Large compost trial

Food and plastics waste (food:non-food = 0.4:1) from a cafeteria trash bin was sorted and distributed to three experimental bins containing 1500 mealworms (83 g) each and one control bin (49 L each) (Figure 5). The trial was terminated at 100 days and observations were made and plastics sorted and weighed. Unfortunately, the need to weigh the starting material as wet weight meant that after 100 days in a semi-arid environment the liquid from the food waste had evaporated and/or food had been consumed, so the light plastic materials did not exhibit significant differences in weight between the mealworm (MW) and control treatments. However, visual inspection showed a different picture with obvious signs of biting and consumption of most PS plastics and paper waste material by the mealworms after 100 days in the experimental groups but not in the control (Figure 6). The small food bowl was the only material that showed a marked decrease in weight in the MW treatment compared to the control (Figure 7). There was no visible consumption of the clear plastics. No food remained in the experimental bins and very little oats

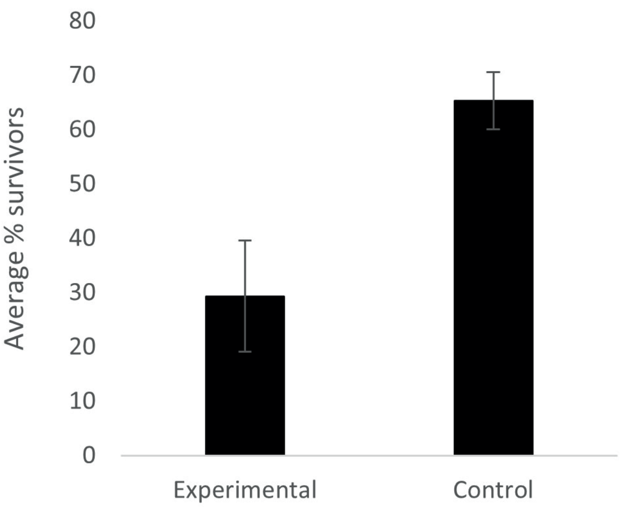

FIGURE 2: Percent survivors after 36-day compost feeding trial.

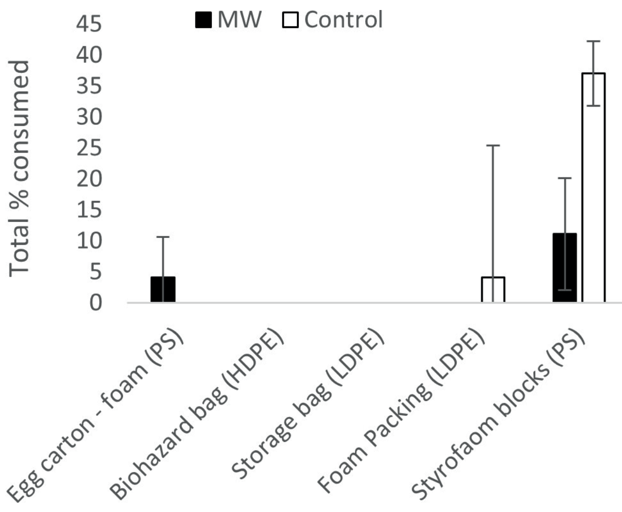

FIGURE 3: Percent of plastics consumed over 36 days.

FIGURE 4: Second compost feeding trial showing (A) food and (B) non-food materials added to bins.

FIGURE 5: A) Food and non-food waste collected from a cafeteria trash bin and sorted into (B) experimental and (C) control bins.

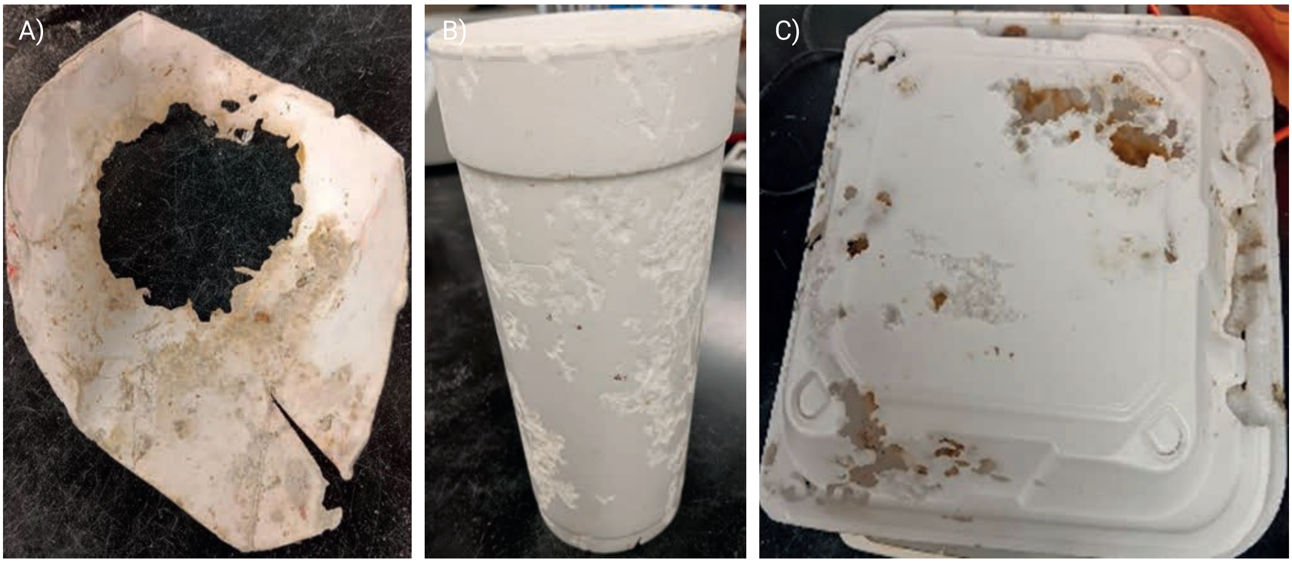

FIGURE 6: (A) Paper and (B&C) PS plastics showing bite marks and consumption of the material by mealworms during the 100-day large compost trial.

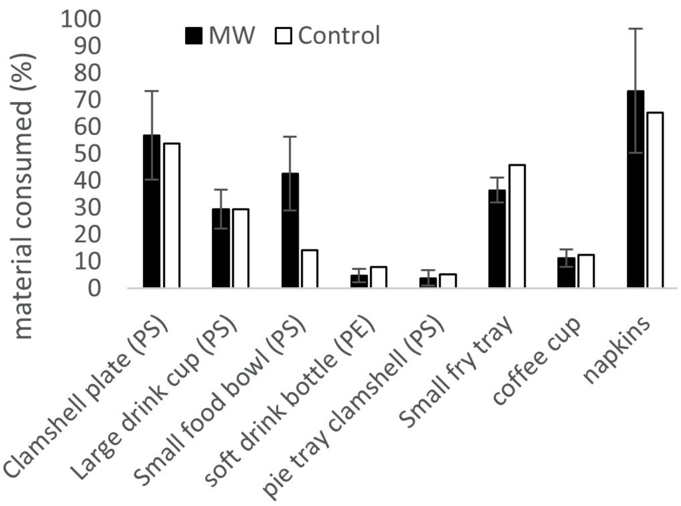

FIGURE 7: Percent consumption by weight of waste material after 100 days in the large compost trial.

remained. Mold/fungi was present, but it was not as overwhelming as the previous compost trials.

Overall, the higher diversity and lower quantity of composted food proved to be a better combination for survival of the mealworms. Although each of the experimental bins started with approximately 2500 larvae, only about 10% remained at a measurable size after 100 days (157 ± 21). There were numerous adults (103 ± 20) and pupae (34 ± 6), and too many early instar larvae (<1mm) to count.

3.4 Modeling for scale-up

Mathematical modeling provides an alternative means for predicting the amount of plastic that will be consumed by a dynamic population of *T. molitor*. Assuming this population is confined to a compost bin in which an excess of edible materials is perpetually maintained, it will eventually reach a steady state wherein the birth of new organisms is offset by the death of older ones. For times much longer than that required to reach this stationary state, the average rate of plastic consumption should remain constant, since it will directly depend upon the steady population of feeding organisms. To estimate this population, we look for the fixed points of the dynamical system defined by Equations 1-6.

This system has only a single physically accessible fixed point (fixed points involving negative organism counts are discarded). We can solve for the value of this fixed point analytically. Defining the steady state subpopulation $N_x(t\to\infty)$ as N_x (with no temporal argument), we find:

$$N_E = \frac{f}{2\eta_e} N_b,$$

$$N_L = \left(\frac{\left(\frac{\alpha\mu_L}{\mu_B} - \rho\right) + \sqrt{\left(\frac{\alpha\mu_L}{\mu_B} - \rho\right)^2 + \frac{2f\mu_L\chi_P\rho}{\mu_B}}}{\frac{2\mu_L\chi_P\rho}{\mu_B}} \right) N_b \equiv \lambda N_b$$

$$N_P = \frac{\rho}{\eta_p} N_L$$

$$N_B = \frac{\alpha N_b^2}{\chi_P\rho N_L - \alpha N_b}$$

$$N_b = \frac{\chi_P\rho\lambda - \alpha}{\frac{\mu_B}{K}\left(\lambda + 1 + \frac{\alpha}{\chi_P\rho\lambda - \alpha}\right)} \qquad (7)$$

We can estimate the various parameters of this population model from the existing body of literature (https://www.breedinginsects.com/yellow-mealworm-life-cycle/), (Eberle et al., 2022; Liu et al., 2020; Palumbo et al., 2024). Under optimal conditions, mealworm eggs are estimated to hatch over the course of 7-10 days, so we can estimate an average hatch rate of η_e=0.824 weeks^{-1}. Female darkling beetles can lay as many as 500 eggs during their lives, mostly during the first two months of adulthood, leading to an estimation of the fecundity as f=62.5 weeks^{-1}. Although the rate of egg laying is not uniformly distributed over this interval, the parameter f represents an average rate across the entire subpopulation of young adult beetles.

Mealworm larvae can start pupating after 10-12 weeks, giving an estimated pupation rate of ρ=0.091 weeks^{-1}. Assuming ample nutrients and space are available to the population, as much as 97% of the larval subpopulation can survive to pupate, suggesting an estimated larval mortality rate of μ_L=0.088 weeks^{-1}. Note that under these optimal conditions, the effects of cannibalism will be minimal. The optimal population density for *T. molitor* has been estimated at 0.25 individuals per square centimeter, suggesting that the carrying capacity of the model should be K=0.25 A, wherein A is the surface area available to the population within the composting container in units of square centimeters. This area will be estimated as the exposed cross-sectional area of the composting container.

Pupae commonly hatch after only about a week, so we estimate η_P=1.0 weeks^{-1}. Only about 75% of pupae actually hatch, under optimal conditions, so we set x_P=0.75. As already mentioned, adult female beetles are only fertile for about two months, so this suggests we set the aging rate α=0.125 weeks^{-1}. Beetles can live between 3-6 months, so we estimate the average beetle mortality rate as μ_B=0.056 weeks^{-1}.

Using the above parameter values and the average rates of larval plastic consumption cited earlier, we plot the estimated total mass of consumed plastic as a function of exposed composting surface area in Figure 8. Because we found that adult beetles consume minimal plastic, and because their steady subpopulation is orders of magnitude lower than the steady subpopulation of larvae, we can estimate the total plastic consumption using N_L rather than N_T.

Despite the complexity of the steady-state subpopulation expressions in Equation 7, a simple linear relationship between the feeding surface area and the average plastic consumption rate emerges. The amount of time required for a system to reach this limiting consumption rate will naturally vary as a function of the initial population size. A smaller seed population of mealworm larvae will take longer to reach steady state than a larger seed population. As mentioned, these estimates also depend upon the composting site being regularly refilled so that edible resources (both food and non-food) remain in excess as well as being maintained at an optimal temperature and humidity level. As our experiments have demonstrated, enforcing this latter constraint without allowing mold and mildew to infiltrate the system represents a significant practical challenge.

4. CONCLUSIONS

The primary objective of this research was to determine the feasibility of breaking down plastics in a novel fashion using plastivorous mealworms in a food and mixed plastics waste compost. The combination of food and plastics waste would be most beneficial to household food composting, but our results demonstrate a need for separate composting, as food waste may be too heavy and wet for the mealworms. The positive takeaway is that unlike recycling, where food scraps need to be removed from the plastics material, food remnants would likely be beneficial in a plastics waste compost, as might small quantities of cardboard and paper waste.

Our coarse-grained continuum population model suggests that even a modestly-sized compost bin, after sufficient time, can support a mealworm population substantial enough to degrade plastics at an appreciable rate—far faster than their natural degradation rate in a landfill. For example, choosing the smallest bin size used in our experiments, our model predicts that it can support a steady population of ~1100 mealworms that can degrade plastic at a rate of ~2 g of polystyrene foam per week. Although it is difficult to understand how this might translate to an individual or household, one estimate states the average American generates approximately 1,800-2,900 g per year (35-56 g per week) (https://www.sandiego.gov/sites/default/files/appendix_b_air_quality_and_greenhouse_gas_emissions_estimates_0.pdf), making composting with mealworms at a household scale seem feasible.

The accuracy of the model predictions remains in question for the present, due to the difficulties in tracking the size of a *T. molitor* population over time, but we can at least say that its predictions represent, at best, a crude upper-bound on the plastic consumption rate of mealworms. This is because our model relies on the assumption of an idealized system environment that, as we have demonstrated, is difficult to achieve even in the laboratory, much less in the compost bin of an Army installation.

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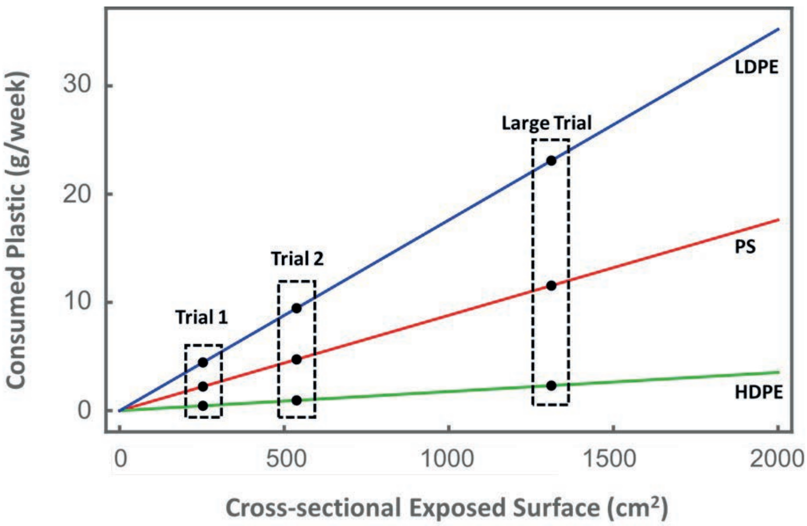

FIGURE 8: Predicted plastic consumption (in grams per day) for steady T. molitor populations as a function of the accessible feeding surface (approximated as the exposed cross-sectional area of the compost bin (in cm2). The curves, from top-to-bottom, represent systems in which the insects are exposed to a mixture of food and either LDPE, PS, or HDPE plastics, respectively. The bin sizes for the three experimental trials are indicated on each curve for reference.

mealworm colony and compost maintenance. Opinions, interpretations, conclusions, and recommendations are those of the authors and are not necessarily endorsed by the U.S. Army.

REFERENCES

Brandon AM, El Abbadi SH, Ibekwe UA, Cho Y-M, Wu W-M, Criddle CS (2020) Fate of hexabromocyclododecane (HBCD), a common flame retardant, in polystyrene-degrading mealworms: elevated HBCD levels in egested polymer but no bioaccumulation Environ Sci Technol 54:364-371. https://doi:10.1021/acs.est.9b06501

Brandon AM et al., (2018) Biodegradation of polyethylene and plastic mixtures in mealworms (larvae of *Tenebrio molitor*) and effects on the gut microbiome Environ Sci Technol 52:6526-6533. https://doi:10.1021/acs.est.8b02301

Eberle S, Schaden L-M, Tintner J, Stauffer C, Schebeck M (2022) Effect of temperature and photoperiod on development, survival, and growth rate of mealworms, *Tenebrio molitor* Insects 13:321. https://doi:10.3390/insects13040321

Jung CM, Carr MR, Cropek DM (2024) Lessons in Rearing Mealworms for Plastics Degradation vol ERDC/EL TN-24-1. US Army Engineer Research and Development Center, Vicksburg, MS. https://doi:10.21079/11681/48551

Kundungal H, Gangarapu M, Sarangapani S, Patchaiyappan A, Devipriya SP (2019) Efficient biodegradation of polyethylene (HDPE) waste by the plastic-eating lesser waxworm (*Achroia grisella*) Environmental Science and Pollution Research 26:18509-18519 https://doi:10.1007/s11356-019-05038-9

Liu C, Masri J, Perez V, Maya C, Zhao J (2020) Growth performance and nutrient composition of mealworms (*Tenebrio molitor*) fed on fresh plant materials-supplemented diets Foods 9:151. https://doi:10.3390/foods9020151

Palumbo B, Cullere M, Singh Y, Pontalti E, Dalle Zotte A (2024) Yellow mealworm: effects of adults breeding density on adults and larvae performances from an industrial perspective Animal 101360 https://doi:10.1016/j.animal.2024.101360

Parker L (2019) The world's plastic pollution crisis explained. (accessed 01-16-2026) https://www.nationalgeographic.com/environment/habitats/plastic-pollution/#close.

Pham TQ, Longing S, Siebecker MG (2023) Consumption and degradation of different consumer plastics by mealworms (*Tenebrio molitor*): Effects of plastic type, time, and mealworm origin J Clean Prod 403. https://doi:10.1016/j.jclepro.2023.136842

Urbanek AK, Rybak J, Wróbel M, Leluk K, Mirończuk AM (2020) A comprehensive assessment of microbiome diversity in *Tenebrio molitor* fed with polystyrene waste Environmental Pollution 262:114281. https://doi:10.1016/j.envpol.2020.114281

Yang J, Yang Y, Wu WM, Zhao J, Jiang L (2014) Evidence of polyethylene biodegradation by bacterial strains from the guts of plastic-eating waxworms Environmental Science and Technology 48:13776-13784. https://doi:10.1021/es504038a

Yang SS et al., (2018) Ubiquity of polystyrene digestion and biodegradation within yellow mealworms, larvae of *Tenebrio molitor* Linnaeus (Coleoptera: Tenebrionidae) Chemosphere 212:262-271. https://doi:10.1016/j.chemosphere.2018.08.078

Yang Y, Wang J, Xia M (2020) Biodegradation and mineralization of polystyrene by plastic-eating superworms Zophobas atratus Science of the Total Environment 708:135233 . https://doi:10.1016/j.scitotenv.2019.135233

Yang Y et al., (2015a) Biodegradation and Mineralization of Polystyrene by Plastic-Eating Mealworms: Part 1. Chemical and Physical Characterization and Isotopic Tests Environmental Science & Technology 49:12080-12086. https://doi:10.1021/acs.est.5b02661

Yang Y et al., (2015b) Biodegradation and mineralization of polystyrene by plastic-eating mealworms: Part 2. role of gut microorganisms Environmental Science & Technology 49:13087-13093. https://doi.org/10.1021/acs.est.5b02663

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WASTE TO ENERGY: HYDROTHERMAL CARBONIZATION OF AGRO-WASTE FOR THE PRODUCTION OF HIGH VALUABLE SOLID BIOFUEL

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ABSTRACT

In the present study, hydrothermal carbonization (HTC) of banana peel waste, showing a moisture content of 85 wt%, was carried out in a 500 mL stirred stainless steel autoclave at a temperature ranging between 180 and 250°C, for a residence time of 0.5 h. The reaction solid residue was recovered via vacuum filtration and washed with distilled water before drying it in a ventilated oven at 105°C, until constant weight, and stored in 50 mL plastic vials, before analytical characterization. Banana hydrochars were characterized in terms of proximate, elemental analysis, FTIR and high calorific value (HHV). Hydrochar mass yield showed a maximum at 220°C (32 wt%), while gas and liquid mass yields increased monotonically with reaction temperature, up to 70 and 2 wt%, respectively. HHV increased with reaction temperature up to 27.13 MJ/kg, corresponding to an energy densification ratio EDR of about 169%. Mass yield and energy properties of banana hydrochar prove that a HTC treatment carried out at 250°C for 0.5 h is able to produce a valuable solid biofuel showing the characteristics of lignite.

1. INTRODUCTION

The urgent need for a sustainable and green production of thermal energy and electricity by renewable sources exploitation, has boosted, in the last few years, the seek for a clean technology to address the climatic challenge and ensure reliable energy solutions (Cotana et al., 2014; Ibarra-Gonzalez and Rong, 2019; Shen, 2020). An adequate thermochemical valorization of waste biomass, globally available in large quantities, may accelerate the transition from fossil fuels to carbon neutral energy systems, reducing both greenhouse gas emissions and waste generation (Lee et al., 2020; Nunes et al., 2020). Between the different technologies investigated in the last decades for the energy exploitation of waste biomass, thermochemical treatments such as pyrolysis and gasification have received high attention due to the possibility of producing energy carriers, either solid (bio-char), liquid (bio-oil) and gaseous (syngas), potentially capable to substitute the mineral fuels (R. Volpe et al., 2016). Unfortunately, the majority of waste biomass has high moisture content (often higher than 50% by weight) that limits the use of such dry thermochemical technologies, given the high energy costs associated with the removal of water before the substrate processing (Messineo et al., 2012; Messineo and Panno, 2008).

More recently, hydrothermal carbonization technology (HTC) has shown the high potential for transforming high moisture waste biomass into a carbon and energy dense material, named hydrochar, with energy characteristics close to the pyrochar (Picone et al., 2021). HTC allows to convert wet biomass in closed reactors and in the presence of additional water at subcritical conditions (180-250°C and 10-50 bar), for a residence time from few minutes to several hours, with high energy efficiency (Becker et al., 2019). Compared to the above mentioned dry thermochemical treatments, HTC ensures several advantages, mostly related to the significant energy saving, due the absence of a biomass pre-drying step, the higher conversion degree at milder reaction conditions, and the generation of solid products with a very low ash content (Heidari et al., 2019; Rodríguez Correa et al., 2018). During HTC, subcritical water plays a crucial catalytic role, as it promotes the splitting of the highly heterogeneous lignocellulosic macromolecules by hydrolysis. The generated soluble biomass fragments are then involved into a multiphase and intricate network of competing reactions, with dehydration, decarboxylation, poly-condensation, and aromatization recognized as the main conversion pathways (Wang et al., 2018). Dehydration and decarboxylation lead to a notable

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reduction of oxygenated species in the initial feedstock, substantially increasing the carbon content in the final solid product, while poly-condensation and aromatization govern the hydrochar forming mechanism by liquid-to-solid and solid-solid (pyrolysis-like) reactions (Ischia et al., 2024; Wu et al., 2025).

In the past years, HTC has been successfully employed for the treatment of a wide range of feedstocks, such as food waste, sewage sludge, residues from the agro-inudstrial sector, and plastic substrates, largely showing to be a valid solution within the realm of waste management and the production of valuable bio-derived materials with immense potential in both energy and environmental applications (Volpe et al., 2020). Numerous scholars investigated the potential use of hydrochar for the generation of clean thermal energy by direct combustion, observing promising performances due to the high energy density (higher heating value in the range of 20-28 MJ/kg), higher thermal stability compared to raw feedstock, prevention of fouling and slagging due to the low ash content, and reduced harmful emissions compared to coal combustion (Wang et al., 2018; Wilk et al., 2020). Tunability of hydrochar properties (surface area, pore structure, surface functionality, and chemical composition) by simply adjusting HTC parameters, allowed for the optimization of its electrochemical performances in specific energy storage applications (Zhou et al., 2021). As well documented in literature, hydrochar has been also effectively used for water remediation, by removing heavy metals and organic persistent pollutants, soil amendment, and carbon sequestration (Regmi et al., 2012; Wu et al., 2025).

Until now, extensive work has been dedicated to studying the influence of feedstock nature, in combination with the operating conditions (reaction severity, starting biomass concentration, use of catalysts, etc), on the process outputs, in terms of mass yield and physico-chemical properties, and therefore on their suitability for specifically selected applications (Hedin and Kruse, 2020; Sharma et al., 2020). Nevertheless, further research activity is essential not only for the advancing of scientific knowledge and a complete understanding of the process fundamentals, but also for optimizing and tailoring the conversion toward HTC products with perfectly controlled and desired features (Ischia et al., 2024).

In the present study, we report novel results regarding banana peel waste energy upgrading via HTC at the temperature range of 180-250°C. The produced hydrochars were analytically characterized to measure the energy densification and observe the evolution of proximate, elemental composition, and chemical functionality, after the thermochemical treatment. This work clarly demonstrates the huge potential of HTC to efficiently handle wet agro-waste biomass and convert it into an energy-dense material that might be used as an alternative, carbon neutral solid fuel, contributing to the reduction of fossil fuels exploitation and to the development of a circular bioeconomy.

2. MATERIALS AND METHODS

2.1 Sample preparation and experimental procedure

Banana peel waste (BPW) was used as feedstock for the HTC treatment. BPW represents a typical agro-food waste and was collected as a food residue of kitchen and restaurant. Fresh BPW was ground by using a knife miller to provide completely homogeneous pulp composition with a reproducible size reduction. The so obtained pulp was portioned and stored in sealed plastic bags in a freezer at -20°C, before their use. The sample was slowly left defrosting down to room temperature, before each HTC experiment. A sample of as above prepared feedstock was dried in a ventilated oven at 105°C until constant weight to quantify the moisture content (MC); BPW showed a MC value of 85.0 ± 0.2 wt%.

BPW defrosted wet feedstock, without pre-treatment (Volpe et al., 2022), was hydrothermally carbonized in a 500 mL stainless steel batch reactor with stirring (200 rpm) at the temperatures of 180, 220 and 250°C with a residence time of 0.5 h. For each experiment the reactor was loaded with approximately 300.0 g of wet BPW (corresponding to 45 g of dry sample) and 200.0 g of distilled water (DW), to reach a prefixed biomass to water mass ratio (B/W) of 0.1. After loading the feedstock, the reactor was vented with pure nitrogen (Alphagaz 1, Airliquid™) and the reaction temperature set at the desired temperature. When the conversion time, at set temperature, has passed, the reactor was cooled down to room temperature by a cooling water circulating system, allocated inside the reactor. The gas phase produced was flowed, through an outlet valve, into a graduated cylinder filled with water, and its mass was evaluated applying the ideal gas law, assuming atmospheric pressure, temperature of 30°C and CO_2 as the sole gaseous product. Indeed, it was seen that the typical CO_2 molar fraction in HTC gas is higher than 0.90, with minor traces of other species such as CO, H_2 and CH_4 (Picone et al., 2021). The hydrochar was separated from HTC process water (PW) by vacuum filtration and dried at 105°C until constant weight prior to mass yield determination. All experimental tests were performed at least three times to evaluate the deviation of data and ensure reproducibility (the results were considered valid if standard deviation < 2.0%).

2.2 Analytical methods and calculation

Composition and energy content of raw feedstock and hydrochars were evaluated by proximate analysis and higher heating value (HHV) measurement, respectively. Proximate analysis was carried out by a LECO Thermogravimetric Analyser TGA 701 (following ASTM D7582-15 standard method). Approximately 400 mg of each solid sample was used to quantify volatile matter (VM), ashes (ash) and fixed carbon (FC) as follows: 5°C/min ramp to 105°C in air, held until constant weight (<±0.05%) to remove moisture in open crucibles; 16°C/min ramp from 105 to 900°C, hold time 7 min, in nitrogen and covered crucibles to determine VM as the mass lost; natural cooling down to 500°C in nitrogen; 30°C/min ramp in air to 800°C and isothermal until constant weight in open crucibles to evaluate ASH as the remaining mass. FC was estimated by subtracting VM and ash percentages from 100%.

Elemental analysis was carried out using a CHN LECO 828 series element analyzer using argon as carrier gas and EDTA as calibration standard.

A calorimeter LECO AC500 was used to measure HHVs according to the CEN/TS 14918 standard. FTIR analysis was carried out on the raw material and corresponding hydrochars samples using a Shimadzu IR Tracer spectrometer in mid-IR mode, equipped with a Universal ATR sampling device containing diamond/ZnSe crystal. The spectra were recorded in the range from 600 to 4000 cm^{-1}, with a resolution of 4 cm^{-1}, by averaging 64 scans, baseline corrected and normalized. The hydrochar mass yield (MY_{HC}) was calculated through Equation 1, where m_{HC} is the mass of hydrochar (d.b.) collected after the conversion and m_{raw} is the mass (d.b.) of solid starting material.

$$MY_{HC} = m_{HC}/m_{raw} \quad (1)$$

Similarly, gas mass yield (MY_{gas}) was evaluated through Equation 2.

$$MY_{gas} = m_{gas}/m_{raw} \quad (2)$$

PW mass yield (MY_{PW}) was calculated by difference through Equation 3.

$$MY_{PW} = 1 - MY_{HC} - MY_{gas} \quad (3)$$

The fuel ratio (FR) was defined as the mass yield ratio of FC to VM to measure the carbonization degree of resulting hydrochar:

$$FR=FC/VM \quad (4)$$

The energy yields (EY) were calculated by Equation 5 to evaluate the energy retention efficiency in the hydrochar after the biomass HTC conversion.

$$EY = (HHV_{HC}/HHV_{raw})\ MY_{HC} \quad (5)$$

3. RESULTS AND DISCUSSION

3.1 Mass yields of BPW HTC residues

Figure 1 shows the mass yields of banana HTC residues. As predictable, gas mass yield slightly increased with HTC reaction temperature, reaching the maximum value of 1.7%, at 250°C. Indeed, since gas formation during the hydrothermal treatment is highly endothermic, it was promoted with the increase of reaction severity, coherently with other studies reported in literature (Wang et al., 2018). Conversely, the hydrochar mass yield did not follow a monotonic behaviour in terms of final mass recovered, with the maximum value (~ 32%) reached at 220°C.

The majority of HTC studies report a monotonic decrease of hydrochar mass yield with the increase of operating temperature, which is ascribed to the dominant hydrolysis and devolatilization of starting feedstock over the formation of stable hydrochar structures (Wang et al., 2020, 2018). However, in some cases, the hydrochar production could be enhanced by increasing the thermal energy supplied to the reactive multiphase mixture inside the reactor. As well known, temperature is the most influential factor in HTC and can indeed significantly affect the conversion pathways leading to solid products (Sharma et al., 2020). HTC relies on a complex and intricate network of both parallel and consecutive reactions that include the formation of solid particles at solid-solid and liquid-solid interfaces (Fu et al., 2025). On the one hand, the so-called primary char is obtained from the unhydrolyzed starting biomass through pyrolysis-like reactions, responsible of the increased carbon content in the recovered solid product (Lucian et al., 2018). On the other hand, the hydrolyzed fraction is involved into competing subreactions, with the formation of highly reactive liquid by-products (Fu et al., 2025). Among these, organic acids, such as acetic, lactic, propionic, levulinic, and formic acids, play a key role in HTC as they lower the pH of reaction medium and spark an autocatalytic process consisting of

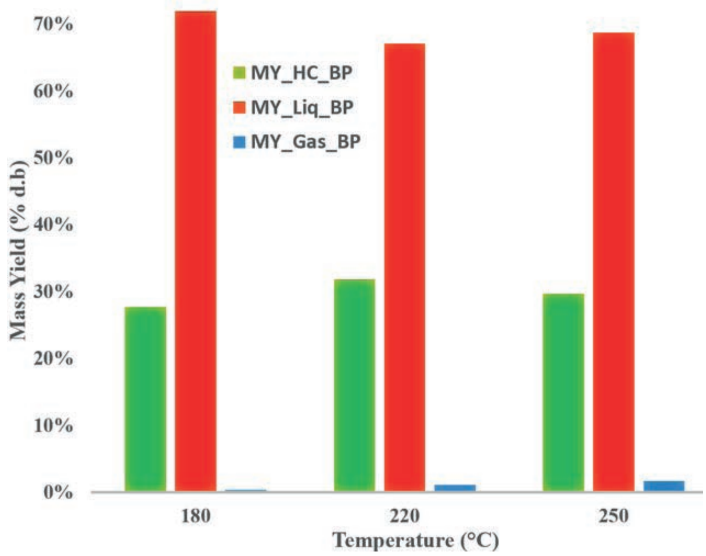

FIGURE 1: Mass yields of banana peel waste HTC residues at different process temperatures (solid, liquid, and gas yields are reported in green, red, and blue, respectively).

a boosted biomass depolymerization. The enhanced PW reactivity may promote the accumulation of hydrochar precursors in the liquid phase, including furans, furfurals, and aldheydes, which could then lead to the endothermic formation of additional solid phases with quasi-spherical structure, named secondary char, via interphase catalyzed reactions. Typically, once formed, the new carbon-based microparticles tend to adhere to the primary char surface and contribute to the increase of hydrochar recovery, especially at severe reaction conditions (Picone et al., 2021; Sevilla and Fuertes, 2009; Wang et al., 2020).

In light of these considerations, the results presented in this section, suggest that the increase of temperature could have promoted the concentration of liquid precursors for secondary char formation and its subsequent deposition of the available primary char sites, leading to a greater hydrochar mass yield at 220 and 250°C, compared to that obtained at 180°C.

It is noteworthy that a fraction between 60 and 70% of the starting feedstock remained in the spent process solvent. Detailed investigations have shown that PW composition can significantly vary according to the feedstock type used for the HTC treatment, however, the most common species detected include organic acids, in high concentration, and other unconverted biomass fragments, heavy metals, nutrients, and organic compounds potentially toxic for the environment (Wang et al., 2020). For this reason, in order achieve the process economic and environmental feasibility, and thus promote HTC upscaling, an adequate valorization of residual PW represents one of the biggest scientific and technical challenges. In the recent past, several scholars demonstrated that PW holds remarkable potential for innovation across energy, agriculture and environmental fields (Ischia et al., 2024), therefore we will focus on the development of effective strategies for PW reuse and/or industrial valorization, in a future study.

3.2 Hydrochars characterization

Table 1 reports the results from the HHV measurement, energy properties (EY, FR) calculation and proximate analysis for BPW raw material and hydrochars obtained at different reaction temperatures.

It can be observed that the energy content of produced hydrochars is substantially higher than that of raw BPW, due to the carbonization reactions. In addition, HHV increased with reaction temperature, up to 27.13 MJ/kg at 250°C, reaching a value approaching that of lignite. The energy yield of hydrochars reached a value of 50.0% at 250°C, with a corresponding EDR of approximately 169%. Proximate analysis data show that, as expected, VM decreased with HTC process temperature, while FC increased up to approximately 31.5%, at 250°C. The ash content was significantly reduced during HTC, from 8.5 for raw BPW to 0.75-1.67% for hydrochars at 180, 220 and 250°C, respectively. This confirms that during the wet thermochemical treatment, most of inorganics are moved to the aqueous phase, thus considerably reducing the risk of slagging and fouling phenomena in the case of combustion (Reza et al., 2013).

Elemental analysis data, shown in Table 2, corroborate the findings of proximate analysis. For hydrochars, the content of elemental carbon increased with increasing temperature, reaching a maximum value of 63.65%, at 250°C. The increase of carbon content with the process severity was accompanied by a drastic reduction of oxygenated species, while, as already reported in HTC literature, hydrogen content is not significantly affected by the reaction, showing stable values between 5.9 and 6.3%.

Thermochemical evolution of BPW undoubtedly prove that coalification occurred during HTC, with a notable densification of carbon and FC contents as the main result (Danon et al., 2014; García-Bordejé et al., 2017). Dehydration and decarboxylation dominate the devolatilization process, leading to the significant decrease of elemental oxygen and VM contents. As both reactions are highly endothermic, they were likely promoted with the increase of reaction temperature, as suggested by the evident reduction of H/C and O/C atomic ratios (Jia et al., 2022; Wang et al., 2022). In this regard, the Van Krevelen diagram reported in Figure 2 clearly shows that dehydration is the main mechanism involved during HTC of BPW. Furthermore, it can be seen that hydrochars progressively assume a fuel behavior resembling that one of peat, at 220°C, and lignite, at 250°C. The carbon densification reflects the higher energy content found for the hydrochars, compared to the parent substrate (raw BPW), definitely proving the effectiveness of HTC technology for the production of solid biofuels with high potential.

Surface chemical functionalities in raw material and hydrochars were studied via ATR-FTIR analysis.

Figure 3 shows the FTIR spectra in ATR mode of raw BPW and corresponding hydrochars obtained at 180, 220 and 250°C.

The spectra show a broad band between 3400 and 3300 associated to the stretching vibration of O-H, involved in hydrogen bonding. It can be seen that this band shifted toward higher wavenumber after the HTC treatment due to the concentration of phenolic compounds in lignin and carboxylic acids groups. The increasing aromatic character in hydrochars with HTC temperature is also

TABLE 1: Energy properties of BPW and correspondig hydrochars.

Sample	HHV* [MJ/kg]	EY [%]	EDR [%]	VM* [%]	FC* [%]	Ash* [%]	FR
BPW_{raw}	16.09 (± 0.18)	100.0	100.0	80.72	10.78	8.50	0.13
BPW_{HC180}	23.48 (± 0.31)	40.5	146.0	75.82	23.43	0.75	0.31
BPW_{HC220}	24.93 (± 0.32)	49.3	155.0	67.59	31.49	0.92	0.47
BPW_{HC250}	27.13 (± 0.26)	50.0	168.6	63.07	35.26	1.67	0.56

** on a dry basis (d.b.)*

TABLE 2: Elemental analysis data, O/C and H/C atomic ratios of BPW and corresponding hydrochars.

Sample	C [%]	H [%]	N [%]	O* [%]	O/C**	H/C**
BPW_{raw}	40.85	5.87	0.84	43.95	1.72	0.81
BPW_{HC180}	54.95	6.59	1.45	36.27	1.44	0.49
BPW_{HC220}	59.55	6.30	1.35	31.89	1.27	0.40
BPW_{HC250}	63.65	6.31	1.33	27.05	1.19	0.32

** O = 100−C−H−N- Ash; ** atomic ratio*

visible with the increase of the aromatic C-H stretching mode at 2950 and 1610 cm^{-1}, together with the bending out of plane vibration at around 720 cm^{-1}. Aliphatic CH_2 and CH_3 vibration modes were also detected in all samples at 2920 and 2850 cm^{-1}, demonstrating that aliphatic character is maintained during HTC due to aromatic structures. C=O stretching vibration at around 1700 cm^{-1} (aldehydic and/or carboxylic compounds) increased with HTC temperature, as visible in the spectra of BPW at 220 and 250°C. The carboxylic C-O stretching mode was revealed by the bands at 1150-1100 cm^{-1}, while the sharp band at 1030 cm^{-1} could be assigned to inorganig C-O vibration. In summary, data from FTIR indicate that HTC strongly promote aromatization and decarboxylation of the BPW sample with temperature.

4. CONCLUSIONS

Hydrothermal carbonization (HTC) is a thermochemical process used for the conversion of high moisture biomass, like agro-industrial wastes, into carbon-dense hydrochars, which could find application as valuable solid biofuels.

In this study, we investigated the thermochemical upgrading of banana peel waste (BPW) via HTC and assessed the effect of reaction temperature on energy properties and composition of the hydrochars produced. Experiments were carried out by using a 50 mL stainless steel batch reactor at the three different temperatures of 180, 220 and 250°C, with a biomass to water mass ratio of 0.1 and keeping a fixed reaction time of 0.5 h. After HTC, the solid products were characterized in terms of higher heating value (HHV), proximate and elemental composition, and surface chemical functionality.

While, as expected, gas production increased with temperature due to the endothermic nature of decarboxylation reactions, differently from the common outcomes reported in HTC literature, solid mass yield did not follow a monotonic decrease, reaching a maximum value of approximately 32%, at 220°C. This peculiar trend might be closely related to the formation of additional solid phases, known as seconary char, at the more severe reaction conditions, with an observable increase of solid mass recovered as the main result. Data from proximate and elemental analyses suggest that dehydration was the main converson pathway and led to a drastic reduction of elemental oxygen content, due to the biomass devolatilization. Correspondingly, a significant carbon and energy densification was found after the BPW thermal treatment, with the hydrochar obtained at 250°C that showed a carbon content of approximately 64%, a fixed carbon of about 35%, and a HHV of 27.1 MJ/kg. HTC also promoted the inorganics removal from the solid phase and

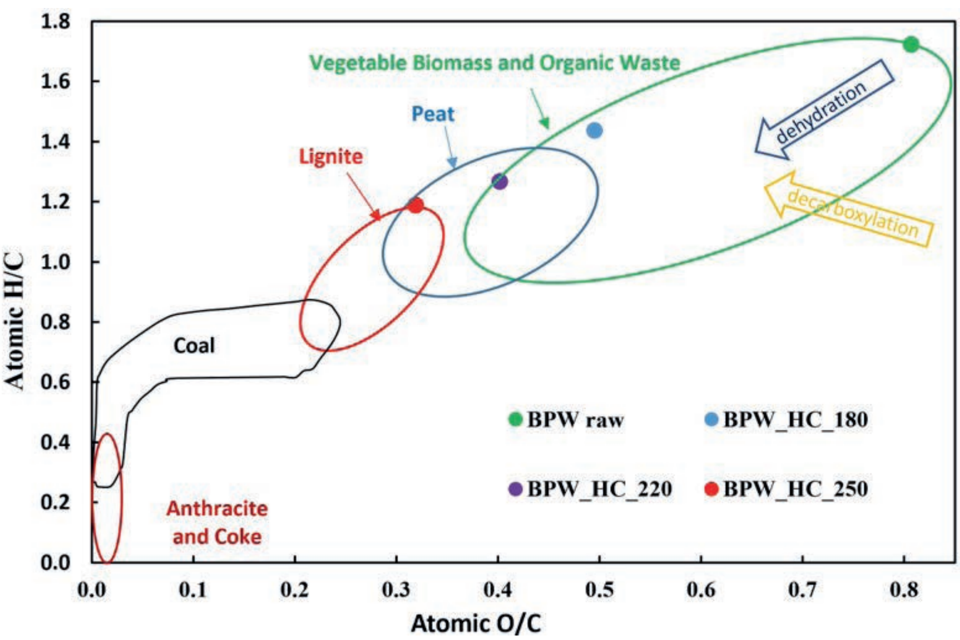

FIGURE 2: Van Krevelen diagram of raw banana peel waste and corresponding hydrochars.

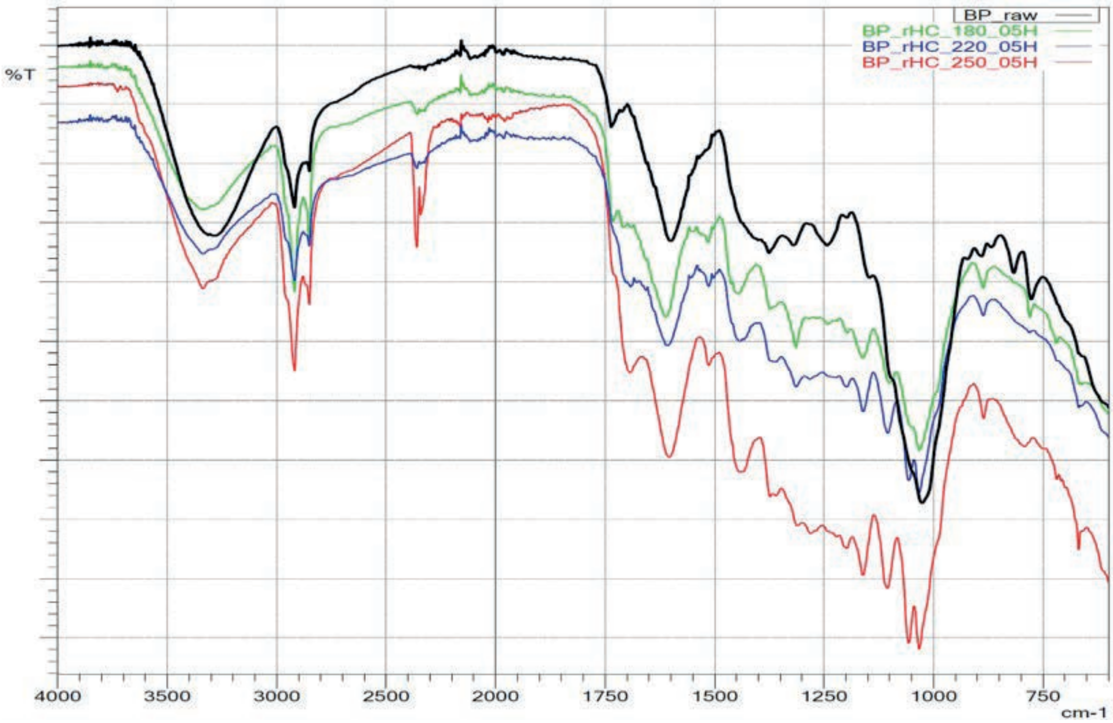

FIGURE 3: FTIR-ATR of raw banana peel waste and corresponding hydrochars.

thus reducing the potential risk of slagging and fouling during combustion.

The obtained results clearly prove that the solid products from BPW HTC should be considered as excellent candidates for co-combustion with mineral coal in existing boilers and for the progressive replacement of fossil fuels. Supplemental hydrochars' characterization, such as the investigation on their chemical composition and structure, would be crucial for a deeper understanding of formation mechanisms and optimization of process conditions, toward tailored target products. A detailed analysis of HTC process water is a lack of our work and will be proposed in a future study to corroborate the experimental findings reported, in terms of chemical reaction pathways, and suggest an adequate valorization method for the residual liquid fraction. Other potential future developments of this study might include the expansion of basic research on unconventional substrates, such plastic waste, investigating the real effects of different substrates interaction during co-HTC, and the implementation of machine learning to assist the process engineering.

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REFERENCES

Becker, G.C., Wüst, D., Köhler, H., Lautenbach, A., Kruse, A., 2019. Novel approach of phosphate-reclamation as struvite from sewage sludge by utilising hydrothermal carbonization. J. Environ. Manage. 238, 119–125. https://doi.org/10.1016/j.jenvman.2019.02.121

Cotana, F., Messineo, A., Petrozzi, A., Coccia, V., Cavalaglio, G., Aquino, A., 2014. Comparison of ORC turbine and stirling engine to produce electricity from gasified poultry waste. Sustain. 6, 5714–5729. https://doi.org/10.3390/su6095714

Danon, B., Hongsiri, W., van der Aa, L., de Jong, W., 2014. Kinetic study on homogeneously catalyzed xylose dehydration to furfural in the presence of arabinose and glucose. Biomass and Bioenergy 66, 364–370. https://doi.org/10.1016/j.biombioe.2014.04.007

Fu, H., Wang, F., Liu, Z., Duan, X., Wang, L., Yi, W., Zhang, D., 2025. Role of secondary char on the fuel properties and pyrolysis behaviors of hydrochars: Effect of temperature and liquid-solid ratio. Fuel Process. Technol. 267, 108167. https://doi.org/10.1016/j.fuproc.2024.108167

García-Bordejé, E., Pires, E., Fraile, J.M., 2017. Parametric study of the hydrothermal carbonization of cellulose and effect of acidic conditions. Carbon N. Y. 123, 421–432. https://doi.org/10.1016/j.carbon.2017.07.085

Hedin, N., Kruse, A., 2020. Assessment of the effects of process water recirculation on the surface chemistry and morphology of hydrochar Renew. Energy 155, 1173-1180. https://doi.org/10.1016/j.renene.2020.04.050

Heidari, M., Dutta, A., Acharya, B., Mahmud, S., 2019. A review of the current knowledge and challenges of hydrothermal carbonization for biomass conversion. J. Energy Inst. 92, 1779–1799. https://doi.org/10.1016/j.joei.2018.12.003

Ibarra-Gonzalez, P., Rong, B.G., 2019. A review of the current state of biofuels production from lignocellulosic biomass using thermochemical conversion routes. Chinese J. Chem. Eng. 27, 1523–1535. https://doi.org/10.1016/j.cjche.2018.09.018

Ischia, G., Berge, N.D., Bae, S., Marzban, N., Rom, S., Farru, G., Kulli, B., Fiori, L., 2024. Advances in Research and Technology of Hydrothermal Carbonization : Achievements and Future Directions. Agronomy 14. https://doi.org/10.3390/agronomy14050955

Jia, J., Wang, R., Chen, H., Xue, Q., Yin, Q., Zhao, Z., 2022. Interaction mechanism between cellulose and hemicellulose during the hydrothermal carbonization of lignocellulosic biomass. Energy Sci. Eng. 10, 1–12. https://doi.org/10.1002/ese3.1117

Lee, M., Lin, Y.L., Chiueh, P. Te, Den, W., 2020. Environmental and energy assessment of biomass residues to biochar as fuel: A brief review with recommendations for future bioenergy systems. J. Clean. Prod. 251, 119714. https://doi.org/10.1016/j.jclepro.2019.119714

Lucian, M., Volpe, M., Gao, L., Piro, G., Goldfarb, J.L., Fiori, L., 2018. Impact of hydrothermal carbonization conditions on the formation of hydrochars and secondary chars from the organic fraction of municipal solid waste. Fuel 233, 257–268. https://doi.org/10.1016/j.fuel.2018.06.060

Messineo, A., Panno, D., 2008. Municipal waste management in Sicily: Practices and challenges. Waste Manag. 28, 1201–1208. https://doi.org/10.1016/j.wasman.2007.05.003

Messineo, A., Volpe, R., Asdrubali, F., 2012. Evaluation of net energy obtainable from combustion of stabilised olive mill by-products. Energies 5, 1384–1397. https://doi.org/10.3390/en5051384

Nunes, L.J.R., Causer, T.P., Ciolkosz, D., 2020. Biomass for energy: A review on supply chain management models. Renew. Sustain. Energy Rev. 120, 109658. https://doi.org/10.1016/j.rser.2019.109658

Picone, A., Volpe, M., Giustra, M.G., Bella, G. Di, Messineo, A., 2021. Hydrothermal Carbonization of Lemon Peel Waste : Preliminary Results on the Effects of Temperature during Process Water Recirculation. Appl. Syst. Innov. 4. https://doi.org/10.3390/asi4010019

Regmi, P., Luis, J., Moscoso, G., Kumar, S., Cao, X., Mao, J., Schafran, G., 2012. Removal of copper and cadmium from aqueous solution using switchgrass biochar produced via hydrothermal carbonization process. J. Environ. Manag. 109, 61-69. https://doi.org/10.1016/j.jenvman.2012.04.047

Rodríguez Correa, C., Ngamying, C., Klank, D., Kruse, A., 2018. Investigation of the textural and adsorption properties of activated carbon from HTC and pyrolysis carbonizates. Biomass Convers. Biorefinery 8, 317–328. https://doi.org/10.1007/s13399-017-0280-8

Sevilla, M., Fuertes, A.B., 2009. The production of carbon materials by hydrothermal carbonization of cellulose. Carbon N. Y. 47, 2281–2289. https://doi.org/10.1016/j.carbon.2009.04.026

Sharma, H.B., Sarmah, A.K., Dubey, B., 2020. Hydrothermal carbonization of renewable waste biomass for solid biofuel production: A discussion on process mechanism, the influence of process parameters, environmental performance and fuel properties of hydrochar. Renew. Sustain. Energy Rev. 123, 109761. https://doi.org/10.1016/j.rser.2020.109761

Shen, Y., 2020. A review on hydrothermal carbonization of biomass and plastic wastes to energy products. Biomass and Bioenergy 134, 105479. https://doi.org/10.1016/j.biombioe.2020.105479

Volpe, M., Fiori, L., Merzari, F., Messineo, A., Andreottola, G., 2020. Hydrothermal carbonization as an efficient tool for sewage sludge valorization and phosphorous recovery. Chem. Eng. Trans. 80, 199–204. https://doi.org/10.3303/CET2080034

Volpe, M., Picone, A., Luz, F.C., Mosonik, M.C. at, Volpe, R., Messineo, A., 2022. Potential pitfalls on the scalability of laboratory-based research for hydrothermal carbonization. Fuel 315, 123189. https://doi.org/10.1016/j.fuel.2022.123189

Volpe, R., Messineo, S., Volpe, M., Messineo, A., 2016. Catalytic Effect of Char for Tar Cracking in Pyrolysis of Citrus Wastes , Design of a Novel Experimental Set Up and First Results 50, 181–186. https://doi.org/10.3303/CET1650031

Wang, R., Jia, J., Jin, Q., Chen, H., 2022. Forming mechanism of coke microparticles from polymerization of aqueous organics during hydrothermal carbonization process of biomass. Carbon N. Y. 192, 50–60. https://doi.org/10.1016/j.carbon.2022.02.030

Wang, R., Jin, Q., Ye, X., Lei, H., Jia, J., Zhao, Z., 2020. Effect of process wastewater recycling on the chemical evolution and formation mechanism of hydrochar from herbaceous biomass during hydrothermal carbonization. J. Clean. Prod. 277, 123281. https://doi.org/10.1016/j.jclepro.2020.123281

Wang, T., Zhai, Y., Zhu, Y., Li, C., Zeng, G., 2018. A review of the hydrothermal carbonization of biomass waste for hydrochar formation: Process conditions, fundamentals, and physicochemical properties. Renew. Sustain. Energy Rev. 90, 223–247. https://doi.org/10.1016/j.rser.2018.03.071

Wilk, M., Magdziarz, A., Kalemba-Rec, I., Szymańska-Chargot, M., 2020. Upgrading of green waste into carbon-rich solid biofuel by hydrothermal carbonization: The effect of process parameters on hydrochar derived from acacia. Energy 202, 117717. https://doi.org/10.1016/j.energy.2020.117717

Wu, S., Wang, Q., Man, H., Wu, D., Lu, Q., Pan, S., Bai, J., Cui, D., Zhang, X., 2025. Recent advances on hydrothermal carbonization of biomass for carbon-negative materials : From mechanistic insights to functional applications. Ind. Crop. Prod. 237, 122142. https://doi.org/10.1016/j.indcrop.2025.122142

Zhou, Y., Shi, W., Engler, N., Nelles, M., 2021. High-value utilization of kitchen waste derived hydrochar in energy storage regulated by circulating process water. Energy Convers. Manag. 229, 113737. https://doi.org/10.1016/j.enconman.2020.113737

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WASTE-TO-ENERGY: REUSE OF OIL SPILLED ON THE BRAZILIAN COAST IN 2019 THROUGH CO-PROCESSING

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ABSTRACT

In August 2019, one of Brazil's largest environmental disasters occurred as a result of a crude oil spill from an oil tanker between August 2019 and March 2020, reaching the northeastern and southeastern coastal regions of the country, known as the Blue Amazon. Eleven states, 130 municipalities, and 1,009 localities were affected, in addition to 10 ecosystems, more than 57 protected areas, and 34 animal species. Despite the extreme circumstances, the disaster did not receive a timely response from the federal government, and this inaction led to more significant long-term consequences, such as the compromise of preservation areas and the temporary suspension of artisanal fishing, damaging the livelihoods of thousands of residents in the affected localities. The waste was mostly collected by volunteer citizens and reused for energy recovery in co-processing as an alternative fuel in cement kilns. Around 5,379.76 tons of waste were collected, most of which were sent to cement industries. It was found that a large amount of the collected oil was reused energetically as fuel in co-processing industries. The present article is a descriptive case report and aims to analyze and offer an overview of the energy reuse of oil collected after spills in co-processing activities. Primary data were obtained from official reports, while secondary data were acquired through a literature review. The results showed that reusing oil collected from spills in the cement industry was considered alternative for waste destination instead of disposal in landfills, and new destinations could be implemented by the official environmental organisms.

1. INTRODUCTION

In August 2019, the largest oil spill in Brazilian coastal history occurred. From August 2019 to March 2020, 11 states, 130 municipalities, and 1,009 localities were affected by the oil spill (IBAMA, 2020a). The disaster caused extensive damage to marine biodiversity, affecting 10 ecosystems, more than 57 protected areas, and 34 species (Soares et al., 2020). At the time, many artisanal fishermen suffered the impacts of crude oil contamination, in particular due to the suspension of fish and shellfish consumption in the Northeast region of Brazil (Pena et al., 2020). There was an immediate reduction in the sale and consumption of fish, severely affecting the main source of income and food for thousands of families, especially female shellfish gatherers (Silva et al., 2021). Furthermore, there was significant damage to tourism activities, a major source of income for the affected coastal areas.

Figure 1 shows the extent (700 km) of the Brazilian coastal area affected by the oil spill, reaching the northern and northeastern areas, above and below the Equator due to the bifurcation of the Brazilian Current. This region is an oceanic frontier of the continent, belonging to an area known for its natural riches as the Blue Amazon.

According to the Brazilian Institute of Environment and Renewable Natural Resources (IBAMA, 2020b), the oil that reached the Brazilian coast in 2019 (Figure 2) presented high viscosity, high density, and low concentrations of volatile compounds due to the oil quality and weathering. Because of its characteristics, manual removal techniques (using shovels) were used. Visible monitoring sensors did not detect oil slicks, indicating that the oil was not drifting on the sea surface but below the surface, and for this reason it was undetectable by remote observation systems (IBAMA, 2022a).

After the spill, much attention was dedicated to discovering the source of the oil, which was later confirmed by Brazilian authorities to be of Venezuelan origin. Initially, most of the oil residue was collected by volunteer citizens without personal protective equipment (PPE) or any type of training. However, as the oil spill spread, there was greater

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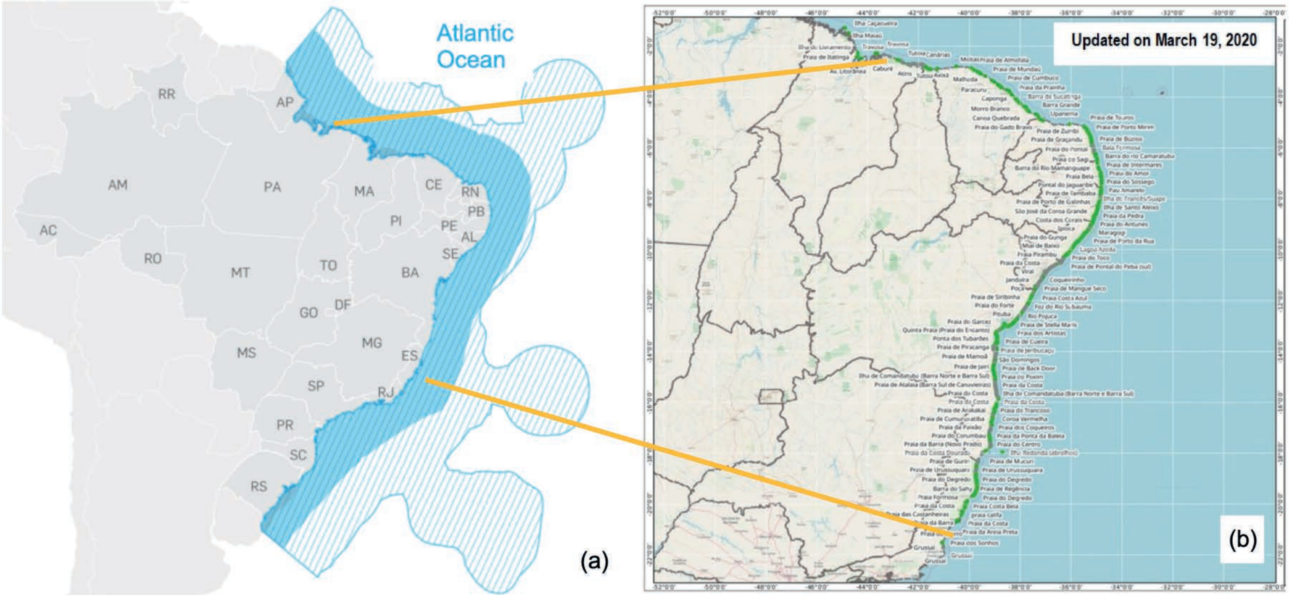

FIGURE 1: (a) Blue Amazon (blue area). Source: IBGE/Estadão; (b) Extent of the oil spill on the Brazilian coast (green area). Source: IBAMA, 2020a.

FIGURE 2: (a) Oil spill in Pau Lagoon, in Coruripe (located at Alagoas State), in 2019. Source: MPF, 2025b; (b) Volunteers attempt to rescue young man trapped in the thick layer of oil at Itapuama Beach (located at Pernambuco State). Source: Leo Malafaia/Folha PE, 2019.

mobilization of state environmental agencies, city governments, and the Brazilian Navy.

According to Tassigny et al. (2024), risk communication in disasters involving marine pollution is essential to prevent further damage and reduce uncertainty – particularly standardized disclosure directed at communities surrounding the disaster, in addition to wide communication by the press directed at society. The people most exposed to the oil were those who worked on coast cleanup activities and workers involved in artisanal fishing (Carmo and Teixeira, 2020). This accident stands out among other oil spills in Brazil due to its extent (over 700 km), and despite its proportions and environmental damage, there have been no measures of compensation for the damage caused by the pollution (Lawand et al., 2021).

A guidebook (IBAMA, 2022b) was published under the guidance of International Tanker Owners Pollution Federation with recommendations to the population regarding personal protective equipment use and with instructions for manual cleaning — which was the most appropriate method for collecting the oil in order to reduce environmental damage.

At the beginning, the Brazilian Navy (2020) indicated that the incident occurred gradually, with the first signs of oil not suggesting a situation of national significance, and in the absence of identification of the polluter, the Brazilian Petroleum Company (Petrobras) was asked to clean up the beaches due to its quantity of equipment and greater presence along the coast.

In this scenario, one of the biggest challenges was managing the large amount of waste generated by the spill. The solution found by the authorities was to send most of the oily waste for co-processing, where it would be reused as fuel in clinker-production kilns. The accident's scale increased and 5,379.76 tons of waste were collected, raising concerns about its destination. Most of it was sent to cement industries, while a smaller amount was sent to other final destinations, such as Class I landfills. The oil sent to cement factories was reused as an alternative fuel in co-processing activities.

Considering the oil spill reached the Brazilian coast in 2019–2020, this study aims to verify the destination of oil spilled in different Brazilian coastal regions, observing the energetically feasible use as a contribution to the improvement of environmental protective action in the future. This work contributes to the dissemination of data on the energy recovery of oily waste in co-processing, as an alternative for waste disposal after the spill that occurred in Brazil in 2019.

One of the biggest challenges during the elaboration of the study was obtaining data related to the destination of oil collected on the coast after the accident, because there was not wide dissemination at the time and most of the information was obtained through the Access to Information Act (legislation for open information). Therefore, this study is highly relevant, since it aims, in the context of this accident, to reveal information regarding the amount and application of the oil collected on the coast, contributing to the improvement of the most effective actions in similar future situations. Among the main limitations found during the study, the following stand out: reliance on official data, the absence of independent environmental performance metrics, the lack of emission measurements, and the constraints of a short communication format.

2. METHOD

This study a short communication presenting a descriptive case report and is based mainly on official reports regarding the energy reuse of oil collected after spills in co-processing activities. The research is performed using scientific bibliographic sources and documents available related to the oil destination from the spill of 2019–2020 that reached the Brazilian coast.

First, a survey was conducted on primary data with public information obtained directly from the authorities involved in the oil spill that occurred in Brazil in 2019 — such as the amount and destination of the oil collected. The work proceeded with consultation of documents under the Access to Information Act (Brasil, 2011) and official websites. Subsequently, secondary data relevant to the object of study were added based on the selection of scientific papers from bibliographic searches. Also, some complementary data were carefully added from news published in the more formal press.

3. RESULTS AND DISCUSSION

According to the Federal Public Prosecutor's Office (MPF, 2025a), based on data from Chico Mendes Institute for Biodiversity Conservation, shown in Table 1, a total of 13 federal conservation units, among others, were affected by the oil spill in 2019. Some of these reserves are extractive, meaning they are home to native communities that depend primarily on the ecosystem services provided by the environment.

TABLE 1: Conservation units affected by the oil spill on the Brazilian coast in 2019. Source: based on data from MPF (2025a).

Location (State)	Conservation Unit
Maranhão (MA)	Cururupu Extractive Reserve
	Lençóis Maranhenses National Park
Piauí (PI)	Delta do Parnaíba Environmental Protection Area
Ceará (CE)	Jericoacoara National Park
	Batoque Extractive Reserve
	Prainha do Canto Verde Extractive Reserve
Paraíba (PB)	Barra do Rio Mamanguape Environmental Protection Area
	Manguezais da Foz do Mamanguape Area of Ecological Interest
	Acaú-Goiana Extractive Reserve
Pernambuco (PE)	Costa dos Corais Environmental Protection Area
Alagoas (AL)	Lagoa do Jequiá Extractive Reserve
	Piaçabuçu Environmental Protection Area
Sergipe (SE)	Santa Isabel Biological Reserve
Bahia (BA)	Cassurubá Extractive Reserve Corumbau Marine Extractive Reserve

IBAMA (2016) presents a National Emergency Action Plan for Wildlife Affected by Oil, which is responsible for assigning the members of a wildlife team to the spill site, with the objective of protecting and developing an appropriate response to wildlife affected by oil. Meanwhile, the National Contingency Plan sets out the responsibilities and guidelines whose competences and actions are assigned to each agency linked to the organizational structure in order to deal with oil accidents in waters under national jurisdiction. Such plans were used in dealing with the accident, the extent of which required a widely coordinated action.

Around 5,379.76 tons of waste were collected, 30% of which was oil. In accordance with technical guidelines for waste management (IBAMA, 2020c), the oil collected along the coastline was stored in big bags. Table 2 shows the reported amount of waste collected by state in the areas affected by the oil spill. The states of Alagoas (2,564.58 tons) and Pernambuco (1,676.26 tons) had the highest amounts of waste collected.

Furthermore, based on external access to IBAMA public data, most of this waste was sent to licensed cement kilns for co-processing, as shown in Table 3, at the Votorantim Cimentos facilities in Laranjeiras (located in Sergipe State) and Sobral (located in Ceará State), Mizu Cimentos (located in Baraúna, Rio Grande do Norte State), InterCement (located in Bahia State), and Cimenteira APODI (located in Ceará State) for energy generation in their respective industries.

In some states, waste transportation and disposal were carried out by the state itself (through the state environmental agency or city halls), while in other locations PETROBRAS was responsible for transporting the waste to cement industries (Brazilian Navy, 2020). In 2019, Mizu Cimentos received approximately 60.3 tons of contaminated material collected from the beaches of Northeast Brazil (Mizu, 2025).

TABLE 2: Amount of waste collected by state after the oil spill on the Brazilian coast in 2019. Source: based on data from IBAMA (2022).

State	Waste Collected by Petrobras (tons)	Waste Collected by Local Authorities (tons)	Total Amount Collected (tons)
Alagoas (AL)	16.09	2548.49	2,564.58
Bahia (BA)	95.03	364.46	459.49
Ceará (CE)	0.00	39.76	39.76
Espírito Santo (ES)	0.00	6.26	6.26
Maranhão (MA)	0.01	13.68	13.69
Paraíba (PB)	0.00	0.85	0.85
Pernambuco (PE)	29.84	1,646.42	1,676.26
Piauí (PI)	0.00	10.46	10.46
Rio de Janeiro (RJ)	0.00	0.00	0.00
Rio Grande do Norte (RN)	12.16	33.83	35.18
Sergipe (SE)	356.19	213.16	569.35
Offshore	1.00	2.88	3.88
TOTAL	510.32	4,880.25	5,379.76

It is possible to verify that, even after all the challenges related to storage and logistics costs, most of the oil collected (as shown in Figure 3) on beaches in Northeastern Brazil was reused energetically as an alternative fuel in co-processing industries, instead of being disposed of in landfills without any prior treatment.

The growth in waste generation and its final disposal are major challenges for Brazilian cities. According to estimates, more than 81.6 million tons of municipal solid waste (MSW) were generated in 2024 (ABREMA, 2025). In a scenario of significant final disposal rates, energy recovery methods have emerged as an important and appropriate environmental solution for final disposal. Federal Law No. 12,305 (BRASIL, 2010), established the National Solid Waste Policy, which implemented energy recovery from solid waste in the country; according to its legal criteria, disposal should be considered only after all possibilities for use, reuse, and treatment of waste have been exhausted.

Since then, waste has been considered material with potential for recovery and/or reuse. Although there are legal instruments aimed at waste recovery and reuse activities, little progress has been made in the country. As a result, the reuse of oily waste collected after the 2019 spill in co-processing activities in cement industries has proven to be highly relevant, as it reuses a large amount of waste instead of simply disposing of it in landfills.

Although it is still unclear from an environmental perspective to quantify emissions, changes in carbon footprint, and the possible effects of replacing petroleum coke with pre-processed waste in co-processing, the technolo-

TABLE 3: Quantity and destination of oil collected on beaches in Northeast Brazil after the 2019 environmental disaster. Source: based on data from IBAMA (2022).

State	Estimated Oil Collected by Petrobras (tons)	Estimated Oil Collected by Local Agencies (tons)	Total Amount of Oil Collected (tons)	Temporary Storage	Transportation	Destination
Alagoas	16.09	2,548.49	2,564.58	Petrobras Waste Management Center in Carmópolis (SE)	Petrobras	Votorantim Laranjeiras (SE) and Mizu Cimentos in Baraúna (RN)
Bahia	95.03	364.46	459.49	Petrobras Waste Management Center in Carmópolis (SE); CETREL (in Camaçari)	State	Cimenteira InterCement and CTR-Bahia
Ceará	0.00	39.76	39.76	-	State	Cimenteira APODI (CE)
Espírito Santo	0.00	6.26	6.26	São Luiz da Barra, Linhares and São Mateus - ES	State	Class I Landfill Site operated by Vitória Ambiental
Maranhão	0.01	13.68	13.69	IBAMA (MA), Tutóia, Port Authority in São Luiz, and Waste Management Center in Parnaíba (PI)	State and Petrobras	Class I Waste Management Center of Tiara and Votorantim Cimentos in Sobral (CE)
Paraíba	0.00	0.85	0.85	Cabedelo	State	Class I Metropolitan Landfill in João Pessoa

FIGURE 3: Cleanup carried out by volunteers: (a) Paiva Beach (located in the city of Recife). Source: Oton Veiga/TV Globo, 2019; (b) Itapuama Beach (located at Pernambuco State). Source: Sidney Vieira/Araújo et al, 2020.

gy for reusing oil waste to produce energy in cement industries was applied as an alternative final destination for waste after the oil spill.

Co-processing is an activity that reuses waste of high calorific value and biomass as alternative fuel in cement kilns. The process proposes the replacement, although partial, of fossil fuels (such as coal and petroleum coke) in clinker production kilns – the main raw material and responsible for CO_2 emissions in cement production.

Co-processing is licensed in Brazil by the National Environment Council under Resolution No. 499 (BRAZIL, 2020), with the following definition: "Co-processing of waste in clinker production kilns: environmentally appropriate final destination involving the processing of solid waste as a partial substitute for raw materials and/or fuel in the clinker production kiln system in cement manufacturing".

The waste mixture is uniform in size and has a high calorific value — generally 18 megajoules per kilogram — recovered after collection, sorting, and treatment (BNDES, 2014). It is estimated that in Brazil, alternative fuels account for 32% of thermal substitution in relation to traditional fossil fuels (such as coal and petroleum coke) in co-processing, with an annual processing of 3,258 million tons of waste in the country (ABCP, 2024).

A simple order-of-magnitude estimate of the potential energy recovery from the oily waste was performed using typical calorific values reported in the literature. Since, approximately 5,379.76 tons of waste were collected after the spill, of which about 30% corresponded to oil-contaminated material, it results in an estimated 1,614 tons of oily waste.

$$M_{oil} = 0.30 \times 5379.76\ t \approx 1{,}614\ t \approx 1.6 \times 106\ kg$$

where: M_{oil} = mass of oily waste; 1t = 1000 kg

Also, assuming a typical lower heating value of 18 MJ kg^{-1} for waste used in cement kiln co-processing, the theoretical energy recovery potential is approximately 2.9×10^7 MJ, equivalent to about 29,000 GJ (≈ 8,05 GWh) of thermal energy. This simplified estimate illustrates the significant energetic potential of the recovered oily waste when used as an alternative fuel.

$$E = M_{oil} \times LHV = 1.6 \times 106\ [kg] \times 18\ [MJ]\ [kg^{-1}] \approx 2.9 \times 107\ MJ$$
$$E \approx 2.9 \times 107\ MJ \approx 29{,}000\ GJ \approx 8{,}05\ GWh$$

where: E = thermal energy; LHV = lower heating value

High calorific value waste can be used for co-processing purposes due to the absence of harmful effects on emissions in an industrial facility, the clinker production process, and the final product quality (FRICKE et al., 2015). The residence time in the kilns is usually from 2 to 4 seconds, during which the waste is completely destroyed. One of the biggest problems faced by the cement industry is the large amount of CO_2 emissions related to the calcination of raw materials for clinker production, and therefore co-processing has shown to be an alternative technological strategy to reduce such emissions.

In this way, the reuse of oil collected on the Brazilian coast after the accident in co-processing activities, when possible, was a final destination solution found by the authorities at the time, instead of disposal in landfills. Based on available official data, co-processing was the main destination; however, a more in-depth assessment of environmental performance is needed.

Smoldering (slow combustion) is a flameless combustion process that has been used to treat oily waste, which is used as fuel and has a heat source used to initiate the reaction, since the combustion is sustained by an air distribution (Sabadell et al., 2018). However, it should be noted that the application of smoldering combustion is a relatively recent technology for waste treatment (Yermán, 2016).

Regarding the political implications related to the 2019 oil spill on the Brazilian coast, we can highlight that: emergency waste management protocols should include predefined industrial reuse pathways; transparency of waste destination data must be institutionalized; and co-processing can function as a contingency waste management mechanism in large-scale spills.

4. CONCLUSIONS

The oil spill on the coast of Northeast Brazil in 2019, known as the largest in the country's history and one of the world's biggest environmental disasters, had several impacts on the local population and damaged biodiversity, affecting protected and sensitive environmental areas, as well as damaging artisanal fishing and the livelihoods of thousands of families.

A total of 5,379.76 tons of waste were collected, mainly by volunteer citizens, most of which were sent to cement industries, where they were reused as an alternative fuel for energy generation in clinker production kilns in co-processing activity.

In a scenario of significant waste disposal rates in Brazil, energy recovery methods have emerged as an important and appropriate environmental solution. In this way, co-processing was the alternative found by the authorities at the time, instead of the large amount of oily waste disposed of in landfills.

However, other strategies for collecting and utilizing oily waste (such as smoldering) resulting from environmental accidents should be identified and included in government protocols so that other alternatives can be adopted, especially those that involve a reduction in CO_2 emissions and do not compromise ecosystem services, the maintenance of life, and health.

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REFERENCES

ABCP – Brazilian Portland Cement Association. Associação Brasileira de Cimento Portland. Panorama do Coprocessamento – Brasil 2024 (Ano base 2023). São Paulo, 2024. ABCP, 2024.

ABREMA - Associação Brasileira de Resíduos e Meio Ambiente. Panorama dos Resíduos Sólidos no Brasil 2025. São Paulo, 2025.

ARAÚJO, M. E.; RAMALHO, C. W. N.; MELO, P. W. Pescadores artesanais, consumidores e meio ambiente: consequências imediatas do vazamento de petróleo no Estado de Pernambuco, Nordeste do Brasil. Cadernos de Saúde Pública, 36(1), 230319. 2020. https://doi.org/10.1590/0102-311X00230319

BNDES Brazilian Development Bank. Banco Nacional de Desenvolvimento Econômico e Social. Análise das Diversas Tecnologias de Tratamento e Disposição Final de Resíduos Sólidos Urbanos no Brasil, Europa, Estados Unidos e Japão. Fundação de Apoio ao Desenvolvimento da Universidade Federal de Pernambuco (FADE/UFPE). BNDES, 2014.

BRASIL. Ministry of Environment. Ministério do Meio Ambiente. CONAMA – Conselho Nacional do Meio Ambiente. Resolução CONAMA/MMA Nº499, de 06 de outubro de 2020. Dispõe sobre o licenciamento da atividade de coprocessamento de resíduos em fornos rotativos de produção de clínquer. Diário Oficial da União. Brasília, 2020. BRASIL, 2020.

BRASIL. Access to Information Act. Lei de Acesso à Informação. Lei no 12.527, de 18 de novembro de 2011. Regula o acesso a informações prevista na Constituição Federal; e dá outras providências. Diário Oficial da União. Brasília, 2011. BRASIL, 2011.

BRASIL. Brazilian National Solid Waste Policy Plano Nacional de Resíduos Sólidos (PNRS). Lei nº 12.305 de 02 de agosto de 2010; altera a lei no 9.605, de 12 de fevereiro de 1998; e dá outras providências. Diário Oficial da União. Brasília, 2010. BRASIL, 2010.

Carmo, E. H., Teixeira, M. G. (2020). Desastres tecnológicos e emergências de saúde pública: o caso do derramamento de óleo no litoral do Brasil. Cadernos de Saúde Pública, 36(2). https://doi.org/10.1590/0102-311X00234419

Estadão. (2024). Estadão. https://www.estadao.com.br/sustentabilidade/amazonia-azul-marinha-mar-brasil-ibge/?srsltid=AfmBOopMQueaOaGH3Z4O-pZl6OyhmauvXIXr0FVqzYrVat_hrEiUQ7p4

FRICKE, K.; PEREIRA, C.; LEITE, A.; BAGNATI, M. Gestão Sustentável de Resíduos Sólidos Urbanos: Transferência de Experiência Entre a Alemanha e o Brasil. Braunschweig: Technische Universität Braunschweig, 2015. TU Braunschweig, Germany, 2015.

G1, Globo. Protocolo internacional para limpeza de óleo nas praias traz alertas para 4 situações enfrentadas no Nordeste. (2019, October 29). G1. https://g1.globo.com/natureza/desastre-ambiental-petroleo-praias/noticia/2019/10/29/protocolo-internacional-para-limpeza-de-oleo-nas-praias-traz-alertas-para-4-situacoes-enfrentadas-no-nordeste.ghtml

IBAMA – Brazilian Institute of Environment and Renewable Natural Resources. Instituto Brasileiro do Meio Ambiente e dos Recursos. DIRETORIA DE PROTEÇÃO AMBIENTAL COORDENAÇÃO-GERAL DE EMERGÊNCIAS AMBIENTAIS. Ibama. 2022a. Available on: https://www.gov.br/ibama/pt-br/assuntos/fiscalizacao-e-protecao-ambiental/emergencias-ambientais/manchasdeoleo/arquivos/2022/2022-12-16_sei_ibama_6898984_relatorio_tecnico_ibama.pdf .Accessed on: 17 november 2025.

IBAMA – Brazilian Institute of Environment and Renewable Natural Resources. Instituto Brasileiro do Meio Ambiente e dos Recursos. Orientações. Ibama. 2022b. Available on: https://www.gov.br/ibama/pt-br/assuntos/fiscalizacao-e-protecao-ambiental/emergencias-ambientais/manchasdeoleo/orientacoes - uso-epi .Accessed on: 17 november 2025.

IBAMA – Brazilian Institute of Environment and Renewable Natural Resources. Instituto Brasileiro do Meio Ambiente e dos Recursos. Localidades afetadas. Brasília. IBAMA, 2020a. Available on:https://www.ibama.gov.br/phocadownload/emergenciasambientais/2020/manchasdeoleo/2020-03-19_LOCALIDADES_AFETADAS.pdf . Accessed on: 11 august 2025.

IBAMA – Brazilian Institute of Environment and Renewable Natural Resources. Manchas de óleo. Litoral Brasileiro. Brasília. IBAMA. 2020b. Available on:https://www.gov.br/ibama/pt-br/assuntos/fiscalizacao-e-protecao-ambiental/emergencias-ambientais/manchasdeoleo/arquivos/ibama-manchasdeoleo-desmobilizacao-cartilha_v2.pdf . Brasília, 2020. Accessed on: 11 august 2025.

IBAMA – Brazilian Institute of Environment and Renewable Natural Resources. Orientação técnica: gestão de resíduos. Brasília. IBAMA. 2020c. Available on:https://www.gov.br/ibama/pt-br/assuntos/fiscalizacao-e-protecao-ambiental/emergencias-ambientais/manchasdeoleo/arquivos/2022/20221216_OT_gestao_residuos_oleo.pdf . Brasília, 2020. Accessed on: 11 august 2025.

IBAMA – Brazilian Institute of Environment and Renewable Natural Resources. Plano Nacional de Ação de Emergência para Fauna Impactada por Óleo. IBAMA. 2016. Available on:https://www.ibama.gov.br/phocadownload/emergenciasambientais/paefauna2016_plano.pdf . Brasília, 2016. Accessed on: 25 june 2025

Lawand, A., Dutra de Almeida Silva, C., & Ferreira de Oliveira, L. P. (2021). Derramamento de óleo no nordeste brasileiro: Responsabilização e desdobramentos. Revista De Direito E Negócios Internacionais Da Maritime Law Academy - International Law and Business Review , 1(1), 84-113. Available on: https://mlawreview.emnuvens.com.br/mlaw/article/view/8 . Accessed on: 25 november 2025.

MPF - Federal Public Ministry. Ministério Público Federal. Dúvidas frequentes - Derramamento de óleo na costa brasileira. Available on: <https://www.mpf.mp.br/grandes-casos/casos-historicos/oleo-na-costa-brasileira/duvidas-frequentes>. Accessed on: 10 april 2025. Brasília, 2025. MPF, 2025a.

MPF - Federal Public Ministry. Procuradoria da República no Estado de Sergipe. ACP - Ação Civil Pública nº 0805679-16.2019.4.05.8500 -assinada por procuradores dos nove estados do Nordeste. Available on: https://www.gov.br/ibama/pt-br/assuntos/fiscalizacao-e-protecao-ambiental/emergencias-ambientais/manchasdeoleo/arquivos/2023/20230118_Of_Ext_30_279_2020_CM_MMA_An_Incidente_Poluicao_Oleo_Relatorio_final.pdf. Accessed on: 10 april 2025. Sergipe, 2025. MPF, 2025b.

Marinha do Brasil - Brazilian Navy. Coordenação Operacional. Incidente de Poluição por Óleo na Costa Brasileira. Relatório Final. Rio de Janeiro, 2020. Available on: https://www.gov.br/ibama/pt-br/assuntos/fiscalizacao-e-protecao-ambiental/emergencias-ambientais/manchasdeoleo/arquivos/2023/20230118_Of_Ext_30_279_2020_CM_MMA_An_Incidente_Poluicao_Oleo_Relatorio_final.pdf . Accessed on: 11 august 2025.

Mizu. Socio & Environmental - Mizu. (2025, February 24). Available on:https://mizu.com.br/en/socio-environmental/ . Accessed on: 22 november 2025.

Pena, P. G. L., Northcross, A. L., Lima, M. A. G. de, & Rêgo, R. de C. F. (2020). Derramamento de óleo bruto na costa brasileira em 2019: emergência em saúde pública em questão. Cadernos de Saúde Pública, 36(2). https://www.scielo.br/j/csp/a/RdpV54PDWjxktvSjhJRCvTP/?lang=pt

Sabadell, G., Scholes, G., Thomas, D., Murray, C., Bireta, P., Grant, G., & Major, D. (2018). Ex Situ Treatment Of Organic Wastes Or Oil-Impacted Soil Using A Smoldering Process. WIT Transactions on Ecology and the Environment. https://doi.org/10.2495/wm180341

Silva, L. R. C. da, Pessoa, V. M., Carneiro, F. F., Andrade, N. S. M., & Meireles, A. J. de A. (2021). Derramamento de petróleo no litoral brasileiro: (in)visibilidade de saberes e descaso com a vida de marisqueiras. Ciência & Saúde Coletiva, 26(12), 6027-6036. Available on: https://www.scielo.br/j/csc/a/hKJZft8YxVVw4yhGYK4kyLD/?lang=pt. Accessed on: 28 August 2025.

Soares, M. de O., Teixeira, C. E. P., Bezerra, L. E. A., Paiva, S. V., Tavares, T. C. L., Garcia, T. M., Carneiro, P. B. de M. (2020). Oil spill in South Atlantic (Brazil): Environmental and governmental disaster. Marine Policy, 115, 103879. https://www.sciencedirect.com/science/article/abs/pii/S0308597X19308346

TASSIGNY , Mônica; ROCHA, Ronald; BEZERRA DE MELO GIRÃO MOTA, Andrea. Poluição Marinha E Direito Dos Desastres: Análise Do Incidente De Derramamento De Óleo No Litoral Do Nordeste Brasileiro Sob A Perspectiva Da Comunicação De Risco. Novos Estudos Jurídicos, Itajaí (SC), v. 29, n. 3, p. 591-616, 2024. DOI: 10.14210/nej.v29n3.p591-616. Available on:https://doi.org/10.14210/nej.v29n3.p591-616 . Accessed on: 2 september 2025.

Yermán, L. (2016). Self-sustaining Smouldering Combustion as a Waste Treatment Process. InTech EBooks. https://doi.org/10.5772/64451

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FROM CORRECT WASTE MANAGEMENT TO HEALTH PROTECTION: A FOCUS ON NANOTOXICS

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ABSTRACT

Waste generation has increased drastically worldwide in recent decades. Less than 20% of waste is recycled each year, and one-third of all food is wasted. While a fair level of knowledge exists concerning the general risks, there are notable gaps in understanding specific aspects of human health risks related to improper waste disposal and recycling, such as plastics in all types of wastes, and metallic nanoparticles from urban and sanitary wastewater, representing both emerging pollutants with a broad range of human health impacts. Although general environmental risks associated with waste mismanagement are well recognized, significant knowledge gaps persist regarding the specific human health impacts linked to emerging particulate pollutants. In particular, microplastics and metallic nanoparticles represent critical contaminants of growing concern due to their persistence, mobility, and broad toxicological profiles. This narrative review synthesizes the current scientific understanding of microplastic and metallic nanoparticle pollution, particularly through urban and sanitary wastewater, their toxicological profiles, and the limitations of current recycling and circular economy strategies in addressing these issues. It examines the environmental occurrence, release pathways, and health effects of these particles, highlighting how improper waste disposal, recycling processes, and wastewater treatment inefficiencies contribute to their dispersion. The findings highlight the need for innovative waste treatment technologies and a more comprehensive approach to risk assessment, particularly within life cycle assessments (LCA). Overall, the review underscores the urgency of integrating particle-specific metrics into monitoring frameworks and LCA methodologies to better protect environmental and human health.

1. INTRODUCTION

Over the past two centuries, humanity's relationship with waste has undergone a radical transformation. With the Industrial Revolution and the urban expansion of the 19th century, waste production began to grow far beyond the capacity of communities to dispose of it sustainably. In the 20th century, the introduction of plastics and synthetic materials further worsened the situation, marking the shift from predominantly organic and biodegradable waste to durable and persistent waste (Alanalp et al., 2026). Globally, waste management has become one of the most urgent environmental challenges, particularly in rapidly developing countries where infrastructure struggles to keep pace with demographic and industrial growth. Despite regulatory efforts introduced in many nations since the 1970s, the amount of waste continues to increase, along with the associated risks to the environment and human health (Ye et al., 2014).

The improper waste management, recycling, and discharge of wastes in the environment contribute to the environmental health impacts at the water, soil, and air levels,

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but with evidence of human health negative effects (Raphela et al., 2024).

Just in Japan, the World Bank has forecasted a rise of 1.3 - 2.2 billion tonnes of waste generated yearly by 2025, highlighting that 50% of waste will be collected, and inefficient management service will leave the area polluted, while the rest of the uncollected waste will be discarded (UN Environment, 2017).

Globally, the problem concerns not only the quantity but also the nature of waste. In recent years, the issue of invisible waste, such as microplastics, has emerged strongly. These originate from a variety of sources, including synthetic clothing, cosmetics, and the degradation of plastic materials. The lack of sufficiently advanced and effective water and wastewater treatment technologies has facilitated the spread of these micro- and nanoparticles in the environment (Cristaldi et al.,2020).

As by-products of both improper waste management and the current absence of efficient water treatment technologies, the large releases in environment of microfibers and microparticles are recognized as a physical risk and concern. Between the last, micro- and nanoplastic particles and nanometallic particles represent the last emergent pollutants, discovered in all environments and already affecting the food chain with several concerns for human health (Garrido Gamarro & Costanzo, 2022).

Considering this evidence, it becomes urgent to adopt more sustainable and integrated approaches to waste management, which should include not only efficient collection and recycling, but also source prevention, technological innovation, and greater social awareness of the issue (Figure 1).

2. METHODOLOGY OF REVIEW

The methodology of this review involved a narrativeand structured search of the scientific literature to capture the most recent and relevant studies related to microplastics, metallic nanoparticles, wastewater contamination, toxicity, and circular economy approaches. We searched multiple authoritative databases, including PubMed, Scopus, and Web of Science, covering publications from January 2010 to January 2026 to include contemporary research and emerging trends. The search strategy combined keywords such as "Waste management", "emergent pollutants", "Microplastics", "Metallic Nanoparticles", "Risk assessment", "Health impacts", and "Prevention", using Boolean operators to refine the results.

We focused on original articles published in English that addressed the environmental occurrence, fate, transport, and toxicological effects of micro- and nanoparticles, as well as studies describing advances in wastewater treatment technologies and waste management practices relevant to these contaminants. Additionally, papers discussing circular economy strategies and policies for mitigating waste and emerging pollutants were included. The review encompassed a broad range of study types, including observational and experimental research, in vitro and in vivo toxicity assessments, modeling studies, and comprehensive review articles.

To ensure the quality and relevance of the included literature, we excluded non-peer-reviewed publications such as conference abstracts, theses, and non-scientific reports, as well as studies that focused exclusively on macroplastics without addressing micro- or nanoparticles. Publications outside the 2010-2024 timeframe were generally excluded, except for seminal or highly cited works considered essential for contextual understanding.

3. MICROPLASTICS RELEASED BY IMPROPER WASTE MANAGEMENT OR RECYCLING

Microplastics released due to improper waste management or recycling represent one of the main contemporary environmental issues.

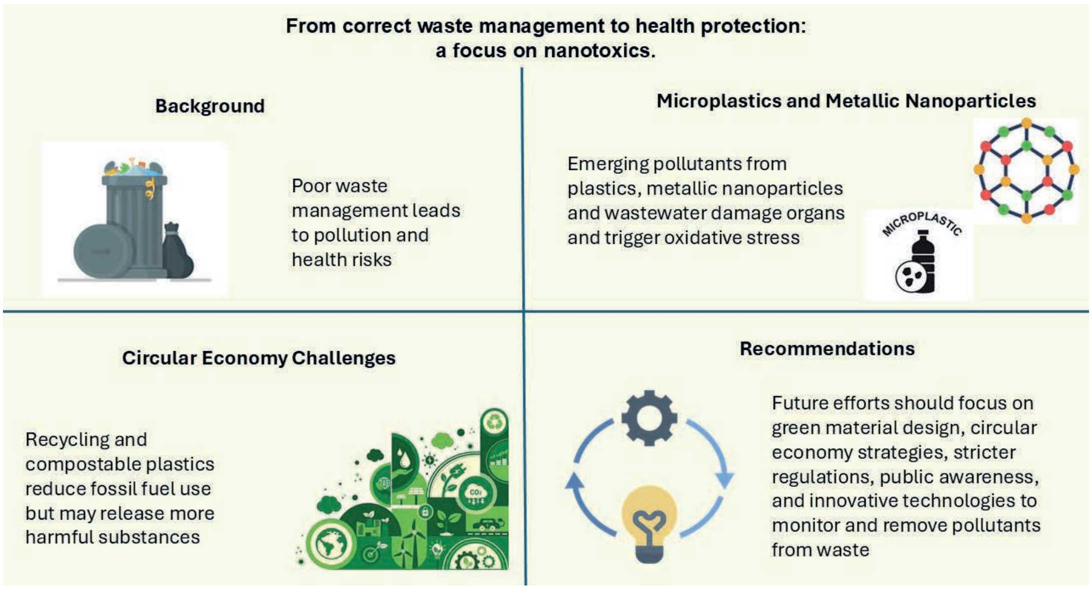

FIGURE 1: Emerging pollutants and health risks in circular waste management.

Plastics are acknowledged as emergent persistent pollutants that are capable (non-biodegradable) of causing contamination for extended periods in all environmental matrices. The plastic polymers produce small fragments or particles through crushing, splitting, and degrading during use. Microplastics refer to plastic fragments and particles with a diameter of less than 5 mm. They are called nanoplastics when the diameter is less than 1 μm (Oliveri Conti et al., 2024). Microplastics (primary microplastics) are also voluntarily produced and added to personal care products such as facial cleansers and cosmetics, hence contributing to wastewater in output effluents rich in microplastics.

According to the United Nations Environment Programme (UNEP, 2024), an estimated 19-23 million tonnes of plastic waste are dumped into aquatic ecosystems each year, contaminating lakes, rivers, and seas. In recent decades, numerous studies have documented a progressive increase in the presence of microplastics in the environment. For instance, Lebreton et al. (2017) highlighted how rivers are releasing increasing amounts of plastic into the oceans, reflecting a rise in global plastic production and ineffective waste management (Lebreton et al., 2017). In marine contexts, research by Cózar et al. (2014) in the Pacific Ocean documented a consistent increase in microplastic concentrations, demonstrating that plastic dispersion in the oceans is a continuously growing phenomenon (Cózar et al., 2014). Even in more sensitive environments, such as the Persian Gulf, increasing microplastic concentrations have been observed, as shown by Kor K & Mehdinia A., 2019, confirming the global spread of this issue (Kor K & Mehdinia A., 2019). A review by Eerkes-Medrano et al. (2019) highlighted that freshwater systems are also experiencing a significant increase in microplastic presence (Eerkes-Medrano et al., 2019), and a recent study further supports this alarming trend, documenting the widespread presence of microplastics across various freshwater compartments, including water columns, sediments, and aquatic biota (Ashamo et al., 2026), indicating that plastic pollution is pervasive across both marine and freshwater ecosystems. Similarly, studies on atmospheric microplastic pollution have shown a significant rise in the deposition of synthetic fibers in the air, suggesting that microplastic contamination affects not only aquatic but also terrestrial environments (Dris et al., 2016). Taken together, these findings illustrate a multi-compartment dispersion of microplastics across water, air, and sediments.

This dispersion not only alters natural habitats but also undermines the ability of ecosystems to adapt to climate change, directly and indirectly contributing to global warming and, consequently, affecting the health of millions of people (Oliveri Conti et al., 2024).

The exposure to microplastics induces a variety of toxic effects, including oxidative stress, metabolic disorder, immune response, neurotoxicity, as well as reproductive and developmental toxicity (Pulvirenti et al., 2022).

Several studies showed the presence of microplastics and nanoplastics in the brain, kidney, lung, liver, blood, heart, etc. The exposure is due to inhalation, dermal absorption, and ingestion. Also, due to the surface energy caused by their small size, microplastics may act as the vector adsorbing other environmental pollutants, especially heavy metals and hydrophobic organic chemicals (HOCs), which may enhance their toxicity (Xiang et al, 2022).

According to the last studies, several foods are already contaminatedby environmental microplastics, including marine and terrestrial food sources, as well as drinking waters (Garrido Gamarro & Costanzo, 2022; Ferrante et al., 2022; Oliveri Conti et al., 2020).

In this context, the first results from the Italian PRIN-PNRR project AQUApLASt (P2022XE857 – Tap wAter QUality and micropIAstics. Countering an ecoLogical threAt with an evidence-baSed approach) assessed microplastic contamination in drinking waters of Catania and Sassari, representing two major urban centers of southern Italy's largest islands. The study revealed microplastics (<10 μm) in tap water at a mean concentration of 4.85×10^6 particles/L in Catania (95% CI: 2.04×10^7 - 8.44×10^7; mean diameter: 3.2 μm), and 6.54×10^6 particles/L in Sardinia (95% CI: 8.16×10^5 - 6.54×10^6; mean diameter: 1.8 μm). Although these levels are lower than those reported in bottled mineral water (3.16×10^{10} particles/L; Zuccarello et al., 2018), the findings raise concerns regarding chronic exposure from tap water sources, especially when superficial waters are used, despite undergoing more advanced treatments than deep-water sources. Notably, particles <2 μm are considered the most hazardous due to their heightened toxicological potential (Oliveri Conti et al., 2024).

The recycling of plastic waste is not the cure. In fact, the toxicity of methanolic extracts obtained from compostable plastics (BPs) and conventional plastics (both virgin and recycled) showed an elevated toxicity of recycled plastics, compostable plastics, and semi-degraded compostable plastics resulting from partial disintegration, as compared to virgin conventional plastic extracts (Wang et al., 2023). The potential hazards that arise from the recycling processes of plastics derive from previous use and the recycling process itself, and from the release of the highest number of microplastics.

Despite the current and innovative technologies to recycle plastic waste, non-recoverable nanoplastics and microplastics cannot easily catch up with existing reclaim or treatment technologies due to their exceptionally small size. Further, the size reduction and washing during mechanical recycling facilities cause the release of a significant quantity of microplastics into the environment, reaching water or air as microplastics from recycling facilities. A study on PET recycling facilities reveals microplastic releases range from approximately 23-1836 mg/L in wastewater, distributed in the effluent (8-83 mg/L) and the sludge (52,166-68,866 mg/L) of the facility (Singh & Walker, 2024).

4. METALLIC NANOPARTICLES RELEASED BY IMPROPER WASTE MANAGEMENT

Metallic nanoparticles are nanomaterials constituted only by one element, such as Au, Ag, Pt, Cu, Ti, Pd, Re, Zn, Ru, Co, Cd, Al, Ni, and Fe. Metallic nanoparticles have remarkable properties, including surface plasmon resonance, high reactivity, and broad electromagnetic spectrum absorption. Currently, metallic nanoparticles are applied in several areas, including electronics, textiles, industry, healthcare, biomedicine, environment, agriculture, and food. Metallic nanoparticles, also, are also used for several applications due to their catalytic, antimicrobial, and finally, cancer and viral therapy characteristics.

Given their widespread use and unique physicochemical properties, their release into the environment has become increasingly relevant.

Increasing amounts of engineered nanoparticles in wastewater can reach the aquatic environment by passing through sewage treatment plants.

So, metallic nanoparticles were components of sanitary and urban wastewaters that contribute through wastewater facilities' output effluents and sewage sludge to discharge these in the environment, reaching the human organs through diet and dermal absorption (Shahalaei et al., 2024; Cervantes-Avilés & Keller, 2021).

These pathways highlight how both domestic and industrial activities contribute to the continuous introduction of metallic nanoparticles into natural systems.

Unlike traditional dissolved heavy metals, engineered metallic nanoparticles exhibit greater mobility and enhanced ability to penetrate cellular membranes due to their nanoscale size and unique surface properties (McGinnity TL et al., 2021). This increased bioavailability allows them to interact more readily with biological tissues, influencing their environmental and biological behavior.

Moreover, their persistence in the environment is often greater than that of conventional pollutants, as they can resist degradation and accumulate in sediments and organisms (Zhao et al., 2021).

Fe or Zn particles holds good potential to act as reducing agents relative to many redox-labile contaminants, and these has been successfully applied in the removal of a large range of wastewater and drinking water contaminants, including halogenated organic compounds, organic dyes, phenols, heavy metals, inorganic anions such as phosphates and nitrates, metalloids, and radio elements (Lu et al.2015).

Their reactivity, while beneficial in engineered applications, also influences their environmental behavior and interactions with pollutants.

In comparison to microplastics, which primarily act as physical carriers of pollutants and can cause mechanical harm to aquatic life, metallic nanoparticles pose chemical toxicity risks through the release of reactive metal ions and the generation of oxidative stress within cells. Pesticides, while biologically active, usually degrade over time or can be broken down by microbial action; metallic nanoparticles, however, can remain stable and continue to exert toxic effects long after their initial release.

Metallic nanoparticles challenge conventional wastewater treatment processes. Unlike dissolved metals, which can be precipitated or adsorbed by standard treatments, nanoparticles often pass through filtration and sedimentation systems due to their small size and colloidal behavior. This unpredictability complicates risk assessment and calls for the development of advanced treatment technologies specifically targeting nanoparticle removal (Azeez et al., 2023).

As a result, significant fractions of these particles can bypass treatment barriers and enter natural waters.

Metallic nanoparticles can induce ecotoxicological effects due to their specific chemical properties, reaching through the water and food chain to the human body, exerting several health effects.

Therefore, metallic nanoparticles can be translocated through the circulatory, lymphatic, and nervous systems to many tissues and organs, including the brain and placenta. Toxic effects encompass tissue inflammation and alter cellular redox balance toward oxidation, leading to abnormal function, DNA mutation, or cell death (Medici et al., 2021).

These biological interactions underscore the potential for both acute and chronic health impacts.

Among the emerging technologies for the removal of metallic nanoparticles from wastewater, nanoporous materials and selective membranes are showing promising results. These advanced filtration systems are designed to capture particles at the nanometer scale through size exclusion and specific surface interactions. Functionalized bioadsorbents, engineered to bind selectively to metal nanoparticles, offer an eco-friendly alternative by enhancing the efficiency of contaminant removal without generating secondary pollution (Nguyen et al., 2023) (Figure 2).

Moreover, bioremediation approaches using selected microorganisms are gaining attention as sustainable solutions. Certain bacteria and fungi have demonstrated the ability to adsorb, transform, or even degrade metallic nanoparticles, reducing their toxicity and mobility. These biological treatments could be integrated with conventional methods to improve overall removal rates and mitigate environmental release, paving the way for greener and more cost-effective wastewater management strategies (Alavi et al., 2022).

Together, these technologies provide additional approaches that may help address some of the limitations of conventional treatment systems.

5. LIFE CYCLE ASSESSMENT LIMITATIONS IN EVALUATING PARTICULATE POLLUTANTS FROM WASTE MANAGEMENT

Life Cycle Assessment (LCA) is a widely adopted methodology for evaluating the environmental impacts associated with products and processes across their entire life cycle, from raw material extraction to end-of-life management. Although LCA is a powerful decision-support tool, its application to contaminants such as microplastics and metallic nanoparticles remains limited by substantial methodological gaps (Nizam et al., 2021). Current LCA frameworks are primarily designed to quantify mass flows, emissions, and resource use, but they struggle to incorporate particulate pollutants whose release is diffuse, non-linear, and strongly dependent on mechanical, chemical, and biological transformations occurring along the waste management chain (Salieri et al., 2018). Moreover, the absence of standardized characterization factors for particle-specific impacts, such as persistence, bioaccumulation, and ecotoxicity, prevents a comprehensive assessment of their environmental burden. A notable limitation in several

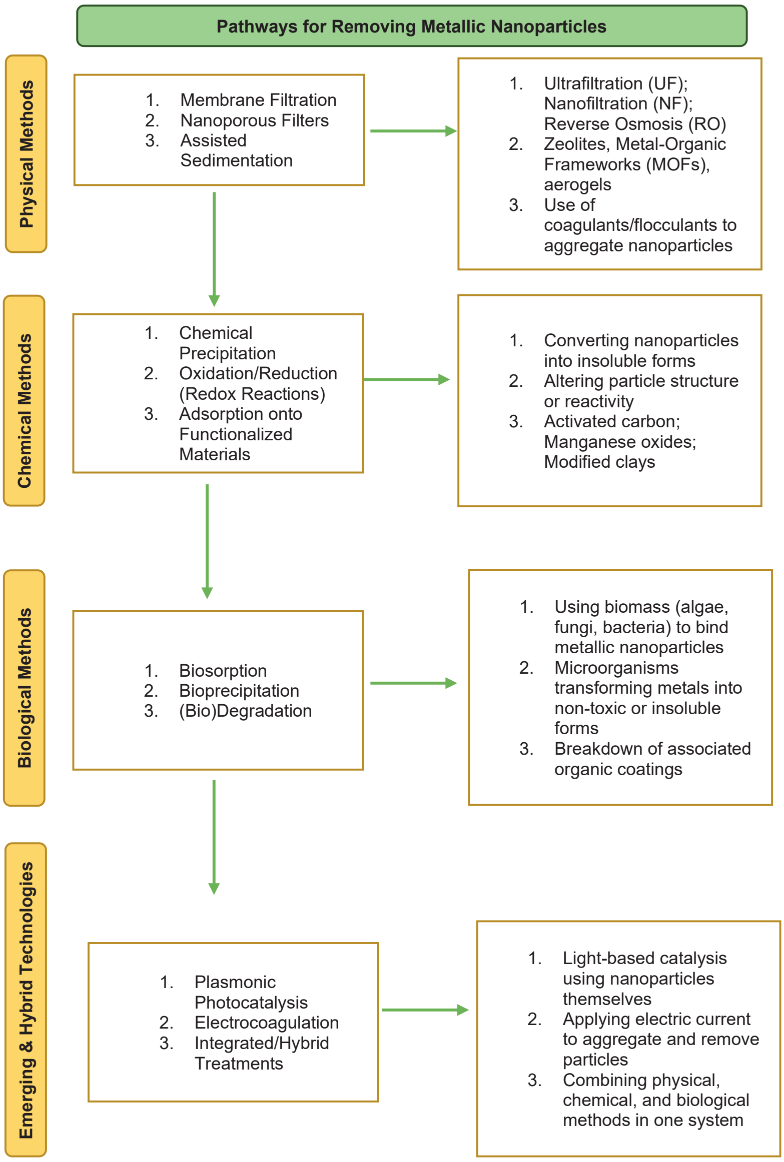

FIGURE 2: Pathways for removing metallic nanoparticles.

standard LCA methodologies lies in omitting the long-term fate of chemicals and particulates released during waste treatment and recycling processes (Singh et al., 2024).

In the case of microplastics, most LCAs of plastic recycling processes do not account for the generation of secondary micro- and nanoplastics during mechanical treatments such as shredding, washing, and extrusion (Rafiq et al., 2023). Plastic recycling shows both positive and negative aspects: while it significantly reduces fossil fuel utilization, power consumption, and landfilling, the size reduction and washing steps in mechanical recycling facilities cause the release of substantial quantities of microplastics into wastewater and air. As a result, the environmental benefits attributed to recycling may be overestimated, since the release of microplastics into wastewater streams or air emissions is rarely quantified or included in impact categories (Gandhi et al., 2021). The omission of short-term LCA studies in disregarding the consequences of chemical and particulate releases raises concerns about the overall efficacy of plastics recycling as a solution to plastic pollution (Table 1).

For metallic nanoparticles, the main limitation lies in the lack of LCA studies addressing their accumulation and fate in wastewater sludge destined for agricultural reuse or energy recovery. Despite the increasing evidence of nanoparticle accumulation in sludge, the absence of short-term LCA studies evaluating their concentrations in wastewater and biosolids raises concerns about the overall efficacy of sludge recycling as a sustainable waste management strategy. Unlike plastic recycling, where the concern is the mechanical fragmentation of polymers, the critical issue for metallic nanoparticles is their persistence in biosolids and their potential mobilization during sludge application to soils (Kim et l., 2010).

Overall, the current LCA framework is not yet equipped to fully capture the environmental implications of particulate contaminants generated or mobilized during waste management processes (Bisinella et al., 2024). Developing

TABLE 1: Plastic recycling, alternatives, and the circular economy each offer unique benefits and challenges in reducing plastic pollution.

	Main Features	Advantages	Disadvantages
Plastic Recycling	Processing used plastic into new materials.	• Reduces waste and pollution • Saves energy and raw materials	• Not all plastics are recyclable • Costly and energy-intensive - Downcycling issues
Plastic Alternatives (Biodegradable, Compostable, Bio-based, Reusable)	Use of biodegradable, compostable, or plant-based plastics.	• Break down faster in specific conditions • Useful for short-term use (bags, cutlery)	• Often requires industrial composting • May compete with food production • Not marine-safe
Circular Economy	A system focused on reuse, repair, recycling, and sustainable product design.	• Reduces overall waste • Encourages innovation and durability • Less resource extraction	• Requires systemic change • Initial implementation can be complex and costly

particle-specific inventory data, fate models, and characterization factors is essential to avoid underestimating the impacts of recycling strategies and to support more informed decisions in circular economy policies.

6. CONCLUSIONS

The widespread use of plastic and metallic nanomaterials has led to an increase in environmental exposure affecting human health. The alarming consequences of this rapid rise have begun to emerge.

Nanomaterials, often not classified as true pollutants, can be transported into the environment through water, accumulate in soil, and eventually enter the food chain. It is well known that nanotoxics generate oxidative stress in cells as a response to the reactive oxygen species and induce toxicity through inflammation.

Considering this, urgent measures must be taken to reduce the use and release of both microplastics and metallic nanoparticles into the environment. The emergence of new nano-pollutants poses a growing and tangible threat to public health and ecosystem stability. Although the circular economy offers a promising framework for sustainable resource management, it is currently insufficient to effectively prevent contamination by nanoscale pollutants. Due to their persistence and often invisible nature, these particles require targeted technological solutions that can safely and definitively remove them (Selim et al., 2025). Advanced technologies such as pyrolysis for plastics and specialized filtration systems for nanoparticles must be developed and implemented on an industrial scale. Moreover, LCA studies should systematically account for the environmental impact caused by pollutant release during waste recycling processes, in order to provide a more realistic and transparent evaluation of environmental balances.

Ultimately, addressing the issue of nano-pollutants demands an integrated approach that combines technological innovation, regulatory updates, social responsibility, and public awareness. Only through collective effort can we harness the benefits of nanotechnology while truly safeguarding human and environmental health.

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REFERENCES

Alanalp MB, Ergin MF, Durmus A. Management of plastic wastes: history, current applications, and future perspectives of recycling, upcycling, and reclaiming technologies. In: Sustainability for Integrated Waste Management. Cham: Springer; 2026. p. 69–97. doi:10.1007/978-3-032-03718-3_5.

Alavi M. Bacteria and fungi as major bio-sources to fabricate silver nanoparticles with antibacterial activities. Expert Rev Anti Infect Ther. 2022 Jun;20(6):897-906. doi: 10.1080/14787210.2022.2045194. Epub 2022 Feb 22. PMID: 35188050.

Ashamo MO, Adu BW, Adeyemi JA. Occurrence and distribution of microplastics in water, sediment, and aquatic insects of the Owena River, Osun State, Nigeria. Environ Monit Assess. 2026;198:138. doi:10.1007/s10661-026-14985-z.

Azeez L, Lateef A, Olabode O. An overview of biogenic metallic nanoparticles for water treatment and purification: the state of the art. Water Sci Technol. 2023 Aug;88(4):851-873. doi: 10.2166/wst.2023.255. PMID: 37651325.

Bisinella V, Schmidt S, Varling AS, Laner D, Christensen TH. Waste LCA and the future. Waste Manag. 2024;174:53-75. doi:10.1016/j.wasman.2023.11.021.

Cervantes-Avilés, P., Keller, A.A. Incidence of metal-based nanoparticles in the conventional wastewater treatment process. Water Res. 2021 Feb 1;189:116603. doi: 10.1016/j.watres.2020.116603

Cózar A, Echevarría F, González-Gordillo JI, Irigoien X, Ubeda B, Hernández-León S, Palma AT, Navarro S, García-de-Lomas J, Ruiz A, Fernández-de-Puelles ML, Duarte CM. Plastic debris in the open ocean. Proc Natl Acad Sci U S A. 2014 Jul 15;111(28):10239-44. doi: 10.1073/pnas.1314705111. Epub 2014 Jun 30. PMID: 24982135; PMCID: PMC4104848.

Cristaldi A, Fiore M, Zuccarello P, Oliveri Conti G, Grasso A, Nicolosi I, Copat C, Ferrante M. Efficiency of Wastewater Treatment Plants (WWTPs) for Microplastic Removal: A Systematic Review. Int J Environ Res Public Health. 2020 Oct 30;17(21):8014. doi: 10.3390/ijerph17218014. PMID: 33143273; PMCID: PMC7663475.

Dris R, Gasperi J, Saad M, Mirande C, Tassin B. Synthetic fibers in atmospheric fallout: A source of microplastics in the environment? Mar Pollut Bull. 2016 Mar 15;104(1-2):290-3. doi: 10.1016/j.marpolbul.2016.01.006. Epub 2016 Jan 17. PMID: 26787549.

Eerkes-Medrano D, Thompson RC, Aldridge DC. Microplastics in freshwater systems: a review of the emerging threats, identification of knowledge gaps and prioritisation of research needs. Water Res. 2015 May 15;75:63-82. doi: 10.1016/j.watres.2015.02.012. Epub 2015 Feb 17. PMID: 25746963.

Fazzo, L., Minichilli, F., Santoro, M. et al. Hazardous waste and health impact: a systematic review of the scientific literature. Environ Health 16, 107 (2017). https://doi.org/10.1186/s12940-017-0311-8

Ferrante M, Pietro Z, Allegui C, Maria F, Antonio C, Pulvirenti E, Favara C, Chiara C, Grasso A, Omayma M, Gea OC, Banni M. Microplastics in fillets of Mediterranean seafood. A risk assessment study. Environ Res. 2022 Mar;204(Pt C):112247.

Gandhi N, Farfaras N, Wang NHL, Chen WT. Life cycle assessment of recycling high-density polyethylene plastic waste. J Renew Mater. 2021;9(8):1463-1483. doi:10.32604/jrm.2021.015529

Garrido Gamarro, E. & Costanzo, V. 2022. Microplastics in food commodities – A food safety review on human exposure through dietary sources. Food Safety and Quality Series No. 18. Rome, FAO. https://doi.org/10.4060/ cc2392en

Kim B, Park CS, Murayama M, Hochella MF. Discovery and characterization of silver sulfide nanoparticles in final sewage sludge products. Environ Sci Technol. 2010 Oct 1;44(19):7509-14. doi: 10.1021/es101565j. PMID: 20839838

Kor K, Mehdinia A. Neustonic microplastic pollution in the Persian Gulf. Mar Pollut Bull. 2020 Jan;150:110665. doi: 10.1016/j.marpolbul.2019.110665. Epub 2019 Oct 23. PMID: 31655300.

Lebreton LCM, van der Zwet J, Damsteeg JW, Slat B, Andrady A, Reisser J. River plastic emissions to the world's oceans. Nat Commun. 2017 Jun 7;8:15611. doi: 10.1038/ncomms15611. PMID: 28589961; PMCID: PMC5467230.

Lu, H., Wang, J., Stoller, M., Wang, T., Bao, Y., & Hao, H. 2015. An Overview of Nanomaterials for Water and Wastewater Treatment. Advances in Materials Science and Engineering, 2016(1), 4964828. https://doi.org/10.1155/2016/4964828

McGinnity TL, Sokolova V, Prymak O, Nallathamby PD, Epple M, Roeder RK. Colloidal stability, cytotoxicity, and cellular uptake of HfO_2 nanoparticles. J Biomed Mater Res B Appl Biomater. 2021 Oct;109(10):1407-1417. doi: 10.1002/jbm.b.34800. Epub 2021 Jan 20. PMID: 33474824.

Medici, S., Peana, M., Pelucelli, A., Zoroddu, M.A. 2021. An updated overview on metal nanoparticles toxicity. Semin Cancer Biol. 2021 Nov;76:17-26. doi: 10.1016/j.semcancer.2021.06.020.

Nguyen NTT, Nguyen LM, Nguyen TTT, Tran UPN, Nguyen DTC, Tran TV. A critical review on the bio-mediated green synthesis and multiple applications of magnesium oxide nanoparticles. Chemosphere. 2023 Jan;312(Pt 1):137301. doi: 10.1016/j.chemosphere.2022.137301. Epub 2022 Nov 18. PMID: 36410506.

Nizam NUM, Hanafiah MM, Woon KS. A content review of life cycle assessment of nanomaterials: current practices, challenges, and future prospects. Nanomaterials (Basel). 2021;11(12):3324. doi:10.3390/nano11123324

Oliveri Conti, G., Ferrante, M., Banni, M., Favara, C., Nicolosi, I., Cristaldi, A., Fiore, M., Zuccarello, P. Micro- and nano-plastics in edible fruit and vegetables. The first diet risks assessment for the general population. Environ Res. 2020 Aug;187:109677.

Oliveri Conti, G., Rapisarda, P., Ferrante, M. 2024. Relationship between climate change and environmental microplastics: a one health vision for the platysphere health. One Health Advances, 2:17 https://doi.org/10.1186/s44280-024-00049-9

Pulvirenti, E., Ferrante, M., Barbera, N., Favara, C., Aquilia, E., Palella, M., Cristaldi, A., Conti, G.O., Fiore, M. 2022. Effects of Nano and Microplastics on the Inflammatory Process: In Vitro and In Vivo Studies Systematic Review. Front Biosci (Landmark Ed). 27(10):287. doi: 10.31083/j.fbl2710287.

Rafiq A, Xu JL. Microplastics in waste management systems: a review of analytical methods, challenges and prospects. Waste Manag. 2023;171:54-70. doi:10.1016/j.wasman.2023.08.015

Raphela, T., Manqele, N., Erasmus, M. 2024. The impact of improper waste disposal on human health and the environment: a case of Umgungundlovu District in KwaZulu Natal Province, South Africa. Front. Sustain., 5:1386047. doi: 10.3389/frsus.2024.1386047

Salieri B, Turner DA, Nowack B, Hischier R. Life cycle assessment of manufactured nanomaterials: where are we? NanoImpact. 2018;10:108-120. doi:10.1016/j.impact.2017.12.003.

Selim MM, Tounsi A, Gomaa H, Shenashen MA. Recent advancement in removal of microplastics as emerging pollutants from aquatic ecosystems: adsorption and separation. Environ Geochem Health. 2025 Oct 16;47(11):503. doi: 10.1007/s10653-025-02790-5. PMID: 41099928.

Shahalaei, M., Azad, A.K., Sulaiman, W.M.A.W., Derakhshani, A., Mofakham, E.B., Mallandrich, M., Kumarasamy, V, Subramaniyan, V. 2024. A review of metallic nanoparticles: present issues and prospects focused on the preparation methods, characterization techniques, and their theranostic applications. Front. Chem. 12:1398979. doi: 10.3389/fchem.2024.1398979

Singh N, Walker TR. Plastic recycling: A panacea or environmental pollution problem. Npj Mater Sustain. 2024;2(1):17. doi: 10.1038/s44296-024-00024-w. Epub 2024 Aug 1. PMID: 39114578; PMCID: PMC11304528

Singh, N., Walker, T.R. 2024. Plastic recycling: A panacea or environmental pollution problem. npj Mater. Sustain. 2, 17. https://doi.org/10.1038/s44296-024-00024-w.

UN Environment. 2017. The Global Garbage Crisis: No Time to Waste. Available at: https://www.unep.org/news-and-stories/press-release/global-garbage-crisis-no-time-waste.

UNEP. 2024. Global Waste Management Outlook 2024. ISBN: 978-92-807-4129-2. Job number: DTI/2619/NA , DOI: https://doi.org/10.59117/20.500.11822/44939

Wang, T., Hosseinzadeh, M., Cuccagna, A., Alakenova, R., Casademunt, P., Reyes Rovatti, A., López-Rubio, A., Porte,C. 2023. Comparative toxicity of conventional versus compostable plastic consumer products: An in-vitro assessment, Journal of Hazardous Materials,459,132123. https://doi.org/10.1016/j.jhazmat.2023.132123

WHO, 2024. Health-care waste. https://www.who.int/news-room/fact-sheets/detail/health-care-waste

Xiang, Y. J.; Jiang, L.; Zhou, Y. Y.; Luo, Z. R.; Zhi, D.; Yang, J.; Lam, S. S. 2022. Microplastics and Environmental Pollutants: Key Interaction and Toxicology in Aquatic and Soil Environments. J. Hazard. Mater. , 422, 126843, DOI: 10.1016/j.jhazmat.2021.126843

Ye, L., Wu, A.M. 2014. Urbanization, land development, and land financing: evidence from Chinese cities. J Urban Aff 36(sup1):354–368. https://doi.org/10.1111/juaf.12105

Zhao J, Wang X, Hoang SA, Bolan NS, Kirkham MB, Liu J, Xia X, Li Y. Silver nanoparticles in aquatic sediments: Occurrence, chemical transformations, toxicity, and analytical methods. J Hazard Mater. 2021 Sep 15;418:126368. doi: 10.1016/j.jhazmat.2021.126368. Epub 2021 Jun 9. PMID: 34329024.

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THE ROLE OF BIOGENIC WASTE AND RESIDUES IN A CLIMATE NEUTRAL WORLD

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ABSTRACT
The paper discusses the role of biogenic waste and residues in a climate-neutral world, highlighting their potential for sustainable energy production, climate protection, and resource conservation. The authors conclude that a significant amount of biogenic waste and residues can be utilized to produce biofuels, biomethane, and other bio-based products, contributing substantially to greenhouse gas reduction and a circular economy (Carrillo-Rodríguez et al., 2025). In Germany, the use of biofuels from waste and residues already prevented emissions of 11.6 million tonnes of CO2-eq in 2022. The paper also emphasizes the importance of a National Biomass Strategy (NABIS) to steer biomass flows and ensure the optimal use of this valuable resource.

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1. INTRODUCTION

Against the backdrop of current wars and other violent conflicts, climate and environmental issues appear less important in the public eye.

Nevertheless, it is not a new insight that we are to change our utilization of raw materials into sustainable consumption on long term. Earth overshoot day in 2025 was July 24th, when globally more resources are consumed as therse could be regenerated.

The share of renewable energies in Germany's energy system is around 20% (Umweltbundesamt, 2024), and the circularity rate (CMU), i.e. the proportion of waste and residual materials generated in production processes, is only around 14% (Eurostat, 2024).

International comparison shows that Germany remains one of the leading countries in waste management and recycling, and makes a significant contribution to resource and climate protection. For example, waste management has been the most successful sector in reducing CO_2 emissions since 1990. In addition, other positive effects are not attributed to waste management, but to the energy industry or other sectors to which the waste management industry supplies secondary fuels and resources.

Over the past years, the German government has created a so-called 'intergenerational climate contract' (Bundes-Klimaschutzgesetz vom 12. Dezember 2019) with several amendments to the Climate Protection Act (Bundesregierung, 2022). This aims to reduce German greenhouse gas emissions by at least 65% by 2030 and by 88% by 2040, compared to 1990 levels. From 2045, the Federal Republic of Germany is to become climate-neutral, i.e. there should be a balance between greenhouse gas emissions and sequestration. After 2050, the German government aims to achieve negative emissions by naturally sequestering more greenhouse gases than are emitted. The new government remains committed to these goals, but has so far failed to publicly define and implement enough of the necessary measures (Figure 1).

What is required to achieve climate neutrality in Germany by 2045 is, to increase further optimised interaction of wind, solar and geothermal energy, bioenergy, and hydropower for electricity, heating/cooling and mobility.

In Parallel, a development from a 'linear' economic system into a true circular economy is required, where used materials are further recycled and at last, the energy remainders shall be recovered. And for the supply of raw ma-

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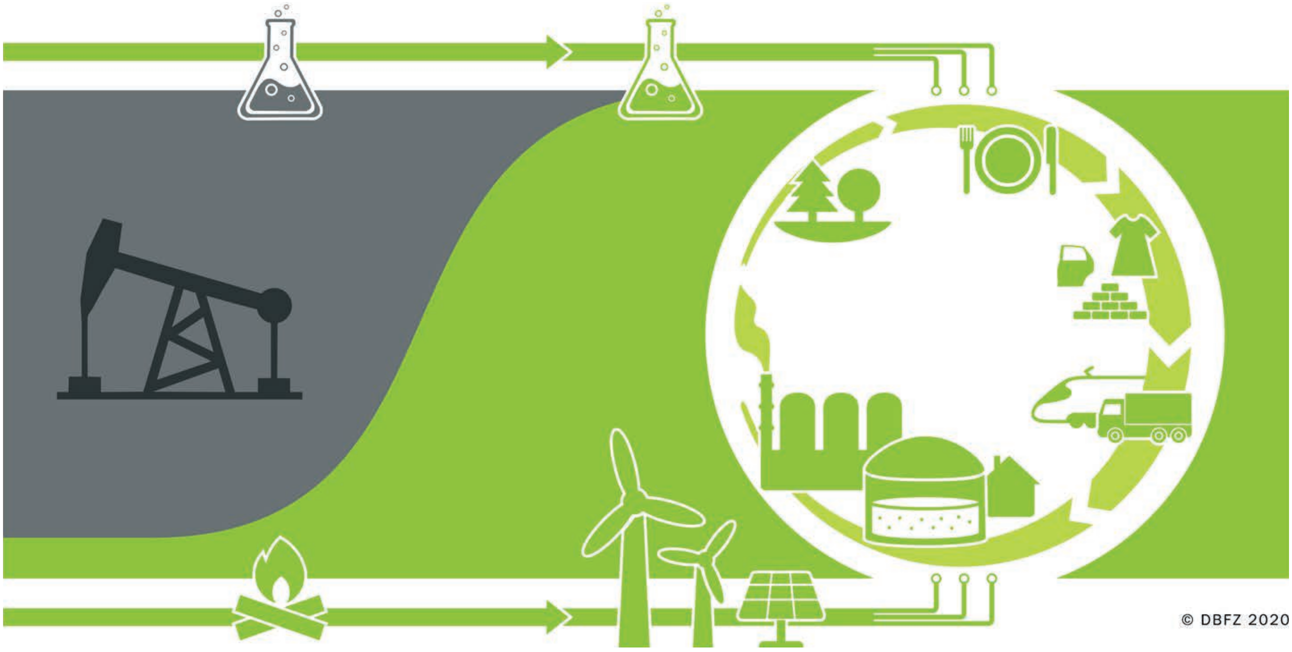

FIGURE 1: Highly simplified representation of future energy supply and (bio-based) circular economy as central building blocks for a climate-neutral society © DBFZ.

terials to the industry, petroleum-based raw materials shall be replaced step by step with biobased materials.

Biomass will therefore be the basis of the bioeconomy and an integral part of a sustainable energy system. This can only be achieved if biomass is produced sustainably and used efficiently, in an environmentally friendly manner and with the greatest possible economic benefit. A sustainable bioeconomy therefore optimises the material and energy recovery of biogenic waste and residues, as the current production technologies are optimized for the conversion of fossil raw materials. This requires new technological concepts and coupled and cascading use. But also 'negative' emissions that can be generated through the storage of 'green' carbon. Biomass is thus increasingly coming into focus as an important 'green' carbon carrier, on the one hand to reduce and bind CO_2 emissions, and on the other hand, as the carbon carrier for the future bioeconomy. This will significantly increase the demand for biomass and the need for sustainable cascade and coupled use in the future.

As an essential basis for the transformation process towards a sustainable, circular bio-based economy and society, the last German federal government undertook in its 2021 coalition agreement to develop a National Biomass Strategy (NABIS). The development of NABIS was led by the Federal Ministry for Economic Affairs and Climate Action (BMWK), Federal Ministry of Food and Agrculture (BMEL) and the Federal Ministry for the Environment, Nature Conservation, Nuclear Safety and Consumer Protection (BMUV), with the original goal of adoption by the Federal Cabinet by the end of 2023. Against the backdrop of climate and biodiversity protection and food security, NABIS aims to steer biomass flows in a targeted manner so that this valuable resource is used in the best possible way (Bundesumweltministerium, 2025). Such a long-term strategy is certainly necessary and helpful in order to make the best possible use of the limited biomass available in Germany in terms of materials and energy in the future.

However, it is clear that the transition to the desired circular bioeconomy can only succeed if the utilisation of biogenic waste and residual materials is given the priority. This applies to both material and energy recovery, which are also the main areas of focus of the cooperation between the DBFZ and the Chair of Waste and Resource Management at the University of Rostock. Against this background, the potential and current use of biogenic waste and residues will first be presented below. This will be followed by an explanation of the current status and necessary developments in the areas of "waste wood and residual wood", "biogas technology", "biofuels and biorefineries" and "negative emissions".

2. POTENTIAL AND USE OF BIOGENIC WASTE AND RESIDUES IN GERMANY

In order to take into account, the availability of biogenic resources as a decisive factor in assessing the opportunities and risks of existing and potential uses, the DBFZ develops raw material monitoring systems for various geographical regions, implements them and makes them available in standardised formats.

Currently, 77 different biogenic residues from numerous sectors are considered. All results, including documentation, are available for individual data analysis in an online database in the DE Biomassemonitor, which can be accessed at https://datalab.dbfz.de. The database includes by-products from agriculture and forestry, municipal waste, sewage sludge, industrial residues and residues from other areas. On this basis, the German theoretical potential (biophysical maximum) ranges from 185.2 to 232.2 million tonnes of dry matter (tDM) in 2020. The annual technical biomass potential, which is defined as the potential that includes futher limiting restrictions of technical, structural and legal nature (Brosowski et al., 2019), amounts to around 91.7 to 128.9 million tonnes of dry matter (tDM) in 2020 in Germany (Wilske et al., 2025) (Naegeli de Torres et al., 2024). Between 68 and 83% is already being used for material and/or energy purposes. The DBFZ assumes that, in addition to optimising existing use, a further 15.7 to 41.8 million tonnes DM can be mobilised for further applications. A material flow aggregation of the annual biogenic waste and residues is shown in Figure 2 for 2020 and the average values of the ranges described above. Regarding the overall average theoretical biomass potential of 208.7 million tonnes DM, the largest contribution sectors are agricultural by-products (77.6 million tonnes DM), forestry by-products (74.4 million tonnes DM), municipal waste and sewage sludge (34.1 million tonnes DM), followed by industrial residues (15.4 million tonnes DM) and residues from other areas (7.2 million tonnes DM). Including the further above outlined restrictions regarding the technical potential, the order of the largest sectors changes, municipal waste and sewage sludge now contributing 33.8 million tonnes DM, forestry by-products 30.2 million tonnes DM and agricultural by-products 27.2 million tonnes DM. Finally, industrial residues and residues from other areas provide 15.3 and 3.8 million tonnes DM respectively.

It should be noted that the quantities are given as dry matter (DM). This means that the quantities actually available for recycling are significantly higher. For example, in animal husbandry in Germany alone, an average of 17.6 million tonnes DM per year are produced in the form of cattle and pig solid manure, cattle and pig slurry, dry chicken manure, etc., whereby the wet mass (WM) is much higher (up to a factor of 7) depending on the DM content of the respective animal excrement. Low dry matter contents (e.g. 4-8% in pig slurry) pose particular challenges for the exploitation of still available feedstocks due to their low transportability (Krause et al., 2020).

Biogenic waste and residues comprise a very broad spectrum of different material flows, and resource management is already being implemented in a highly differentiated manner, especially outside waste legislation. The very large quantities of biogenic residues mentioned above, which are already collected separately from municipal waste generated from private households and businesses, are recycled in a variety of ways. For example, there are highly differentiated recycling processes for slaughterhouse waste, which end up in the pharmaceutical, feed and food industries and contribute to the substitution of so-called primary raw materials. Large quantities also arise in agriculture and fisheries and in their processing stages of the value chain. Production residues from the food industry are also reused in a similar way for animal feed production or as a basis for further production processes.

Especially if material flows cannot be used in applications with higher economic and environmental added value (e.g. material use), they can be used either for thermochemical, biological or mechanical conversion into biogas, solid or liquid biofuels, electricity or heat (cascading principle REDIII) (EU, 2023). In order to mobilise additional potentials for those applications, municipal biowaste from households, gardens and parks account with a technical potential of around 14 million tonnes of dry matter (average) for a significant but by no means dominant proportion of total biogenic waste and residues. They are of particular importance within municipal waste, both for sanitation and for meeting specific recycling and recovery targets.

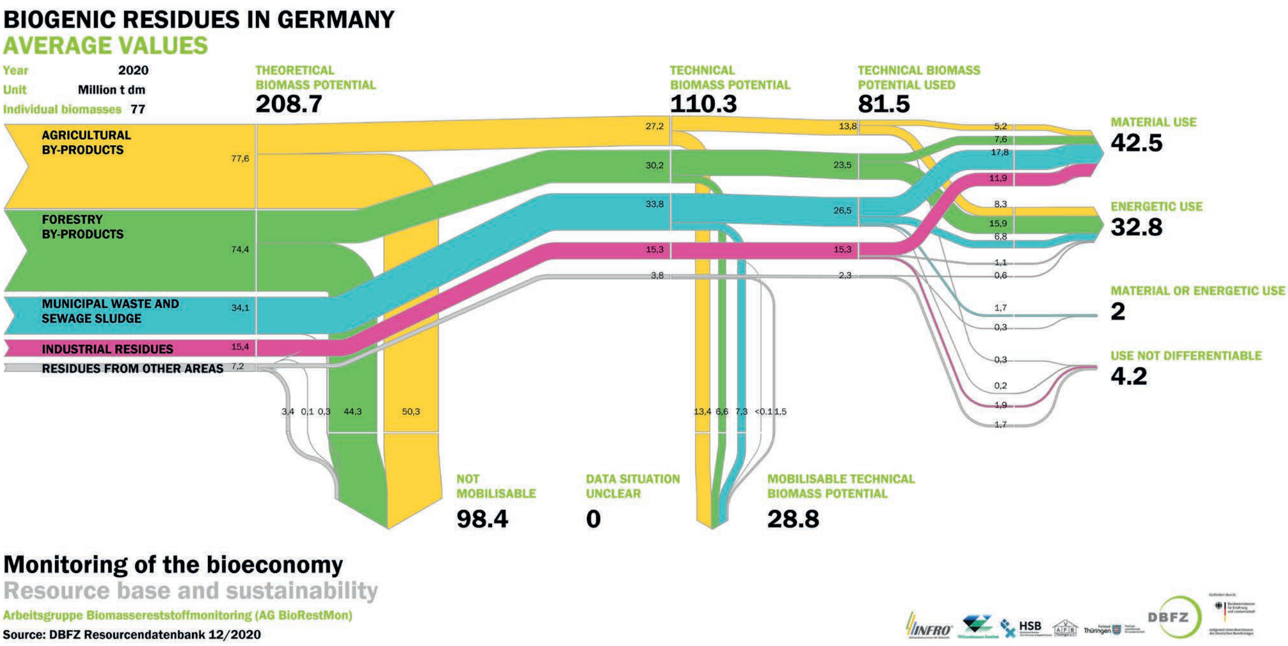

FIGURE 2: Aggregated material flow of biogenic residues, by-products and waste as average values in millions of tonnes of dry matter (Million tDM). Source: DBFZ resource database, https://datalab.dbfz.de. © (Wilske et al., 2025, p.18).

Other individual biogenic wastes and residues relevant in terms of quantity are shown in Figure 3. In 2020, the total technical biomass potential mainly consists of 15 types of biomasses, which together account for 93.7 million tonnes of dry matter (DM) or 85% of the total technical potential of biogenic waste and residues of 110.3 million tonnes DM. In terms of quantity, waste paper (14.5 million tonnes DM), green waste (11.2 million tonnes DM) and cereal straw (9.7 million tonnes DM) are particularly noteworthy.

However, in addition to animal excrement, wood-based residues and by-products are particularly well represented among the top 15 biomasses. The five biomasses from the wood and forestry by-products sector, i.e. sawmill by-products and wood shavings, waste wood, coniferous forest residues, deciduous forest residues and black liquor, account for a total of 27.9 million tonnes DM or 29% of the technical potential. However, it should be noted that relevant quantities of this potential are already in use. For example, 76% (average) of the five biomasses from the wood and forestry by-products sector are already being used for material or energy purposes. Additional mobilisable potential exists only for forest residues (coniferous and deciduous), as sawmill by-products and wood shavings, waste wood and black liquor are already fully utilised. However, greater use of forest residues should be examined, particularly in view of future sustainability requirements (e.g. EU biodiversity and sink targets (LULUCF)), which could limit the amount that can be extracted in the future, depending on the location. However, there is also considerable potential for sawmill by-products and wood shavings, waste wood and black liquor, as these could be used in alternative pathways. In particular, due to the expiry of EEG subsidies for waste wood, large quantities (~ 50% of the volume covered by EEG subsidies) could potentially be used for other material or energy purposes (Matschoss, 2020). Due to increasing cascade factors for wood and rising material use of wood, e.g. through bans on single-use plastics and by timber construction initiatives, it can be assumed that the volume of waste wood will continue to rise until 2030 (Mantau, 2023).

3. RECYCLING OF WASTE WOOD AND RESIDUAL WOOD

In practice, it is not always easy to distinguish between primary wood and residual and waste wood. While there is at least a formal legal classification for waste wood, the situation is much more complicated for residual wood. Ultimately, the practical application makes the distinction by wood fractions (logs) that are processed in a sawmill, whereby, depending on the processing method, 30 to 50% of bark, slabs and splinters, offcuts, sawdust and wood shavings are produced as production residues. In addition, further wood fractions are produced in the forest and during landscape maintenance as part of clearing, thinning, timber harvesting and the partial removal of calamity wood. These fractions cannot be used in the sawmill industry, but have so far been mainly used for energy purposes and assigned to the category of residual wood. Primary wood is mainly used for energy when sawable wood from the forest is processed into firewood for end customers – either privately or by commercial wood processors. In principle, however, all the quotas listed here in Germany are characterised by the fact that there has been no economically advantageous demand from material use to date. It should be noted that funding incentives (e.g. the Renewable Energy Sources Act or investment subsidies for wood-fired heating systems) can distort the economic situation between material and

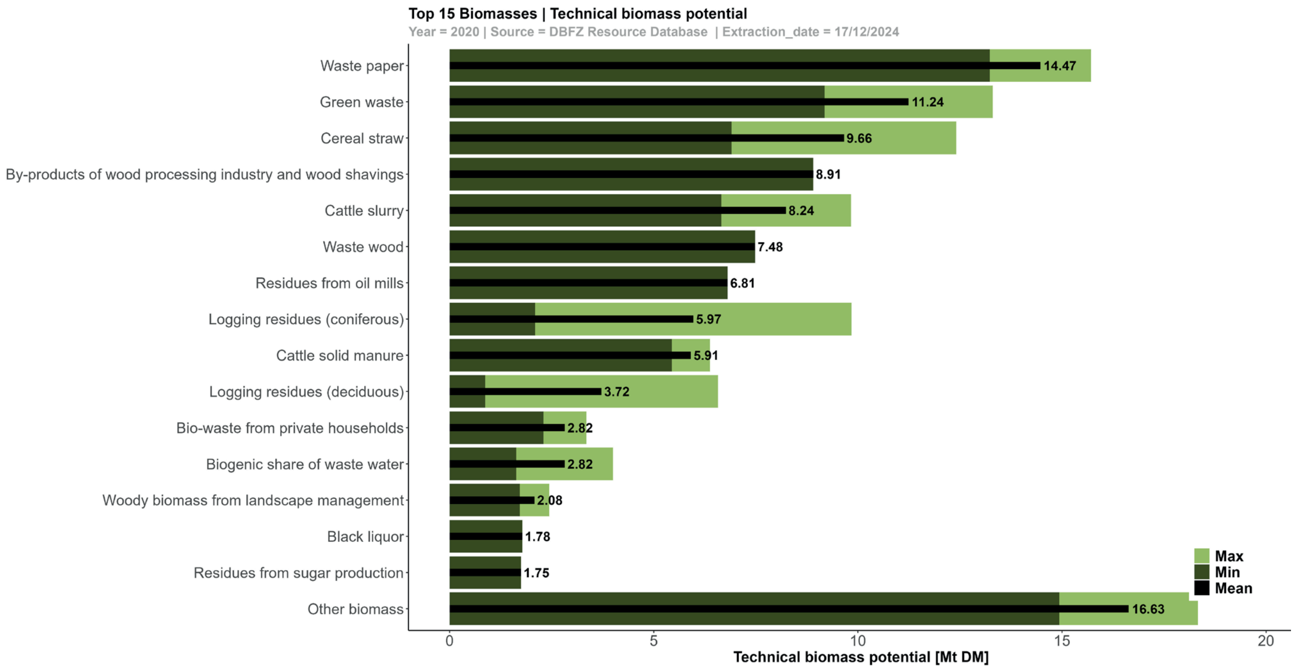

FIGURE 3: Top 15 biomasses with technical potential in 2020 according to minimum, maximum and average values in million tonnes of dry matter (Million tDM). Source: DBFZ resource database, https://datalab.dbfz.de © (Wilske et al., 2025, p.20).

energy use. Conversely, it has also been observed in the past that without financial incentives for energy use, prices in the timber sector fall to such an extent when material and energy use are linked that timber supply is reduced. Although this would lead to an increase in the amount of carbon stored in forests in the short term, it would increase the risk of climate-damaging total loss through forest fires with high soot emissions in the foreseeable future as a result of climate change and the resulting delay in forest conversion.

Until now, residual wood and waste wood have mainly been used for energy purposes to provide full or base load heating and, in case of waste wood, to generate electricity in waste wood power plants within the EEG. The combustion plants are more expensive than gas or oil-fired plants, both in terms of heat and electricity and in the CHP sector. It is therefore important to recoup the higher investment costs by ensuring that the plant is utilised as much as possible with cheaper wood fuels. Apart from wood pellets (over 90% of which is wood from wood processing residues in Germany), bark, wood chips from sawmill by-products or waste wood and black liquor are mainly used in larger plants (over 100 kWthermal, in some cases significantly over 1 MW). In addition, the use of waste wood and scrap wood for energy purposes has become established in industry, particularly where these fuels are produced during the manufacturing process, i.e. in sawmills and the wood-processing industry, including paper manufacturing. Due to the demand for electricity and heat for cooking and drying processes, temperatures required are usually below 200°C. In German speaking countries, wood pellets are mainly used in the end customer sector for space heating at a maximum temperature of 90°C. Overall, waste wood and scrap wood are mainly used for energy in the lower temperature range, with the result that the available capacities are only sufficient for a limited number of combustion plants. At the same time, the development of the legally required municipal heating plans shows that many heating networks rely on significant proportions of wood for defossilisation, which in total amounts to much too high wood consumption for the future in Germany.

In this respect, the use of residual and waste wood must be re-thought and transformed from a systemic perspective.

(1) Material cascades must be extended. Sawdust, wood shavings and wood chips are suitable raw materials for chemical base materials, as they are a very homogeneous biomass with significantly less than 1% impurities (mineral ash). Bark can be used as a proven raw material for peat substitutes (Fachagentur Nachwachsende Rohstoffe, 2022), as can wood chips from various tree species with and without treatment in biogas plants (Fachagentur Nachwachsende Rohstoffe, 2022). For these natural biomasses, processing into disposable food packaging and disposable tableware is also standard. In the area of waste wood, classes AI (wood without any additions despite very small ones) to AIII (waste wood with colours, ceilings and so on including halogen organic ingredients, but without wood protection poisons) are suitable for the production of building components which, depending on waste wood class (occupational health and safety), can be used either for free distribution to craftsmen or restricted to industrial use. Last but not least, there are approaches to further utilise black liquor for chemical raw materials instead of directly recycling it thermally (Fraunhofer-Gesellschaft, 2025). Figure 4 provides an overview of these options.

Besides these main utilization pathways there are also some innovative concepts for reusing wood waste e.g.for nanocellulose production to be used in advanced materials (Ghamari, 2025).

(2) Wherever energy use still appears advantageous for technical or economic reasons, care must be taken to generate maximum system benefits. This means that biomass should primarily be used in peak load applica-

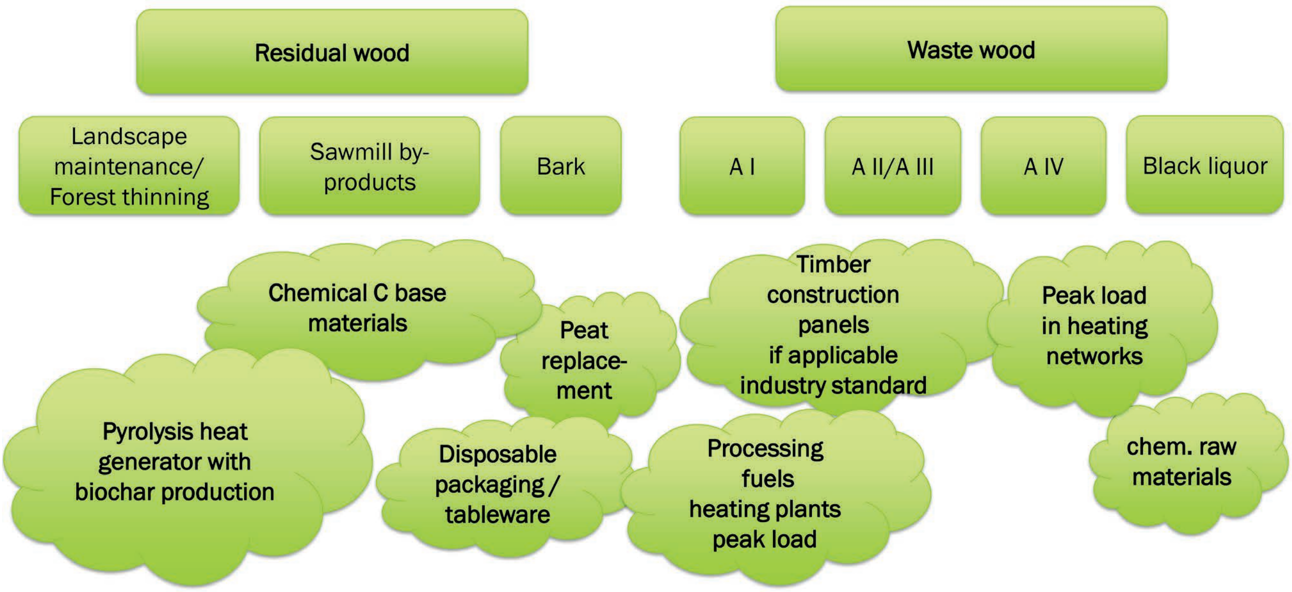

FIGURE 4: Development options for innovative utilisation concepts for residual and waste wood. © DBFZ.

tions in combination with other renewable heat generators (hybrids, especially with heat pumps) and in high-temperature applications for industrial process heat (Jordan et al., 2019). Hybrid concepts should be designed in such a way that the proportions of biomass and electricity (ambient heat) can be flexibly adjusted depending on the price situation and the availability of renewable electricity. In the medium term, options for negative emissions should be integrated. For example, materials from landscape management, which are often very heterogeneous and tend to occur locally in small quantities, could be used in small-scale pyrolysis plants for heat supply and biochar production as a future option (Thrän et al., 2025). In addition to covering any necessary peak heat loads, this would also produce stable biochar for material applications (e.g. soil improvers, additives for road construction, concrete, etc. (RWTH Aachen, 2024) (Vaupel, 2023) (Wollnik et al., 2023). The use of AIV waste wood (waste wood contaminated with wood protection poisons) in waste-fired heating plants remains interesting, although in future, the focus here should also be on the storability of the biomass and combustion to supply heating networks during winter. In addition, CO_2 capture (BeCCS) should also be planned for these rather large plants in the medium term. In this context, use in the cement industry is likely to be particularly promising, as this industry will not be able to avoid CO_2 capture due to processrelated CO_2 emissions. In addition, AI and, if necessary, AII waste wood (glued and/or painted wood without halogen organic ingredients), provided it cannot be used for other purposes, can be processed for peak load heating plants in local heating networks. Here, pyrolysis heating plants with simultaneous biochar production should also become more relevant in the medium term. Besides these established thermochemical conversion routes, research worked on process to produce hydrogen from wood residues by thermos-catalytic conversion (Jelschow, 2024).

In a glance, due to legislation in Germany all wood residues and wastes are already used, either for material purposes or for energy supply. Up to now, this is oriented very much on cost efficiency and not on climate efficiency. That means, for a full renewable material and energy world, limited wood potentials have to be redirected in a much more climate efficient way, like prolonged material cascades, energetic uses for peak loads and higher temperatures and combined with negative emission generation, perfectly.

4. BIOGAS TECHNOLOGY FOR THE UTILISATION OF BIOGENIC WASTE AND RESIDUES

Biogas technology plays an important role in the sustainable utilisation of biogenic waste and residues, particularly from agriculture and food production (IEA, 2020; Weinrich & Nelles, 2021). Anaerobic digestion involves the breakdown of organic materials by microorganisms in the absence of oxygen. The resulting biogas consists mainly of methane and carbon dioxide. This technology not only provides an environmentally friendly source of energy, but also helps to reduce greenhouse gas emissions associated with conventional storage and disposal, especially of agricultural residues (Fachagentur Nachwachsende Rohstoffe, 2022). To date, energy crops (such as silage maize) and farm manure (primarily slurry) have been the main inputs used in biogas plants (Rensberg et al., 2023). In 2023, the energy-related input of renewable raw materials was approximately 68% from cultivated biomas and 19% from farm manure. Municipal biowaste accounts for approximately 4% and other waste materials from industry and commerce for approximately 9% of energy-related consumption. From 2010 to 2023, the energy-related share of renewable raw materials has decreased slightly in favour of farm manure and industrial residues (Figure 5) (Rensberg et al., 2023).

The cultivation of energy crops is generally considered to compete with food production and to have negative environmental consequences (e.g., land-use competition and biodiversity trade-offs) (IEA, 2020; Theuerl et al., 2019). Using biogenic waste and residues as input materials reduces these negative effects, and is therefore increasingly supported by politics and public opinion alike (IEA, 2020). Residual materials produced in agriculture include manure, slurry, harvest residues and landscape conservation materials. However, landscape conservation materials can have waste characteristics, especially roadside vegetation, which is decisive for the legal status of a biogas plant and has an impact on requirements and permits (Bundesgütegemeinschaft Kompost e.V., 2014). Slurry and manure are particularly suitable substrates for biogas production because they are continuously available and their fermentation reduces emissions of methane, a potent greenhouse gas (Umweltbundesamt, 2019). Crop residues such as straw offer further potential but are often underutilised, mainly due to the high cost of harvesting and pre-treatment. Physical, chemical or biological pre-treatment processes can break down the structure of these materials, making them more accessible to microorganisms (Yu et al., 2019). This improves both the biogas yield and handling during the process by improving mixability and reducing layer formation in the fermenter (Yu et al., 2019). In the case of straw, there is also competition for material uses such as bedding in animal husbandry, meaning that, depending on the region, straw is a very cost-intensive substrate with an average purchase price of €120/t (top agrar., 2024). This example shows that biogenic waste and residues can be very expensive, which contradicts the often-held assumption that they are always cost-neutral. This needs to be examined on a case-bycase basis.

Waste from food production and the processing industry for agricultural raw materials often has a high energy content and is therefore well suited for biogas production (IEA, 2020). This sector generates significant amounts of biogenic waste and residues, including production waste or unsaleable products and calmity batches, such as grain contaminated with mycotoxins. However, residues that previously had other sales markets but are now disappearing, such as the use of residues from starch production for pig fattening, are also of interest. Stagnating livestock numbers require new utilisation solutions in this area. Depending on the starch factory, quantities of residual materials

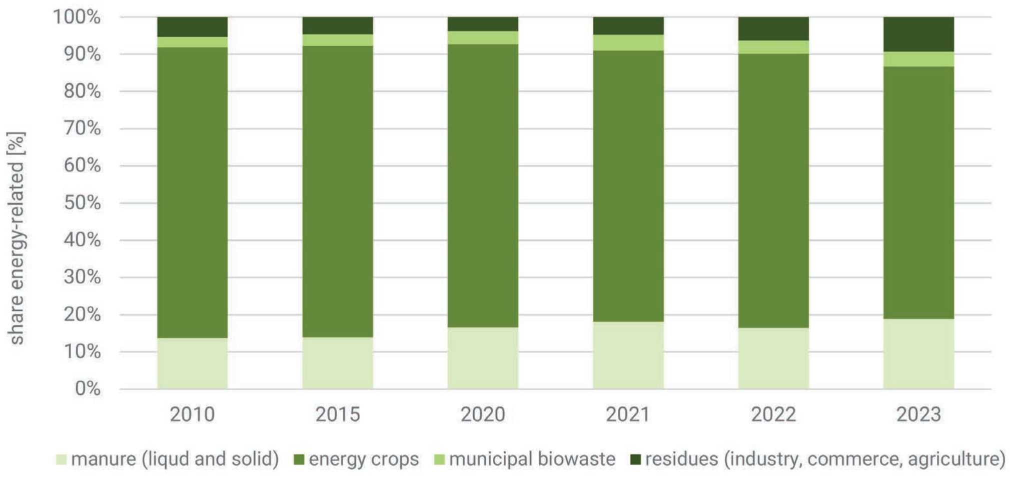

FIGURE 5: Energy-related substrate input in German biogas plants. Source: Non-public database: DBFZ operator surveys from 2011, 2016, 2021-2024 (reference value for the previous year). © (Daniel-Gromke et al., 2025, p.55).

ranging from 100,000 to 500,000 tonnes per year (based on fresh mass) may be available at a single location. The low dry matter content of approximately 20% poses economic challenges in terms of logistics. Direct discharge into the wastewater system without pre-treatment is not possible due to the high organic load (Umweltbundesamt, 2019). To add value to wastewater treatment, biogas can be produced by integrating anaerobic fermentation (IEA, 2020). Furthermore, added value can be generated by processing and recycling the fermentation residues, either as fertiliser or as substances that can be isolated from the fermentation residues as basic chemicals (Drosg et al., 2015). New input streams from the manufacturing industry in particular require the adaptation of biogas technology, which has mainly been used in the agricultural sector to date. The fermentation of homogeneous individual substances poses particular challenges in terms of supplying the microorganisms with micronutrients, for example, or regulating the pH value in the process (Weinrich & Nelles, 2021).

Technological innovations play an important role in the efficient use of these heterogeneous waste streams. Modern biogas plants are capable of processing a wide range of substrates while maintaining optimal process conditions (Theuerl et al., 2019). Advances in process monitoring and control make it possible to dynamically adapt fermentation to the properties of the materials used, thereby maintaining process stability with heterogeneous substrate streams. This is crucial for biogenic waste and residues, as their composition and quantities can vary greatly depending on the season.

In addition to energy production, biogas technology offers further advantages. The fermentation residues produced during fermentation are nutrient-rich fertilisers that can be used in agriculture, thereby reducing the use of mineral fertilisers (European Biogas Association, 2024). Furthermore, the use of biogenic waste and residues contributes to closing material cycles and reduces the amount of waste to be disposed (IEA, 2020). However, the environmental performance of biogas systems increasingly depends on (i) minimising methane slip along the chain and (ii) managing digestate to avoid nitrogen and phosphorus losses to air and water (IPCC, 2021; Umweltbundesamt, 2019).

Innovative utilisation of digestate is therefore a key future priority. Solid digestate can be upgraded via solid–liquid separation followed by composting, drying and pelletisation to produce transportable organic fertilisers and soil improvers, or it can be converted into biochar via (co-)pyrolysis/hydrothermal routes to stabilise carbon, reduce odour, and create sorbent/soil-amendment products (Drosg et al., 2015; Fu et al., 2024; Dincă et al., 2025). In parallel, the liquid fraction (often rich in readily available ammonium) requires careful treatment and application management to prevent contamination of water bodies (e.g., nitrate leaching and runoff), consistent with water-protection requirements such as the EU Nitrates Directive (EU, 1991). Promising approaches include ammonia stripping/scrubbing (e.g., ammonium sulfate), membrane filtration or evaporation/concentration, and phosphorus recovery as struvite, thereby turning the liquid fraction into standardised, marketable fertiliser products and reducing nutrient surpluses in vulnerable regions (Drosg et al., 2015; EU, 2019). Application methods that increase nutrient use efficiency (e.g., injection or trailing-shoe/band application, adapted timing and rates) further reduce emissions and leakage risks (Korba et al., 2024).

Despite the advantages mentioned above, there are challenges involved in integrating biogenic waste and residues into biogas production. The heterogeneous composition of the substrates requires flexible plant technologies and careful process monitoring (Weinrich & Nelles, 2021). Logistics such as the collection, transport and storage of the materials must also be taken into account. In addition, legal aspects of the joint use of biogenic waste and residues often stand in the way and are a major obstacle (Bundesgütegemeinschaft Kompost e.V., 2014). In the future, it will be crucial to further increase the efficiency of biogas technology and tap into new, previously unused waste and residual material streams. Research and development must focus on optimising process control, developing new pre-treatment methods and integrating biogas production into existing agricultural and industrial systems.

Overall, biogas technology can make a robust contribution to the energy transition and circular economy if it prioritises residues over dedicated crops, reduces methane slip, and couples energy services with nutrient recovery and safe digestate management as central design goals (IEA, 2020; IPCC, 2021; European Biogas Association, 2024).

5. BIOFUELS AND BIOREFINERIES BASED ON BIOGENIC WASTE AND RESIDUES

In 2022, the use of biofuels in Germany prevented emissions of 11.6 million tonnes of CO_2-eq. Of the total 140 PJ, 46% came from biofuels from waste and residues and 54% from cultivated biomass (Bundesanstalt für Landwirtschaft und Ernährung, 2022). The legal framework for bio-based fuels has already been adapted in recent years to significantly increase the share of waste and residues as feedstock for production. Based on the requirements of the EU Renewable Energy Directive (RED II), the following biomass classes for biofuels can be distinguished (Bundesanstalt für Landwirtschaft und Ernährung, 2025):

1. Conventional biomass: so-called food and feed crops;
2. Advanced biomass: raw materials in accordance with Annex IX A of RED II that are not suitable for food or feed;
3. Waste-based biomass: raw materials in accordance with Annex IX B of RED II;
4. Biomass with a high risk of indirect land use change: palm oil;
5. Other biomass: raw materials not explicitly assigned to any of the other categories.

In a comparative meta-study of sustainable biomass and substitution potentials, the amount of biogenic waste and residues available for processing into biofuels in Germany and Europe was investigated (Naumann et al., 2024). Relevant studies were identified based on the following criteria: transparent methodology, recency (no older than ten years) and potential levels considered (at least bioenergy potential or mobilisable potential). Furthermore, the focus of all studies was quantifying the potential of biogenic by-products, waste and residues. The comparison of the thus identified five studies – (Fehrenbach et al. 2018), (Panoutsou & Maniatis 2021), (Brosowski et al., 2019), (Ruiz et al., 2015) and (Searle & Malins 2016). In addition to quantifying biomass potential, its mobilisation potential is also of central strategic importance. This means that suitable production technologies that are sufficiently developed for market establishment are also required. In the best-case scenario, processing capacities are already available or existing facilities from the fossil resource processing sector can be converted or upgraded.

Based on the five studies the possible production quantities of biobased diesel, kerosene, naphtha, ethanol, methane and methanol were calculated. The analysis was made for Germany and Europe, to determine the quantity of biogenic substitutes that could be produced. According to these calculations, 7-28% of current refinery output could be substituted in Germany and 10-45% in the European Union (Figure 6). Looking at sufficiently mature conversion processes, there is substitution potential resulting primarily from the conversion of biogenic by-products, waste and residues into biomethane and biomethanol.

The wide range of the estimated substitution potentials is due to differences between the biomass potential studies. These arise from the potential levels considered, the number of biomasses considered, the assumptions made about the mobilisation and use of biomasses, the data used and the reference year. However, assuming a significant reduction in demand for liquid and solid fuels in the future, mainly due to electrification, the substitution shares determined would be even higher. The use of biogenic waste and residues for the production of biofuels therefore already makes a significant contribution to climate protection in the transport sector. Further biogenic waste and residues can also be used to produce fuels for the combustion engines that will remain in use in the medium term.

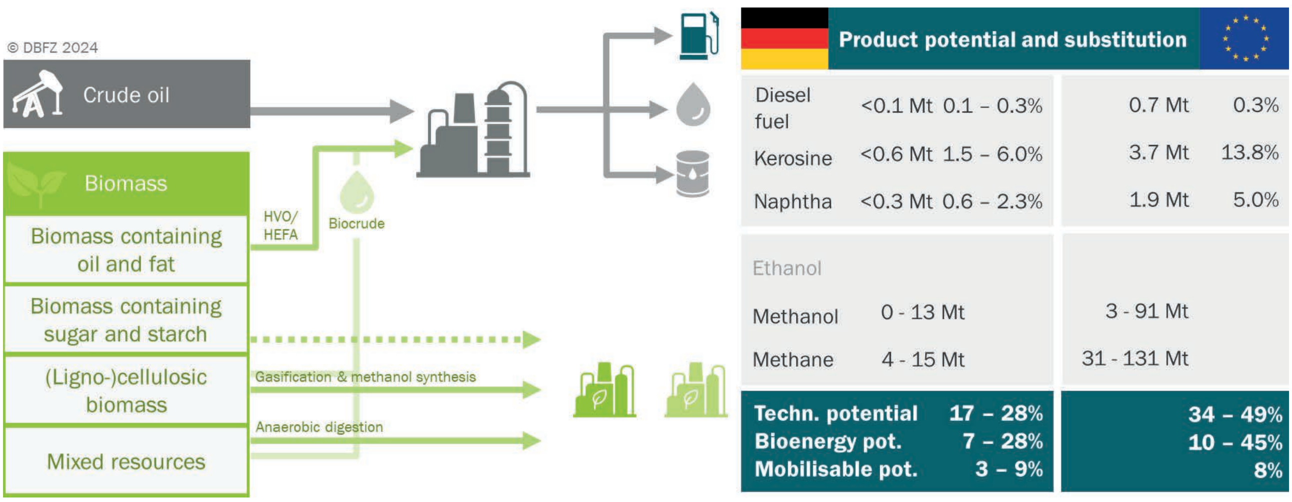

FIGURE 6: Substitution potential for mineral oil-based fuels by bio-based alternatives, for Germany and the EU. © (Naumann et al., 2024, p. 3).

For example, the DBFZ is investigating the efficient conversion of agricultural and urban residues into biomethane for transport (Röder et al., 2024). In an integrated pilot plant (Pilot-SBG), modules for biomass pre-treatment, anaerobic digestion, digestate processing and catalytic methanisation of CO_2 are being developed. Parallel to the ongoing experimental campaigns on developing the mentioned techniques, techno-economic assessments and sustainability assessments were already conducted. The costs for a commercial scale plant for the conversion of straw and manure to liquified biomethane were estimated for an exemplary plant concept with all of the mentioned modules. One of the findings is, that the catalytic methanation of the CO_2 in the biogas could double the methane production. However, the catalytic methanation module significantly adds to the costs, both to the CAPEX (33% of the entire investment) and to the OPEX (hydrogen would make 51% of the entire consumption related costs) (Röder, et al., 2024). A more thorough techno-economic assessment of benefits and drawbacks of the different modules and their combinations will be elaborated, when the experimental campaigns are completed.

The production of hydrogen from biomass is another approach to substitute fossil fuels. While competition with large-scale electrolysis is seen difficult, bio-hydrogen was found particularly promising through routes like steam reforming of biomethane and thermochemical gasification to serve as a decentralized pillar in the transition to a climate-neutral energy system (Dögnitz, 2022). Hydrogen could then be used to meet regional demands in transport and municipal fleets or be part of integrated concepts, where biogenic carbon is coupled with green hydrogen to create high-value synthetic fuels. Conversely, direct applications for the heating market or injection into the national gas grid are deemed inefficient, suggesting that the role of biogenic hydrogen is best realized in localized, circular value chains where it can provide high greenhouse gas reduction potential from otherwise underutilized waste streams.

The production of basic and fine chemicals from biobased waste and residues is also increasingly being pursued (Nitzsche et al., 2021). In principle, higher specific revenues can be achieved with these products, but at the same time the quality requirements for product purity are significantly higher and usually require longer processing chains. Since often only parts of the biomass can be processed into the desired products, the concept of biorefineries plays an important role in the production of bio- based raw materials for material use in the chemical industry. In biorefineries, the raw biomass is processed into various products as well as biogenic waste and residues through a combination of conversion and separation processes (Bundesregierung, 2012).

Another example of producing chemicals from regional biomass residues is the production of medium chain carboxylic acids by fermentation. The fermentation process has successfully demonstrated that caproic and caprylic acid can be produced from moist residues such as fruit pomace, harvest residues or other residues from the food industry or even mixed organic waste (Xu et al., 2018), (Chen et al., 2017). The entire process, including the purification of the carboxylic acids, has been demonstrated on a pilot scale (Braune et al., 2021). Sample quantities of the products were produced for application tests in the lubricant sector. The processing method is designed to be integrated into biogas plants so that, in addition to the carboxylic acids, biomethane is produced and the nutrients in the remaining digestate can ultimately be returned to agriculture as secondary fertiliser. This turns biogas plants into biorefineries that can process biogenic waste and residues with high added value (Braune et al., 2021).

6. NEGATIVE EMISSIONS

BECCS stands for Bioenergy with Carbon Capture and Storage. Bioenergy is based on biomass, the formation of which involves CO_2 capture from the air through plant growth. BECCS involves the technical conversion of biomass into bioenergy products (e.g. biofuels, heat/cooling, electricity) while simultaneously capturing carbon in the form of carbon dioxide or solid carbon products, including its permanent storage. This makes BECCS one of the better-known technologies in the field of carbon dioxide removal (CDR).

What contribution can BECCS options make to CO_2 capture in Germany in the coming years? The focus here is primarily on biogenic waste and residues, the utilisation of which is associated with a high climate protection contribution.

The significant bioenergy flows (> 10 PJ) shown in Figure 7 are converted into CO_2 flows. To this end, the CO_2 potential that would be released if the biomass were completely converted into CO_2 is calculated. This calculation is based on the carbon content of the biomass and the bioenergy products, as well as the efficiency of the associated bioenergy plant. In total, the CO_2 potential fed into the bioenergy system by biomass is approximately 150 million tonnes of CO_2/year (based on 2021 figures). Of this, just under 50 million tonnes of CO_2/year comes from biogenic waste and residues.

However, one restriction applies: not all bioenergy supply is equally suitable for expansion with BECCS components. The most promising routes are as follows:

CO_2 captured from biogas and biofuel production: around 2 million tonnes of CO_2 are already being captured each year from the processing of bioethanol and biogas. However, CO_2 capture could be doubled if large biogas plants that use waste and residual materials switch to biomethane production and set up appropriate processing with CO_2 capture. Around 3 million tonnes of CO_2 per year appear to be possible (Rensberg et al., 2023).

CO_2 that can be captured from the use of solid fuels in large-scale plants: 25-35 million tonnes of CO_2 per year could be captured from the combustion of waste and residual materials (e.g. waste wood, industrial wood residues) in existing waste incineration plants and large biomass heating (combined heat and power) plants. Assuming a capture rate of 80% (Markewitz et al., 2018, own estimates

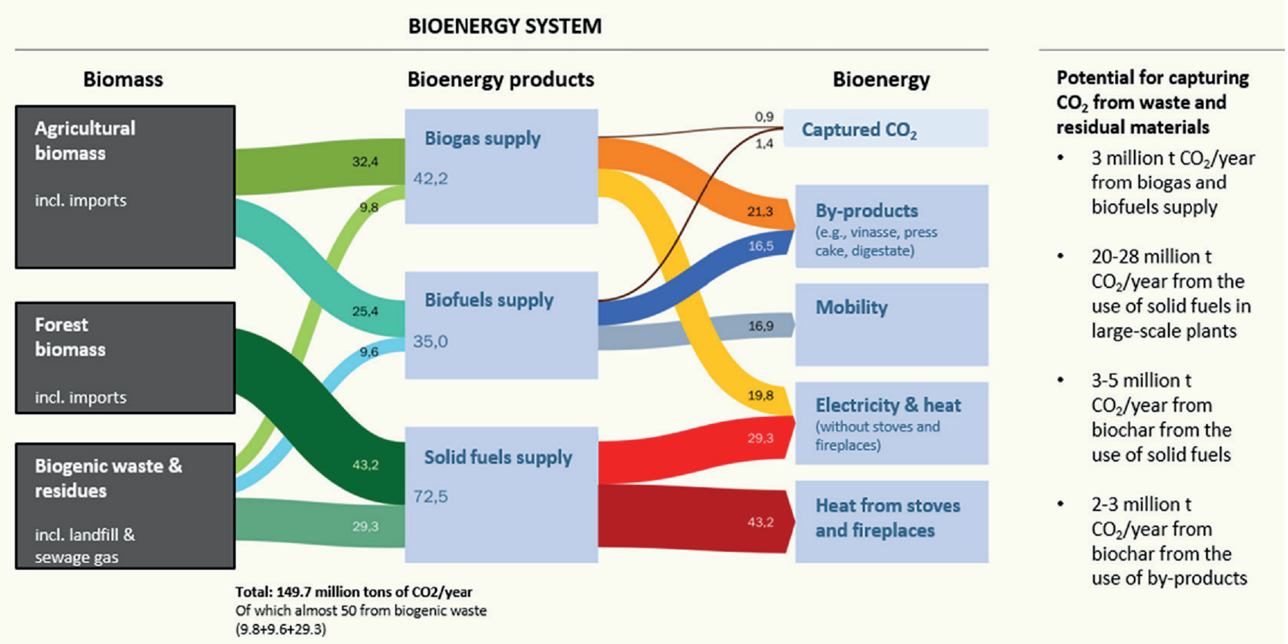

FIGURE 7: Main CO_2 flows (million tonnes CO_2/year) from biomass to bioenergy for 2021 (own calculations, based on energy flows > 10 PJ). © (Thrän, D. et al. (2025), p. 4).

of further losses in practice), this could contribute 20-28 million tonnes of CO_2 per year.

CO_2 that can be captured through biochar from the use of solid fuels: biochar can also be developed from the current bioenergy system. Assuming the further development and large-scale conversion of heating plants and combined heat and power plants to pyrolysis gas, up to 10 million tonnes of CO_2/year could be converted from forestry residues in these systems. Sewage sludge, which can also be converted into biochar due to its very highwater content but does not provide any energy, is not included in this estimate. Assuming a capture rate of 30-50% (Kaltschmitt et al., 2024), a further 3-5 million tonnes of CO_2/year could be captured in this way.

CO_2 that can be captured using biochar from the use of by-products: approximately 10% of the by- products generated from biogas and biofuel production (38 million tonnes of CO_2/year) are based on waste and residues. They could provide an additional 2-3 million tonnes of CO_2/year of permanently stored CO_2 in the form of biochar.

In total, based on the waste and residual materials used and the establishment of various BECCS options, it is conceivable that 28-39 million tonnes of CO_2/year could be removed in the current bioenergy system. It should be noted that the conversion will not be technically and economically feasible in all cases in terms of fuel or plant technology. More information can be found in (Thrän et al., 2025).

7. CONCLUSIONS

The article discusses the role of biogenic waste and residues in achieving a climate-neutral world, particularly in Germany. The main findings include:

1. Substitution potential: 7-28% of current refinery output in Germany could be substituted with bio-based alternatives, and 10-45% in the European Union.
2. CO_2 reduction: The use of biofuels from waste and residues prevented emissions of 11.6 million tonnes of CO_2-eq in Germany in 2022.
3. Recycling of residual and waste wood: The use of residual and waste wood must be re-thought and transformed from a systemic perspective to achieve a more climate-efficient utilization, focusing on extended material cascades, peak load applications, and integrated negative emission generation.
4. Biomass potential: Biogenic waste and residues can provide a significant amount of biomass for energy production, with approximately 150 million tonnes of CO_2/year potentially available in Germany.
5. BECCS (Bioenergy with Carbon Capture and Storage): This technology has the potential to capture around 2 million tonnes of CO_2/year from biogas and biofuel production, which could be doubled if large biogas plants switch to BECCS.
6. Biogas technology: Anaerobic digestion can convert biogenic waste and residues into biomethane, providing an environmentally friendly source of energy and reducing greenhouse gas emissions.

Overall, the article highlights the importance of utilizing biogenic waste and residues in a sustainable manner to achieve climate neutrality and reduce greenhouse gas emissions

REFERENCES

Braune, M., Yuan, B., Sträuber, H., McDowall, S. C., Nitzsche, R., & Gröngröft, A. (2021). A Downstream Processing Cascade for Separation of Caproic and Caprylic Acid from Maize Silage-Based Fermentation Broth. Frontiers in Bioengineering and Biotechnology, 9. https://doi.org/10.3389/fbioe.2021.725578

Brosowski, A.; Krause, T.; Mantau, U.; Mahro, B.; Noke, A.; Richter, F. et al. (2019): Arbeitsgruppe Biomassereststoffmonitoring (AG BioRestMon). Schlussbericht. Hg. v. Deutsches Biomasseforschungszentrum (DBFZ). Deutsches Biomasseforschungszentrum (DBFZ). Leipzig. Online verfügbar unter https://www.fnr.de/ftp/pdf/berichte/22019215.pdf.

Bundesanstalt für Landwirtschaft und Ernährung.(2024). Nachhaltigkeitsverordnung/ Biokraftstoff- Nachhaltigkeitsverordnung https://www.ble.de/SharedDocs/Downloads/DE/Klima-Energie/Nachhaltige-Biomasseherstellung/Evaluationsbericht_2022.pdf?__blob=publicationFile&v=1

Bundesanstalt für Landwirtschaft und Ernährung. (2025). 2 KRAFTSTOFF Nabisy-Biomassearten Stand 05.2025. https://www.ble.de/SharedDocs/Downloads/DE/Klima-Energie/Nachhaltige-Biomasseherstellung/Nabisy/Biomassearten.xlsx

Bundesanstalt für Landwirtschaft und Ernährung. (2025). Evaluations- und Erfahrungsbericht für das Jahr 2022: Biomassestrom-Nachhaltigkeitsverordnung/ Biokraftstoff-Nachhaltigkeitsverordnung. https://www.ble.de/SharedDocs/Downloads/DE/Klima-Energie/Nachhaltige-Biomasseherstellung/Evaluationsbericht_2022.pdf

Bundesgütegemeinschaft Kompost e.V. (2014). Landschaftspflegematerial in der Vergärung und Kompostierung (legal/permit relevance of landscape conservation material as waste). https://www.kompost.de/uploads/media/Landschaftspflegematerial_in_der_Vergaeung_und_Kompostierung.pdf?utm

Bundes-Klimaschutzgesetz vom 12. Dezember 2019 (BGBl. I S. 2513), das zuletzt durch Artikel 1 des Gesetzes vom 15. Juli 2024 (BGBl. 2024 I Nr. 235) geändert worden ist: KSG (21.11.24). https://www.gesetze-im-internet.de/ksg/KSG.pdf

Bundesregierung (2012). Roadmap Bioraffinerien. Bundesregierung.

Bundesregierung (2022). Generationenvertag für das Klima: Pressemitteilung zur Änderung des Klimaschutzgesetzes 2022. www.bundesregierung.de/breg-de/service/archiv/klimaschutzgesetz-2021- 1913672.pdf

Bundesumweltministerium (2024). Eckpunkte für eine Nationale Biomassestrategie (NABIS). https://www.bundesumweltministerium.de/download/eckpunkte-fuer-eine-nationale-biomassestrategie-nabis

Carrillo-Rodríguez, L. A., Romero, M. C., & Galindo Larrahondo, J. A. (2025). The use of agro-industrial waste towards a sustainable circular economy: A systematic review. Detritus, 31, 45–61. https://doi.org/10.31025/2611-4135/2025.19491

Chen, W.-S., Strik, D. P. B. T. B., Buisman, C. J. N., & Kroeze, C. (2017). Production of Caproic Acid from Mixed Organic Waste: An Environmental Life Cycle Perspective. Environmental Science & Technology, 51(12), 7159–7168. https://doi.org/10.1021/acs.est.6b06220

Daniel-Gromke, J., Rensberg, N., Denysenko, V., Mazlum, L., Dotzauer, M., Dögnitz, N., Naumann, K. (2025): Wissenschaftliche Analysen zu ausgewählten Aspekten der Statistik erneuerbarer Energien und zur Unterstützung der Arbeitsgruppe Erneuerbare Energien-Statistik (AGEE-Stat). Fachbericht Biomasse. Climate Chang 55/2025. ISSN:1862-4359. Dessau Roßlau. abrufbar unter: https://www.umweltbundesamt.de/publikationen/wissenschaftliche-analysen-zu-ausgewaehlten

Dincă, M. N.; et al. (2025). Review on digestate valorisation pathways (solid/liquid management and resource recovery). https://www.mdpi.com/2076-3417/15/16/8939?utm

Dögnitz, N.; Hauschild, S.; Cyffka, K.-F.; Meisel, K.; Dietrich, S.; Müller-Langer, F.; Majer, S.; Kretzschmar, J.; Schmidt, C.; Reinholz, T.; Gramann, J. (2022). Wasserstoff aus Biomasse: Kurzstudie im Auftrag des Bundesministeriums für Ernährung und Landwirtschaft. (DBFZ Report, 46). Leipzig: DBFZ. III, 4-147 S. ISBN: 978-3-946629-88-7. DOI: 10.48480/b4wn-c154

Drosg, B.; et al. (2015). Nutrient Recovery by Biogas Digestate Processing. IEA Bioenergy. https://task37.ieabioenergy.com/wp-content/uploads/sites/32/2022/02/NUTRIENT_RECOVERY_RZ_web2.pdf

Eurostat. (2024). Circular material use rate database. https://ec.europa.eu/eurostat/databrowser/view/env_ac_cur/default/table?lang=en

European Biogas Association. (2024). Exploring digestate's contribution to healthy soils (report). https://www.europeanbiogas.eu/wp-content/uploads/2024/03/Exploring-digestate-contribution-to-health-soils_EBA-Report.pdf?utm

EU (1991) Council Directive 91/676/EEC (EU). Protection of waters against pollution caused by nitrates from agricultural sources (Nitrates Directive). (EUR-Lex).

EU (2019) Regulation (EU) 2019/1009. EU rules for fertilising products (incl. conditions for placing fertilising products on the market).

Fachagentur Nachwachsende Rohstoffe. (2022). Biogas aus landwirtschaftlichen Rest- und Abfallstoffen. https://biogas.fnr.de/biogas-gewinnung/gaersubstrate/rest-und-abfallstoffe

Fachagentur Nachwachsende Rohstoffe. (2022). Zwei Fliegen mit einer Klappe: Pappelholz für

Biomethan und Torfersatzstoffe. FNR-Pressemitteilung. https://news.fnr.de/fnr-pressemitteilung/zwei-fliegen-mit-einer-klappe-pappelholz-fuer-biomethan-und-torfersatzstoffe

Fachagentur Nachwachsende Rohstoffe. (2025). Holzfasern: Gärtnern ohne Torf: Mit Holzfaserstoffen. https://torfersatz.fnr.de/torfminderung/torfalternativen/holzfasern

Fachagentur Nachwachsende Rohstoffe. (2025). Rindenhumus: Gärtnern ohne Torf: Mit Rindenhumus. https://torfersatz.fnr.de/torfminderung/torfalternativen/rindenhumus

Fraunhofer-Gesellschaft. (2025). Lignin – eine nachhaltige Alternative zu fossilen Rohstoffen. www.fraunhofer.de/de/forschung/fraunhofer-strategische-forschungsfelder/biooekonomie/lignin.html

Fehrenbach, H.; Giegrich, J.; Köppen, S.; Wern, B.; Pertagnol, J.; Baur, F. et al. (2018): BioRest: Verfügbarkeit und Nutzungsoptionen biogener Abfall- und Reststoffe im Energiesystem. (Strom-, Wärme- und Verkehrssektor). Abschlussbericht. Hg. v. Umweltbundesamt. Institut für Energie- und Umweltforschung GmbH (ifeu); Institut für Zukunftsenergie- und Stoffstromsysteme (IZES); Öko - Institut e.V. Dessau-Roßlau. Online verfügbar unter https://www.umweltbundesamt.de/sites/default/files/medien/1410/publikationen/2019-0924_texte_115-2019_biorest.pdf

Fu, Z.; Zhao, J.; et al. (2024). A comprehensive review on the preparation of biochar from digestate sources and its application in environmental pollution remediation. Science of the Total Environment, 912, 168822. doi:10.1016/j.scitotenv.2023.168822.

Global Footprint Network. (2024). Footprint Calculator. https://www.footprintnetwork.org/resources/footprint- calculator/

Ghamari, M., Suvish, Hwang See, C., Yu, H., Anitha, T., Balamurugan, V. T., Velusamy, S., Hughes, D., Sundaram, S. (2025) Nanocellulose Extraction from Biomass Waste: Unlocking Sustainable Pathways for Biomedical Applications. The chemical record. https://doi.org/10.1002/tcr.20240024

Intergovernmental Panel on Climate Change (IPCC). (2021). AR6 Working Group I: The Physical Science Basis (methane as a potent greenhouse gas). https://www.ipcc.ch/report/ar6/wg1/downloads/report/IPCC_AR6_WGI_Chapter06.pdf?utm

International Energy Agency (IEA). (2020). Outlook for biogas and biomethane: Prospects for organic growth. Paris. https://www.iea.org/reports/outlook-for-biogas-and-biomethane-prospects-for-organic-growth

Jelschow, V., Mais, F., Gramberg, T., Gross, E., Fajardo, E.A.G., Kiemel, S., Full, J., Rögele, S., Schabernack, N., Gümbel, T., Wendt, J. (2024) H2Wood – BlackForest: Wasserstofferzeugung aus Rest- und Altholz in der Region Schwarzwald. Potenziale, Barrieren und Handlungsmaßnahmen. Fraunhofer-Institut für Produktionstechnik und Automatisierung IPA. https://www.ipa.fraunhofer.de/de/Publikationen/studien/h2wood-blackforest.html

Jordan, M., Lenz, V., Millinger, M., Oehmichen, K., & Thrän, D (Daniela) (2019). Future competitive bioenergy technologies in the German heat sector: Findings from an economic optimization approach. Energy. Advance online publication. https://doi.org/10.1016/j.energy.2019.116194

Kaltschmitt, M., Hofbauer, H., & Lenz, V. (2024). Energie aus Biomasse: Band 2 (4th ed.). Springer Vieweg. https://doi.org/10.1007/978-3-658-41216-6

Korba, J.; et al. (2024). Digestate Application Methods and Rates with Regard to Greenhouse Gas Emissions and Crop Conditions. Agronomy, 14(2), 336. doi:10.3390/agronomy14020336.

Krause, T., Mantau, U., Mahro, B., Noke, A., Richter, F., Raussen, T., Bischof, R., Hering, T., Thrän, D., & Brosowski, A. (2020). Nationales Monitoring biogener Reststoffe, Nebenprodukte und Abfälle in Deutschland: Teil 1 : Basisdaten zu Biomassepotenzialen. https://doi.org/10.48480/6MZ1-ZS78

Mantau, U. (2023). Holzrohstoffbilanzierung - Kreislaufwirtschaft und Kaskadennutzung: 20 Jahre Rohstoffmonitoring Holz. Schriftenreihe nachwachsende Rohstoffe: Vol. 40. Fachagentur Nachwachsende Rohstoffe e.V. (FNR). https://edocs.tib.eu/files/e01fn24/1905202296.pdf

Markewitz, P., Zhao, L., & Robinius, M. (2018). Technologiebericht 2.3 CO_2-Abscheidung und -Speicherung innerhalb des Forschungsprojekts TF_Energiewende. Wuppertal Institut für Klima, Umwelt, Energie. https://epub.wupperinst.org/frontdoor/index/index/docId/7051
Matschoss, P. (2025). Keynote: Bioenergie und Energiewende: Die „Post-EEG"-Frage im größeren energiepolitischen Rahmen. Folgekonzepte für die post-EEG-Phase von Bioenergieanlagen (BE-20plus & NextGenBGA), Berlin. https://www.dbfz.de/fileadmin/user_upload/Fachgespraeche/Sonstige/Vortraege/2020-02_Bioenergie-Post-EEG_Vortraege.pdf
Naegeli de Torres, F. et al. (2024)DBFZ Resource Database : DE-Biomass Monitor. Biomass Potentials and Utilization of Biogenic Wastes and Residues in Germany 2010-2020. https://zenodo.org/records/14273285
Naumann, K., Cyffka, K.-F., Karras, T. (2024). Bio2x: Vergleichende Analyse zu nachhaltigen Biomasse- und Substitutionspotenzialen. Hintergrundpapier. DBFZ. https://www.dbfz.de/fileadmin/user_upload/Referenzen/Statements/Hintergrundpapier_Bio2x_Okt_2024.pdf
Nelles, M., Backes, R., Cyffka, K.-F., Gröngröft, A., Kornatz, P., & Lenz, V. (2025). Verwertung biogener Abfälle und Reststoffe: Kohlenstoffquelle, Bioenergie & negative Emissionen. Müll und Abfall, 57(4), 180. https://doi.org/10.37307/j.1863-9763.2025.04.03
Nitzsche, R., Gröngröft, A., Köchermann, J., Meisel, K., Etzold, H., Verges, M., Leschinsky, M., Bachmann, J., Saake, B., Torkler, S., Patzsch, K., Rößiger, B., Pufky-Heinrich, D., Unkelbach, G. (2021). Platform and fine chemicals from woody biomass: demonstration and assessment of a novel biorefinery. Biomass Conversion and Biorefinery, 11(6), 2369–2385. https://doi.org/10.1007/s13399-020-00769-z
Panoutsou, C.; Maniatis, K. (2021): Sustainable biomass availability in the EU, to 2050. Ref: RED II Annex IX A/B. Hg. v. Concawe. Imperial Consultants, Imperial Collage London. London. Online verfügbar unter https://www.concawe.eu/wp-content/uploads/Sustainable-BiomassAvailability-in-the-EU-Part-I-and-II-final-version.pdf
Rensberg, N., Denysenko, V., Daniel-Gromke, J. (2023). Biogaserzeugung und -nutzung in Deutschland: Report zum Anlagenbestand Biogas und Biomethan. DBFZ-Report: Vol. 50. DBFZ. https://doi.org/10.48480/zptb-yy32
Röder, L., Nitzsche, R., Etzold, H., Oehmichen, K. (2024). Beispielkonzept zur Bereitstellung von erneuerbarem LNG aus biogenen Rest- und Abfallstoffen und erneuerbarem Wasserstoff im kommerziellen Maßstab. DBFZ Deutsches Biomasseforschungszentrum gemeinnützige GmbH. https://doi.org/10.48480/JSCT-Z879
Ruiz, P.; Sgobbi, A.; Nijs, W.; Thiel, C.; Dalla Longa, F.; Kober, T. et al. (2015): The JRC-EU-TIMES model. Bioenergy potentials for EU and neighbouring countries. Luxembourg: Publications Office (EUR, Scientific and technical research series, 27575).
RWTH Aachen. (2024). Biochar – Neue Generation von mit Biokohle modifizierten, durchlässigen Straßenbelagsmaterialien. www.isac.rwth-aachen.de/cms/isac/forschung/projekte/strassenbau-laufende-projekte/abgeschlossene-forschungsprojekte-strassenbau
Searle, S. Y.; Malins, C. J. (2016): Waste and residue availability for advanced biofuel production in EU Member States. In: Biomass and Bioenergy 89, S. 2–10. DOI: 10.1016/j.biombioe.2016.01.008.
Theuerl, S.; et al. (2019). The Future Agricultural Biogas Plant in Germany: A Vision. Energies, 12(3), 396. https://www.mdpi.com/1996-1073/12/3/396?utm
Top Agrar. (2024). Welchen Preis erzielen Heu und Stroh derzeit? Kostenlos mit Rechner ermitteln. https://www.topagrar.com/markt/news/hier-koennen-landwirte-kostenlos-den-strohpreis-berechnen-k-20007220.html
Thrän, D., Borchers, M., Lenz, V., Jordan, M., Markus, T., Matzner, N., Oehmichen, K., Otto, D., Radtke, K. S., Reshef, N., Sadr, M., Siedschlag, D., Wollnik, R. (2025). The role of BECCS in Germany: A key to sustainable and permanent CO_2 removal? Environmental Research Communications. Advance online publication. https://doi.org/10.1088/2515-7620/ae02ee
Umweltbundesamt. (2019). Aktuelle Entwicklung und Perspektiven der Biogasproduktion aus Bioabfall und Gülle (Texte 41/2019). https://www.umweltbundesamt.de/system/files/medien/1410/publikationen/2019-04-15_texte_41-2019_biogasproduktion.pdf?utm
Umweltbundesamt. (2024). Erneuerbare Energien in Zahlen. https://www.umweltbundesamt.de/themen/klima-energie/erneuerbare-energien/erneuerbare-energien-in-zahlen#waerme
Umweltbundesamt. (2025). Biogasproduktion aus Gülle und Bioabfall ausbauen. https://www.umweltbundesamt.de/themen/biogasproduktion-aus-guelle-bioabfall-ausbauen
Vaupel, M. (2023). Pflanzenkohle in Beton: Gebäude und Infrastruktur werden zur Kohlenstoffsenke. Technik in Bayern - das Regionalmagazin für VDU und VDE(Jan/Feb), 26. https://www.technik-in-bayern.de/bauen-und-gebaeudetechnik/nachhaltiges-bauen/mensch-raum-und-pflanze
Weinrich, S.; Nelles, M. (2021). Basics of Anaerobic Digestion: Biochemical Conversion and Process Modelling (DBFZ-Report 40). https://www.dbfz.de/fileadmin/user_upload/Referenzen/DBFZ_Reports/DBFZ_Report_40.pdf?utm
Wilske, B.; Cyffka, K.-F.; Kalcher, J.; Karras, T.; Selig, M.; Radtke, Kai et al. (2025): Schlussbericht zum Projekt MoBi II. Aufbau eines systematischen Monitorings der Bioökonomie – Konsolidierungsphase. Teilvorhaben 2: Aktualisierung Reststoffmonitoring. https://dx.doi.org/10.2314/KXP:192896544X
Wollnik, R., Borchers, M., Seibert, R., Abel, S., Herrmann, P., Elsasser, P., Hildebrandt, J., Mühlich, M., Eisenschmidt, P., Meisel, K., Henning, P., Radtke, K. S., Selig, M., Kazmin,S., Thrän, D., Szarka, N. (2023). Steckbriefe für biobasierte Kohlenstoffdioxidentnahme-Optionen in Deutschland. https://datalab.dbfz.de/bionet/
Xu, J., Hao, J., Guzman, J. J. L., Spirito, C. M., Harroff, L. A., & Angenent, L. T. (2018). Temperature-Phased Conversion of Acid Whey Waste Into Medium-Chain Carboxylic Acids via Lactic Acid: No External e-Donor. Joule, 2(2), 280–295. https://doi.org/10.1016/j.joule.2017.11.008
Yu, Q.; et al. (2019). Review on straw/ lignocellulosic pre-treatment for anaerobic digestion (improving accessibility and methane yield). https://www.sciencedirect.com/science/article/pii/S1364032119301133?utm

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ENVIRONMENTAL LIABILITY ALLOCATION - PRINCIPLES

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1. INTRODUCTION

The past year has witnessed several significant developments in fixing liability for climate and environmental harm, signalling a decisive shift from abstract notions of responsibility to concrete legal and financial accountability, even in areas as complex and contested as climate change. From California's proposed Polluters Pay Climate Superfund Act, which seeks to impose financial liability on major fossil fuel companies for climate-related damages (California State Legislature, 2025), to the landmark ruling of the Hamm Higher Regional Court in Saúl Luciano Lliuya v. RWE, affirming the proportional liability of major emitters for climate impacts (Hamm Higher Regional Court, 2025), the legal landscape is rapidly evolving. Parallel policy proposals, such as those advanced by Eurodad and the Global Alliance for Tax Justice advocating additional taxes on fossil fuel profits (Trilling, 2025), further reflect a growing consensus that the emphasis is no longer on achieving perfect certainty in attribution, but on ensuring that those who have contributed meaningfully to environmental degradation bear an enforceable share of the economic burden of its impacts.

While these contemporary developments appear novel, the concept of holding individuals accountable for environmental harm is not new. Early civilisations recognised rudimentary forms of environmental responsibility. Plato, in The Laws, suggested that those who polluted water sources should not only compensate for the damage but also restore the resource (Jowette, 2008). In ancient India, Kautilya's Arthashastra prescribed financial penalties for acts such as polluting public spaces and water bodies, with fines serving compensatory and deterrent purposes (Rangarajan, 1992). Roman law similarly developed principles of nuisance and liability for harmful acts affecting neighbouring property and community resources (Ibbetson, 2003). These early formulations, though limited in scope, reveal an enduring concern for regulating environmental harm through liability.

Environmental liability, however, began to take explicit legal shape during the Industrial Revolution, when large-scale industrial activity caused widespread pollution and harm to human health and property. Initially, liability was addressed through traditional tort law, focusing primarily on personal injury or property damage, with little recognition of harm to the environment as an independent legal interest. It was only in the mid-twentieth century, following major environmental disasters and increased scientific understanding of ecological damage, that environmental liability emerged as a distinct legal concept. The 1972 Stockholm Conference on the Human Environment marked a turning point, and the OECD's articulation of the Polluter Pays Principle in the same year provided a normative foundation for modern environmental liability regimes (United Nations, 1972).

This evolution found an early and influential expression in statutory frameworks such as the United States' Comprehensive Environmental Response, Compensation, and Liability Act, 1980 (CERCLA), the first law to expressly impose strict and retroactive liability for environmental contamination. Enacted against the backdrop of disasters such as Love Canal (USEPA & NYSDEC, 2013) and the inadequacy of existing legal remedies, CERCLA firmly established environmental liability as an independent and enforceable legal mechanism. The principles governing environmental liability have continued to expand and adapt in response to new forms of environmental harm. Against this backdrop, this column examines the principles underlying the allocation of environmental liability and their growing relevance in addressing contemporary environmental challenges.

2. NEED FOR ENVIRONMENTAL LIABILITY

Environmental liability allocation is essential for protecting the environment and human health by ensuring accountability for environmental harm. Natural resources such as air, water, land, and ecosystems are fundamental to life, and unchecked pollution can cause irreversible damage. By enforcing the polluter pays principle, environmental liability places the responsibility for prevention, control, remediation, and restoration on those who cause pollution, rather than shifting the burden to society or future generations. Importantly, it also serves as a tool of environmental justice by ensuring that communities affect-

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ed by pollution—often economically weaker and socially marginalized groups—receive compensation and legal protection instead of bearing the costs of harm they did not cause.

A central justification for environmental liability lies in the internalization of environmental costs. Industrial activities frequently generate pollution, resource depletion, and health hazards that disproportionately impact poorer communities living near industrial zones, landfills, or polluted water bodies. These harms are usually treated as external costs and excluded from the market price of goods. Environmental liability converts such external costs into internal costs by legally requiring industries to account for environmental damage, health impacts, and social consequences. As a result, expenses related to pollution control, waste management, cleanup, compensation, and environmental restoration become part of regular business operations, closely reflecting the polluter pays principle.

By internalizing environmental costs, environmental liability promotes fairness, deterrence, and sustainable development. Industries are encouraged to adopt cleaner technologies and safer production methods to minimize liability, thereby reducing environmental risks at the source. At the same time, when polluters pay the true cost of environmental damage, product prices better reflect their real social costs, preventing unfair competition based on environmentally harmful practices. This approach protects the public—especially economically weaker sections—from absorbing hidden costs of pollution and ensures that environmental protection is integrated into industrial decision-making, supporting both social justice and long-term environmental sustainability.

3. PRINCIPLES OF ENVIRONMENTAL LIABILITY

The principles of environmental liability allocation determine who should bear responsibility for environmental damage and how that responsibility is shared. The main principles are discussed below.

3.1 Polluter Pays Principle

The polluter pays principle is a foundational concept in environmental liability allocation, placing responsibility for environmental harm squarely on the person or industry that causes pollution. It requires polluters to bear the costs of preventing, controlling, and remedying environmental damage, ensuring that such burdens are not transferred to the public or the state. The principle emerged in response to the limitations of traditional fault-based liability systems, which proved inadequate for addressing widespread and long-term environmental harm. Its origins lie in economic theory of the 1960s (OECD, 1992), and it was formally articulated by the Organisation for Economic Co-operation and Development (OECD) in 1972 to promote efficient resource use through the internalization of environmental costs (United Nations, 1972). Subsequently, the principle gained international recognition through instruments such as the Stockholm Declaration (1972) and the Rio Declaration on Environment and Development (1992), marking its evolution from a policy guideline to a recognized legal norm.

Over time, the polluter pays principle has been incorporated into binding legal frameworks and judicial decisions across jurisdictions. In the European Union, it is enshrined in Article 191 of the Treaty on the Functioning of the European Union and underpins environmental liability directives. In the United States, legislation such as the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA) gives effect to the principle by imposing cleanup costs on responsible parties. In India, the Supreme Court has firmly embedded the polluter pays principle within environmental jurisprudence, through its decisions in cases such as Indian Council for Enviro-Legal Action v. Union of India (1996) (1996 SCC (3) 212), where polluting industries were held liable for the full cost of remediation, and Vellore Citizens' Welfare Forum v. Union of India (1996) (1996 AIR SCW 3399), which recognized the principle as an essential component of sustainable development. Through these legal developments, the polluter pays principle has become a cornerstone of environmental liability allocation and a key instrument for ensuring environmental justice.

3.2 Precautionary Principle

The precautionary principle operates as an important principle of environmental liability allocation by allowing responsibility to be imposed even in the absence of complete scientific certainty regarding environmental harm. Where an activity poses a serious or irreversible risk to the environment, the principle requires the actor to take preventive measures in advance, and failure to do so may attract liability. A distinctive feature of this principle in the context of liability allocation is the shifting of the burden of proof. Instead of requiring affected communities or regulatory authorities to establish actual environmental damage, the onus is placed on the developer or industry to prove that the proposed activity is environmentally safe. This approach reflects the understanding that waiting for conclusive scientific evidence may result in irreversible harm. The principle originated in the German Vorsorgeprinzip of the 1970s (ERPS, 2015), which emphasized anticipatory action, and gained international recognition through Principle 15 of the Rio Declaration on Environment and Development (1992).

As a principle of environmental liability allocation, the precautionary principle shifts environmental governance from a reactive, damage-based model to a preventive framework that prioritizes risk avoidance. It has been widely incorporated into national and international legal systems, particularly in areas involving environmental and public health risks. In the European Union, it forms a core component of environmental policy and guides regulatory decision-making in fields such as chemical control and food safety (European Commission, 2022). However, its implementation and codification vary among Member States, leading in practice, to differing interpretations and outcomes. For example, Sweden applies a comparatively "strong" version of the precautionary principle. Under Swedish law, the burden of proof rests with the operator, and regulatory action may be triggered by a risk of "damage" as such, rather than being limited to cases involving a risk of "irreversible damage," which is often the threshold applied elsewhere (Chapter 2, s. 3 p. 2, Swedish Environmental Code). In India, the Supreme Court has expressly adopted the precautionary principle as part of environmental jurisprudence in Vellore Citizens' Welfare Forum v. Union of India (1996) (AIR 1996 SC 2715), where the Court held that the burden of proof lies on the industry to demonstrate environmental safety. This position was reinforced in A.P. Pollution Control Board v. Prof. M.V. Nayudu (1999) (AIR 1999 SC 812), which emphasized preventive action in situations of scientific uncertainty.

3.3 Strict Liability Principle

The principle of strict liability plays a significant role in environmental liability allocation by allowing responsibility to be imposed without the need to prove fault, negligence, or intent. Under this principle, when an inherently dangerous or hazardous activity causes environmental harm, the operator or person in control of the activity is held liable irrespective of the care taken. The rationale behind strict liability is that certain activities, by their very nature, pose a high risk to the environment and public safety, and those who undertake such activities must bear the consequences of any resulting damage. This principle evolved to address the inadequacy of fault-based liability systems in situations where environmental harm occurs despite the absence of negligence.

The origins of strict liability can be traced to English common law, particularly the landmark decision in Rylands v. Fletcher (1868) (LR 3 HL 330), where the House of Lords held that a person who brings onto their land something likely to cause mischief if it escapes must keep it at their peril and is liable for any damage caused by its escape. This marked a decisive shift away from fault-based liability and laid the foundation for holding operators accountable for harm arising from hazardous substances and activities. Internationally, strict liability has been widely incorporated into environmental law. In the United States, statutes such as the CERCLA (1980) impose strict liability on parties responsible for hazardous waste contamination. Similarly, strict liability is reflected in environmental liability regimes and international conventions addressing oil pollution and nuclear damage, including the International Convention on Civil Liability for Oil Pollution Damage (1969). Through these frameworks, strict liability ensures effective compensation and environmental protection by placing responsibility on those best positioned to prevent harm and absorb the costs of environmental damage.

3.4 Absolute Liability Principle

The principle of absolute liability represents a more rigorous and advanced form of environmental liability than strict liability and is a distinctive contribution of Indian environmental jurisprudence. While strict liability may allow certain exceptions or defences, absolute liability permits none and imposes complete responsibility on industries engaged in hazardous or inherently dangerous activities for any harm caused. This principle was evolved by the Supreme Court of India in the landmark case M.C. Mehta v. Union of India (1987 AIR 1086, 1987 SCR (1) 819) (Oleum Gas Leak Case, 1987), where the Court held that such enterprises owe an absolute and non-delegable duty to the community to ensure that no harm results from their operations. Unlike strict liability, which is rooted in common law and is subject to limitations, absolute liability extends the scope of responsibility to cover compensation to victims and also the full cost of environmental damage and remediation, thereby reinforcing a stronger commitment to public safety and environmental protection in the context of modern industrial hazards.

3.5 The Public Trust Doctrine

The Public Trust Doctrine is a significant principle applicable to environmental liability, based on the idea that certain natural resources—such as air, water, forests, rivers, and coastal areas—are held by the State in trust for the benefit of the public. Under this doctrine, the State is not the absolute owner of natural resources but a trustee with a fiduciary duty to protect, preserve, and manage them for present and future generations. Consequently, environmental liability may arise not only from direct harm caused by the State's actions but also from its failure to regulate, supervise, or prevent environmental damage by private actors. When public authorities neglect their statutory or constitutional duties, especially in permitting environmentally harmful activities or failing to enforce environmental laws, the State can be held accountable for breach of its public trust obligations.

The application of the public trust doctrine to environmental liability has been strongly affirmed through judicial decisions, particularly in India. In M.C. Mehta v. Kamal Nath (1997) ((1997) 1 SCC 388), the Supreme Court explicitly recognized the public trust doctrine and held that the government is duty-bound to protect natural resources and cannot permit their use for private or commercial purposes if it results in environmental degradation. The judicial pronouncements, read alongside constitutional mandates such as Article 48A and Article 51A(g), establish that failure of the State to act as a responsible trustee can give rise to environmental liability. Thus, the public trust doctrine serves as a vital mechanism for holding governments accountable and integrating environmental protection into the core functions of governance.

4. PRINCIPLES OF CIVIL LIABILITY ALLOCATION APPLICABLE TO ENVIRONMENTAL LIABILITY

Environmental harm often involves complex factual situations, multiple actors, and overlapping responsibilities, making the allocation of civil liability a crucial aspect of environmental governance. To address these complexities, environmental law draws upon general civil liability allocation principles such as joint and several liability, equitable allocation, proportional liability, etc. These principles provide legal frameworks for determining how responsibility for environmental damage is shared among different actors, ensuring effective compensation, timely remediation, and fairness in burden-sharing.

4.1 Joint, Several and Joint & Several Liability

In the context of environmental liability allocation, the concepts of joint, several, and joint & several liability determine how responsibility for environmental harm is shared among multiple polluters or responsible parties. These doctrines are particularly important where contamination is caused by cumulative or overlapping activities, and where prompt remediation is a regulatory priority. The choice of liability model directly affects enforcement strategies, risk allocation, and post-remediation contribution claims between responsible parties.

Joint liability treats multiple parties as collectively responsible for a single environmental obligation. Each party is legally liable for the full extent of the environmental harm, but the obligation is discharged once remediation or compensation is completed by any one of them. Procedurally, enforcement actions often require that all jointly liable parties be brought before the authority or court together. While this approach reflects the collective nature of environmental harm, it may hinder effective enforcement if one or more responsible parties are insolvent, untraceable, or otherwise unable to participate in proceedings.

Under several liability, environmental responsibility is apportioned among parties according to defined criteria, such as degree of fault, volume of pollutants contributed, or duration of involvement. Each party is liable only for its allocated share of cleanup costs or damages, and compliance by one party has no impact on the obligations of others. This model promotes fairness and proportionality but can create enforcement challenges, particularly where environmental harm is indivisible or where some parties lack the resources to meet their share, potentially resulting in under-remediation.

Joint and several liability represents a hybrid approach and is widely adopted in environmental regulation due to its effectiveness in ensuring remediation. Each responsible party is individually liable for the entire environmental harm, while also being collectively liable with others. Regulators or claimants may pursue any one party, multiple parties, or all parties together for full recovery. This maximizes the likelihood that environmental damage will be promptly addressed, leaving issues of equitable cost-sharing to be resolved through contribution or indemnity claims among the responsible parties.

In environmental law, joint and several liability is often preferred where harm is indivisible, the polluters' contributions are difficult to quantify, or public health and ecological protection demand immediate action. Although it may result in one party bearing a disproportionate initial burden, this approach prioritizes environmental remediation while preserving private rights of recourse between parties to achieve a more equitable final allocation of liability.

4.2 Proportionate and Equitable Liability

Proportionate liability in environmental law is based on allocating responsibility according to a party's actual contribution to the overall environmental harm. Each polluter is assigned a share of liability that reflects factors such as the nature and extent of its activities, the quantity and toxicity of pollutants released, and the duration of its involvement (Gilead et al., 2013). This approach is grounded in fairness, as it avoids imposing the entire burden of remediation on a single party when harm has resulted from multiple sources. Importantly, in a proportionate liability framework, if one responsible party is unable to meet its share due to insolvency or other incapacity, the remaining parties' liabilities may be reassessed and redistributed in proportion to their contribution to the total harm, ensuring that environmental remediation is not stalled.

Equitable liability complements proportionality by allowing courts or regulators to adjust liability allocation to achieve a just outcome in light of the circumstances of each case. While proportional contribution remains the guiding principle, equity permits flexibility where strict apportionment would undermine effective remediation or lead to unjust results. This stands in contrast to several liability, where each party's share is fixed and cannot be increased even if another party defaults (Staab v. Diocese of St. Cloud) (1575 A12 1972 (2013)). Equitable and proportionate allocation, therefore strikes a balance between fairness and effectiveness: it ensures that each party is held responsible for its role in causing environmental damage, while also allowing liability to be redistributed where necessary to secure cleanup and restoration in the public interest.

4.3 Vicarious Liability

Vicarious liability, though applied more selectively than strict or absolute liability, plays an important supporting role in addressing environmental harm. The principle operates where a legally recognised relationship exists—such as employer–employee, principal–agent, or contractor–subcontractor—and the environmentally harmful act occurs in the course of employment or the performance of assigned duties. In such cases, liability is attributed not only to the individual who directly caused the harm but also to the person or entity exercising control or authority over that individual. This reflects the understanding that environmental damage often arises from organised industrial or commercial activities rather than isolated personal conduct.

In environmental contexts, vicarious liability ensures that organisations remain accountable for pollution or environmental damage caused by those acting on their behalf, even where the organisation did not directly engage in the wrongful act. By imposing responsibility at the institutional level, the principle discourages delegation of risky activities without adequate oversight and promotes better training, supervision, and compliance systems. Indian courts have implicitly recognised this approach by holding industries responsible for environmental harm arising from their operations (M.C. Mehta v Union of India ((1987) 1 SCC 395), Indian Council for Enviro-Legal Action v. Union of India, ((1996) 3 SCC 212), Vellore Citizens Welfare Forum v. Union of India, ((1996) 5 SCC 647)) while international regimes, such as under the U.S. Clean Water Act, similarly impose liability for violations committed by employees or contractors under a company's control (USEPA, 2002). Although vicarious liability does not displace strict or absolute liability, it reinforces environmental protection by closing accountability gaps and preventing entities from evading responsibility through organisational structures.

5. ENVIRONMENTAL LIABILITY IN COMMON LAW VS CIVIL LAW SYSTEMS

Environmental liability principles operate differently under the two major legal systems in practice - common law systems and civil law systems. Countries that were once under British rule, such as India, generally follow the common law system, while most European countries adhere to the civil law system. The key distinction between these systems lies in the source and development of law. In common law systems, statutes coexist with judicial decisions, and court rulings themselves constitute a source of law. This enables courts to actively shape environmental liability principles through interpretation and precedent. As a result, principles such as absolute liability, strict liability, and the polluter pays principle have evolved through judicial creativity, allowing the law to adapt to new environmental challenges even in the absence of codified rules. For instance, principles such as the "polluter pays" were part of the law of the land in India through judicial pronouncements even before they were explicitly incorporated into some of the later statutes. This flexibility allows common law systems to respond more rapidly to environmental challenges.

In civil law systems, by contrast, the law is primarily codified, and legal certainty is achieved through comprehensive statutes and codes enacted by the legislature. Environmental liability principles in such systems derive their authority from formal legislative recognition rather than judicial innovation. Courts play a more restrained role, focusing on the application and interpretation of codified provisions. While this may appear less flexible, it ensures clarity, uniformity, and predictability in the enforcement of environmental liability.

6. CONCLUDING REMARKS

The existence of clear rules and guidelines governing environmental liability is essential for any society seeking to effectively address environmental harm and hold polluters accountable. However, the mere availability of laws and guidelines based on these principles is not sufficient; equal importance must be given to the manner in which these laws are implemented and to the institutional mechanisms that support their enforcement. It is often argued that law operates reactively by imposing liability only after environmental damage has occurred, whereas long-term environmental protection requires proactive societal action focused on prevention rather than remediation. While such proactive measures are undeniably crucial, the deterrent effect of legal consequences cannot be overlooked, as the prospect of liability plays a significant role in discouraging environmentally harmful conduct. Well-conceived principles, their effective incorporation into law, proper enforcement, and proactive societal engagement together form the foundation of a robust and effective environmental redress mechanism, ensuring long-term environmental protection.

REFERENCES

California State Legislature, 2025. Polluters Pay Climate Superfund Act of 2025 (Senate Bill 684).

European Commission, 2022. Report from the Commission to the European Parliament and the Council: Application of EU health and environmental standards to imported agricultural and agri-food products. Brussels: European Commission.

European Parliamentary Research Service (EPRS), 2015. The precautionary principle: Definitions, applications and governance. Brussels: European Parliament.

Gilead, I., Green, M.D. & Koch, B.A. (eds.), 2013. Proportional liability: Analytical and comparative perspectives. Berlin/Boston: De Gruyter.

Hamm Higher Regional Court, 2025. Judgment in Lliuya v. RWE AG.

Ibbetson, D., 2003. How the Romans did for us: Ancient roots of the tort of negligence. University of New South Wales Law Journal, 26(2), pp.475–514.

Organisation for Economic Co-operation and Development (OECD), 1992. The polluter-pays principle: OECD analyses and recommendations (OCDE/GD(92)81). Paris: OECD.

Jowett, B. J. 2008. Translation of Laws by Plato. Book VIII. Project Gutenberg eBook No. 1750.

Rangarajan, L.N., 1992. Kautilya: The Arthashastra. New Delhi: Penguin Books.

Trilling, M., 2025. Make polluters pay: Proposal for a surtax on fossil fuel industries' profits. Global Alliance for Tax Justice (GATJ) & European Network on Debt and Development (Eurodad).

United Nations, 1972. Report of the United Nations Conference on the Human Environment, Stockholm, 5–16 June 1972. New York: United Nations.

USEPA, 2002. The United States Environmental Protection Agency. Federal Water Pollution Control Act (Clean Water Act), as amended through Public Law 107–303. Washington, DC: EPA.

USEPA & NYSDEC, 2013. The United States Environmental Protection Agency & New York State Department of Environmental Conservation. Love Canal Superfund site report prepared for the Honorable John T. Curtin, U.S. District Court for the Western District of New York (Report No. 534007).

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DETRITUS & ART / A personal point of view on Environment and Art

by Rainer Stegmann

The Garbage Story
Rainer Stegmann and Cristina van der Westhuyzen

By chance, one of us noticed an overflowing bin in a street in Venice and was drawn to the aesthetic of its colorful contents. A photo was taken, and thoughts were turned into words.

What is the relevance of sharing this photo and the reflections it inspired? Is it just an overflowing waste bin, or is there more to discover? We see it as a way to ignite curiosity and interest in the discarded object, the beauty it can reveal, and the powerful message it sends. Waste can inspire our imagination and uncover many stories behind it. Waste can become a source of creativity.

Viewed from the perspective of waste management, a close look at the bin tells us that several items are recyclable, such as the two plastic bottles, the can, and perhaps the single-use paper-and-plastic cups. The remaining waste materials may be difficult to sort and should be disposed of; due to their high calorific value, they should be incinerated. Some items in the bin could have been avoided: the single-use cups could have been replaced with reusable cups, the drink in the can could have been served in a deposit-return bottle, and plastic bottles could have been replaced with refundable ones.

Another often overlooked aspect of Waste Management is organizing waste collection in public spaces. Therefore, there should be enough trash cans, emptied regularly, to prevent littering. Key questions include how many to install, their capacity, and how frequently they should be emptied to avoid overflows, which often happen during events. This is a significant challenge due to high costs. Trash cans should have lids to prevent litter from being blown away by the wind, to deter animals, and to protect against rain. Plastic cigarette butts should be collected in a designated compartment inside the trash can to prevent them from ending up in the oceans. Although the trash container in Venice inspired us for the Garbage Story, the City of Venice may reconsider the design of the trash containers presented here.

"Every item in this garbage bin has a story to tell: plastic, paper and tin. The remains of a busy day in a city. Water bottles, a fizzy drink, an ice cream cup smudged in melted chocolate, an invitation to an evening of music and wine, a scrunched-up ticket. Was it for a museum? The trail of leisurely time spent strolling around in the warm sun. The air is vibrant with sound. The piazza is buzzing with many people, colourful and confused like the jumbled display in the bin. Happy memories will be taken but this will remain."

In the next volume of DETRITUS, I will present the artist Diedel Klöver, who lives in Varel, Germany. He creates wonderful sculptures from wasted metal items. In this example, these are screws and nuts.

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