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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 IWWG (International Waste Working Group), a non-profit organisation established in 2002 to serve as a forum for the scientific and professional community and to respond to a need for the international promotion and dissemination of new developments in the waste management industry.

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Editorial

SUPPORTING YOUNG RESOURCE MANAGEMENT PROFESSIONALS

Introduction

Society faces numerous pressing problems, one of which is the development of a sustainable approach to waste (resource) management. Modern society involves consumption on a massive scale, and this leads to massive resource use and massive waste generation. Enabling effective resource management requires advanced higher education (Williams, 2014) alongside appropriate infrastructure, service provision and active public engagement, and this is hugely challenging (Timlett and Williams, 2011). Many political, environmental, social, technological, legal, and economic approaches have been trialed, but only slow progress has been achieved.

Everyone was young once. Everyone will remember the frustrations of having ideas, often promising ideas, that simply did not get heard, for complex (and sometimes valid) reasons. Sometimes - because no-one was listening - clever ideas were not acted upon. What a waste!

Academic members of the waste management community are often older than those in other academic fields. Waste management has been a critical area of study for a long time, driven by the need to address public health and environmental issues (Williams, 2014). This long-standing focus means that pioneers and early researchers in the field are now very senior academics. The interdisciplinary nature of waste management, which combines elements of engineering, environmental science, public health, and policy, often attracts professionals who have long careers in both academia and industry. This can lead to a higher average age within the community, with its leading experts having decades of experience. Senior academics and companies that have contributed over time to the development of now well-established and reliable technologies are generally reluctant to consider entirely new and innovative ideas. However, the field of environmental engineering is still a relatively young sector compared to others, with significant potential for further innovation. It is thus crucial that we should do more with and for our younger colleagues, as their contributions could play a pivotal role in driving innovation within this sector. We should listen actively to their viewpoints and perspectives and provide more assistance with their careers. We should learn from them and support their progressive ideas with energy and vigor.

What can the young teach the old?

Young resource management researchers/professionals (henceforth "young RMs") have valuable insights they can offer to older waste management researchers/professionals ("older WMs"). Learning is a two-way street. Older

WMs can learn from their younger colleagues through a process known as "reverse mentoring" (Gadomska-Lila, 2020), creating a dynamic exchange of knowledge and experience. Here are some ideas:

- **Fresh Perspectives:** Young RMs can provide fresh perspectives and innovative ideas (Wagner and Compton, 2012). They can challenge established norms and encourage creative thinking. They can bring a freshness that encourages creativity and adaptability (Warnell, 2015).
- **Identifying Research Gaps:** Young RMs can identify research issues and questions that older WMs may miss or not prioritize. Their unique experiences and viewpoints allow them to spot gaps in existing knowledge.
- **Language Clarity:** Young RMs can help ensure that research tools (such as questionnaires, interview schedules, leaflets, and reports) are worded in a language that is clear and relatable to their peers. This clarity can improve communication and data collection and ensures wider reach and impact (Warnell, 2015).
- **Redressing Power Imbalances:** Involving young RMs in research and professional decision-making can help redress power imbalances between different age groups (Jolles M et al, 2022). It fosters more positive relationships and mutual learning.
- **Interdisciplinary Collaboration:** Young RMs often have interests and skills in emerging areas. Collaborating with them can lead to interdisciplinary approaches and novel solutions (Warnell, 2015).
- **Technology Savvy:** Young RMs are often adept at navigating the digital landscape; having grown up with computers and the Internet they tend to "have a natural aptitude and high skill levels when using new technologies" (Cross et al, 2010). However, Kent (2013) highlights that these young professionals, from lack of experience, often naively believe that articles promoting social media, how to employ it and its importance, contain "facts". Nevertheless, these so-called "digital natives" can teach older colleagues (so-called "digital immigrants") about emerging technologies, social media platforms, efficient digital tools, use of advanced tools, data analytics, and computational methods.
- **Adaptability:** In a rapidly changing world, young RMs can excel at adapting to new situations. They can share strategies for resilience and flexibility and excel at adapting to new methodologies, technologies, and research paradigms (Warnell, 2015). Their agility could help the resource (waste) sector stay responsive to changing needs.

- **Collaboration and Inclusion:** Young RMs typically understand the importance of diversity, inclusion, and collaboration (Warnell, 2015). They can foster a more inclusive work environment by bridging generational gaps and embracing collaboration across disciplines. Young RMs can teach older colleagues to break down silos and work together better for strong holistic solutions.
- **Work-Life Balance:** Balancing work and personal life is crucial and young professionals regard work-life balance as especially important (Mihelič & Aleksić, 2017; Smith, 2010). Young RMs can share insights on maintaining a healthy work-life balance.
- **Enthusiasm and Change:** Young people can infuse workplaces with energy and a desire to create positive change (Warnell, 2015). Their enthusiasm can catalyze innovation and disruption. They can be willing to challenge old systems and seek fresh solutions.
- **Entrepreneurial Thinking:** It was widely anticipated that millennials would be more entrepreneurial than previous generations (Liu et al, 2019). Whilst this is true of high-profile entrepreneurs such as Mark Zuckerberg, founder of Facebook, in fact, millennials have been less entrepreneurial than previous generations (Wilmoth, 2016). Nevertheless, young RMs can be innovative in their thinking (Twenge et al, 2010), can generate fresh ideas and can be willing to disregard traditional models to meet customer needs.
- **Flexibility, Speed and Technology:** Young RMs regard themselves as flexible and feel they excel at multi-tasking, leveraging innovative technology for efficiency (Folarin, 2021). This suggests they can quickly tackle tasks using high-tech shortcuts and resources, with this speed complementing older WMs' patience and experience.
- **Diverse Perspectives:** Young RMs can bring diversity in thought and experience (Folarin, 2021). Their backgrounds and viewpoints can enrich problem-solving and decision-making processes.
- **Storytelling:** Young people understand the power of narratives. They can teach older people to communicate complex ideas effectively and engage stakeholders.
- **Innovation and Creativity:** As digital natives, younger people can be innovative and creative in use of IT. They can build upon newer ideas and interdisciplinary studies, injecting fresh perspectives into research.
- **Communication Skills:** Young RMs understand the importance of IT-based communication. They can teach older WMs new and more effective ways to convey complex ideas to diverse audiences.
- **Lack of Funding:** Insufficient financial support poses a major obstacle for young RMs. Securing grants and funding for research projects can be highly competitive and discouraging. Experienced WMs often have established skills in planning, organizing, writing, and winning research- and private sector funding.
- **Gender Bias:** Gender disparities persist in the WM community. UNEP (2022) reports that "gender inequalities and norms are embedded in almost every aspect of waste management." Young female RMs often face bias, unequal opportunities, and barriers to advancement. Addressing gender inequities is crucial for fostering diversity and inclusion.
- **Mentorship Gap:** Access to effective mentorship is essential for professional growth since mentoring is one of the most effective ways of helping individuals to develop (Stoeger et al, 2021). Some young RMs struggle to find experienced mentors who can guide them through professional challenges and career decisions.
- Although young researchers often bring brilliant and innovative ideas, these ideas frequently require the insight and expertise of senior researchers to identify potential challenges and refine them, ensuring they are well-adjusted and capable of achieving a promising future.
- **Research and Writing Skills:** Developing strong research and writing skills is critical. Young RMs may require assistance to overcome difficulties in effectively communicating their findings, navigating the intricacies of professional and scholarly writing, and getting published.
- **Interest from Policymakers:** Policymakers' engagement with the resource (waste) management community, especially with research outcomes, is vital for driving positive change. However, young RMs often face challenges in capturing policymakers' interest and translating research into actionable policies.
- **Heavy Workload:** Balancing research, coursework, and other responsibilities can be overwhelming. The demanding workload leaves little time for e.g., professional development; networking; focused research efforts. Older WMs can provide tips on managing time, responding to the inevitability of change, balancing activities, the value of learning from mistakes, and being persistent.
- **Dual Narrative:** Young RMs may struggle with reconciling their cultural backgrounds and experiences with dominant sectoral narratives. Bridging this gap is essential for meaningful contributions from young RMs and older role models can help in this regard. Kricorian et al (2020) have highlighted the importance of being mentored by those of the same gender and ethnicity, either in person or through media.
- **Mobility:** Opportunities for international collaboration and mobility are crucial for scientific and professional growth. However, barriers related to visas, travel, and cross-border research can hinder young RMs' mobility.

Learning is a continuous process. All generations of resource (waste) managers will benefit from mutual exchange of knowledge and experience.

How can the old help the young?

Young RMs will encounter challenges as they embark on their scientific journeys. These barriers can significantly impact their research, professional development, and career progression. Young RMs can be supported and encouraged by older colleagues. Here are challenges that older WMs can help younger RMs to overcome:

Support for young professionals

Addressing the challenges faced by young RMs is fundamental for fostering a vibrant and inclusive community. There are many strategies we can employ to support and empower them; here are a few ideas:

- **Increased Funding Opportunities:** Advocate for increased funding from institutions, governments, and private organizations specifically for young RMs. Financial support enables young RMs to pursue research and professional development without unnecessary constraints.
- **Promote Equity:** Combat gender and other bias by creating awareness and implementing policies that ensure equal opportunities for all WMs, regardless of gender, age, or ethnicity. Encourage mentorship and networking for young RMs, especially by those of their same gender and ethnicity.
- **Effective Mentorship Programs:** Establish robust mentorship programs that connect older, experienced WMs with young RMs. Mentorship provides guidance, career advice, and emotional support during critical stages of a young RM's professional journey.
- **Global Collaboration and Mobility:** Facilitate international collaboration by promoting cross-border research. Exposure to diverse environments enhances young RMs' skills and perspectives.
- **Skill Development Workshops:** Organize workshops on research and professional methodologies, scientific writing, and communication skills. These enhance young RMs' abilities to present their work effectively.
- **Promote Interdisciplinary Research:** Encourage collaboration across WM-related disciplines. Interdisciplinary projects foster creativity, innovation, and holistic problem-solving.
- **Recognize Non-Traditional Career Paths:** Acknowledge that WM careers can take various forms. Highlight opportunities in industry, policy, academia, communication, and entrepreneurship.
- **Emphasize Mental Health and Well-Being:** A demanding workload and competitive environment can take a toll on young RMs. Encourage work-life balance and self-care.
- **Advocate for Inclusive Research Environments:** Create spaces where diverse voices are heard. Inclusion fosters creativity, resilience, and innovation.

The resource sector has a leading role in enabling the transition to a sustainable economy. Our industry needs young professionals who are innovative; creative; superb communicators; scientifically, mathematically and IT literate; and who engage with problems in a systematic, "can-do" fashion. Supporting young RMs is crucial for fostering a vibrant community. Investing in young RMs today will ensure a brighter future tomorrow.

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TOWARDS A CIRCULAR ECONOMY: A COMPREHENSIVE POLICY FRAMEWORK FOR PACKAGING WASTE MANAGEMENT IN THE EUROPEAN UNION

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ABSTRACT

Directive 94/62/EC on packaging and packaging waste (PPWD) aimed at harmonizing regulations and coherence of the European Union (EU) internal market and preserving the environment and natural resources. Despite its economic importance, packaging production poses significant environmental challenges, including the generation of solid waste and the depletion of natural resources. This paper addresses the current policy framework that the EU intends to implement to minimize the growing generation of packaging waste, promote circularity, and integrate recycled materials in the EU. The proposed regulatory measures, including the transition from a directive to a Regulation, the establishment of mandatory targets for the reduction of solid waste and recycled content, and the strengthening of reuse systems, aim to promote a resilient value chain, mitigate environmental degradation, and stimulate economic growth. By adopting a circular economy approach, the EU strives to champion sustainable development in the internal market and improve social well-being in various sectors.

1. INTRODUCTION

Packaging is essential in today's commercial activity to protect goods during transport, storage, and consumption. However, the rapid expansion of packaging production and use poses significant environmental challenges, which require a paradigm shift in regulations to ensure sustainability and mitigate adverse environmental impacts. Directive 94/62/EC, which regulates packaging and packaging waste (PPWD), constitutes one of the fundamental pieces of the European Union's (EU) commitment to harmonizing and regulating the sector, preserving the environment, and promoting coherence in the international market.

Recognizing the complexity between economic relevance and environmental sustainability, the EU seeks to reconcile these two divergent aspects. Only the EU plastics value chain generated a turnover of more than €400 billion in 2021 (Plastics Europe, 2023), most of that in the packaging sector. This symbolizes the economic vitality of the market while representing a significant environmental challenge. From resource depletion to ecosystem contamination and greenhouse gas emissions, the environmental effects of the proliferation of packaging are very substantial (Zhuo et al., 2023).

The EU legislative approach to packaging waste management incorporates a holistic vision that aims to reconcile economic issues with the interests of safeguarding environmental values, for example, the Circular Economy Action Plan (Resolution P9_TA(2021)0040). The essential requirements stipulated by Directive 94/62/EC entailed strict standards for packaging manufacturing and sustainability criteria, underlining the EU's commitment to promoting a circular economy. However, despite these regulatory efforts, the truth is that packaging waste generation continues to increase, waste circularity is negligible, and the integration of recycled materials requires further improvements (Tallentire and Steubin, 2020; Ncube, 2021). Using recycled materials in packaging is an essential strategy, but additional improvements are needed in the collection, sorting, and recycling processes (Eygen et al., 2018; Balwada et al., 2021). The transition to a circular economy, particularly in the plastic packaging industry, is crucial and requires a holistic approach involving all stakeholders (Billiet and Trenor, 2020).

The current EU initiative to transition from a Directive to a Regulation comes from regulatory gaps, market inefficiencies, and uneven applications across member states, which have consequently worsened the packaging waste problem with far-reaching environmental consequences.



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Therefore, there is an urgent imperative for comprehensive regulatory action beyond mere application, requiring fundamental legislative reforms so that economic expectations can best be met without neglecting environmental aspects and ecological sustainability.

On the other hand, we seek to align the new Regulations on packaging and packaging waste with the objectives of the European Ecological Pact and the Circular Economy Action Plan to make the EU legislative framework compatible and modernized (Foschi and Bonoli, 2019; Stankevičius et al., 2020). The EU thus seeks to facilitate the transition to a circular economy in the packaging sector by harmonizing regulations, defending sustainable practices, and promoting stakeholder engagement. This article describes the challenges, policy objectives, and regulatory measures proposed to mitigate packaging waste, promote circularity, and promote sustainable development in the EU.

2. CHALLENGES IN PACKAGING AND PACKAGING WASTE MANAGEMENT

2.1 Packaging waste generation

Effective packaging waste management presents several challenges to the EU, especially with the exponential increase in packaging consumption and production (Foggia and Beccarello, 2022). These challenges include a very significant increase in the generation of packaging waste, circularity hampered by the characteristics of some types of packaging, inadequate integration of recycled materials, and regulatory and market inefficiencies (Vogt et al., 2021).

From 1990-2015, there was an average 26% reduction in unit weight. However, variations exist within material groups, often due to marketing reasons. E-commerce exacerbates over-packaging, with excess outer packaging and space. While light-weighting reduces weight, there is a shift to materials with worse environmental footprints, from metal/glass to plastic/paper. Reuse of packaging, mainly primary and tertiary, has decreased over the past decades, with exceptions in specific beverage sectors. One-way packaging, significantly single-use plastic, is rising due to evolving retail practices. This trend will likely continue despite recent signals of a slowing decline in reusable packaging (Coelho et al., 2020; Diprose et al., 2023).

Some EU member states are promoting reuse through targets, green procurement, and Extended Producer Responsibility schemes, but challenges to internal market integrity and proportionality exist.

The EU faces a significant challenge in managing the unstoppable production of packaging waste, particularly in materials such as plastic and paper. Factors contributing to this increase include the widespread adoption of disposable packaging, driven by social trends and the proliferation of e-commerce activities. The onset of the COVID-19 pandemic further boosted this trend, as the increased demand for packaged products led to an unprecedented increase in the production of packaging waste (Jacobs et al., 2022). Consequently, the EU faces a growing waste crisis with environmental and socio-economic implications. According to Eurostat statistical data, the evolution of per capita production of packaging waste in EU countries went from 154 kg in 2010 to 188.7 kg in 2021, and the quantity of recycled material from 98.5 to 120.7 kg in the same period (Figure 1).

This quantity varied between 73.8 kg per inhabitant in Croatia and 246.1 kg per inhabitant in Ireland. By packaging material, for 2021, the distribution of the most common materials in the EU are paper and cardboard (40.3%), plastic (19.0%), glass (18.5%), wood (17.1%), and metal (4.9%). Other materials represented only 0.1% of the total volume of packaging waste generated in 2021 (Figure 2). In 2021, the EU generated an estimated total volume of 84.3 million tons of packaging waste – an increase of 6.0% (or 4.8 million tons) compared with 2020. The growth in 2021 was mainly due to the rise in waste from packaging made of wood by 2.3 million tonnes (+19.0%). There was also a notable increase in paper and cardboard packaging waste compared with 2020, up 1.3 million tonnes (+3.9%). The increases in waste material were smaller for packaging made of plastic (up 598 thousand tonnes; 3.9%), glass (up 467 thousand tonnes; 3.1%), and metal (up 181 thousand tonnes; 4.5%). Over the 11 years from 2010 to 2021, paper and cardboard were the primary packaging waste material, contributing 34.0 million tonnes to the total packaging waste generated in 2021; since 2010, this waste stream has grown by 23.5%. Plastic packaging reached a total of 16.1 million tonnes as the second most crucial packaging

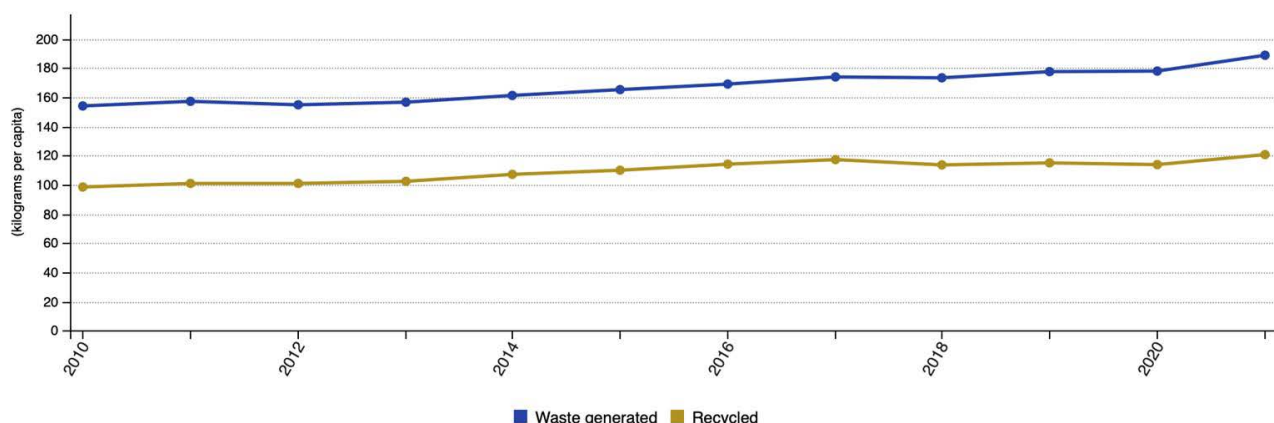


FIGURE 1: Evolution of packaging waste production and recycling in EU (2010-2021). Source: Eurostat (online data code: env_waspac).

material (+29.9% compared with 2010). Glass packaging waste had a volume of 15.6 million tones (+18.1%), wood 14.4 million tones (+32.7%) and metal packaging 4.2 million tones in 2021 (+10.6%).

In conclusion, the EU faces various challenges in effectively managing packaging waste, ranging from increasing waste production to hampered circularity and inadequate integration of recycled materials. Regulatory and market inefficiencies further aggravate these challenges, requiring comprehensive and decisive action to achieve a good relationship between the economic development of the packaging sector and environmental sustainability. By identifying and addressing these challenges, the EU can pave the way for a more sustainable and resilient future supported by circularity, efficient use of resources, and environmental management and protection.

2.2 Packaging and packaging waste problems

The escalation in non-recyclable packaging from 2012 to 2020 has posed significant challenges to recycling efforts. Despite the technical recyclability of many packaging types, collection, sorting, and demand issues in end markets have led to incineration, landfilling, or exportation as standard disposal methods. The prevalence of lightweight materials like multi-layer flexible packaging further complicates matters, as re-processors often reject them due to inadequate incentives, exacerbating environmental litter.

Notably, plastic packaging presents a formidable challenge, with 44% encountering design issues affecting recyclability. Although the EU reports a 40.6% recycling rate for plastic packaging, a mere 14% is effectively recycled when the waste exported outside the EU was not considered (25% when assuming that all the waste exported was also reprocessed), highlighting significant environmental repercussions such as increased greenhouse gas emissions, re-

source depletion, and biodiversity pressure (Antonopoulos et al., 2021).

Efforts to promote circularity in the packaging sector encounter barriers such as material types and design complexities, hindering efficient recycling and reuse. Design features impeding recyclability, cross-contamination of recycling streams, and hazardous substances in packaging pose considerable challenges to achieving circularity objectives. Recycling rates have stagnated, with many Member States at risk of failing to meet recycling targets set by current legislation, perpetuating the linear “take-make-dispose” model (Vogt et al., 2021; Hondroyiannis et al., 2024). The inadequate integration of recycled materials into the packaging value chain perpetuates reliance on virgin resources, exacerbating pressure on natural resources. Market dynamics, regulatory inconsistencies, and logistical challenges impede the widespread adoption of recycled materials in packaging production, hindering the EU’s sustainability objectives (Antonopoulos et al., 2021).

Regulatory and market inefficiencies exacerbate challenges associated with packaging waste management in the EU, leading to legislative fragmentation and suboptimal waste management practices. Additionally, the demand for bio-based and compostable plastics has surged, leading to confusion regarding end-of-life management due to inconsistent labeling practices across the EU. Hazardous substances in packaging materials pose risks to human health and the environment, undermining confidence in recycled materials (Geueke et al., 2023). Waste sorters and recyclers express dissatisfaction with packaging producers’ choices, complicating waste treatment processes and limiting potential economies of scale. For example, packaging producers often design products focusing on aesthetics, durability, or cost-efficiency without fully considering how easily the materials can be sorted and recycled. This also

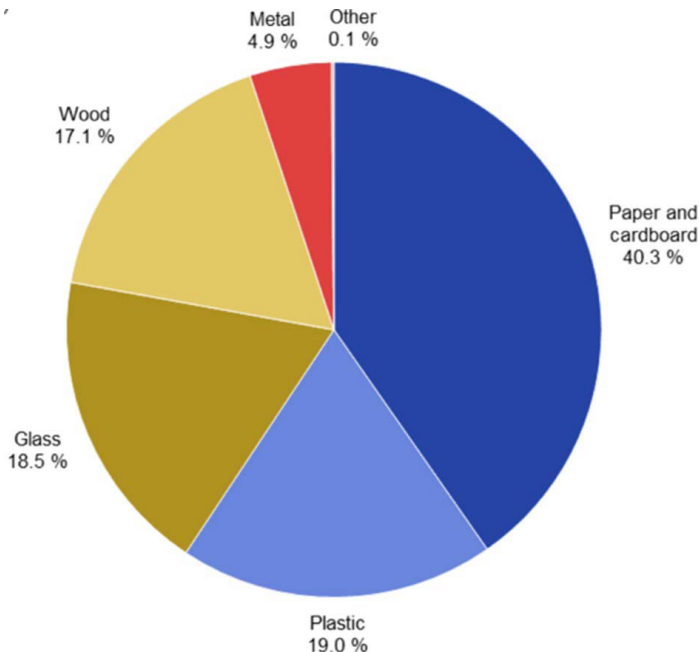


FIGURE 2: Waste generation by packaging material in 2021 in the EU.

might include using non-recyclable plastics, multi-layer packaging, or materials that are difficult to separate. When packaging is not designed with recyclability, the sorting process is more complex, time-consuming, and less efficient. Waste treatment processes become harder when packaging materials are diverse and not standardized. Economies of scale in recycling occur when there is a large quantity of uniform materials to process, making recycling more cost-effective. However, when there is too much variation in packaging materials, recyclers cannot efficiently process large batches of the same material. As a result, recyclers may miss opportunities to save on costs or achieve higher profitability through large-scale processing (Bassi et al., 2020; Rutkowski, 2020; Colelli et al., 2022).

A clear legal framework incentivizing circularity risks is needed to deter investment in recycling technologies. Moreover, confusion persists among consumers regarding waste sorting due to diverse labeling practices, hindering correct waste separation and recycling efficiency. The current predominance of downcycling in waste recycling and varying uptake rates of recycled material in packaging underscores the need for comprehensive strategies to promote circularity and mitigate environmental impacts. These challenges jeopardize the EU internal market, impede the transition to a circular economy, and generate adverse environmental consequences. Addressing these issues requires harmonizing regulations, enhancing circularity in packaging design and production (ecodesign regulations), and implementing sustainable waste management practices. Firstly, these problems negatively affect the functioning of the internal market by posing challenges to the free circulation of packaged goods. Divergent legislations across Member States lead to additional costs for producers to adapt, hinder economies of scale, and impede investment in reprocessing capacity and innovation.

Fragmentation in the market also results in efficiency losses in the supply chain due to reduced demand and additional stock holding. Secondly, the demand for packaging, coupled with low reuse, recyclability, and recycled content, hinders progress towards a circular economy. Ambiguities in the definition of recyclable packaging and other barriers impede Member States' ability to meet recycling targets, increase the use of recycled materials, and create jobs in recycling sectors. Thirdly, the manufacture of packaging has significant adverse environmental impacts, including carbon emissions and litter pollution (Ncube et al., 2020; Pilli et al., 2021).

Littered plastics, especially in marine environments, pose risks to ecosystems and human health as they degrade into micro and nano plastics and enter the food chain (Mofijur et al., 2021; Gündoğdu et al., 2024). Greenhouse gas emissions from packaging are projected to increase unless measures are taken to reduce them, undermining the EU's goal of carbon neutrality by 2050 (Beghetto et al., 2023). Additionally, packaging waste contributes to soil and land pollution and marine littering, exacerbating environmental degradation. In summary, addressing these issues requires comprehensive strategies that promote harmonization of regulations, enhance circularity in packaging design and production, and mitigate environmental

impacts through sustainable practices and waste management.

2.3 Packaging waste impacts

The multifaceted impacts of packaging and packaging waste on society, consumers, brands, manufacturers, and waste management companies include environmental pollution, resource depletion, emissions, economic loss, and public health risks. Consumers need more transparent and harmonized information about packaging options, hindering their ability to make informed decisions and manage packaging waste effectively. Brands must balance packaging functions with increasing consumer demand for environmentally conscious solutions amidst regulatory uncertainties. Packaging manufacturers must meet brand demands and regulatory requirements while maintaining competitiveness through innovation and cost-efficiency. Waste management companies and recycling industries need more profitability due to fragmented markets and regulatory gaps, calling for a harmonized economic framework and legal clarity to improve coordination and efficiency in waste management.

Environmental degradation is at the forefront of the consequences of insufficient packaging waste management. The proliferation of packaging materials, often derived from non-renewable resources, exacerbates resource depletion and contributes to ecosystem contamination. Improper packaging waste disposal, including landfilling, leads to visual pollution and habitat destruction, disrupting delicate ecological balances. Plastic packaging poses a significant threat to marine environments, where it fragments into nano-microplastics (Boone et al., 2023), infiltrating marine food chains and causing harm to aquatic life. Additionally, packaging waste incineration releases greenhouse gases and pollutants into the atmosphere, exacerbating climate change and air pollution (Zhuo et al., 2023).

The economic consequences of inadequate packaging waste management are multiple, including significant costs for municipalities, companies, and taxpayers, as resources are diverted to waste collection, transport, and disposal (Deniz et al., 2023). Furthermore, the reliance on virgin materials for packaging production perpetuates a linear economic model characterized by resource extraction, production, consumption, and disposal, leading to missed resource efficiency and cost savings opportunities (Zhu et al., 2022). Moreover, the environmental externalities associated with packaging waste, such as pollution clean-up and health impacts, impose additional financial burdens on society, undermining the economic resilience and competitiveness of the EU (Dechezleprêtre and Sato, 2017).

The health consequences of inadequate packaging waste management extend beyond environmental degradation to encompass direct risks to public health. Improperly disposed packaging waste, particularly plastics, can leach harmful chemicals and additives into soil and water, contaminating food and water supplies and posing risks to human health. Moreover, the accumulation of packaging litter in urban and natural environments creates breeding grounds for pests and disease vectors, increasing the

prevalence of vector-borne diseases and compromising public health and sanitation standards (Wright and Kelly, 2017; Krystosik et al., 2020). In addition to environmental, health, and economic consequences, inadequate packaging waste management engenders social disparities and inequities. Vulnerable communities, including marginalized populations and low-income households, bear a disproportionate burden of the negative externalities associated with packaging waste, such as exposure to environmental pollution and health hazards.

In summary, the consequences of inadequate packaging waste management within the EU are far-reaching and profound, encompassing environmental degradation, economic losses, public health risks, and social disparities. Addressing these consequences necessitates concerted efforts to transition towards sustainable packaging practices, promote circular economy principles, and enhance waste management infrastructure and governance mechanisms. By mitigating the adverse impacts of packaging waste, the EU can foster environmental sustainability in the internal market, promote economic prosperity, and safeguard public health and well-being for present and future generations.

3. POLICY OPTIONS DESCRIPTION

3.1 General description

A new regulatory approach to address shortcomings in current legislation governing packaging and packaging waste management is necessary. Despite existing controls, environmental sustainability objectives have yet to be achieved. Regulatory failures, such as delayed transposition and uneven application, reveal the necessity for harmonization through Regulation rather than revising the current directive. A comprehensive set of regulatory solutions is proposed to promote a transition to a low-carbon and circular economy. Placing obligations on economic operators ensures consistent implementation across Member States, addressing the challenge of uneven national regulatory frameworks. A Regulation would provide legal certainty, reduce competition distortion, and send clear signals to non-EU market actors. Implementing measures can be swiftly developed to flesh out the Regulation further. This approach aligns with industry calls for harmonization and political commitments outlined in the European Green Deal and the Circular Economy Action Plan to ensure all packaging is reusable or recyclable by 2030. The EU faces complex challenges in managing packaging waste while also seeking to promote a circular economy. In the legislative change process in the EU, several policy options were proposed to face these challenges, each offering different approaches to promote the reduction of packaging waste production, its circularity, and the integration of recycled materials in the packaging sector (Farmer, 2020). Extensive analysis, stakeholder consultations, and applicability and effectiveness considerations inform these policy options. The first set of options focuses on standardizing and clarifying the essential requirements for packaging, establishing the basis for subsequent measures. The Essential Requirements would be revised and expanded to include clear definitions

and performance criteria to minimize excess packaging and promote recyclability. By providing regulatory clarity and consistency, this option seeks to increase the effectiveness of packaging minimization requirements and facilitate compliance across Member States. Although this option may involve minimal administrative and economic costs, its main objective is establishing a robust regulatory framework that promotes waste reduction and circularity as, for example:

- **Minimization of space in packaging:** Sets maximum void space limit to reduce packaging waste in various sectors.
- **Update of Essential Requirements:** Defines over-packaging and specifies packaging performance criteria to minimize excessive packaging.
- **Revision of CEN standard for reusable packaging:** Updates standards for defining and improving reusable packaging.
- **Clarity on the definition of reuse activity:** This section clarifies when reusable packaging becomes waste to facilitate reuse systems.
- **Packaging to be reusable or recyclable by 2030:** Establishes enforceable requirements prioritizing reusable packaging to be recyclable by 2030: Requires recyclability for reusable packaging.
- **Qualitative definition of recyclable packaging:** Defines recyclable packaging to improve recycling rates.
- **Updates of Essential Requirements and EN 13432 - Biodegradable Packaging Certification:** Clarifies biodegradability and compostability concepts in packaging.
- **Allowing compostable and conventional plastics for selected packaging types:** This policy permits compostable materials for specific packaging.
- **Definition of Recycled Content and Measurement Method:** Establishes methodology for calculating and verifying recycled content levels in packaging.
- **Update of current material-based labeling:** This update removes alphanumeric codes for waste sorters and introduces more competent sorting labeling requirements.
- **Update of 'hazardousness' definition:** Reviews packaging legislation objectives and updates hazardous substance requirements.
- **Expanding information on hazardous substances:** Identifies and prioritizes relevant dangerous substances in packaging for potential limitations or restrictions.

The second set of options introduces mandatory targets for waste reduction, reuse, and minimum recycled content in plastic packaging, along with requirements for full recyclability by 2030 and harmonized product rules. These goals encourage economic operators to adopt sustainable packaging practices, minimize packaging waste, and integrate recycled materials into their production processes. These measures seek to drive systemic changes in packaging design, production, and consumption, establishing clear objectives and deadlines and promoting a transition to a circular economy. While these options may entail implementation challenges and economic adjust-

ments for the packaging sector, their ambitious goals reflect a commitment to advancing sustainability objectives and promoting resource efficiency. These measures seek to accelerate progress toward circularity by imposing more stringent targets and requirements on economic operators, such as:

- **Definition of Recyclable Packaging:** This measure establishes criteria for packaging design and recyclability assessment procedures. It aims to increase recycling rates and improve recycled material quality.
- **Harmonization of Extended Producer Responsibility (EPR) Fee Modulation Criteria:** This involves harmonizing EPR fee modulation criteria to incentivize improved packaging design and recycling. It aims to make recycling targets more efficient and cost-effective.
- **Mandatory Compostability for Certain Plastic Packaging Types:** Lightweight plastic carrier bags, tea bags, and labels on fruits and vegetables must be made from compostable plastic polymers. This measure aims to reduce waste contamination and increase the usage of compostable plastic.
- **Broad Targets for Plastic Packaging Recycling:** Mandatory targets for postconsumer recycled content in plastic packaging for 2030 and 2040 are set to increase the use of recycled content and reduce environmental impacts.
- **Mandatory Deposit Refund Systems (DRS) and Minimum Requirements:** DRS systems for plastic bottles and metal cans are made compulsory, with voluntary systems for other beverage containers. This measure aims to increase waste collection and purity rates.
- **Harmonized Labelling of Products and Waste Receptacles:** Uniform labeling systems are introduced to facilitate waste sorting and harmonize collection systems across the EU.
- **Restrictions on Confusing Labels:** This measure restricts how information on packaging is communicated to reduce consumer confusion.
- **Labelling Criteria for Recycled Content:** Standardized labeling requirements for recycled content in packaging are introduced to prevent misleading claims.
- **Harmonized Labelling for Reusable Packaging:** A mandatory symbol for reusable packaging is introduced to promote reuse and prevent fragmentation of labeling schemes.
- **Notifying Substances of Concern in Packaging:** Legal obligations have been introduced to inform substances of concern in packaging to enhance safety and transparency.
- **Mandatory Minimum Green Public Procurement Criteria for Packaging:** Mandatory minimum criteria for environmentally friendly packaging in public procurement are established to promote sustainability.
- **Harmonization of EPR Reporting System for Packaging Producers:** Reporting requirements for Extended Producer Responsibility systems are harmonized to streamline data processing and reporting.
- **Reporting Obligation on Plastic Carrier Bags:** To ensure comprehensive data collection, reporting requirements

for plastic carrier bags are extended to include bags of different thicknesses.

Finally, a third set of options represents a more ambitious approach, covering higher mandatory targets for waste reduction, reuse and recycled content, and additional product requirements. These measures can produce more significant environmental benefits and resource savings. However, they can also imply implementation burdens and higher economic costs for industries in the sector. However, the potential long-term benefits of these measures, in terms of reduced environmental impacts and greater resource efficiency, justify their consideration as a policy option. The key measures are:

- **Mandatory high-level targets to increase packaging reuse and refill by 2030/2040 in selected sectors.** The targets aim to reduce packaging waste and increase reusable packaging, with estimated economic impacts and environmental benefits.
- **Banning the heaviest packaging for selected items by 2030 based on existing lighter alternatives.** This measure initially targets plastic and glass bottles and aims to reduce overall waste generation.
- **Establishing a mandatory target of a 23% reduction in packaging waste per capita by 2030 compared to the baseline.** This measure aims to reduce waste and has significant economic and environmental impacts.
- **Quantitatively defining recyclable packaging based on actual recycling rates.** This measure aims to increase recycling rates, but its implementation needs to be improved due to technological limitations.
- **Mandating compostability for selected plastic packaging types.** While this measure eliminates consumer confusion, it poses challenges for the packaging industry and faces opposition from waste management sectors and Member States.
- **Introducing higher ambition and broad targets for recycled content in plastic packaging by 2030 and 2040.**

This measure aims to increase recycling rates but may need help due to tighter market conditions. Including prioritized use of recycled packaging from Deposit Refund Systems (DRS), waste collection schemes alternative to DRS, notification of all substances in packaging, restrictions of substances under the reviewed PPWD, mandatory minimum Green Public Procurement (GPP) criteria for packaging, and compulsory reporting requirement for recycled content for all packaging. These measures aim to support recycling, reduce environmental impact, and ensure packaging sustainability.

3.2 Sustainable Packaging Regulation

The proposal for a Regulation of the European Parliament and of the Council is made up of twelve chapters and 64 articles that can generally be described as follows:

- **Chapter I (Articles 1 to 4)** outlines the general provisions and defines the regulation's scope, including requirements for the entire packaging lifecycle, environmental sustainability, labeling, extended producer

responsibility, and packaging waste management. The regulation aims to facilitate the placement of packaging on the market while addressing environmental concerns and promoting the transition to a circular economy. Specifies that the regulation applies to all packaging and packaging waste types, irrespective of the material used. Provides definitions necessary for understanding the regulation, including terms adopted from existing legislation and new definitions related to recyclable packaging, recycled content in plastic packaging, reusable packaging, and packaging waste prevention. Establishes the principle of free movement within the single market for packaging that meets the sustainability and labeling requirements outlined in the regulation.

- Chapter II (Articles 5 to 10) addresses the regulation details sustainability requirements for packaging and imposes restrictions on the concentration levels of certain substances in packaging, such as lead, cadmium, mercury, and hexavalent chromium. The Commission can adjust these restrictions and grant exemptions through delegated acts. Mandates that packaging must be recyclable, with specific criteria to be met in two stages. By January 1, 2030, packaging must comply with design criteria for recycling, and by January 1, 2035, it must also meet requirements for effective collection, sorting, and recycling ("recycled at scale"). The Commission will establish criteria and methodologies for assessing recyclability and recycling scale through delegated acts. Financial contributions by producers for extended producer responsibility will be based on recyclability performance grades, which will include thresholds for recyclability at scale from 2035 onwards. Innovative packaging and specific medical device packaging are granted exemptions until 2034. Stipulates that plastic packaging must contain a minimum amount of recycled content from postconsumer plastic waste by January 1, 2030, with increasing amounts required by January 1, 2040. The Commission will establish a methodology for calculating and verifying recycled content percentages and may adjust the minimum percentage through delegated acts. Defines conditions for compostable packaging and specifies certain items, such as filter coffee pods and sticky labels on fruits and vegetables, that must be compostable within 24 months of the regulation's entry into force. Other packaging, except for lightweight plastic carrier bags, must be eligible for material recycling. The Commission can amend the list of compostable packaging through delegated acts. Mandates minimizing packaging weight and volume while ensuring safety and functionality, with compliance demonstrated through technical documentation. Outlines requirements for reusable packaging, including being designed for multiple uses and part of a reuse system meeting specified conditions outlined in Annex VI of the regulation.
- Chapter III (Articles 11 to 13) of the regulation focuses on labeling, marking, and information requirements for packaging. Article 11 mandates that packaging contain a label containing information about its material com-

position to aid consumer sorting. Similar labels should be placed on waste receptacles to assist consumers in identifying appropriate disposal routes. Harmonized labels may also include information about recycled content in plastic packaging, and reusable packaging should feature a QR code or other data carrier facilitating reuse. The Commission has the authority to establish harmonized labeling requirements and formats, including digital marking technologies.

- Chapter IV (Articles 14 to 28) outlines obligations for economic operators, including provisions regarding information exchange between suppliers and manufacturers, maximum space in packaging, and restrictions on specific packaging formats and purposes. Economic operators must ensure systems for reuse are in place for reusable packaging and comply with refill requirements for products offered for purchase through refill stations.
- Chapter V (Article 29) limits the annual consumption of lightweight plastic carrier bags and provides exemptions for certain types of bags.
- Chapter VI (Articles 30 to 34) addresses conformity assessment procedures for packaging.
- Chapter VII (Articles 35 to 51) focuses on managing packaging and packaging waste, including requirements for Member States to designate competent authorities, establish waste management plans, and progressively reduce packaging waste generation. It also mandates the establishment of producer registers, extended producer responsibility, and deposit and return systems for specific single-use packaging.
- Chapter VIII (Articles 52 to 56) covers safeguard procedures.
- Chapter IX (Article 57) allows for the adoption of delegated acts to establish green public procurement requirements.
- Chapter X (Articles 58 and 59) includes delegated and implementing acts provisions.
- Chapter XI (Articles 60 and 61) contains amendments to existing regulations.
- Chapter XII (Articles 62 to 64) contains final provisions, including penalties, an evaluation of the regulation after eight years, and transitional provisions.

4. RESULTS AND DISCUSSION

Considering the application of the proposed Regulation of the European Parliament and the Council, environmental impacts result mainly in reducing greenhouse gas (GHG) emissions and water use. Thus, by 2030, there could be a reduction of 1.1 million tons of CO₂e emissions and a decrease in water use by 31,000 m³. Considering the perspective that all packaging must be reusable or recyclable, such a change could be responsible for significant savings, which could result in a reduction of around 812,000 tons of CO₂e emissions and a reduction of 29,000 m³ in water use due to the decrease in the amount of packaging waste sent for incineration. On the other hand, diverting packaging waste from incineration and landfills for recycling could save 550,000 tonnes of CO₂e emissions and

reduce water use by 18,000 m³. Given its current absence, the Commission's definition of recyclable packaging is crucial. Although the impact of introducing only a qualitative definition on recycling rates is not globally significant, certain types of packaging could see an increase in recycling rates of 3 to 5%. This change could generate savings of 1.2 million tons of CO₂e emissions and a reduction of 44,000 m³ in water use. However, while recycling can reduce greenhouse gas emissions and resource depletion, mechanical recycling may increase microplastic emissions (Gabisa et al., 2023). MNPs can enter the food web, affecting marine organisms and potentially human health through cytotoxicity and immune responses (Mofijur et al., 2021; Yee et al., 2021). Advanced imaging techniques have shown direct evidence of nanoplastic release from recycled plastics (Zhang et al., 2020). MNPs can interact with other contaminants, increasing toxicity and potentially biomagnifying in the environment (Singh et al., 2021). Current research gaps include developing quantification techniques, understanding degradation mechanisms, and creating sensors for environmental detection (Singh et al., 2021).

Table 1 describes the sectors and types of packaging and products with the most significant economic and environmental impacts with the targets for 2030 and 2040 on the new Regulation. The adoption of reusable packaging is expected to result in reductions in GHG emissions of 1.25 million tons of CO₂e in 2030 (5.54 million tons of CO₂e in

2040) and a decrease in water consumption of 69,000 m³ in 2030 (212,000 m³ in 2040). The progressive elimination of avoidable or unnecessary packaging suggests a reduction of 2.18 million tons of CO₂e emissions and 91,000 m³ of water use. Measures such as reducing space, phasing out unnecessary packaging, and reuse targets contribute to almost 60% of waste reduction. EU member states are tasked with lowering absolute packaging waste per inhabitant, targeting a 19% reduction compared to the 2030 baseline, equivalent to a 5% reduction compared to 2018 values. In terms of reduction of GHG emissions in 2030 compared to the base scenario, they are estimated at 11.8 million tons of CO₂e. The definition of recyclable packaging is based on design criteria for recycling, complemented by a recyclability assessment procedure and a negative list of non-recyclable packaging characteristics. This measure is expected to result in environmental benefits, with savings equivalent to 4.2 million tons of CO₂e emissions and a reduction of 150,000 m³ in water use. Mandatory compostability is required for certain types of selected plastic packaging, while compostable or conventional plastics are permitted for others. This measure includes lightweight plastic bags, tea bags (and similar items such as capsules or coffee bags), and fruit and vegetable stick-on labels, all of which must be produced from compostable plastic polymers. The estimated model predicts savings of 1.1 million tons of CO₂e emissions. However, we cannot fail to mention that bioplastics, introduced as envi-

TABLE 1: Reuse/refill targets for particular packaging for 2030 and 2040 in the EU.

Sector	Packaging type	Packaging groups and products	Target 2030 / 2040	Target 2030 / 2040
Food and beverage (HORECA)	Primary	Beverage (cold and hot) served into a container at the point of sale for take-away, to be sold in packaging within a system for re-use or refill	20% / 80%	30% / 95%
	Primary	Food for take-away, to be sold in packaging within a system for re-use or refill	10% / 40%	20% / 80%
Food and beverage (Retail)	Primary	Alcoholic beverages (wine and spirits not included) and products based on wine, spirits, or other fermented beverages mixed with non-alcoholic beverages, to be sold in packaging in a reuse or refill system	10% / 25%	20% / 80%
	Primary	Wine, sparkling wine, spirits, and other spirituous beverages are to be sold in packaging within a system for reuse or refill	5% / 15%	20% / 80%
	Primary	Non-alcoholic beverages are to be sold in packaging within a system for re-use or refill	10% / 25%	20% / 80%
	Tertiary	Large household appliances, e.g., washing machines or fridges, to be sold in reusable packaging	90% / 90%	20% / 80%
Commercial and Industrial	Tertiary	Goods sold using pallets, crates, foldable boxes, pails, and drums for the conveyance or packaging of the goods, to be sold in reusable packaging	30% / 90%	20% / 80%
	Tertiary	Non-food goods sold via e-commerce using packaging for transport and delivery, to be sold in reusable packaging	10% / 50%	20% / 80%
	Tertiary	Pallet wrappings and straps for stabilization and protection of goods during transport, to be sold in reusable packaging	10% / 30%	20% / 80%
	Tertiary	Grouped packaging boxes used for wholesale (excluding cardboard) to group a certain number of goods to create a stock-keeping packaging unit are classified as reusable packaging within a system for re-use.	8% / 25%	20% / 80%
	Secondary	Grouped packaging boxes use outside sales packaging to group a certain number of goods to create a stock-keeping packaging unit, which is classified as reusable packaging within a system for re-use.	8% / 25%	20% / 80%

ronmentally friendly alternatives to conventional plastics, are defined as biodegradable if they can be decomposed by microorganisms into carbon dioxide, methane, inorganic compounds, and biomass (Lavagnolo et al., 2024). However, the biodegradability of bioplastics is influenced by their physical and chemical structure, environmental conditions, and microbial populations (Folino et al., 2020). Standard methods for assessing biodegradability often fail to provide clear targets and may not accurately represent real-world conditions (Folino et al., 2023; Lavagnolo et al., 2024). Research has shown significant variability in biodegradation rates across different environments, including compost, soil, and aquatic habitats (Kliem et al., 2020). The comparison of results from different studies is challenging due to variations in methodologies and experimental conditions (Lavagnolo et al., 2024). Future research should focus on long-term field experiments and standardized testing methods to better understand the environmental fate of bioplastics (Polman et al., 2021; Afshar et al., 2024).

Regarding measures regarding the recycled content of plastic packaging, recycling efforts and incorporating recycled content in the EU alleviate uncertainty associated with oil supplies related to geopolitical issues. The 2030 medium ambition target is expected to reduce the EU's fossil fuel needs by 3.1 million tons annually. Increasing the recycled content of plastics is estimated to save 6.5 billion tons of CO₂e emissions compared to the 2030 baseline.

Mandatory high-level targets are set to increase packaging reuse and replenishment by 2030/2040 in selected sectors. These targets are expected to reduce packaging waste by 7.8% in 2030 (26.4% in 2040). Furthermore, this measure aims to increase the percentage of reusable packaging from 2.4% in the 2030 baseline to 8.6% (19.2% in 2040). The predicted increase in reusable packaging for 2030 is expected to result in a decrease in GHG emissions of 2.67 million tons of CO₂e (15.76 million tons of CO₂e in 2040) and water consumption of 118,000 m³ (451,000 m³ in 2040). By 2030, heavier packaging for selected items is expected to be banned in favor of existing lighter alternatives. This measure estimates a reduction of 766 thousand tons of CO₂e emissions and 39,000 m³ of water in 2030. A mandatory target of a 23% reduction in packaging waste per capita by 2030 compared to the baseline is set, equivalent to a 10% reduction compared to waste generated in 2018. The environmental impact of this measure includes a decrease estimated in 2030 relative to the baseline of 14.6 million tons of CO₂e emissions.

Regarding measures on recyclability and compostability in the policy option, a quantitative definition of recyclable packaging is implemented, with estimated savings of 8.3 million tons of CO₂e emissions and 287,000 m³ in 2040. Mandatory compostability for all selected types of plastic packaging is applied, with an estimated saving of 2.1 million tons of CO₂e emissions. With greater ambition and broad targets for recycled content in plastic packaging, reducing the EU's fossil fuel needs would amount to 4.5 million tons per year. The model predicts savings of 12 million tons of CO₂e emissions and a reduction of 930,000 m³ of water use.

5. CONCLUSIONS

The proposed Regulation of the European Parliament and the Council on packaging and packaging waste management presents a comprehensive approach to addressing packaging materials' environmental and economic challenges. The EU aims to promote sustainability, circularity, and resource efficiency in the packaging sector through policy options and regulatory measures. Key measures include increasing the reuse and recyclability of packaging, eliminating unnecessary packaging, and boosting the recycled content in plastics. By 2030, these initiatives are expected to significantly reduce more than 1 million tons of CO₂e emissions and decrease water consumption, with even more significant reductions projected by 2040.

The transition to reusable packaging and the gradual elimination of superfluous packaging are central strategies for achieving these environmental goals. By reducing the amount of waste sent for incineration and landfills, alongside higher recycling rates, the proposed measures will lead to substantial savings in CO₂e emissions and water usage. However, the successful implementation of these measures will depend on clear and comprehensive definitions of recyclable packaging to maximize their effectiveness.

While recycling offers many environmental benefits, the text also highlights potential risks, such as the increased release of microplastics during mechanical recycling, which can threaten ecosystems and human health. These microparticles may interact with other contaminants, amplifying toxicity and spreading through the food chain. Additionally, the biodegradability of bioplastics, which are promoted as a sustainable alternative to conventional plastics, varies significantly depending on environmental conditions and material structures. There is substantial variability in biodegradation rates across different environments, such as composting, soil, and aquatic ecosystems, and a lack of standardized methodologies makes it challenging to compare results between studies. As a result, ongoing and more extensive research is needed, particularly in long-term field experiments and the development of standardized testing methods, to better understand the environmental fate of bioplastics and the impacts of plastic recycling.

The ambitious targets for increasing recycled content in plastic packaging and reducing per capita packaging waste underscore a commitment to reducing the EU's reliance on fossil fuels while promoting a more sustainable circular economy. By increasing recycled content, Europe is projected to save millions of tons of CO₂e emissions and large volumes of water. Therefore, these measures are crucial for the European Union to meet its broader environmental objectives, advancing the transition to a future with lower environmental impact and greater resource efficiency.

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USING EXERGY TO ADDRESS A CIRCULAR ECONOMY

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ABSTRACT

The use of exergy is described as a tool for addressing a circular economy so the benefits can be appreciated and attained. Exergy can be used to understand circular economy measures and to assess and improve energy systems, and can help better understand the benefits of utilizing sustainable energy by providing more useful and meaningful information than energy provides. Exergy clearly identifies efficiency improvements and reductions in wastes and environmental impacts attributable to sustainable energy. Exergy can also identify better than energy the environmental benefits and economics of energy technologies. Exergy should be applied in addressing a circular economy.

1. INTRODUCTION

The concept of a circular economy has been investigated by many, from various perspectives and in many domains. These investigations include contributions by the present author in the fields of business and engineering (Alvarez-Risco et al., 2022), the plastic industry (Alvarez-Risco et al., 2020), the food industry (Del-Aguila-Arcentales et al., 2022), and companies in pandemic times (Alvarez-Risco et al., 2021).

The use of energy as a tool for addressing a circular economy and measuring the benefits of sustainable energy systems can be misleading and confusing. However, the thermodynamic quantity exergy, which can be used to help address a circular economy and to enhance energy systems, can help better understand the potential benefits of relevant measures.

Exergy can clearly identify efficiency improvements and reductions in wastes and environmental impacts. Exergy can also identify better than energy ways to improve environmental benefits and economics. Hence, many researchers suggest that the impact of energy use on the environment, the achievement of increased efficiency, and the economics of energy systems are best addressed by considering exergy (Dincer and Rosen, 2021; Demirel and Rosen, 2023). Thus, many exergy methods have been developed, e.g., exergy analysis for efficiency improvement, and exergoeconomics for improving the economics of energy systems. Thus, exergy has an important role to play in addressing a circular economy.

This article discusses these points with the objective of demonstrating how exergy can help improve understand-

ing and applying exergy methods for addressing a circular economy.

2. USES OF EXERGY

2.1 Efficiency improvement and emissions reduction

The primary use of exergy analysis is as a methodology for understanding the behavior of and improving the efficiency of energy systems. The exergy method is useful for improving efficiency because more meaningful efficiencies are evaluated with exergy rather compared to energy, since exergy efficiencies are always a measure of the approach to the ideal.

2.2 Government and public policy

Facets of exergy relate to government policies, strategies and priorities in areas such as environment, energy, natural resources, industry and economic development. Governments can use exergy methods in establishing public policies in such areas to increase the benefits. Some of these follow:

- Environment policies. In recent decades, various exergy-environment methods have evolved, which can be integrated with government environmental policy.
- Energy and natural resources policies. Exergy can provide a useful perspective for governments to establish natural resource and energy security, for a country or societies within it.

- Economic and industry policies. Exergy methods can be integrated with economics to address financial and economic problems in industry.
- Global, national and regional policies. Evaluations have been carried out of exergy flows through the world, countries and regions, and help in understanding the true efficiency of energy and resource use in these jurisdictions.

3. EXERGY AND ECONOMICS

Economic issues are important in assessing systems. In designing systems, technical disciplines (especially thermodynamics) are combined with economics to achieve preferred and optimal systems. For energy system, costs are usually based on energy, although many suggest costs are better distributed among outputs based on exergy.

Numerous exergy-economic methods have been developed, e.g., exergoeconomics, thermoeconomics and second-law costing. Some researchers have critically reviewed costing methods using exergy, while others describe thermoeconomic optimization. Some even use exergy in describing price-driven economic order systems from a thermodynamic perspective. One reason costs are better distributed among outputs when based on exergy is that exergy often is a consistent measure of economic value, while this is true only sometimes for energy.

4. EXERGY, ENVIRONMENT AND SUSTAINABILITY

Although energy and resources help meet human needs and improve living standards and quality of life, they generally lead to environment impacts. The UN Sustainable Development Goals (2015-2030) reinforce and expand on these ideas (UN, 2015; Rosen, 2021b). In particular, carbon dioxide emissions mitigation can be achieved by introducing sustainable and renewable energy sources, reducing the fossil fuel portion of the energy mix and increasing efficiency.

4.1 Exergy and the environment

Environmental actions can reduce impacts like climate change by lowering resource and energy losses. This raises exergy efficiency and reduces exergy losses. One link between environmental impact and exergy stems from the latter being a measure of the departure of the state of a system from the environment (Rosen, 2021a). The exergy of a system, which depends on the states of both the system and the environment, is zero only when the system and its environment are in equilibrium.

Recently, activity regarding improving understanding of the relation between exergy and the environment has expanded (Nižetić and Papadopoulos, 2018). Kılış and Kılış (2017) propose exergy metrics for energy, the environment, and the economy. Exergoeconoenvironmental analysis is considered for improving understanding of systems and for developing thermodynamically, economically, and environmentally sound energy conversion systems (Esmaeilian et al., 2024; Moharamian et al., 2023). Also, Nwodo et al. (2020) have recently reviewed exergetic life cycle assessment.

Research on the relation between exergy and ecology has also increased in recent years. Valero et al. (2018) describe the theory of exergy cost and thermo-ecological cost, and assess the exergy degradation of natural capital. Some consider thermo-ecological cost for quantifying the exergy and ecology connection. Sciubba (2019) has reviewed exergy-based ecological indicators. Some consider eco-exergy as an indicator of ecosystem complexity, and Hother use exergy as an index for system-level ecological risk assessment of contaminants. Issues have also been examined, e.g., Nielsen (2019) discusses issues arising when shifting the concept of exergy among ecological levels and domains.

Some studies have focused on climate change. For example, some examine limits imposed by the second law of thermodynamics on reducing greenhouse gas emissions, while Hernandez and Cullen (2019) describe exergy as a universal metric for measuring resource efficiency to address industrial decarbonisation. Veiga and Romanelli (2020) examine the mitigation of greenhouse gas emissions using exergy, and Zhang and Reistad (1998) analyze energy conversion systems with exergy methods, including material and global warming aspects. Khajepour et al. (2019) consider responsibility allocation of climate change based on exergy. Arango-Miranda et al. (2018) investigate the role of exergy in carbon dioxide emissions, energy consumption and economic growth, while Wu and Wang (2020) explore avoidable carbon emissions by reducing exergy destruction based on advanced exergy analysis.

4.2 Exergy and sustainability

The United Nations Sustainable Development Goals for 2015-2030 (UN, 2015) spell out a broad view for moving the world towards sustainability, and energy is a key aspect of many of the 17 goals. Sustainability requires sustainable energy resources be used, both cleanly and efficiently (Rosen, 2021b). Exergy methods aid in improving efficiency, thereby enhancing the benefits derived from resources while mitigating problems like environmental impact and lengthening the lifetimes of finite resources. When striving for sustainability, one seeks energy resources that cause no environmental impact. This probably involves using energy resources that lead to no environmental waste emissions or only waste emissions having no environmental impact, the latter relevant when environmental emissions are inert or in equilibrium with the environment. In actuality, resource use entails some environmental impact, and increased efficiency can offset some of the limitations imposed on sustainability by environmental emissions. Even if inexpensive and widely available, it will be advantageous to use energy resources efficiently to reduce the corresponding resource needs (energy, material, etc.) to build the relevant systems and environmental impacts.

Others have examined exergy and sustainability relations. Nielsen (2020) describes an exergy-based approach to sustainable development indicators, while Lucia and Grisolia (2019) consider exergy inefficiency as an indicator for sustainable development. Whiting et al. (2017) describe the use of exergy to evaluate the sustainability of fossil fuels and non-fuel mineral depletion and Zhang et al. (2018)

improve understanding of fuel sustainability from the exergy viewpoint. Stanek et al. (2019) describe exergy as a measure of sustainability through thermo-ecology. Exergy is used as an element of a method to assess the environmental impact and sustainability of energy systems (Kiehbadrudinezhad et al., 2023; Jalili et al., 2024).

5. EXERGY AND A CIRCULAR ECONOMY

5.1 Resources and resource depletion

Many investigations focusing on the application of exergy methods to resources, from the perspective of a circular economy have been reported. Many of these studies relate to energy resources, due to their widespread usage and the many concerns related to them, regarding environmental impacts, resource supplies, economics and other factors. Numerous studies have also been carried out on other resources such as metals and plastics.

An interesting general study is the one by Llamas et al. (2024), which examines the quantification of resource dissipation using an exergy approach, as a means of supporting a circular economy. They write, “exergy analysis highlights irreversibilities in circular economy systems, and it provides an appropriate and complementary resource efficiency indicator,” and that through the irreversibilities obtained with exergy analysis, “their dissipative role in the circular economy can be quantified.”

5.1.1 Energy resources

Energy resources have been examined using exergy methods for decades, and recently many have focused on a circular economy. The advantages of exergy in addressing energy issues as part of a circular economy have been examined by the present author (Rosen, 2022; Dincer and Rosen, 2021; Demirel and Rosen, 2023). For instance, Dincer and Rosen (2021) describe eco-industrial clusters that integrate circular economy with industrial ecology based on industrial symbiosis. Cooper et al. (2017) provide insights on a circular economy from a thermodynamic perspective focused on exergy. The study involves analyses of the impact of a range of circular economy approaches on energy use, which proves to be a particularly important application area for exergy methods with respect to circularity.

Waste to energy: Sharma et al. (2020) investigate the waste-to-energy nexus as part of a circular economy including environmental protection, including an exergy perspective. Their focus on the employment of biological methods for waste materials is a key part of their waste-to-energy approach, which helps in addressing concerns about waste management, energy demand and environmental impact.

de Almeida and Borsato (2020) extended the RİPEX exergy-based method, which quantifies energy and material flows in terms of exergy and considers environmental impact, efficiency and cost, yielding a method they claim may support waste to energy as part of a circular economy. The inclusion of exergy in the method is central to its advantages.

Energy conservation: Alvarez et al. (2022) propose a product innovation method based on exergy and knowledge management that they claim is essential for energy

conservation and environmental protection. The method helps to raise exergy efficiencies by lowering exergy destruction rates, i.e., irreversibilities, to support the circular economy. Their approach is transdisciplinary and contributes to climate change mitigation, which is an important area in general as well as for circularity, and which benefits notably from applying exergy concepts.

Heating: North and Jentsch (2021) formulate an approach to heating processes supportive of a circular economy that focuses on exergy. The application of exergy analysis helps identify resource options that are efficient and conserve primary energy resources greatly. They focus on applications to buildings, although heating for other applications benefits similarly from the application of exergy methods.

Renewable energy: Much research has focused on renewable energy sources as alternatives to fossil fuels in support of a circular economy, often from exergy viewpoints. In a general manner, Alvarez et al. (2022) investigate the modeling product innovation from renewable energy based on an exergy methodology. Through energy conservation and environmental protection, the method helps to shift towards a circular economy.

Different types of renewable energy have received attention, given they all have applications where they can be advantageous. Cano et al. (2020) apply life cycle assessment based on exergy indicators to contrast the use of renewable and non-renewable resources in processes in a circular economy. Various renewable resources from nature are considered, including hydraulic, solar and wind, allowing for a reduction in consumption of non-renewable resources. Alvarez et al. (2022) consider the innovation of products utilizing renewable energy to satisfy environmental protection regulations. Their focus on exergy helps in climate change mitigation and achieving a circular economy. Novak (2017) focuses on a sustainable energy system based on renewable energy in support of a circular economy, following exergy methods. The author stresses how the circular economy necessitates exergy considerations in technology and economic patterns.

Solar energy has received much attention in recent years and applications have grown considerably, especially in the area of photovoltaic (PV) systems. Bartie et al. (2020) focus on the role of metallurgical infrastructure in their analysis of the resource efficiency of the circular economy, focusing on cadmium telluride PV technology and its life cycle. The work focuses on thermodynamic limits that are made clear with exergy methods. Bartie et al. (2021) assess solar resources from an exergy perspective through photovoltaic technology, and examine the opportunities it provides for a circular economy. The authors focus on opportunities for recovery of resources to sustain PV industry growth through consideration of the PV life cycle, and point out how the technical limits of circularity are highlighted via exergy methods.

Alternative fuels: Various fuels that are alternatives to fossil fuels have been considered from exergy perspectives, in support of a circular economy. These fuels facilitate the use of renewable energy sources in most cases, thus making energy systems more circular.

One important alternative fuel is hydrogen, which has received increasing interest in recent decades. Martínez-Rodríguez and Abánades (2020) assess and compare hydrogen production techniques (e.g., electrolysis and pyrolysis) with exergy methods. The work can support a hydrogen economy, in which the main energy carriers are electricity and hydrogen, which in turn can support a circular economy. Sharma et al. (2020) investigate hydrogen energy trends as part of a waste-to-energy nexus and thereby part of a circular economy. They focus on hydrogen yields, economics, research and development needs, commercial viability and sustainable development, all in support of a circular economy. They note the imperative need for a circular economy and sustainable development. Novak (2017), in considering exergy as sustainability measure for energy systems, proposes hydrogen production from solar and planetary energy at high exergy efficiency as a means to a circular economy that can support the development of societies. The work focuses on known chemical processes and existing infrastructure, with an aim of reducing the harm caused by fossil fuels and their resource limitations.

Another important alternative fuel is methanol. Vilardi et al. (2020) investigated methanol production with exergy methods via biogas upgrading routes. The authors claim these are key to a circular economy by permitting reuse of wastes. Novak (2017) proposes methanol from solar and planetary energy as part of a circular economy, partly in extension of work on hydrogen production. The work, based on exergy methods, incorporates carbon recycling.

Novak (2017) proposes synthetic methane produced from solar and planetary energy as part of a sustainable energy system in support of a circular economy. The work is founded on an exergy approach. The work provides an interesting focus, given methane is a key constituent of natural gas, but the synthetic nature of the product methane avoids issues associated with natural gas utilization.

Vilardi et al. (2020) focus on biogas production and treatment to generate alternative bio-derived fuels for a circular economy, drawing on waste sources and exergy methods. Following an exergy approach, Novak (2017) proposes using solar and planetary energy like biomass within a circular economy to produce gaseous and liquid fuels, via chemical processes.

5.1.2 Non-energy resources

Non-energy resources have been examined using exergy methods, sometimes focusing on a circular economy. North and Jentsch (2021) adopt a circular economy approach that involves decoupling resource consumption from resource depletion. Gai et al. (2021) examine the recycling of multiple resources, by considering materials recovery, exergy demand and economics as well as the trade-offs between them. They apply the work to industrial and urban processes. In support of sustainable production and consumption, Hernandez and Cullen (2019) propose exergy as a universal metric for increasing resource efficiency and reducing global carbon emissions. Their work is developed under the rubric of material efficiency or circular economy, and has the potential for widespread appli-

cation in many disciplines given its “universal” intent and approach.

Much work on non-energy resources relates to metallurgy and metals. Reuter et al. (2019) examine material and metallurgical issues from the viewpoint of product design, within a circular economy. They utilize a product (mineral)-centric approach and clarify limitations of material-focused considerations. Abadías Llamas et al. (2020) report an exergy analysis of zinc production for large circular economy systems. The exergy-based methodology demonstrates resource efficiency limits through exergy dissipation, in terms of material losses. Hannula et al. (2020) examine aluminum recycling and related processes with exergy, focusing on separation processes. Through the work, they demonstrate that recycling and process metallurgy are the main enablers of a circular economy. Cano et al. (2020) examine gold mining using exergy indicators within a life cycle assessment. They consider both open-pit and alluvial mining. Applying a circular economy focus, the helps in reducing resource consumption. Focusing on cadmium telluride production, Bartie et al. (2020) examine the roles of mineral processing, extractive metallurgy and metallurgical infrastructure in a circular economy, while considering thermodynamic limits via exergy methods. Making metal processes more circular is an important need in a circular economy, given the pervasiveness of the usage of metals. Finally, Llamas et al. (2024) use exergy to quantify resource dissipation for metallurgical processes and point out how it provides an appropriate and complementary resource efficiency indicator for them.

Other work on non-energy resources involves non-metal materials and, like metal processes, making them more circular is an important need in a circular economy given their extensive usage. Reuter et al. (2019) examine material factors, within a circular economy and identify limitations of material-focused considerations through exergy. Cano Londoño and Cabezas (2021) present their perspectives on a circular economy via a supply chain strategy, considering the thermodynamic limitations in terms of exergy factors. The work focuses on maintaining materials in a useful state in products, and avoiding losses and dissipation. Specific materials have also been examined. Textiles like cotton have been assessed by Popescu et al. (2022) for their exergy footprint in terms of recycling. Popescu et al. (2022) also examine polyethylene production in a related manner. The work involves the entire process chain of polyethylene synthesis by recycling cotton waste, and demonstrates how circular economy approaches tend to decrease the exergy footprint. Russo et al. (2021) examine the production and recycling of polymers and plastic based on exergy. The work is useful for developing recycling processes for plastics. Ortego et al. (2023) extend this work focusing on plastic car parts, highlighting the recycling challenges because of the wide variety of plastics used. Huysman et al. (2017) examine post-industrial plastic waste and relevant performance indicators for a circular economy, accounting for cumulative exergy extraction. Some of the indicators are observed to be useful for guiding how to use waste streams, based their quality and environmental considerations.

5.2 Chemical processes

Many investigations have been reported of chemical processes a circular economy, following exergy methods, and this is not surprising given the particular benefits of exergy to processing operations of a chemical nature in particular. Focusing on a chemical engineering context, Cano Londoño and Cabezas (2021) examine various views on a circular economy and sustainable development, considering the thermodynamic limitations through exergy. A wide range of diverse chemical processes have been considered. The benefits are demonstrated of efficient resource use and efficient technology. As well, the valorization and minimization of wastes is rationalized and quantified via exergy.

Many specific chemical processes have been investigated. Sharma et al. (2020) investigate hydrogen energy trends, in support of a circular economy, considering biological and chemical processes for wastes. They consider such processes as dark fermentation, photofermentation, and combined dark and photofermentation, as well as pre-treatment methods including heat treatment, acid or base treatment, ultrasonication, aeration and electroporation. Vilardi et al. (2020) examine several processes for biogas upgrading to biomethane (amine scrubbing, water scrubbing and membrane separation) with exergy methods, consistent with the precepts of a circular economy. The upgraded biogas is developed to meet the standards for injection into a grid, and is also usable directly. Vilardi and Verdone (2022) also examine with exergy the processes involved in waste incineration, including oxy-combustion, air separation and flue gas recirculation. Waste reuse is confirmed through the results to be an important aspect of a circular economy. In examining with exergy methods the polymers and plastics production and recycling, Russo et al. (2021) consider the life cycle of nine commercial polymers and the resource consumption during all processes to generate the crude polymeric material. By considering such a variety of polymers, the generality of the exergy-based approach is demonstrated to have great breadth. In investigating polyethylene production via the exergy footprint, Popescu et al. (2022) consider all processes polyethylene synthesis by recycling waste, highlighting the advantages of a circular economy approach. Hannula et al. (2020) examine the production of alloys with exergy-based and LCA methods, focusing on physical separation (e.g., using an eddy current separator) and re-melting processes, illustrating how exergy methods support circularity through more efficient resource use.

5.3 Waste utilization, recycling and end-of-life processes

Much research has been reported on waste utilization, recycling and end-of-life processes, incorporating exergy methods. Such work is founded on the rationale values of the usefulness of wastes and inputs provided with exergy methods. de Almeida and Borsato (2020) investigate waste to energy processes and their contribution to a circular economy, following an exergy-based method that considers exergy flows. They obtain broad results in the

areas of environmental impact, efficiency and economics, and consider the trade-offs involved in reuse, recycling and incineration.

Many specific waste utilization, recycling and end-of-life processes have been investigated in line with the concepts of a circular economy. In examining biogas upgrading processes, Vilardi et al. (2020) examine with exergy analysis the reuse of industrial by-products and agro-industrial wastes, for several energy intensive sectors. Vilardi and Verdone (2022) also examine by exergy methods the use of municipal solid waste to contribute to a circular economy, through incineration. Waste to energy processes have also been investigated from an exergy perspective as an important aspect of a circular economy by Sharma et al. (2020), providing an avenue for about waste management in a manner that supports a circular economy. Huysman et al. (2017) consider plastic waste processing for a circular economy, accounting for indicators to quantify the circular economy performance based on exergy.

Finally, many studies have focused on applications of recycling to contribute to the circular economy through exergy approaches. Gai et al. (2021) examine trade-offs in recycling multiple resources, as well as the advantages in terms of circularity. The criteria applied are derived from exergy through a circular exergy use rate, and these are straightforwardly contrasted with the results of more conventional approaches. In assessing via exergy methods resource recovery opportunities as a means of circularity, Bartie et al. (2021) use neural networks to investigate a system's response to variations in end-of-life processes and recycling. They consider primary resource and electricity use and environmental emissions. Popescu et al. (2022) assess with an exergy approach the use of recycling in textile production, drawing on a procedure for industrial chemical recycling. The widespread pervasiveness of textiles globally, in countries of greatly varying levels of wealth, make this application particularly impactful across the planet. de Almeida and Borsato (2020) study end of life processes in waste treatment of industrial products, considering reuse and recycling, via an exergy-based method, that supports a circular economy. Polymer and plastic recycling is examined based on exergy by Russo et al. (2021), via thermodynamic recycling indexes that are founded in exergy. Reuter et al. (2019) examine design for recycling with a product-centric approach using exergy to address circular economy challenges. The product-centric nature of this approach makes it different from many other methods, but it still is contingent on applying exergy concepts. Llamas et al. (2024) quantify resource dissipation using an exergy approach and indicate that it provides an appropriate and complementary resource efficiency indicator for recycling processes.

5.4 Advanced exergy-based methods for a circular economy

Many investigations have been reported focusing on the development of advanced exergy-based methods in support of a circular economy, and their applications in numerous areas. Many of these studies relate only to

thermodynamic techniques, while others go beyond thermodynamics, covering such areas as the environment and economics and more broadly design and optimization. All of these can support a circular economy, either directly or indirectly.

5.4.1 *Methods for resources and energy*

Numerous studies have examined means of increasing efficiency. North and Jentsch (2021) apply exergy methods to buildings heating processes to identify efficient resource options and conserve energy resources in support of a circular economy. Hernandez and Cullen (2019) use exergy methods to increase resource efficiency in support of sustainable production and consumption via a circular economy. Cano et al. (2020) investigate means to increase efficiency through reducing exergy wastes by applying exergy methods and exergy life cycle assessment to support a circular economy, and apply them to mining processes. Although a unique type of operation, mining is nonetheless widespread globally, rendering the work described here important for more circular use of mined resources.

Hernandez and Cullen (2019) propose a resource efficiency metric based on exergy, in a sense as a universal metric. They find that such an exergy-based resource efficiency metric has high relevance and robustness, and can help raise awareness and understanding about exergy and its benefits for a circular economy. The benefit of a universal approach in this approach is quite significant, as one criticism of exergy methods can be that they have to be tailored to individual processes or operations. A universal approach once widely accepted could therefore advance the exergy-based resource efficiency metric rapidly.

Gai et al. (2021) propose a multi-resource integration map for processes (e.g., recycling) as a way of managing the trade-off between process circularity and the expenditure of resources (energy, economic, etc.). The technique considers multiple resources, and such activities as materials recovery.

Gai et al. (2021) also propose a total circularity index, developed by integrating circular exergy use rate and circular material use rate. The technique constitutes a way of providing guidance for the integration of multi-resource systems, and an understanding of economic and exergy performance of circular economy options. A good degree of practicality is reflected in this approach as well as the one described in the prior paragraph through their recognition of the necessary trade-offs when striving for circularity.

To focus on sustainability in a realistic manner, Huysman et al. (2017) propose a performance indicator for a circular economy based on cumulative exergy extraction from the natural environment, and apply it to waste streams and their treatment options for plastics. The indicator is useful for valorization of waste streams and attaining and comparing environmental benefits. The applicability of this approach is great, due to the broad use of plastics, providing significant potential in moving towards a circular economy.

5.4.2 *Methods for the environment*

To aid in environmental protection and energy conser-

vation, Alvarez et al. (2022) propose eco-innovation, i.e., a technique involving product innovation focusing on renewable energy use. The transdisciplinary technique involves exergy methods and knowledge management, and bolsters a circular economy. This is particularly evident in the substitution it envisions of renewable energy for fossil fuels.

Popescu et al. (2022) develop and utilize the exergy footprint concept and apply it to recycling. The process stages are identified and examined, with subsequent efforts made to reduce or eliminate wastes and reduce exergy input, thereby decreasing the exergy footprint. Doing so enhances circularity directly.

Numerous researchers consider life cycle assessment and related techniques as key to a circular economy. Hanula et al. (2020) consider aluminum recycling using exergy and life cycle assessment, as a means to enable a circular economy. Resource efficiency and environmental impacts are determined through exergy analysis and life cycle assessment, along with modeling and simulation. The significant benefits of assessing and addressing environmental issue via a life cycle assessment approach are notably enhanced by the incorporation into the approach of exergy concepts. Cano et al. (2020) apply exergy life cycle assessment and related indicators to investigate the mining sector.

Many researchers consider decarbonization as part of a circular economy, exploiting the advantages of exergy methods. North and Jentsch (2021) formulate an approach focusing on exergy to decarbonizing building heating. Xie et al. (2023) examine the nexus between low-carbon and circular economies. They show how low-carbon actions and circular practices can work together at various scales and dimensions, and suggest bio-waste management and renewable energy use, and adjusting the mix of circular actions to follow the transition to a low-carbon energy. Hernandez and Cullen (2019) propose exergy metrics for reducing global carbon emissions as well as increasing resource efficiency, and consider numerous efficiency metrics as a means of powering a low-carbon transition. Novak (2017) examine and promote industrial decarbonisation through the utilization of exergy as a universal metric for resource efficiency renewable energy in support of a circular economy, following exergy methods. The author emphasizes how renewable energy supports a circular economy. This approach, and the others noted in this paragraph, further point out the benefit of exergy methods in efforts for a circular economy.

In a somewhat related way, numerous researchers consider climate change using exergy methods, in support of a circular economy. In one example, Alvarez et al. (2022) apply eco-innovation along with renewable energy to mitigate climate change. This approach is extremely noteworthy, given the significance and urgency of challenges related to climate change.

5.4.3 *Methods for economics, design and optimization*

Abadías Llamas et al. (2020) report and apply a thermoeconomic technique that combines exergy methods (i.e., a thermodynamic perspective) with economics. The technique demonstrates resource efficiency limits, in terms of material losses and exergy dissipation and resource-ef-

ficient alternative to minimize wastes, and is shown to be useful for circular economy systems.

Reuter et al. (2019) examine challenges of a circular economy material from a product design perspective factors, including design for recycling following a product-centric approach and assisted by exergy considerations. The authors focus on using wastes as resources and extending product life cycles for maximizing resource efficiency, and apply the research to metallurgical processing.

Bartie et al. (2020) consider factors related to optimization within a circular economy and demonstrate how the true performance is related to thermodynamic limits established with exergy methods, and apply the work to photovoltaic systems.

Given the importance of design in engineering, and the great value of optimization, especially when accounting for economics, these methods are potentially very important to the engineering community.

5.5 Broader applications of exergy methods for a circular economy

Studies have been undertaken on broader applications of exergy methods for a circular economy, often focusing on the large scale, including research in support of policy making and sustainability. These broader applications are significant, as broader and larger applications of exergy approaches to a circular economy, can have a multiplier effect compared to applications to single processes or systems.

North and Jentsch (2021) examine the use of exergy in policy making related to a circular economy, in particular to prioritise the reuse of these freely available heat sources from wastes and renewable sources. The approach helps in decoupling resource consumption from depletion. The authors consider building heating along with heat networks, and examine the use of exergy methods in city energy planning to identify and prioritize resource efficiency and energy resource conservation.

The topics of sustainability and sustainable development, which are attracting increasing attention in recent years, are in line with a circular economy and can be enhanced with exergy methods. Cano Londoño and Cabezas (2021) identify and investigate perspectives on a circular economy in the context of and sustainable development. The work aims to materialize the environmental sustainability of supply chains, considering exergy to acknowledge thermodynamic limitations, and seeks to make production processes more sustainable, especially in the domain of chemical engineering. In examining hydrogen production options with exergy methods, Martínez-Rodríguez and Abánades (2020), point out the importance of a hydrogen economy and how hydrogen is produced affects the sustainability of a hydrogen economy. Xie et al. (2023) demonstrate how a circular economy and low-carbon economy can be viewed as pillars of sustainable development. They show how practices in these areas can be mutually beneficial and support sustainable development of various entities of many sizes. They go beyond economic and environmental sustainability by pointing out the importance of considering social sustainability. In examining chal-

lenges to a circular economy, Reuter et al. (2019) show how integrating product-centric considerations in various domains (e.g., energy, water, industry, transport) can help in improving or maximizing the resource efficiency and make cities sustainable in the future, thereby contributing to the Sustainable Development Goals of the United Nations. Since humanity is now more than half way through the 2015-2030 time frame set for the Sustainable Development Goals, and most experts suggest there is significant effort still needed since progress notably lags what was expected, the impact of this method may be high and widespread.

5.6 Fields of application of exergy methods for a circular economy

Many fields have been the focus of investigations on the application of exergy methods to support a circular economy, particularly in industry and regions. A selection of these is shown in Figure 1, and details are provided below.

Industries have been examined with exergy methods to enhance how they can support a circular economy, both at the general level and for specific industries. Industrial processes in general are examined from an exergy viewpoint by Hernandez and Cullen (2019), particularly regarding decarbonisation and resource efficiency. They show that, to reduce carbon emissions, industry needs to increase both energy efficiency and material efficiency. In discussing the need for exergy in design education, Dincer and Rosen (2021) describe a circular economy and education model incorporating exergy for a dairy facility.

Cano Londoño and Cabezas (2021) examine the chemicals sector and its industries to identify circularity factors and opportunities, using exergy methods. These industries are primarily involved in transforming raw materials into products.

North and Jentsch (2021) focus on buildings and the building sector in developing an exergy approach supportive of a circular economy. They point out the importance of the building sector by pointing out that building heating accounts a significant portion of the energy consumption and greenhouse gas emissions in many countries.

Russo et al. (2021) and Ortego et al. (2023) apply exergy-based assessments to automobiles and the automotive sector. The work has implications for the material content of vehicles and their components, recycling, efficiency and decarbonization. That work also demonstrates the importance of car part materials on design and end-of-life management to promote circularity.

The electricity generation sector has been the subject of numerous studies using exergy methods, in line with its role in a circular economy. North and Jentsch (2021) describe the burdens on electricity infrastructures if heating processes becomes electrified in the future and if renewable energy is increasingly used to provide electricity, hydrogen and heating. Xie et al. (2023) investigate low-carbon and circular economies, focusing issues affecting a circular economy in relation to carbon-intensive electricity. Novak (2017) presents a vision for a sustainable energy sys-

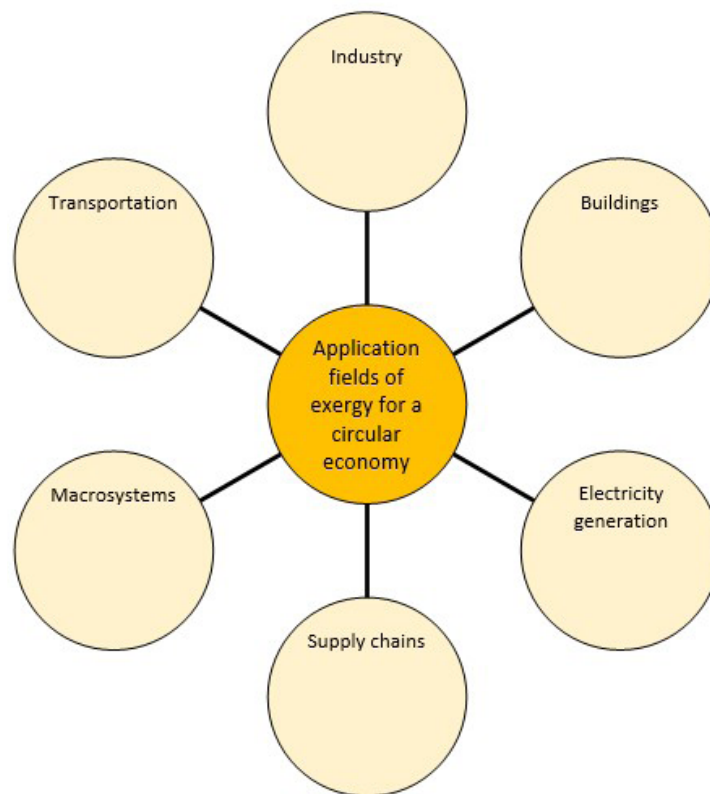


FIGURE 1: Selected fields of application of exergy methods to support a circular economy.

tem premised on solar and other natural energy being used to produce renewable electricity and where appropriate hydrogen. The vision is consistent with a circular economy and is developed using exergy methods. Bartie et al. (2020) and Bartie et al. (2021) examine photovoltaic technology over its life cycle, utilizing exergy methods, and how such technology is a key component of a circular economy.

Supply chains have received attention, as applied to various industries. In examining energy use and its relation to a circular economy, Cooper et al. (2017) identify supply chain impacts through input-output analyses. This work enables prioritization of the areas in which circular economy approaches have the strongest potential for supply chains. Cano Londoño and Cabezas (2021) propose a supply chain strategy that includes limitations identified through exergy methods, as a means of supporting a circular economy. The strategy allows for the materialization of the environmental sustainability of supply chains. While examining low-carbon and circular economies, Xie et al. (2023) describe the need for regional supply structure and circular economy chains to address the trade-offs arising from carbon-intensive energy and raw material scarcity.

Regions have also been studied using exergy methods to enhance their contribution to a circular economy. As pointed out earlier, such broader applications of exergy approaches to a circular economy can have a much more widespread impact compared to applications to single processes or systems, making the activity for regions notable. Xie et al. (2023) investigate with exergy methods the benefits of urban-industrial symbiosis and the need to address

urban-industrial planning and challenges. They demonstrate the importance for decision-makers to minimize trade-offs while maximizing synergies to support a circular economy. Gai et al. (2021) develop a method involving urban symbiosis where product demands are satisfied by a set of waste resources, with or without processing. Applicable to industrial and urban processes, the method can help guide the integration of multi-resource systems by determining economic and exergy performance of circular economy measures. Cooper et al. (2017) provide insights on a circular economy for macrosystems like countries and regions, from an exergy perspective. Although focused on energy use, the implications are broader and permit prioritizing areas where circular economy measures can have noteworthy impacts.

6. GUIDELINES FOR EXERGY ANALYSES RELEVANT TO A CIRCULAR ECONOMY

As a consequence of the critical review presented in this article, guidelines for exergy analysis can be developed relevant to a circular economy. This is done in this section, first by presenting steps for exergy analyses for a circular economy, and then by comparing exergy analysis with conventional approaches.

6.1 Steps for exergy analyses for a circular economy

Exergy analysis uses these concepts to specify quantitatively limitations by expressing losses in terms of exergy. As only a portion of the input exergy to a system or process can be recovered, exergy analysis provides theoretical limi-

tations for a system. Exergy analysis always indicates that exergy cannot be conserved in a real system, but is conserved in the limiting case of a reversible or an ideal system. All of these ideas pertain to a circular economy.

A procedure for performing energy and exergy analyses of a process or system in the context of a circular economy can be developed. It involves the following steps:

- Subdivide the process or system into sections to match level of detail and information desired in the context of assessing circularity for a circular economy.
- Perform conventional mass and energy balances, and determine relevant quantities (e.g., work, heat, mass) and properties (e.g., temperature, pressure), ensuring all needed values to assess a circular economy are accounted for, both for the system or process being examined and modified versions to improve circularity.
- Select a reference-environment model, accounting for characteristics of process/system, information sought, accuracy required and analysis complexity for assessing contribution to a circular economy.
- Determine energy and exergy values, relative to selected reference-environment model.
- Develop exergy balances, including determination of exergy consumptions, to allow the circularity to be assessed for existing and modified versions of the system or process.
- Select and evaluate appropriate measures of merit (e.g., exergy efficiencies or indices) that assess level of circularity attained and the contribution to a circular economy.
- Interpret results, draw appropriate conclusions and make recommendations in the context of improving the system or process through the principles associated with a circular economy.

Some of these steps are common with exergy analyses in general, while others are unique for a circular economy.

A breakdown of the steps in the procedure for performing energy and exergy analyses of a process or system, in the context of a circular economy with emphasis on those aspects, is presented in Table 1.

Note that it is important to recall the definition of exergy at all times, and to differentiate it from energy. Exergy

is a measure of the ability to perform work. This can be applied to exergy a flow or commodity. Either way, exergy is the maximum work can be obtained from it as it passes reversibly to the environmental state, exchanging materials and heat only with the surroundings.

6.2 Comparison of exergy analysis with conventional approaches

A comparison of exergy analysis with conventional approaches for assessing and improving circularity can be useful and instructive. Hence one is provided here, through the following points:

- Exergy analysis allows true measures of thermodynamic efficiency and loss to be attained, whereas energy analysis does not. This making assessments regarding the advantages of circularity based on energy can lead to erroneous or skewed results. Assessments based on exergy values avoid such problems and are always based on efficiencies that measure approach to ideality and losses that measure deviations from ideality.
- System improvements based on enhanced circularity are more meaningfully developed and assessed based on exergy, as the true values of losses are identified in terms of their exergy. This allows rational decisions about which losses are worth recovering at all, and what the priority should be for recovering and reusing separate loss quantities or flows. Also, the true advantage of the circularity efforts in terms of modified systems are clearly assessed in this manner.
- Circularity improvements are quantified in all respects when exergy analysis is applied, since exergy losses are identified in all respects (e.g., causes, locations, quantities). Thus, more comprehensive information is provided and enhanced decisions can be correspondingly made.

7. ILLUSTRATIVE EXAMPLE OF EXERGY ANALYSES FOR A CIRCULAR ECONOMY: TRIGENERATION

An illustrative example based on a trigeneration process is presented to help explain the method of exergy

TABLE 1: Procedure for exergy analysis of a process or system in the context of a circular economy.

Step	Actions	Circularity aspects
1	Subdivide process or system into appropriate sections	Select sections to provide information desired for assessing circularity
2	Perform conventional mass and energy balances, and determine relevant quantities and properties	Ensure values for assessing a circular economy are evaluated, for existing system or process and modified versions to improve circularity
3	Select a reference-environment model for evaluating exergy	Account for characteristics of process or system and information sought for assessing contribution to a circular economy
4	Determine energy and exergy values, relative to selected reference-environment model	Ensure all values necessary to evaluate circularity are attained.
5	Develop exergy balances, including exergy consumptions	Obtain accurate balances for all system or process parts that affect circularity and its assessment for existing and modified versions
6	Select and evaluate appropriate measures of merit (e.g., efficiencies)	Select appropriate measures for assessing level of circularity attained and the contribution to a circular economy
7	Interpret results, draw appropriate conclusions and make recommendations	Carry out interpretation steps from the perspective of improving the system or process through circular economy principles

analyses for a circular economy, and the benefits it can provide.

7.1 Trigeneration

Trigeneration is a process that produces three products simultaneously from an input, rather than in separate processes for each product. Often the products are electricity, useful heat and useful cooling.

A general trigeneration system is shown in Figure 2, where three distinct products are generated from a single input in a single process.

A specific trigeneration process in which the three products are electricity, useful heat and useful cooling is presented in Figure 3.

To avoid confusion and provide clarity, it is noted that cogeneration is a simplified and more common form of trigeneration. Cogeneration is the generation of two products, typically electricity and useful heat, in a single process rather than in separate electricity generation and heating processes. Trigeneration can thus be seen to be an expanded form of cogeneration, which typically yields useful cooling as a third product in addition to electricity and useful heat.

Interest in both cogeneration and trigeneration has expanded as concerns grew about energy supplies, costs and environmental impacts have grown, as such systems can be more efficient than separate systems for each product.

7.2 Trigeneration to improve circularity

The manner by which trigeneration contributes to a circular economy can be seen most readily by considering the conventional alternative to it, which is three separate processes (one for electricity generation, one for heating and one for cooling). Three such processes are shown in Figure 4.

When these three processes are integrated in a manner to enhance circularity, several options become possible:

- The loss from the electricity generator (waste heat) can be redirected to provide the product heat, thereby avoiding the loss and eliminating the need for a resource input to the heater.
- The loss from the electricity generator (waste heat) can be redirected to be the input to the chiller (if an absorption chiller, which is a heat-driven device is used, thereby avoiding the loss and eliminating the need for a resource input to the chiller.
- Some of the product electricity from the electricity generator can be redirected to be the input to the heater, thereby eliminating the need for a resource input to the heater (although at the sacrifice of some of the electrical product from the electricity generator).

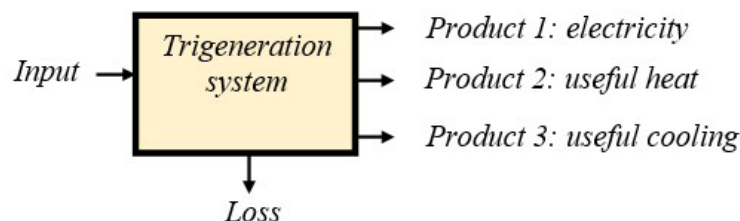


FIGURE 3: A trigeneration system for electricity, useful heat and useful cooling.

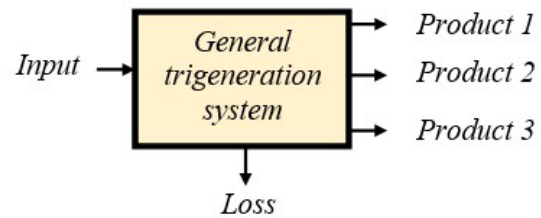


FIGURE 2: A general trigeneration system.

- Some of the product electricity from the electricity generator can be redirected to be the input to the chiller (if a vapor compression chiller, which is typically electrically driven), thereby eliminating the need for a resource input to the chiller (although at the sacrifice of some of the electrical product from the electricity generator).

Any of the four options listed above can improve circularity compared to separate processes for electricity generation, heating and cooling. Also, various combinations of the above four options can also help improve circularity. The best option depends on the characteristics of the inputs, wastes and products, including their values, and these are often best determined using exergy methods, rather than the more conventional energy methods.

It is pointed out that this illustrative example is not just theoretical, but quite practical. In many locations across the globe, power plants driven by on fossil fuels are replaced or modified so that waste heat from these power plants is no longer wasted, but instead utilized to produce heating and/or cooling. The benefit of trigeneration can be reduced input fuel use and wastes, in line with the circularity improvements. That is, trigeneration is able to mitigate environmental impact by facilitating the use of waste heat for cooling and/or heating, thereby offsetting either burning fossil fuel or consuming electricity. The production of three products from a single input via trigeneration boosts efficiency notably. In addition, there can also be a reduction in operating costs, and when carbon taxes are applied, trigeneration can lower the carbon tax paid by recovering waste heat and converting it to additional products.

8. PERSPECTIVE AND LONG-TERM VIEW OF EXERGY ANALYSES RELEVANT TO A CIRCULAR ECONOMY

Based on the findings in the literature presented in this article, a perspective and long-term view can be developed

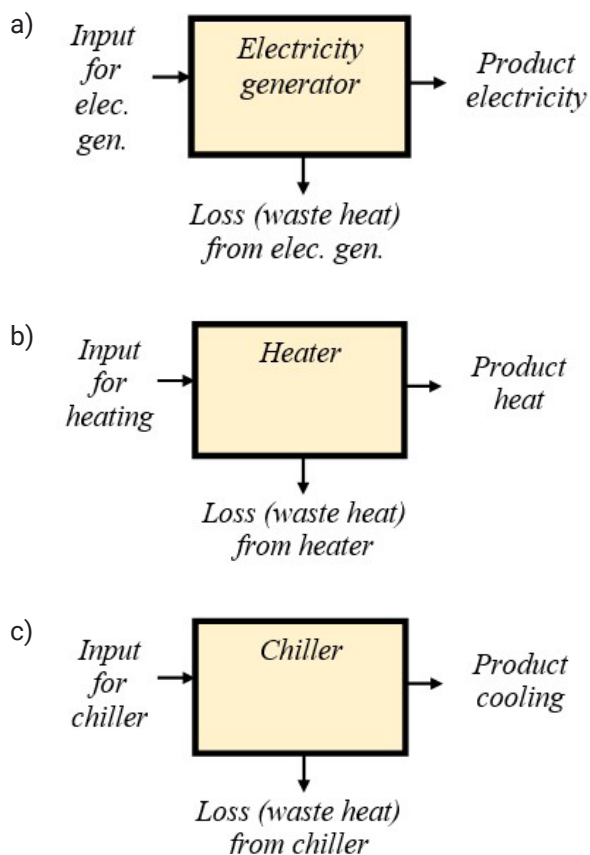


FIGURE 4: Separate systems for electricity (a), useful heat (b) and useful cooling (c).

of the topic of exergy analyses relevant to a circular economy. This is done in this section.

The following points cover several key aspects of such a perspective and long-term view the topic of exergy analyses relevant to a circular economy:

- First, exergy methods are important in general for understanding thermodynamic efficiencies and losses for processes and systems, and for making decisions on system improvements. When exergy methods are not employed, and more conventional methods like energy analysis are used, the results can be misleading and sometimes erroneous.
- Second, given the importance of exergy methods, as raised in the last point, it is perhaps not surprising that exergy is proving to be important and relevant to a circular economy. The same reasoning on why exergy methods are important in general, also apply to methods to improve circularity of systems and processes.
- Third, elaborating on the last point, it is emphasized that based on the literature review decisions regarding improving the circularity of processes and systems are made in terms of meaningful efficiencies and meaningful loss avoidance when such decisions are based on exergy, through the application of relevant exergy methods.
- Finally, it is anticipated, based on the review presented here and the numerous benefits pointed out by applying

exergy methods, that applications of exergy methods to enhance circularity of systems and processes, and thereby contribute to a circular economy, will grow in number and expand in disciplines reflected and regional areas of employment.

9. CONCLUSIONS

The benefits of using exergy as a tool for addressing a circular economy, in particular by identifying and explaining the benefits of sustainable energy, has been demonstrated. It is concluded that the concepts encompassing exergy have a significant role to play in addressing a circular economy and increasing sustainable energy use. Exergy clearly identifies efficiency improvements and reductions in wastes and environmental impacts attributable to sustainable energy, and enhances understanding of environmental benefits and economics. Exergy should prove useful in such activities to engineers and scientists, as well as decision and policy makers in addressing a circular economy.

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A REVIEW OF POTENTIAL PHYSICAL AND CHEMICAL MARKERS FOR TYRE AND ROAD WEAR PARTICLES

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ABSTRACT

Plastic pollution poses a substantial environmental challenge on global scale. Recently, tyre and road wear particles (TRWPs) have been recognized as a source of microplastic pollution to the freshwater environment. Whilst there is a growing concern regarding the potential environmental effects of microplastics, TRWPs are especially concerning because of the additives they have. These additives are utilised in the manufacturing of tyres; persist in the final product; become environmentally available; and may pose significant threats to an ecosystem. A current issue is the identification of specific constituents of TRWPs responsible for these threats. A comprehensive review of the existing literature is presented focusing on the physical and chemical characteristics of TRWPs with the aim to identify suitable marker(s). Wear particles derived from tyre tread possess distinctive a sausage shape that is exclusive to TRWPs. A range of chemical additives linked to tyres have been employed to quantify TRWPs, overlooking other potential sources such as brake wear and exhaust emissions. We found that significant amounts of 6PPD is used for the formulation of tyres, which is why 6PPD, and a comparatively stable transformation product 6PPD-quinone, could be used for the identification of TRWPs. We recommend that sampling and analysis methods be thoroughly documented to enhance the reproducibility.

1. INTRODUCTION

TRWPs are non-exhaust particles related to traffic, generated due to rolling resistance between a tyre tread and the surface of a road (Mattonai et al., 2022). TRWPs are one of several non-exhaust emissions that are of concern due to evidence of associated health problems and premature mortality (Boogard et al., 2023). Others are road wear and brake wear (Charron et al., 2019). When a brake is applied, brake pads enforce pressure against the rotors to slow down the vehicle and alter the vehicle's kinetic energy into thermal energy. Forced deceleration results in frictional heat, leading to wearing of the brake pads and the rotors (Güney and Ali, 2020; Hussain et al., 2017). The generation of brake wear depends upon the severity of braking, driving style, prevailing road conditions and composition of the brake system (Rahimi et al., 2021). However, brake wear emissions are predicted to decrease in future due to the emergence of regenerative braking systems (Liu et al., 202). A regenerative braking system takes the energy produced while slowing down a vehicle and uses it to recharge the

batteries of motor vehicles instead of wearing the brake pads. Hoofman et al. (2018) found that regenerative braking could reduce the wear of brake emissions by up to 66%. Similarly, Nopmongcol et al. (2017) suggested that a regenerative braking system could reduce brake wear emissions by 25%. However, the problem of TRWPs remains until this technology is globally widespread.

Road wear particles (RWPs) are tiny particles that wear away from road surfaces due to friction between a tyre's surface and that of road surface. These particles are found embedded in the tyre wear particles (TWPs), thus called TRWPs (Gustafsson and Johansson, 2012). However, there is inconsistency in the literature regarding the terms used for such particles. In this paper, the term tyre wear particles (TWPs) will be used for the particles generated in the laboratory without road surfaces whilst the term TRWPs for the particles generated with road surfaces and those found in the environment.

An increasing number of TRWPs and their persistence in the environment has gained the attention of scientists



because of the potentially adverse health and environmental impacts. The global emission of tyre wear particles is ~5,917,518 tonnes/year. The UK contributes about 1% (63,000 tonnes/year) of the total emission (Kole et al., 2017). However, this number is likely to grow due to an anticipated increase in the numbers of electric cars (e-cars). Timmer and Achten et al. (2016) and Kole et al. (2017) found that e-cars, as they are heavier, produce 20% more TRWPs. Despite this large emission rate, little information is available about their occurrence, pathways and sinks in the environment. This is because of limitations in present analytical methods, which have typically been designed for conventional microparticles. A lack of verified reference material for identifying TRWPs further hinders the validation of analytical methods. As tyres are comprised of organic and inorganic constituents, use of constituent(s) as a marker seems like a promising approach for the quantification of the TRWPs in the environment. While benzothiazole and zinc have been traditionally employed as markers for the identification of TRWPs, their widespread presence does not necessarily qualify them as suitable markers (Jeong and Woo, 2022; Luongo et al. 2016; Unice et al., 2013). Hence, it is necessary to completely understand the composition of tyres and the particles they shed so that their environmental risks may be evaluated.

Quantification of the mass concentration of TRWPs is a challenging task. Therefore, the identification of distinctive markers, to accurately assess the actual concentration of TRWPs, has become a focal point of research. This paper aims to identify and critically review the markers (physical and chemical) identified so far and propose potentially suitable markers for the identification of TRWPs.

2. CONTEXT

2.1 Production of tyres

An ongoing increase in the number of motor vehicles has led to an extensive production of tyres worldwide. Global vehicle ownership has reached 14.46 million vehicles (Luo et al., 2021) and correspondingly tyre production has increased to 17.85 million tyres in 2019 (Luo et al., 2021). The production of tyres reached 2000 million in 2021 and is likely to reach 5 billion by 2030 (Grammelis et al., 2021). China, as the largest tyre manufacturer, produced ~449 million tyres in first half of 2021 (Yang et al., 2022); thus contributed ~22% of the world's production of tyres (Mavukwana and Sempuga, 2022).

The primary function of tyres is to bear the weight of a vehicle and to grip the road while a vehicle is moving, accelerating, braking, and cornering (Wik and Dave, 2009). A tyre has a complex structure as shown in Figure 1. Briefly, the typical structure of a tyre comprises tread, crown ply (nylon), steel belt, body ply, carcass, and inner lining (Luo et al., 2021; Dobrotă et al., 2020). The outer layer, tread, contacts the road surface while driving and provides directional stability. Softeners, vulcanizing agents, plasticizers, carbon black and fillers are mixed in different proportions, to give the desired characteristics to the tyre composite.

The specific composition of each tyre type is different for required features. For example, winter tyres contain more natural rubber, as it provides grip on wet and snowy roads (Continental, 2019). The exact recipe of tyre composition is not usually publicly released by tyre manufacturers, but a typical general composition includes rubber polymers, softeners and anti-degradants (Luo et al., 2021; Dobrotă et al., 2020) (Table 1).

Kole et al. (2017) estimated that about 6 million tonnes of TRWPs are released in the environment annually world-

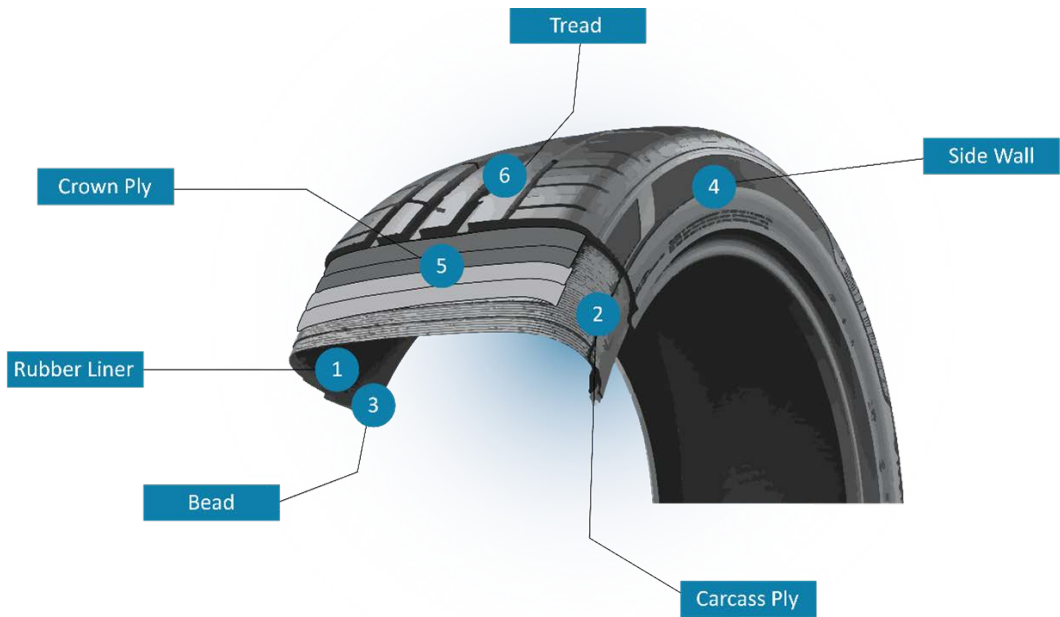


FIGURE 1: Structure of a typical tyre: 1. Rubber liner - airtight layer of synthetic rubber 2. Carcass ply- thin fibers fused within rubber 3. Bead - the boundary of tyre that contacts the rim. 4. Sidewall - vertical part of tyre that connects the tread to the bead. 5. Crown ply - provides rigid base for tyre tread. 6. Tread - designed to make contact with road and maintain optimal friction (Adapted from Dobrotă et al., 2020).

wide. The generation of TRWPs depends on a number of factors, including type and weight of vehicle, speed and road type (Table 2) (Unice et al., 2019; Kole et al., 2017; Kumata et al., 2000). This suggests that the characteristics of TRWPs are different across the world (Mennekes and Nowack, 2022). For instance, asphalt roads in European countries are reported to have a higher wear rate as compared to concrete roads (Pohrt, 2019; Gustafsson and Johansson, 2012), probably due to asphalt having particles of mineral aggregate, filler and bitumen binder (Pohrt, 2019). Bitumen particles wear away with TRWPs, increasing surface voids (Pohrt, 2019). The roughness of this texture increases with time. The rough surface area increases friction between a tyre and a road surface, leading to increased wearing of both (Pohrt, 2019). Concrete roads are made of sand and stone (granite) bound with cement (Kole et al., 2017; Lee et al., 2013) and has a comparatively smooth surface. Seasonal variations influence wear rate. Wear rate has been reported to increase up to 4% by 1°C increase in temperature. This has been confirmed by a reported 31% increased life span of passenger car tyres during the winter season (Pohrt, 2019). Similarly, wear rate has been reported to decrease during rainfall (Parker-Jurd et al., 2018). This might be due to the aggregation of particles already present on the road during rainfall, resulting decrease friction between tyre tread and the surface of a road (Changarnier et al., 2018).

2.2 Estimation of wear particles

Various emission models, employing different approaches, are used to calculate the release of TRWPs in different countries (Table 3). A commonly used approach is based on calculation of an emission factor, which is mass release of wear particles per tyre over a driven distance.

TABLE 2: Factors effecting generation of TRWPs, Adapted from Luhana et al., 2003.

Characteristics	Factors affecting the characteristics
Tyre	Size(radius/width/depth) Tyre age Tread depth Construction Tyre pressure and temperature Tyre hardness Size of tyre Tread grooves Chemical composition Rim size Tyre alignment Total distance travelled
Vehicle specifications	Vehicle type Wheel alignment Vehicle weight and distribution of load Location of driving wheels Engine power Suspension system Power/unassisted steering Axle configuration (e.g. 2 or 4 wheel drive) Electronic braking systems
Road surface characteristics	Material: bitumen/concrete/ asphalt Slope Texture pattern Texture wavelength - micro/macro/mega Presence of debris on road Porosity Condition including rutting and camber Road surface wetness Silt loading of road surface Surface dressings Temperature
Vehicle operation	Speed (average or max) Load Linear acceleration/ deceleration Radial acceleration/ deceleration Frequency and extent of braking Driving style and driving characteristics
Weather	Rain Sunlight Snow Temperature

TABLE 1: Composition of typical tyre tread and the road surface. Adapted from Sommer et al., 2018 and Krieder et al., 2010.

Particle from	Component	Percentage	Main ingredient	Characteristics
Tyre tread	Rubber polymer	40-50%	Natural rubber Isoprene rubber Butadiene rubber Styrene Butadiene rubber	- Gives elastic characteristics to the tyre - Provides directional stability
	Reinforcing agents	25-35%	Carbon black or silica	Increases the life span of tyres
	Softeners	15-20%	Humic acid (HA) oil	Helps the tyre to have grip on the road
	Vulcanizing agent	1-2%	Sulphur Zinc oxide Stearic acid	Helps in increasing the elasticity of the tyres
	Vulcanization accelerators	0.5-2%	Chemical compounds, e.g.: 2-mercapto-benzothiazole (MBT), N-Cyclohexyl-2-benzothiazole sulfenamide (CBS), N tertiarybutyl-2-benzothiazole sulfenamide (TBBS), N-Oxydiethylent-2-benzothiazole sulfenamide (MBS), Hexamethylene Tetramine (HMT), Mercaptobenzothiazole disulfide (MBTS), N,N-Dicyclohexyl-2-Benzothiazole sulfenamide (DCBS) and Tetramethyl-thiuram-Disulfide (TMTD)	- Act as a catalyst - Increases the vulcanization process
	Activators	2-5%	Zinc oxide, zinc and fatty acid	Helps in vulcanization process
	Antidegradants	1-2%	Antioxidants and antiozonants, e.g. 2,2,4 trimethyl-1,2-dihydroquinoline (TMQ), N-isopropyl-N'-phenyl-p-phenylenediamine (IPPD), N - (1,3-dimethylbutyl)- N -phenyl-p -phenylenediamine (6PPD) and N,N'-Bis (1,4-dimethylpentyl)-p-phenylenediamine (77PD)	Helps in improving durability of rubber

TABLE 3: Tyre and road wear particles in different countries, Adapted from Baensch-Baltruschat et al. 2020; Wik and Dave, 2009.

Matrix	Concentration	Country	Reference
Road dust (mg/ vehicle km)			
Road surface	72000	USA	Hopke et al. (1980)
Inside tunnel	2700	Japan	Saito (1989)
Outside tunnel	2100	Japan	Saito (1989)
20m from roadside	900	Japan	Saito (1989)
Road surface	35000	USA	Rogge et al. (1993)
Road surface	1100	USA	Reddy and Quinn (1997)
Road surface	700	Denmark	Fausser et al. (1999)
Inside tunnel	210000	Japan	Kumata et al. (2000)
Outside tunnel	124000	Japan	Kumata et al. (2000)
Road surface	56000	Japan	Kumata et al. (2002)
Calculations t/annum			
Emission factor	1327000	Germany	Wagner et al. (2018)
Emission factor	98400	Germany	Baensch-Baltruschat et al. (2020)
Emission factor	6514-7660	Denmark	Kole et al. (2017)
Emission factor	4200-6600	Denmark	Lassen et al. (2012)
Emission factor	7310	Denmark	Kole et al. (2017)
Emission factor	37646	France	Unice et al. (2019)
Laboratory wear tests	50000	Italy	Millani et al. (2004)
Emission factor	17300	Netherlands	Verschoor et al. (2016)
Emission factor	15030	Netherlands	Kole et al. (2017)
Emission factor	42000-84000	Great Britain	Kole et al. (2017)
Emission factor	20000	Australia	Millani et al. (2004)
Emission factor	294011	Brazil	Kole et al. (2017)
Emission factor	756,240	China	Kole et al. (2017)
Emission factor	292674	India	Kole et al. (2017)
Emission factor	239762	Japan	Kole et al. (2017)
Emission factor	1524740	USA	Kole et al. (2017)

Emission factors are different for different vehicle classes. For instance, Mennekes and Nowack (2022) found that a passenger car tyre loses about 117 mg of tread particles per km but this value is much higher for truck tyres. Similarly, in the study by Kole et al. (2017), the emission rate for 14 countries was estimated to range from 230,000 to 1.9×10^6 mg/capita/year. England's Environment Agency reported the average wear rate of a typical tyre of a passenger car to be 140,000 mg/metre per tyre in Great Britain (Wik and Dave, 2009). However, this wear rate has been noted to be significantly higher in city areas, owing to increased accelerating and braking (Lee and Kim, 2020; Wik and Dave, 2009). In this way, 10-40% of tread rubber (from passenger cars) is emitted into the environment in the form of wear, over the life of a tyre. Wik and Dave (2009) estimated the concentration of TRWPs at different sites (Table 3).

For Japan, it ranges from 900-210,000 mg/km. Similarly, for USA it is 1,100-72,000 mg/km. The variation in concentration of TRWPs might be due to distance from the source or due to different method used for estimation. In addition, Baensch-Baltruschat et al. (2020) calculated yearly emissions of various European and non-European coun-

tries (Table 3). The variation is probably due to different road infrastructure, different vehicle characteristics and number of cars owned by people.

A second approach is the use of material flow analysis (MFA). This method involves estimating production of tyres (by import and export) and the number of old tyres collected (Mennekes and Nowack, 2022). The average life span of a tyre is about 60,000 km before coming to an end (Mennekes and Nowack, 2022). In context of these studies, it has been found that calculation of TRWPs is based on reports and reviews instead of actual experimental work. Therefore, it is important to accurately determine the quantity and composition of emissions of TRWPs.

Furthermore, ~100 million used tyres end up as a waste globally every year and the quantity is expected to grow because of vehicle development (Wang et al., 2019). End-of-life tyres are either dumped in landfills (Gigli et al., 2019) or shredded to produce tyre crumb rubber (TCR) (Hüffer et al., 2019). TCR is produced by first removing the steel belt and then grinding the tyres (Hüffer et al., 2019). TCR is used for various applications. For example, to construct artificial turf fields in schools, colleges, practice fields and

stadiums, in mulches, to produce modified asphalt having crumb rubber, for civil engineering applications and for military installations (Benson et al., 2019; Hüffer et al., 2019). TCR may include potentially toxic constituents that possibly leach or volatilize in the environment (Sibeko et al., 2020). Therefore, human exposure to these crumb rubbers may cause various health problems; e.g. Watterson, (2017) reported that these crumb particles cause respiratory and skin problems. Similarly, Gryniewicz-Bylina et al. (2022) found respiratory problems due to inhalation of particles. According to the European Chemical Agency (ECHA), there are more than 13,000 artificial turf fields in Europe (Watterson, 2017). Millions of people are estimated to play on these artificial pitches. Compared to grass, these pitches allow people to play year-round, irrespective of weather conditions (Verschoor et al., 2016). Watterson (2017) and Benson et al. (2019) evaluated that 1,500-5,000 kg of tyre crumb is used in creating synthetic fields and ~500 kg of refill is required for each field every year, indicating that 10% of the used tyre crumb rubber from each field is released into the environment (Verschoor et al., 2016). Most of the released tyre crumb rubber is coarse in size (>50 µm) (Mattonai et al., 2022; Wagner et al., 2018).

2.3 Change in the characteristics of wear particles

Once released into the environment, TRWPs are subject to change in their physical and chemical structure due to changing environmental conditions. It has been observed that shear stress is expected to cause breakage or aggregation of TRWPs, involving both road dust and vehicle tyres (Wagner et al., 2022). Aggregation may include homo-aggregation or hetero-aggregation. In homo-aggregation, TRWPs combine with other TRWPs whilst, in hetero-aggregation, TRWPs bind with road dust and mineral particles (Luo et al., 2021). This might be due to the rough surface of the rubber particles that offers a large surface area to form aggregation (Sommer et al., 2018). The particles possess a circular cross section, enabling them to move over the surface of the road easily, thus collecting other particles (Sommer et al., 2018). Consequently, this aggregation leads to change in the density and size of the TRWPs. In addition, changing environmental conditions lead to chemical changes in the TRWPs that ultimately lead to changes in physical properties. For example, an increase in temperature leads to increased oxidation, resulting in either breaking chains or further crosslinking reaction (Wagner et al., 2018). Subsequently, cracks may develop in the structure.

2.4 Sinks and impacts of TRWPs

The size and density of TRWPs determine their fate in the environment. Being coarser in size (>50 µm), most of the TRWPs make their way to roadside soil. According to Wagner et al. (2018), about 90% of the generated particles are non-airborne and suspected to be present in roadside soil while only 10% become airborne. Järnskog et al. (2020) estimated that ~50% of TRWPs present in roadside soil wash off through precipitation, of which half reach the freshwater system and remainder goes to wastewater treatment systems (Wagner et al., 2018) and gully-pots (a part of surface water drainage system that receives sur-

face runoff (Rødland et al., 2022). Similarly, Baensch-Baltruschat et al. (2020) and Unice et al. (2019) observed that ~2% of TRWPs reach the ocean. Once in the water system, TRWPs particles may distribute in the water column depending upon the density and size of the particles. It may also depend on the density of the water (freshwater or saline water) and on the energy of the system. All these factors determine the availability of TRWPs to aquatic animals (Cunningham et al., 2022).

TRWPs may be taken up by range of organisms from algae to crustaceans and fish (Cunningham et al., 2022; Huang et al., 2021). Adverse effects have been reported in organisms and these effects might be due to physical interaction between particles and the organisms or due to the constituents (organic or inorganic) released from the particles (Wagner et al., 2018). For example, Parker et al. (2020) found ingestion of tyre particles in Striped and Spot Mullet fish. LaPlaca and van den Hurk (2020) observed mortality in mummichogs (*Fundulus heteroclitus*) and Fathead minnow (*Pimephales promelas*) following exposure to various concentrations of TRWPs ranging from 38 to 355 µm. Similarly, Cunningham et al. (2022) observed slow involuntary movements associated with the development of nervous system, change in swimming behaviour, axis malformation, hatching delays and mortality in embryos of Zebrafish (*Danio rerio*) and freshwater crustacean (*Daphnia magna*). These effects were observed following a 48h exposure to nano-sized (<1 µm) TWP (Cunningham et al., 2022). Craniofacial defects, deformed jaws, and blunt faces were observed in Fathed minnow (*Pimephales promelas*) when exposed to tyre wear particles-derived leachates. Marwood et al. (2011) reported inhibition in growth of freshwater algae (*Pseudokirchneriella subcapitata*) when exposed to different concentrations (0-10000mg/L) of leachate of TRWPs. All these factors combined make it necessary to identify constituent(s) of TRWPs that may potentially affect the ecosystem.

3. METHODS

The methodological approach for this literature review followed the PRISMA 2020 guidelines (Way et al., 2022; Page et al., 2021). In December 2023, an extensive search of database was conducted using PubMed, Scopus and the Web of Science search platforms. The search query included terms such as "elemental composition" OR "elemental marker" AND "chemical composition" OR "chemical marker" AND "physical characteristics" AND "tyre wear particles" OR "tyre and road wear particles". The search criteria was refined to include only English language papers, with a publication date ranging from 2000 to 2023.

After the database search, 638 research articles were identified, and duplicates were removed using the reference manager (Endnote), resulting in 162 papers for screening, based on titles and abstracts. Following this screening, 40 full text articles were included for further in-depth analysis. Preference was given to papers explicitly discussing the physical and chemical characteristics of TRWPs. Subsequently, two articles were excluded from the analysis as they did not clearly discuss the characteristics of TRWPs.

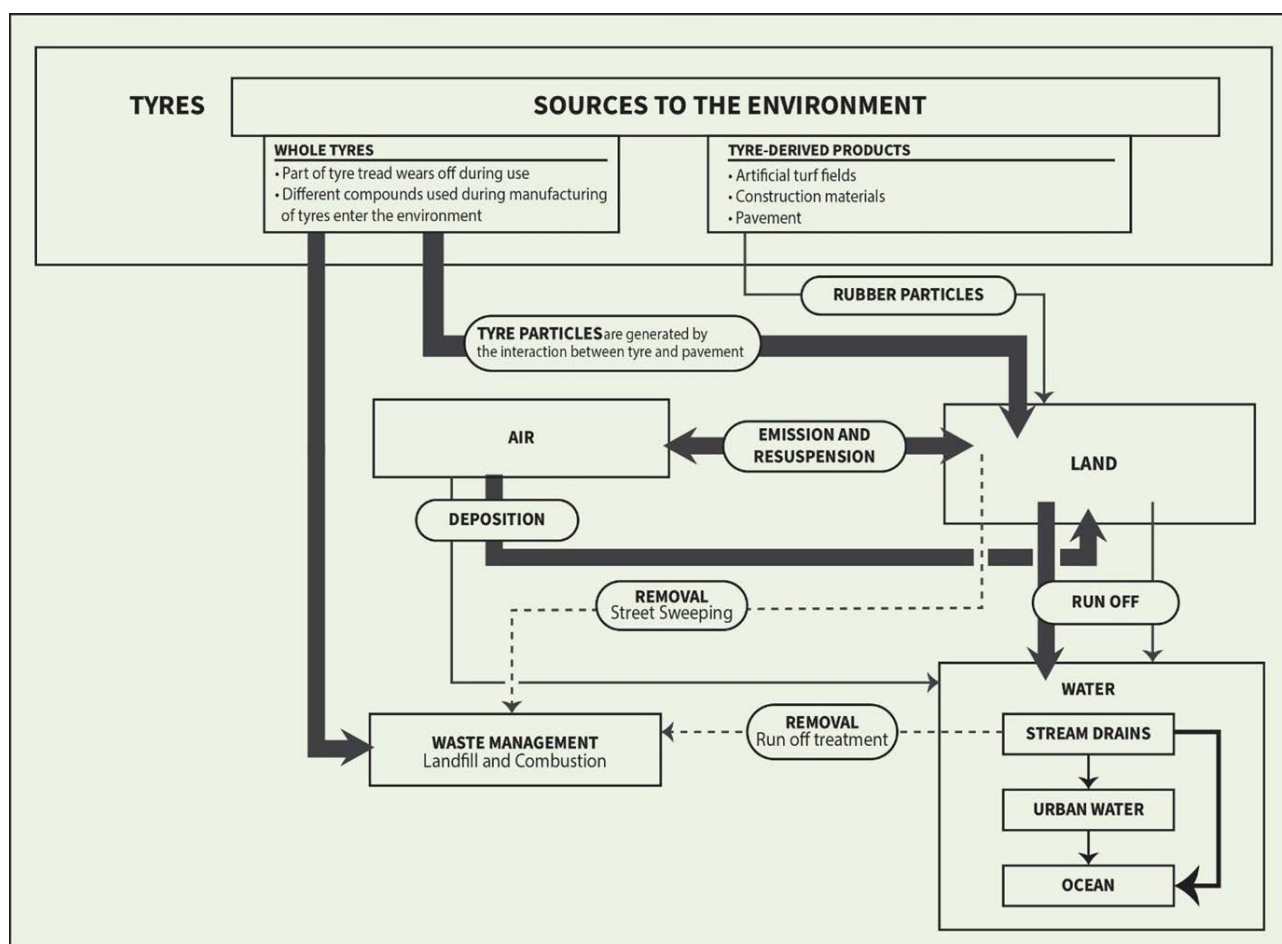


FIGURE 3: Transportation pathways of tyre and road wear particles in different environmental matrices (adapted from Mayer et al., 2024).

It was noted that the articles chosen for analysis primarily focused on particulate matter. It is important to highlight that the majority of TRWPs are typically larger. Comprehensive database search also highlighted that most of the studies on chemical characterization of TRWPs focused on vulcanizing agents and antidegradants, which is justifiable considering that ~1-3% and ~1-5% of tyre weight is attributed to protective molecules and vulcanizing agents respectively. To achieve an extensive understanding, a subsequent search using the Web of Science search engine was conducted during January 2024, without constraints on the publication date. This expanded search incorporated additional terms such as fillers, antioxidants and vulcanizing agents.

4. RESULTS AND DISCUSSION

4.1 Physical characteristics

For several years, research has focused on the identification of marker(s) for TRWPs, to evaluate the associated environmental risks. Notably, in the literature, physical characteristics have been employed for the identification. In this context, particles arising from the interaction between tyre and the surface of a road typically vary in size from a few nanometres to micrometres (Alves et al., 2020). Wear particles from tyres are produced either by tangential forces between a road surface and a tyre tread or by

volatilization (Alves et al., 2020). The initial mechanical process primarily generates coarse particles (Alves et al., 2020). For instance, Kreider et al. (2010), Kwak et al. (2014) and Knight et al. (2020) reported wear particles in the range of 50µm to 100µm. Thermo-mechanical processes also lead to generation of extremely fine aerosol particles. Mathissen et al., (2011) hypothesized that the local hot spots on the surface of a tyre tread attain elevated temperature while driving, leading to evaporative loss of volatile content of the tyre tread. Additionally, it has been noted that ultrafine wear particles are released at higher slip angle and increased speed (Alves et al., 2020).

TRWPs, based on their morphology, have been characterized into four groups. The first category comprises sausage-like/cigar-like or tube-shaped particles, commonly identified as representative forms of TRWPs (Kovochich et al., 2021; Yan et al., 2021; Kwak et al., 2014). Kovochich et al (2021) and Park and Lee, (2018) reported sausage shaped wear particles with encrustations from the road surface. Knight et al. (2020) also identified TRWPs from the roadside based on the sausage shaped appearance. These particles typically maintain their original shape in the absence of secondary wear processes (Yan et al., 2021). The second type consists of spherical particles, originating from the secondary wear processes (Yan et al.,

2021; Kwak et al., 2014). Initially, long particles are generated and later broken into smaller fragments. These particles form through shear forces resulting from friction (Yan et al., 2021; Kwak et al., 2014). Subsequently, secondary wear processes contribute to smoothing the surface of these particles (Yan et al., 2021; Kwak et al., 2014).

A third type of TRWPs consist of irregular shapes, likely formed due to intense wear processes resulting from high friction situations such as severe cornering and braking situations (Kwak et al., 2014). A fourth type is of small spherical clusters, as reported by Park and Lee (2018), with diameters <50nm. These clusters may form aggregate or attach to larger particles, making it challenging to categorize based on apparent size (Dahl et al., 2006).

Although a sausage-shaped assembly is distinctive of TRWPs, other shapes, such as spherical, irregular, and spherical clusters are not exclusive to TRWPs. These shapes may also originate from different sources, such as the wearing of brake pads and soot particles. This suggests that relying only on physical characteristics may be inadequate for identifying TRWPs.

4.2 Rubber as a marker

Given that 40-50% mass of a tyre consists of rubber polymer (Table 1), scientists found it suitable for the identification of TRWPs in environmental matrices. For instance, based on the combined content of natural rubber (NR) and styrene butadiene rubber (SBR), the reported concentration of TRWPs in road dust ranges from 23.2 to 55.0×10^4 µg/g (Miller et al., 2022). Similarly, More et al. (2023), Miller et al. (2022) and Müller et al. (2022a) observed 4-vinylcyclohexene (4-VCH) and dipentene (DP), pyrolysis products of styrene butadiene rubber (SBR) and natural rubber (NR) respectively. It is important to note that *Hevea brasiliensis* is a primary contributor of natural rubber (Venkatachalam et al., 2007). However, studies using natural rubber as a marker for TRWPs often overlook the proximity of vegetation to rule out the potential contribution from other sources of natural rubber.

Rubber undergoes vital modification to customize its properties for the production of tyres. This involves the addition of various additives to achieve the desired characteristics, a procedure termed as compounding (Rodgers and Waddell, 2005). However, additives are not chemically bonded to the polymer material, enabling them to enter the environment and potentially serve as markers for the identification of TRWPs (Johannessen et al., 2021). These additives include inorganic and organic substances. For example, metals, aliphatic and aromatic compounds, anti-oxidants/antiozonants, and vulcanizing agents.

4.3 Elemental composition

Zinc, in the form of zinc oxide, is added to tyres to help the vulcanizing process (see section 4.6) and accounts for ~1% mass of the tyre tread (Unice et al., 2013). Therefore, zinc has consistently been used as a key elemental marker for identifying TRWPs in environmental samples (Beddows et al., 2023; Beji et al., 2023; Müller et al., 2022a; Rausch et al., 2022; Choi et al., 2020; Kovoichich et al., 2021; Alves et al., 2020; Klöckner et al., 2019; Dall'Osto et al., 2014; Kwak

et al., 2013; Adachi and Tainosho, 2004). For instance, Beauchemin et al. (2021) recorded 1,258 mg Zn/ kg in a road dust sample, using inductively coupled plasma mass spectrometry (ICP-MS) as the analysis technique. However, Beauchemin et al. (2021), Ghaidan et al (2019) and Hulskotte et al. (2014) reported the presence of zinc in the wear from brake pads.

Cobalt (Co), titanium (Ti), lithium (Li), and chromium (Cr) serve as catalysts in the transformation of butadiene rubber into polybutadiene rubber (O'Loughlin et al., 2023). Barium (Ba), in the form of barite (BaSO₄), serves as a filler to enhance the ageing resistance of tyres (O'Loughlin et al., 2023). However, metals identified in TRWPs may have multiple sources on roads. For example, Co, Ti and Li are used in the engines of electric cars (Iglesias et al., 2020). Cr has been reported to be in yellow road paints (Adachi and Tainosho, 2004). Similarly, Ba is also found in paints and brake materials (Adachi and Tainosho, 2004). Therefore, Klöckner et al. (2020) raised concerns about the use of elemental constituents as markers for TRWPs and suggested considering organic constituents for their identification.

4.4 Aliphatic and aromatic compounds

Aliphatic hydrocarbon, characterized by the presence of linear or branched carbon chains, are considered essential in the formulation of tyres (Küçükosman et al., 2023). This is because hydrocarbons shield the rubber of tyre from cracking induced by UV light. Therefore, Alves et al (2020) suggested the use of aliphatic compounds (>C₃₅) for the identification of TRWPs. However, reliance on aliphatic compounds for the identification of TRWPs is questionable owing to its ubiquitous presence in gasoline. Gasoline, commonly referred to as petrol, is used as a fuel for combustion engines (Nzila, 2018). Gasoline is a liquid blend of aliphatic hydrocarbons obtained through the fractional distillation of crude oil (Vulimiri et al., 2017). Crude oil consists of a mixture of various molecules having different densities (Hsu and Robinson, 2017). The mixture is heated to extract individual fractions based on their boiling points (Hsu and Robinson, 2017). It includes heating a mixture to produce vapours which are subsequently cooled and condensed to separate the individual liquid (fractional distillation) (Hsu and Robinson, 2017). Burning of gasoline/petrol in combustion engines emits aliphatic hydrocarbons as part of the combustion process.

Aromatic hydrocarbons are groups of organic compounds that contain one or more benzene rings. These compounds are characterized by having alternating single and double bonds in their ring structure, resulting in unique stability (Küçükosman et al., 2023). Due to the stability, PAHs have been used in the formulation of tyres as softeners. Previously, polycyclic aromatic hydrocarbons (PAHs) associated with TRWPs have also been of interest. PAHs have been reported to be in fine or ultrafine TRWPs. For instance, Sharma, (2010) used phenanthrene (Phe), anthracene (Anth) and range of other PAHs to evaluate the emission of TRWPs and reported 378ng/tyre/km of PAH in PM₁₀ particles. In 2003, tyres were alleged as a major contributor of PAHs to the environment. The potential triptogenic effects (mutagenic, carcinogenic and teratogenic) of PAHs

(Yao et al., 2024) drew the attention of the European Union and subsequently it banned the use of high aromatic oils for use in tyres, as they contain PAHs (Kreider et al., 2010). Recently, Alves et al. (2020) reported PAHs in TRWPs. This might be coming from petrogenic sources. PAHs and their derivatives primarily result from the pyrolysis of fuels within motor vehicle engines (Yao et al., 2024; Ogbunuzor et al., 2020). On the surface of soot, formed during combustion, smaller PAH molecules can undergo further reactions, leading to the formation of larger and complex PAH molecules (Yao et al., 2024). They may also adsorb onto the surface of particulate matter (Ogbunuzor et al., 2020).

Considering the widespread presence of aliphatic and aromatic compounds in fuel and subsequent use as markers for the estimation of gasoline emission, these compounds may not be suitable for the identification of TRWPs.

4.5 Antioxidants/antiozonants

Antioxidants/antiozonants play a vital role in inhibiting or slowing down the degradation process of tyre rubber due to oxidative aging (Sharj-Sharifi et al., 2020). Antioxidants/antiozonants function by reacting with and neutralizing the free radicals formed during oxidative aging. This process is triggered when rubber is subjected to environmental stress factors such as light, mechanical shear and heat (Sharj-Sharifi et al., 2020). Frequently used antioxidants are N-phenyl-N'-(1,3-dimethylbutyl)-phenylenediamine (6PPD) and 2,2,4-Trimethyl-1,2-Dihydroquinoline (TMQ) (Zhang et al., 2023; Zhao et al., 2023; Hiki and Yamamoto, 2022; Klöckner et al., 2021). It has been estimated that ~54% of the yearly production of 6PPD is used as a vulcanizing agent in tyres (Zhou et al., 2023), suggesting its widespread presence in most tyres.

In static conditions, 6PPD gradually comes to the surface of a tyre or TRWPs and reacts with oxygen or ozone, creating a protective layer on the rubber surface (Sharj-Sharifi et al., 2020). The protective film effectively prevents ozone from infiltrating and interacting with the rubber (Sharj-Sharifi et al., 2020). This process of moving an antiozonant (i.e. 6PPD) to the surface of a tyre or wear particles is termed as blooming (Sharj-Sharifi et al., 2020). 6PPD may also leach into the environment and become environmentally available (Varshney et al., 2022). For instance, Hiki and Yamamoto, (2022) observed 45-1175ng of 6PPD in one gram of road dust. In a study by Klöckner et al., (2021), the presence of 6PPD was documented in road dust. Klöckner et al., (2021) collected road dust samples from a tunnel having an 80km/h speed limit. However, their sampling methodology was not discussed, compromising the reproducibility of method to verify the results. It is also necessary to analyse TRWPs to confirm the source of 6-PPD. For this reason, Unice et al., (2015) formulated an industry standard test tyre using 6PPD as an antioxidant. TWPs from a test tyre were produced through cryogenic milling, underwent weathering and were subjected to analysis using liquid chromatography-mass spectrometry (LC/MS/MS) in the mode of electrospray ionization (ESI). Analysis confirmed the presence of 1,3-dimethylbutylamine (1,3-DMBA), 4-aminodiphenylamine (4-ADPA) and 4-Nitrosodiphenylamine (4s DPA), aligning with the findings of

Zhaou et al (2023). However, tyre formulation at pilot scale may not accurately reflect the composition of wear particles present in the natural environment.

Fohet et al. (2023) exposed TRWPs (collected directly from the tyres on road) to natural aging conditions for a period of three months and found that the concentration of 6PPD decreased by 98%. It was also found that 6PPD ($C_{18}H_{24}N_2$) transformed into 6PPD-Q ($C_{18}H_{22}N_2O_2$) after 21 days of aging, suggesting that the composition of TRWPs may help in determining the age of the wear particles and the corresponding effects on the environment.

Varshney et al., (2022) found that 6PPD might negatively affect the neural system of larvae of zebrafish. Similarly, 6PPD-Q may induce mortality in Coho salmon (*Oncorhynchus kisutch*) (Greer et al., 2023). However, whilst evaluating the effects of TRWPs owing to its constituents, other possible sources of those additives should also be considered. For example, 6PPD has reported use in hair dyes and cosmetic products (Du et al., 2022).

Zhang et al. (2023) analysed the surface water samples for the targeted study of 6PPD and its transformation compounds. Water samples were supplemented with internal standards to improve the extraction efficiency and selected compounds were extracted using methanol. 6PPD-Q (maximum 1562 ng/L), along with 6PPD (max 7.52ng/L), was observed in the analysis. This not only represents the transportation of 6PPD to water channels through runoff but also indicates the potential transformation product, which is 6PPD-Q. Zhang et al. (2023) reported the presence of other potential transformation products of 6PPD namely DPPD (N,N'-Diphenyl-1,4-phenylenediamine), IPPD (N-Isopropyl-N'-phenyl-1,4-phenylenediamine), DTPD (N,N'-Bis(methylphenyl)-1,4-benzenediamine), because of the structural similarity to 6PPD. However, these identified compounds cannot be exclusively attributed to TRWPs for many reasons. Firstly, tyre wear particles have not been used as a reference, raising concerns about the reliability of the study. Secondly, the study has not considered the surrounding non-point sources of the identified compounds, indicating that the sources of the compounds has not been thoroughly investigated. Thirdly, the internal standard used may have impurities, which may cause false positive results.

Therefore, to understand the transformation mechanism of 6PPD, different researchers conducted in vitro studies. For example, Zhao et al., (2023) subjected slides, coated with industrial grade 6PPD (98% purity), to gas-phase ozone to evaluate the transformation products. Several transformation products have been identified, using liquid chromatography- tandem mass spectrometry (LC-MS/MS), such as 1,3 DMBA, 4-ADPA, 6-PPD, 4-Nitrosodiphenylamine (4s-DPA), 4-Nitrodiphenylamine (4-NDPA) and 4-Hydroxydiphenylamine (4-HDPA). For environmental applicability, road runoff samples were also evaluated; 1,3-DMBA was highest (2200ng/L) followed by 4-HDPA (150ng/L), 6PPD-Q (140ng/L), 4ADPA(110ng/L). Seiwert et al. (2022) also reported similar transformation products in aqueous phase reaction with different relative proportion, suggesting that TRWPs may be a source of these compounds in the environment. However, most of the observed transformation products of 6-PPD have various additional sources. For ex-

ample, 4-ADPA has reported use as precursor in lubricants and fuel, 4s DPA is used during production of polyvinylchloride (PVC) to inhibit polymerization reactions (Fohet et al., 2023). Therefore, 4s DPA and 4-ADPA may have multiple transformation pathways than 6PPD. Furthermore, relative high proportion of 1,3-DMBA in TRWPs and that of runoff samples indicates significant contribution from tyres to the collective environmental concentration of this compound. However, further research is required to assess the stability and significance of this component.

It is to be noted that another antioxidant that has been used in tyres, 2,2,4-Trimethyl-1,2-Dihydroquinoline (TMQ), has not been extensively examined in the existing literature.

4.6 Vulcanizing agents

Vulcanization, also known as crosslinking, is a crucial process in the tyre industry (Lubura et al., 2023; Marković et al., 2020). During vulcanization, the rubber compound transforms into a resilient product known as vulcanized rubber (Marković et al., 2020). This is achieved through sequential and simultaneous alterations of both chemical and physical characteristics of the rubber (Marković et al., 2020). The fundamental aspect of vulcanization involves establishing cross-linkages between rubber macromolecules, resulting in the formation of a three-dimensional network within rubber structure (Marković et al., 2020). The components of a cross-linked network include physical bonds (non-covalent bonds), such as hydrogen bonds, among individual macromolecule (Lubura et al., 2023; Marković et al., 2020). Interactions, including polar or dispersive forces, play a crucial role in the formation of the network (Lubura et al., 2023; Marković et al., 2020). Non-covalent interactions give flexibility and dispersive forces enhance the mechanical strength of the rubber, all crucial for the performance of a tyre (Marković et al., 2020).

Various vulcanizing agents are used to aid in the formation of chemical bonds within rubber structure. Three of the frequently used vulcanizing agents are N-cyclohexyl-benzothiazole-2-sulfenamide (CBS), benzothiazole (BT), and 1,3-Diphenylguanidine (DPG) (Weyrauch et al., 2023), as shown in Figure 3. However, all of the mentioned compounds are unstable in environmental settings (Fohet et al., 2023). It is unclear in the literature, whether CBS is a transformation product of benzothiazole or its parent compound. Fohet et al. (2023) stated that CBS transforms into benzothiazole (BT) immediately after the making of tyres (Fohet et al., 2023). Weyrauch et al. (2023) and Unice et al. (2015) reported a negligible amount (0.04%) of CBS in the extract of the particles and attributed it as a primary compound in the formulation of tyres. Zhang et al. (2018) reported various transformation products of benzothiazole and classified CBS as one of the transformation products of BT. The difference in the behaviour of CBS might be due to variations in analytical approaches and instruments used for the analysis. For instance, Weyrauch et al. (2023) used methanol for extraction of constituents from wear of tyres and used ultra performance liquid chromatography-high resolution mass spectrometry (UPLC-HRMS) for the analysis. Unice et al. (2015) used toluene as an extraction solvent and quantified the extracts using liquid chromatogra-

phy with tandem mass spectrometry (LC-MS/MS), implying the need to standardize the methodological approaches.

Similarly, BT has been widely reported in different environmental samples (Weyrauch et al., 2023). For instance, Knight et al., (2020) reported N-cyclohexyl-2-benzothiazolamine (NCBA), a derivative of benzothiazole, in drainage sumps in the United Kingdom. Wet sediments were subjected to analysis using gas chromatography mass spectrometry (GC/MS) (Knight et al., 2020). However, the specific ionization mode employed was not reported in the study. The ionization mode in analytical equipment such as GC/MS may influence the selectivity and sensitivity of the analysis of the environmental samples (Amirav et al., 2020). This could potentially impact the reproducibility and interpretation of the results (Amirav et al., 2020).

2-hydroxy benzothiazole (2-OH-BT), another derivative of BT, has been recognised as a product resulting from the oxidation of BT (Zhang et al., 2018). Laboratory studies have confirmed that the elevated temperature leads to the conversion of BT into 2-OH-BT (Borowskaa et al., 2016). Consequently, 2-OH-BT may originate from the oxidation of BT during driving, from high-speed friction (Zhang et al., 2018). However, BT and its derivatives have also been found in indoor dust as reported by Wang et al. (2013). Additionally, these compounds have been reported in textile industries (Avagyan et al., 2015).

BT has various applications (Kloepfer et al., 2004). For instance, it has reported use as fungicide and biocide (Luongo et al., 2016). Fungicides, a type of pesticide, are considered a crucial element in managing pests and diseases in horticulture (Wightwick et al., 2010). This occurs because fungal species, such as botrytis and downy mildew, can damage crops, making them unfit for sale (Wightwick et al., 2010). Nevertheless, eliminating most fungal diseases proves challenging, as outbreaks can persist over multiple growing seasons despite efforts to eradicate them (Wightwick et al., 2010). This occurs because spores may become dormant during harsh environmental conditions and start growing again when conditions become favourable (Wightwick et al., 2010). These devastating impacts of fungal diseases led to the development of fungicides. For example, experiments carried out in the United Kingdom revealed that the application of fungicide to winter wheat led to a yield increase of up to 89% (Wightwick et al., 2010). Similarly, biocides are biologically active substances employed to prevent the activities of infectious agents. These substances have various applications, ranging from maintaining hygiene in hospitals to agriculture practices, serving purposes such as disinfection and quality preservation (Kähkönen and Nordström, 2008). Additionally, these substances have frequently been detected in wastewater (Asimakopoulou et al., 2013), municipal wastewater (Kowalska et al., 2019) and household wastewater (Avagyan et al., 2015) unaffected by road runoff (Luongo et al., 2016). The extensive use of benzothiazole and its derivatives across diverse fields suggests that it may not serve as a suitable marker for identifying TRWPs.

1,3-diphenylguanidine (DPG), a bicyclic amine, is commonly used in the production of rubber as an accelerator of vulcanization. It has been associated with the pollution of

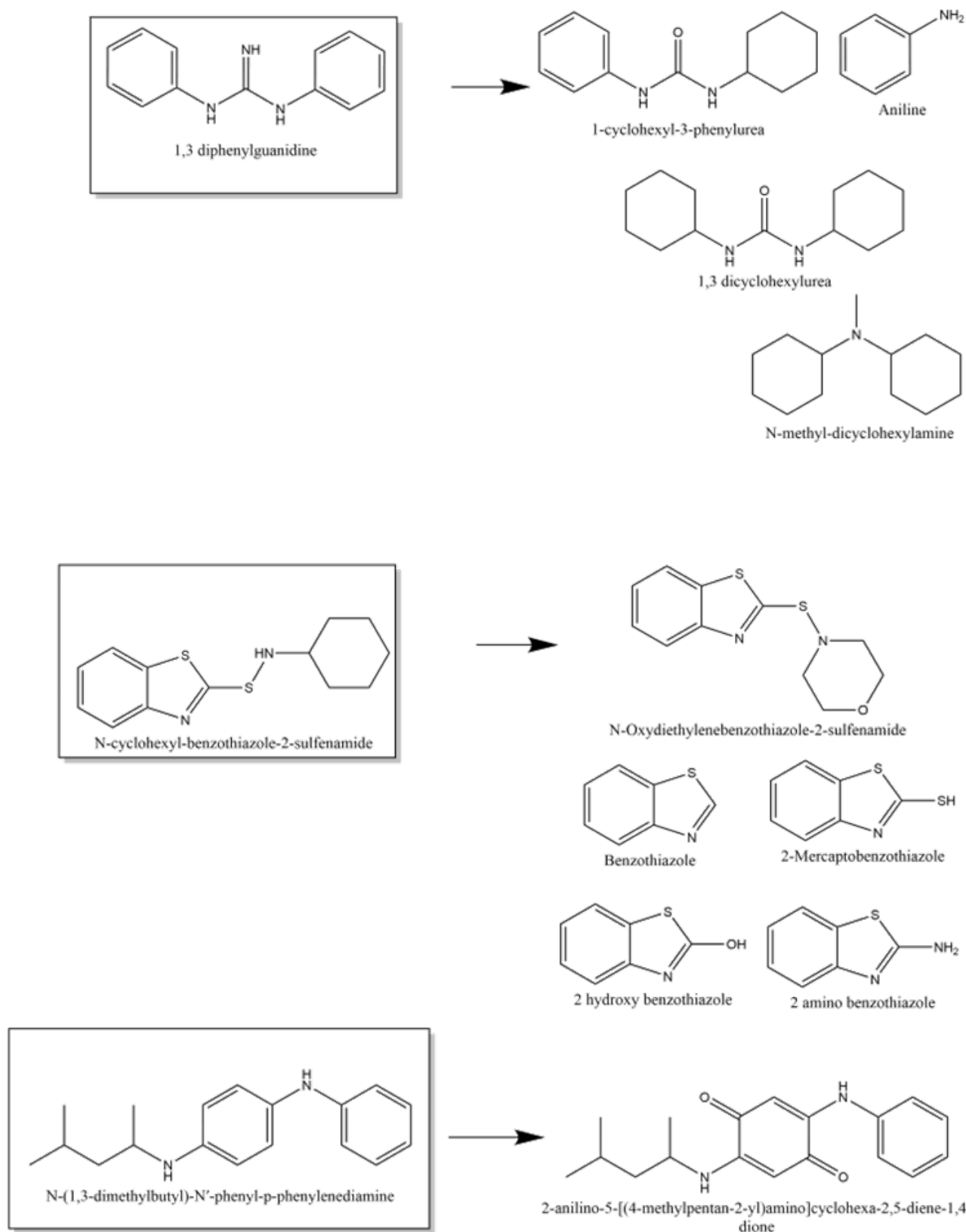


FIGURE 3: Transformation products of 1,3 Diphenylguanidine (DPG), N-cyclohexyl-benzothiazole-2-sulfenamide (CBS) and N-(1,3 dimethyl-butyl)-N'-phenyl-p-phenylenediamine (6PPD). Made by using ChemDraw software.

TRWPs (Müller et al., 2022). Johannessen et al. (2021) reported DPG in the surface water that received storm-water runoff from highways in µg/L. Moreover, Peter et al. (2018)

evaluated the leachate of tyre wear particles for DPG. Numerous compounds structurally similar to DPG were detected in the leachate, suggesting that DPG may transform

into different constituents (Peter et al., 2018). These transformed constituents include 1,3 dicyclohexylurea (1,3-DCU), N-methyl-dicyclohexylamine and N, N'-dicyclohexyl-2-benzothiazole sulfonamide (DCBA) (Peter et al., 2018). Other than these compounds, Seiwert et al. (2020) reported dicyclohexylamine in road runoff. Dicyclohexylamine is likely to be a product of hydrolysis of DCBA. DCBA consists of two cyclohexylamine groups attached to benzothiazole sulfonamide. Hydrolysis is considered to break down the bond between them, releasing dicyclohexylamine. (Peter et al., 2018). However, due to high vapour pressure and half-life of 2.9h dicyclohexylamine readily reacts with oxidizing agents (Johannassen et al., 2021), suggesting that this compound is not an ideal marker for the identification of TRWPs. Furthermore, aniline compound is also considered to be a typical transformation product of DPG (Peter et al., 2018; Unice et al., 2015). Aniline is also used as anti-knock agent in petrol (Viayna et al., 2021). Engine knocking, also known as knocking combustion, occurs when the air-fuel mixture in an internal combustion engine ignites spontaneously before the engine spark plug fires (Wang et al., 2017). This unintended ignition causes pressure waves that collide, leading to a distinctive knocking sound. Engine knocking may cause damage to the engine, causing decrease in the overall performance. Anti-knock agents, such as additives in fuels (e.g Aniline), are used to minimize knocking. These agents alter the combustion characteristics of the fuel, raising its resistance to spontaneous ignition and help to maintain engine operations (Wang et al., 2017).

Overall, it has been found that various constituents of tyre have been used for the identification and quantification of TRWPs in the environmental samples. However, most identified constituents are not ideal for this purpose. For examples, aliphatic and aromatic hydrocarbons that have been associated with TRWPs have origin from exhaust emissions as well. Similarly, benzothiazole and its derivatives have widespread use ranging from textile sector to hospitals. However, tube shaped/sausage shaped particles is an exclusive physical feature of TRWPs. Similarly, most of the world production of 6PPD is used in the manufacturing of tyres, making it somewhat suitable for the identification of TRWPs in the environment.

5. CONCLUSIONS

It is evident that identifying a single effective marker for quantification of TRWPs poses a considerable challenge. Consequently, a suite of physical and chemical markers could be used to enhance identification, quantification and impact assessment. For instance, particles having a sausage-shaped assembly could be used for the identification of TRWPs, since this shape is distinct to wear particles of tyres. As section 4.1 details the variation in shape of TRWPs, from elongated particles to spherical ones, spiking experiment should be carried out to evaluate the inadequate representation of particles other than sausage shaped. Furthermore, whilst using physical characteristics for the identification of TRWPs, zinc could function as a confirming marker to verify if the particles originated from the wearing of tyres. Similarly, 6PPD and its comparatively

stable transformation product 6PPD-Q could be used for the identification and quantification of TRWPs as more than half of the global production of 6PPD serves as an antioxidant in tyres, establishing tyres as a primary source of this compound. However, when evaluating the impact of 6PPD, it is important to consider the potential overrepresentation of this compound, recognizing that the prevalence might influence the overall representation.

Furthermore, this review of literature identified several limitations in identifying a marker for TRWPs that must be addressed when quantifying them:

- The reporting of sampling methods across studies showed significant variability. For instance, some studies have not documented the sampling method at all. The lack of comprehensive documentation regarding the sampling methods raises questions about the reliability of the collected data. Transparent reporting of sampling methodologies is important to enhance the reproducibility of the studies investigating TRWPs in the environmental matrices.
- Most of the studies included in this review did not explicitly disclose the ionization techniques used in conjunction in mass spectrometry. The absence of this vital information poses challenges for the reproducibility of the method by other researchers, as the ionization step is a fundamental aspect of the mass spectrometry process. This involves converting sample molecules into ions, which can then be separated and detected based on their mass to charge ratio (Awad et al., 2015).
- Studies focusing on the analysis of the environmental samples for targeted markers of TRWPs have not incorporated laboratory-generated tyre wear particles as a standard. This omission raises concerns regarding the potential influence of other environmental sources, as the use of laboratory generated particles serves as a critical benchmark and rules out external contribution. Incorporating such standards is essential for ensuring specificity and reliability of the identified markers in the environmental samples.

6. RECOMMENDATIONS

Based on the findings of thorough literature review, there is an urgent need to advance the future research in terms of better understanding of TRWPs. Therefore, following recommendations are proposed:

- The use of multiple thermo-analytical methods by different researchers features the inconsistency for analysing TRWPs in literature. This highlights the importance of standardizing the analytical instruments for the comparability of results.
- Artificial weathering experiments, in literature, has been conducted at 60°C; however, Kreider et al. (2010) reported the max temperature at contact point of tread and road surface to be ~180°C. Thus, future studies should also evaluate the composition and transformation products at higher temperature.
- Most of the studies have analysed targeted constituents of TRWPs without considering the possibility of

wide range of substances. Therefore, subsequent work should consider studying broader spectrum of the constituents.

- Transformation products of 6-PPD should be thoroughly investigated in aquatic ecosystem so that the potentially harmful effects and ecological risks can be assessed.
- A relatively high concentration of 1,3-DMBA in the leachate of TRWPs shows its significance. Further work is required to test if it is a degradation product of 6-PPD or a parent compound of other constituents.
- Much information about the antioxidant, TMQ, could not found in the literature, suggesting further analysis of this compound for evaluating its environmental behaviour.
- Researchers/engineers should also focus on formulating tyres that minimize TRWPs.

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CASE REPORT: MUNICIPAL SOLID WASTE TREATMENT AND ENERGY RECOVERY FOR ALTERNATIVE FUEL PURPOSE IN GERMANY

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ABSTRACT

This present case report aimed to analyze the waste sector in Germany regarding the pre-processed municipal solid waste (MSW) application for alternative fuel purposes, whose monitoring and specification of performance parameters are aligned with environmental and regulatory requirements, in order to obey emission limits and ensure the quality standards. Therefore, unlike other studies in the literature, it was made the correct differentiation about the denomination of alternative fuels in terms of their final quality, since this is directly related to the type of equipment and technology to be applied. It was observed that the lighter fractions (< 25 mm) of pre-processed waste are the most desirable for the high grade SRF (Solid Recovered Fuel) purposes in cement kilns, because they have less potential for contamination and can achieve higher calorific value and also. While the heavier fractions (< 80-100mm) are used as RDF (Refuse Derived Fuel) to denote the alternative fuel with lower burning performance. Besides, it was found that this successful performance in Germany was achieved in a long-term evolutionary process based on technological development, stricter legislation, educational actions, and cooperation between industries and society.

1. INTRODUCTION

Germany is known worldwide as a technological reference in the waste sector, however, this good performance was built on a progressive process of implementing safety measures over the years, since the late 1980s, such as the prohibition of waste disposal in landfills without treatment, the reinforcement of mechanical-biological treatment, and stricted emission limits. In addition, the proper functioning of the entire waste chain happens because of the alignment of the roles played by each sub-sector, which act in synergy and cooperation, from the consumer to the industries. Thus, the concept of circular economy, by the introduction of a more sustainable proposal of conscious consumption of energy and natural resources to the society, as well as the attribution to producers of the responsibility for products from their manufacture to their disposal by the polluter-pays principle.

This study presents an extensive overview over the most successful practices of municipal solid waste treatment and energy recovery in Germany. It is based on a qualitative approach, whose primary data was obtained by an operator survey that was carried out for the main facilities

in the country, while secondary data was obtained from literature sources and official data. In terms of methodology, a survey was first carried out regarding the main official data on the subject, and it was added secondary data from scientific articles. In addition, in order to provide new relevant information to the research, technical visits were made to six large facilities known in Germany for their excellence in quality, which technical data was provided by operators.

2. WASTE POLICY IN GERMANY

By the end of the 1980s, waste management in Germany was facing the following problems:

- Strong increasing of waste volumes;
- Large number of uncontrolled landfills with risk of contamination;
- Lack of available volume for disposal in sanitary landfills;
- Low standards for emissions avoidance (landfill gas and ground water pollution);
- Low capacities for recycling and municipal waste incineration.



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Therefore, it was urgently recognized the need of safety measures (such as emission limits for incineration activity), followed by the implementation of an efficient management that would lead to waste reduction and recovery, and therefore producers will become responsible according to the polluter-pays principle from production to waste disposal (BMUV, 2023).

Figure 1 shows the main measures adopted in Germany from 1991 to 2012, which contributed to the implementation of the circular economy concept in the country. Thus, the Packaging Ordinance was established in 1991 in order to assign the packaging return obligation to the producer, which was extended in 1996 by the 'Closed Substance Cycle and Waste Management Act' and replaced in 2012 by the Circular Economy Act. Another important milestone in the sector was the encouragement of mechanical-biological treatment and also incineration with the ending of untreated waste disposal in 2005, which was arranged by the Waste Disposal Ordinance in 2001 and was later incorporated into the Landfill Ordinance in 2009.

The Circular Economy Law is another important milestone in waste valorization, aiming to extend the life cycle of products and avoid waste generation, in order to keep materials inside the chain by recycling (European Parliament, 2023). Thus, while in the traditional linear model the products become obsolete and are replaced more quickly, the concept of circular economy introduces a more sustainable proposal regarding a conscious consumption of energy and natural resources, and consequently, lower emissions. Circular economy strategies are based on three aspects: eliminating waste, recirculating materials, and regenerating ecosystems; and can be used to combat climate change by transforming recycling solid waste into circulating materials and also, reducing waste disposal and eliminating greenhouse gas emissions with the increasing of the entire product life cycle and resulting in long-term sustainability benefits, such as the efficient use of natural resources, smaller carbon footprint and food waste reduction (Yang et al., 2023).

New plans have been adopted by the European Parliament through the implementation of compulsory goals and strict rules in order to accelerate the transition to a low-carbon economy, such as measures related to waste generation reduction, packaging reuse and biodegradable raw materials.

The European Waste Framework Directive 2006/12/EC (EC, 2006) of the European Parliament and the Council of the European Union, which was updated in 2008 by the

Directive 2008/98/EC (EC, 2008), establishes the waste reuse and recovery through the concept of hierarchy concept, which follows the order of prevention, reuse, recycling, energy recovery and disposal, and also, aims to prevent from negative impacts on the environment by establishing the obligation of waste treatment.

In Germany, the prohibition of the final disposal of waste without prior treatment in landfills in 2005 led to an increase in plants for mechanical biological treatment (MBT) and MSW incineration, as well as a growth in recycling and selective collection efforts. It is necessary to mention that such results were achieved in Germany from the awareness of companies and the population through the adoption of educational measures, both regulatory and enforcement, as well as the application of strict fines in case of noncompliance with the laws, which have been implemented and improved during the years. It is also important to understand that the successful performance of the waste sector in Germany was built from a progressive process over the years and the entire system works because of the alignment of each sector roles, which have been acting in synergy in a cooperation chain from the consumers to the industries. The municipal solid waste (MSW) separation is a successful practice that is initially done at the generating source, i.e., in the households themselves, through segregated waste bins by four types: recyclables (such as metal, plastic and packaging), paper and cardboard, compostable/biodegradable organic waste, and residual materials. According to the Global Waste Index (Sensoneo, 2022), Germany recycles 47.8% of the total waste produced and is in third place in terms of waste management regarding recycling (302kg per capita), incineration (204kg per capita), landfill (5 kg per capita) and waste production (632 kg per capita), after South Korea (1st place) and Denmark (2nd place).

Organics represents 30-40% of the municipal waste volume in Germany and have been collected separately since 2015 through the 'Biotonne', playing an important role in the circular economy by aligning recycling with the segregation of green (yard trimmings) and food wastes for composting and/or energy production by fermentation (BMUV, 2020). Although the separation of organics is mandatory and established by legislation, exceptionally in some municipalities this still does not happen. Regarding the electronic waste, batteries, textiles, lamps and glass, as well as for construction and demolition waste, there are special collection systems (collection and/or delivery) under the responsibility of the municipality.

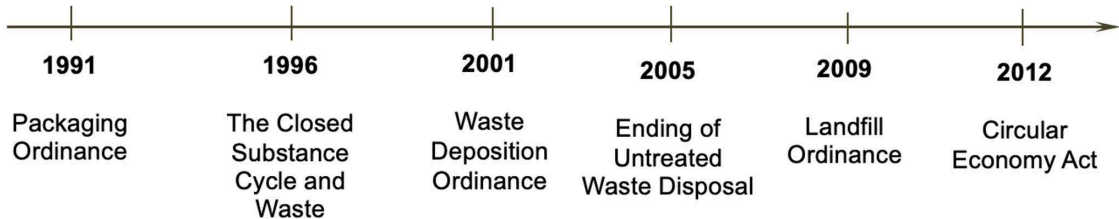


FIGURE 1: Main measures implemented in Germany's waste sector from 1991 to 2012. Source: Elaborated from information available on BMUV (2023).

TABLE 1: Waste balance in Germany in 2020. Source: Destatis, 2022.

Waste Type	Waste Generation (1,000 tons)	Disposal Operations (1,000 tons)			Recovery Operations (1,000 tons)		Recovery Rate (%)
		Landfilling	Incineration	Treatment for Disposal	Energy Recovery	Recycling	
Municipal Solid Waste (MSW)	50,993	209	365	521	15,518	34,379	98

The mandatory deposit collection system for cans and bottles was first introduced in Germany through the Packaging Ordinance in 1991 for disposable packaging, and was later adapted and implemented in 2003 with the aim of increasing the recycling rate in the country. The regulation involved much resistance from large beverage producers due to the high cost of investment in reverse logistics machinery, as well as, it was also met with annoyance by consumers due to the absence of a uniformly regulated return system (Tagesschau, 2015). The return system for beverage packaging involves the payment of a deposit (from German, 'Pfand') whose value can vary from 8 to 25 euro cents, depending on the type of use (single and/or reusable), which is later returned to the consumer when the material is returned, in which, in general, the deposit value for glass bottles equivalent to 8 cents, PET bottles equivalent to 25 cents, while single-use containers (used only once before recycling) equivalent to 25 cents (Ruíz & Cwienk, 2021).

Each consumer is responsible for his own waste, and therefore, such initiatives of separating materials in bins and collection points, although seem to play a small role, are extremely relevant to the functioning of the entire system by reinforcing the recycling activity and preserving the circular economy.

3. BRIEF OVERVIEW OF MUNICIPAL SOLID WASTE TREATMENT AND MANAGEMENT IN GERMANY

According to the German Federal Statistical Office (Destatis, 2022), it was generated an amount of 50 million tons of Municipal Solid Waste in 2020, as shown in Table 1, and regarding the Federal Ministry for the Environment, Nature Conservation, Nuclear Safety and Consumer Protection (BMUV, 2023), there are 15,000 waste treatment facilities in Germany and an estimated turnover of 80 billion euros in the waste management sector.

Mechanical-biological treatment is one of the principles of waste management in Germany (UBA, 2015b), in

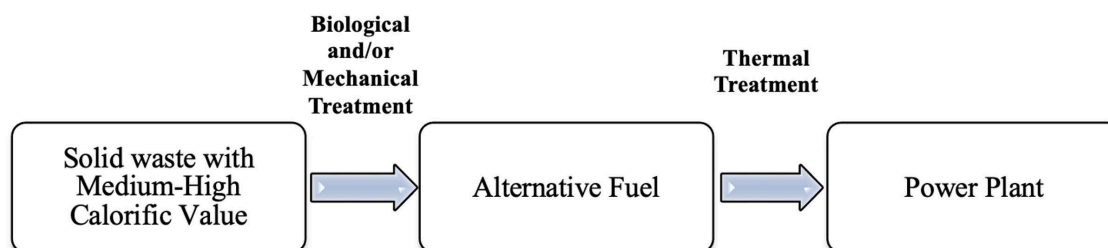
which organics, recyclables and high-calorific fraction from household and commercial origins are separated, recycled and co-incinerated (in case of no recycling potential) in coal-fired power plants or thermal production industries such as lime, cement and WtE (Waste-to-Energy) plants. The waste transformation into alternative fuel is shown in Figure 2. Table 2 lists the number of facilities for the main waste treatments in Germany in 2021.

Germany's thermal treatment facilities follow the standards of the European Directive 2010/75/EC (EC, 2010), whose incineration and co-incineration activities were first regulated for small and medium combustion units by the Federal Emission Control Act (BGBl, 2010), which was updated in 2019, and subsequently, extended to all facilities from the Ordinance on the Incineration and Co-incineration of Waste (BGBl, 2013). The contribution of energy from waste in Germany represents approximately 3.7% of final energy consumption, with most of the energy supply from waste is provided by incineration plants, along with RDF power plants (WEBER et al., 2020). According to Statista (2024), Germany had the greatest installed capacity of energy from municipal waste in Europe in 2023, of approximately 1,023MW, followed by the United Kingdom (773 MW), France (625MW) and Italy (396MW).

According to The Federal Ministry for the Environment, Nature Conservation, Nuclear Safety and Consumer Pro-

TABLE 2: Main waste treatment facilities in Germany. Source: Adapted from Destatis, 2021.

Facility Type	Number of Facilities	Waste Input (1.000 tons)
Thermal Waste Treatment Facility	155	25.493,6
Combustion Facility with Energy Recovery	481	22.088,9
Physical-Chemical Treatment Facility	390	7.711,6
Biological Treatment Facility	1.190	16.132,0
Mechanical-Biological Treatment Facility	50	3.591,2
Sorting Facility	957	24.973,6
Total	4,222	139,747.3

**FIGURE 2:** Simple scheme of solid waste transformation of medium to high calorific residues into alternative fuel.

tection (BMUV), waste is incinerated in 155 facilities in Germany, whose total incinerated waste in 2020 was 25 million tons. In these facilities the generation of heat and electricity (combined heat and power generation (CHP) or at least only electricity is mandatory. After the separation of metals (circa 10%) and impurities (inert and unburnt materials) about 95% of the remaining bottom ashes (slags) are reused as a raw material in further processes, in particular in the road construction area. The residues from flue gas cleaning are recycled or deposited in special landfills for hazardous waste. Grate fired incineration is the most commonly technology in Germany, although fluidized bed combustion is also available, while hazardous waste incineration is carried out exclusively in special licensed plants (BMUV, 2021).

According to the German Federal Environmental Agency (UBA, 2015a), there are 29 facilities in operation and 2 others in construction, as shown in Table 3, with an annual capacity of 4.7 million tons from medium to high calorific waste.

According to the CEN/TS 15359 specification from the European Standardization Commission (CEN, 2006), non-hazardous waste can be pre-processed and used as alternative fuels for energy recovery purposes in incinerators, lime and cement industries. In the past, five classes tried to classify, according to the parameters of net calorific value (NCV), chlorine content (Cl) and mercury content (Hg), as shown in Table 4.

This classification doesn't allow to re-declare waste as a product and define it for the authorities. In reality, each alternative fuel must be adapted to the respective production needs and is subject to the waste legislation and air pollution control measures equal to waste incineration plants – including power plant and cement industry.

With regard to the specification, both emission-relevant parameters (volatile until non-volatile dust bound heavy metals) and performance parameters such as calorific value, residual moisture, chlorine, ash, portion of biomass, etc. must be agreed prior and monitored during conditioning and use.

4. RESULTS AND DISCUSSION

For the purpose of this case report, visits of the main facilities in Germany have been made in order to identify the main technological practices of waste treatment and energy recovery:

- ALBA (located in Berlin) – pre-processing of waste with physical-mechanical stabilization;

- ALBA (located in Braunschweig) – waste sorting and recycling;
- KAVG in Pohlsche Heide (located in the district of Minden-Lübbecke) – mechanical biological treatment of waste and biodrying;

TABLE 3: Power plants in Germany. Source: UBA, 2015a.

Power Plants in Operation		
Name	Location	Capacity [Mg a ⁻¹]
EBS-KW	Bernburg	500,000
EBS-KW	Eisenhüttenstadt	340,000
EBS-KW	Hürth-Knapsack	300,000
KW	Glückstadt	270,000
MHKW	Heringen	270,000
HKW	Witzenhausen	270,000
HKW	Schwedt	250,000
EBS-KW	Großräschen	240,000
HKW	Rostock	230,000
IKW	Rüdersdorf	226,000
MKK	Bremen	226,000
EVE	Premnitz	150,000
TEV	Neumünster	150,000
DZ	Weener	140,000
IHKW	Andernach	114,000
TRB	Bitterfeld	110,000
IKW	Premnitz	100,000
HKW	Stavenhagen	90,000
HKW	Hagenow	80,000
IHKW	Korbach	70,000
HKW	Gersthofen	70,000
HKW	Bremen-Blumenthal	66,000
TVA	Schwarza	60,000
IKW	Amsdorf	60,000
IKW	Amsdorf	60,000
HKW	Meuselwitz	50,000
HKW	Minden	40,000
EVA	Essen	26,000
TREA	Gießen	25,000
Power Plants in Construction		
EBS-KW	Frankfurt Hoe	675,000
HKW	Brunsbüttel	165,000

TABLE 4: Classification of Alternative Fuels from waste. Source: CEN, 2006.

Parameters	Statistical Measures	Classes				
		1	2	3	4	5
Net Calorific Value	Median	≥25	≥20	≥15	≥10	≥3
Chlorine Content	Median	≤0,2	≤0,6	≤1,0	≤1,5	≤3
Mercury Content	Median	≤0,02	≤0,03	≤0,08	≤0,15	≤0,05
	80 Percentile	≤0,04	≤0,06	≤0,16	≤0,30	≤1,00

- A.R.T. (located in the village of Mertesdorf) – mechanical biological treatment of waste with biodrying;
- MBA ECOWEST (located in Ennigerloh) – pre-processing of waste to customized alternative fuels with biodrying and natural gas drying;
- HOLCIM Cement Plant (located in Beckum) – applies customized SRF via kiln end burner – one example of co-processing in the cement industry.

It must be emphasized that the financing is covered by the polluter pays principle, which was introduced for three reasons: 1) to force the polluter to pay for his large amounts of waste through high disposal fees, and 2) to make the entire chain from collection to disposal more sustainable and harmless, 3) an important factor in economy.

It is necessary to emphasize that the operations in the waste treatment and energy recovery facilities involve extremely high costs, which are financially viable due to the payment made by the municipalities from the fees paid by taxpayers (population) and waste generators (such as commerce and industry). Thus, the execution of such activities would not be feasible if these operations were paid unilaterally by these facilities.

4.1 Mechanical – Physical Stabilization – operated by ALBA, located in Reinickendorf

The facility in Reinickendorf (located in Berlin, operated by ALBA) has an annual production capacity of 180,000 tons from two Mechanical-Physical Stabilization (MPS) systems, in which municipal solid waste is transformed into an alternative fuel to replace fossil fuels in power plants and industrial facilities (BSR, 2023). The treatment process involves shredding, sorting (by material type and particle size) and drying, as can be seen in Figure 3.

In the end, the waste is transformed into alternative fuel with a particle size of less than 30 mm and the majority part is applied as a fuel in the cement industry while the other part is used for incineration at the Jänschwalde Coal-Fired Power Plant (located in Brandenburg).

4.2 Waste Sorting – operated by ALBA, located in Braunschweig

The sorting and waste recovery facility of the company ALBA in the city of Braunschweig receives only yellow bin's waste from household and commercial sources, with the capacity to recycle up to 50% of all received material, while the remaining 50% are materials with no recycling potential, which are destined for incineration. The recyclables (segregated in the yellow bins) are driven by the producer industry, especially with regard to packaging, and have a different system than commercial materials and household waste, since they are still separated at source by consumers, collected and sent to sorting units - supported by the private sector.

The unit has 72 employees and receives about 120,000 tons/year, whose sorting capacity corresponds to 60,000 tons/year. The operations are distributed over two kilometers of belts, involving the waste sorting (by type and size), with a view to separating the recyclable from the non-recyclable.

Once the waste is placed in the facility's moving floor, the packages are opened with a grinder equipment, and then the recyclables are mechanically sorted by granulometry (respectively from the largest to the smallest: greater than 300 mm, 300-150 mm, 50-150 mm, 1-50 mm) and by composition.

Among the recyclables, it is also done the separation of metals and tetra pak (with presence of aluminum and without aluminum). The ballistic separator is responsible for the segregation of the lighter residues from the heavy ones, whose mechanism operates with screening paddles that move in opposite positions and in an inclined way, allowing a more optimized classification of the material fractions. Next, such materials are reallocated to two different belts, one for receiving materials destined for recycling, while the other for disposal in incineration. The sequence of such operations is shown in Figure 4.

Additionally, it is done the manual sorting of non-reusable materials, in order to recover eventual waste with recycling potential and to dispose of only rejects to incineration.



FIGURE 3: (a) material sorting by granulometry; (b) metal separation; (c) crushing; (d) drying. Source: Author's personal collection, 2023.



FIGURE 4: Operation flow: (a) waste separation by ballistic separator; (b) reallocation of light and heavy materials on two different belts; (c) separation of reusable materials for recycling and non-recyclable materials for incineration. Source: Author's personal collection, 2023.

Although it has been observed an increase in the recycled material rate from 30% to 50% since the beginning of the activities due to the optimization of the operations, it is still very difficult to reduce the significant number of non-recyclables, possibly due to the inadequate material separation in the source, causing high costs of rejects' incinerating, which price range is from 50 to 60 euros per ton, according to ALBA's Braunschweig. In addition, it is estimated that 6 to 7% of all incoming material is equivalent to organic fraction, which is also sent to incineration. At the end of all the operation flow, the recyclable waste is baled for transport according to the type of composition, as: (a) high density polymer; (b) low density polymer; (c) paper; (d) metal; (e) tetra pak.

One of the problems in the facility is the risk of explosion resulting from Lithium batteries, causing a temporary interruption of operations from three to six times a day. Therefore, real-time monitoring is made by heat detection sensors in order to prevent fire.

4.3 Mechanical Biological Treatment – operated by KAVG, in Pohlsche Heide

Pohlsche Heide is a waste management center operated by the public company KAVG, responsible for receiving residual materials from black bins (residual materials) from eleven cities and municipalities in the Minden-Lübbecke district. The facility has about 130 employees and includes a landfill, two recycling centers, a composting plant for organic and greens, a biological treatment plant, and a mechanical biological treatment plant segregated into two processing lines, being one for household waste, while the other for commercial waste (KAVG, 2023).

The plant has the capacity to process 300,000 tons of waste per year, whose treatment steps involve shredding, size separation and sorting. Magnetic separation is also performed, distinguishing ferrous and non-ferrous materials, which are destined for a smelting plant. Waste with a higher calorific value is used as an alternative fuel (HCF

– High Calorific Fraction) for thermal energy purposes in KAVG's cogeneration Energos plant in Minden, thereby partially replacing fossil fuels (such as gas or coal).

The organic waste is treated in the anaerobic digestion unit to produce biogas, which is converted into heat and electricity in the thermal power station and reused for power generation at the Pohlsche Heide facility itself. The output of this unit is together with the treated non-recyclable waste disposed of in the landfill.

The facility has the capacity to receive 100,000 tons of household and commercial waste per year. After receiving the material, the waste must follow strictly distinct routes, one for household and another for commercial, since they involve different fee systems. Thus, in case of losses in the commercial waste line, for example, it does not cause impacts on the household waste line. Although the maximum capacity of the facility for the commercial waste route is 40,000 t/year, the unit has been receiving only 14,000 t/year because of the low price (disposal fee) paid to the incinerators. For the unit that treats commercial waste the ideal would be to receive approximately 135 euros per ton of waste, however, it has been paid to the incinerators from 80-90 euros per metric ton of waste. On the other hand, new investments have been made in the commercial waste treatment line in order to adapt the techniques and equipment to comply with the standards established in regulations. All the operation costs are financed by disposal fees (polluter-pay principal), which are proportional to the number of residents in a household, and also by the industries.

The mechanical biological treatment unit is responsible for pre-processing the waste higher than 60 mm through the stages of sorting, grinding, drying, magnetic separation and ballistic separation, which are destined for incineration in order to produce steam for the pharmaceutical industry. The pre-processed residues with a final particle size of 60-120 mm and a calorific value of 12,000 J (about 2,866 Kcal) are destined for incineration, of which 35,000 t/year are destined for a local incinerator in Minden (in a distance of

15 km) and about 15,000 t/year for an external incinerator. Approximately 3-5% of the total amount of waste received are metals, which are separated by magnets into ferrous and non-ferrous by eddy-current separator, and sold to the market.

In the aerobic biological treatment process, the waste below 60 mm is stabilized by bacteria in boxes under a controlled environment to be later disposed of in the landfill. While in the anaerobic treatment process, fermentation and biogas production of the smaller fractions of waste is performed, which is treated and distributed to the gas network.

The inadequate separation of materials by the population causes some problems to the facility, such as the high concentration of plastics due to the packaging of organics in inappropriate bags and also, the risk of explosion during the waste shredding process, due to the presence of lithium batteries, which can motivate from two to three weekly stops in the operations, even if eventually.

It has been a long history until Germany becomes a world reference in the waste sector, coming from overloaded landfills to presenting treatment and energy recovery facilities. Despite successful recycling activities, landfills are still needed for the disposal of materials without thermal and recycling potential according to the disposal criteria of the Landfill Ordinance. One of the future goals is to enable recovery alternatives for such materials without reuse, so that the numbers of sanitary landfills can be reduced furtherly. However, there is a consensus about the value of such materials and, therefore, the will to return them, somehow, to the reverse logistics cycle. Even though there is the practice of waste avoidance and separation in the source (households), most of such materials are still destined for incineration, because the infrastructure is still insufficient for more advanced recycling capacities or because the quality is still poor and costs for reuse are still too expensive and thus it becomes cheaper to use new ("virgin") materials than recycled ones. In a way, many materials even containing recyclable components are still going to incineration.

Thus, there is still a long way to go in the country before it is possible to extend the life cycle of products and comply with circular economy principles. Some industry conversations involve discussions about how producers can in the future bring such materials back into the circular economy circle from their material components - such as chemical recycling - without the energy recovery process, for example.

4.4 Waste Stabilization by Biodrying – operated by A.R.T., located in Mertesdorf

The A.R.T.'s facility in the locality of Mertesdorf is one of the plants with biodrying technology in Germany. The unit has 300 employees and receives daily 450 tons of residual materials from a district corresponding to 530,000 inhabitants and is responsible for the sorting of waste from black bins (non-recyclables) on 75 hectares (750,000 m²). Besides the landfill, there are also units for the mechanical biological treatment and biowaste composting, a waste collection point and an energy recovery unit.

First, the truck is weighed on the scale and according to the measure it is identified the gate to be unloaded, and the waste is then relocated for the shredding procedure.

Afterwards, the drying process is started through biological stabilization boxes (Figure 5), which lasts from 9 to 12 days, whose technology allows the removal of 30 to 35% of the residues' humidity for the purposes of calorific value increasing and mass reduction and consequently, lower transportation costs to the incinerators.

Finally, is conducted the sorting of materials, such as organics (later disposed of in landfills) and metals - which are separated into ferrous and non-ferrous, and then sold to a company, enabling the generation of revenues.

One of the problems faced by the facility is the risk of self-combustion of batteries, which are found due to the incorrect waste segregation still at the source (households). Thus, in order to prevent fire, a real-time monitoring is performed by heat sensors located both in the biological drying area (to detect the risk of temperature rise) and in the mechanical part (to detect eventual batteries).

4.5 Waste Pre-Processing – operated by MBA ECOWEST, located in Ennigerloh

The MBA ECOWEST facility in Ennigerloh receives about 150,000 tons per year of waste, of which 60,000 t/year is presorted commercial waste and 90,000 t/year household waste. Because of the different compositions, the flow of operations is planned for two process routes

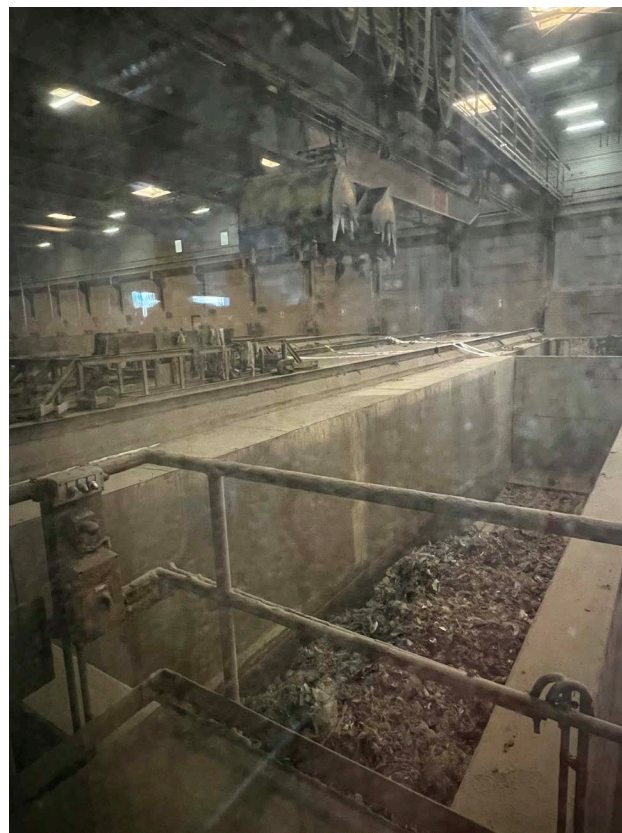


FIGURE 5: Biological drying stabilization. Source: Author's personal collection, 2023.

on separate feeding belts, and therefore, it is possible to ensure the most appropriate treatment for the type of incoming material and consequently achieve better quality for the alternative fuel production.

At the beginning of the process, the waste is unloaded at the facility's moving floor and then relocated to two different types of the shredders on separate routes, for household and commercial wastes. During the passage of the waste, a magnetic separation is also mandatory here (Figure 6), whereby attention was paid to a sensible positioning. The orientation, which is important for the extraction performance and the quality of the iron, was in the conveying direction, which means that the magnetic field captures the ferromagnetic particles more cleanly in their point of weightlessness and without being covered by impurities.

The facility has replaced the useless compost screens against three large flat screens followed by proper wind shifters that are responsible for receiving two different waste streams on three lines. From each, lighter materials are separated at the top as the high-calorific fractions and which are used in the downstream pre-processing, while the unsuitable heavier materials at the bottom will be sent to waste incineration in the cities of Hamm and Bielefeld.

Next, the ferrous materials (corresponding to 2-3% of the total input stream) are removed by the magnets, and at the end, the non-ferrous materials are also removed by an eddy-current separator. In accordance to the local cement industry, which does not operate precalciners and the local fossil-fired power plants, RDF cannot be produced or even used, here. So, the fuels have to be cleaned of heavy materials, which is done by sufficient air classifier. If necessary, the SRF is dried in a drum dryer using natural gas.

The use of appropriate equipment for conditioning will be monitored by sampling which enables a more accurate and representative knowledge in relation to the quality of the pre-processed waste, since the collection is performed randomly by the machine, instead of manual selection by an operator. This procedure is very important to ensure that

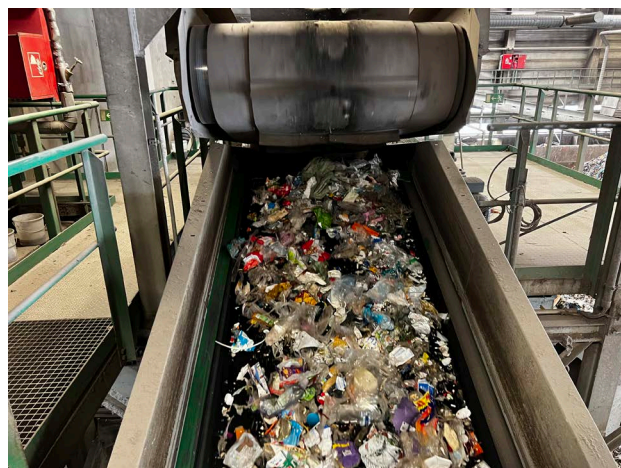


FIGURE 6: Magnetic in the same direction as the belt is conveying. Source: Author's personal collection, 2023.

the limits pre-determined by law are achieved, in accordance to the permission of the customers, and thus obtain a calorific value appropriate to the technology to which it will be applied.

Therefore, the collection of samples in the MBT, as shown in Figure 7, takes place regularly and combined into a daily mixed sample. It will be analyzed in the laboratory in order to compare them with the qualities in the final product agreed with the customer to be supplied. This MBT is the one of the plants that has been awarded the RAL quality mark 724, which means that the heavy metal content in the SRF corresponds to the permissible emission limit values of co-processing and can directly supply without any blending prior to delivery.

The lighter fractions of pre-processed waste are the most desirable for the high grade SRF purposes in cement kilns, because they have less potential for contamination and can achieve higher calorific value, allowing a better burning performance in suspension after leaving the kiln



FIGURE 7: Sample collecting for quality standardization purposes. Source: Author's personal collection, 2023.

end burner or vessel burners. While the heavier fractions are used as RDF with lower burning performance, eventually becoming ash.

It is very important to have the correct differentiation about the denomination of alternative fuels in terms of their final quality, since this is directly related to the type of equipment and technology to be applied. Therefore, the term “alternative fuel” is usually more broadly used within the German technical area, while the following specific ones are considered:

- SRF (Solid Recovered Fuel), as shown in Figure 8, for the alternative fuel composed of fine fractions free of 3-dimensional heavy particles (< 25 mm) and primary quality, similar to lignite with application in the sinter zone burner in the cement industry and the vessel burners in coal fired power plants;
- RDF (Refuse Derived Fuel) to denote the alternative fuel composed of larger fractions (< 80-100mm) and of poorer quality, with application in the calciner in the cement industry or Fluidized Bed Combustion.

In the cement industry, the alternative fuel composed of finer particles and primary quality (SRF) is used in the main burner, while the alternative fuel of larger particles and poor quality (RDF) is used in the calciner, while very large particle (HCF-High Calorific Fraction) are fed to the pre-combustion chamber or processed to RDF, as it is shown in Figure 9.

4.6 Waste Energy Recovery – operated by HOLCIM Cement Plant, located in Beckum

The cement plant in Beckum-Kollenbach (Figure 10) was founded in 1911 and has been part of the Holcim Group since January 2015. Around 14 tons per hour of SRF through the kiln end burner. A thermal substitution rate (TSR) is equivalent to 80% is achieved, due to the use of customized alternative fuel and animal meal (biomass).

In general, the cement manufacturing process starts first with the raw materials (limestone, sand as silicon



FIGURE 8: SRF, alternative fuel composed of light fractions. Source: Author’s personal collection, 2023.

donator, iron ore and aluminum oxide containing clay) are mixed and milled, and then preheated respectively pre-calcined. Then the decomposed raw meal will be burned in a rotary clinker kiln at 1,450°C (Holcim, 2023). In the main burner, coal, alternative fuel (SRF) and animal mill are added, while in the calciner the secondary alternative fuel (RDF) is used. After burning, the clinker is immediately cooled and ground. It should also be noted that the ash is also chemically integrated as a component in the clinker burning process, which is why it is also referred to as “co-processing”.

The decarbonization process in cement works, in which CO₂ is removed from the lime-based raw material, takes place at high temperatures (from 800 to 850°C) and is responsible for one part of the CO₂ emissions. The other part is caused by the combustion of the fuels, which is why seeking for alternatives as to the use of limestone (calcium carbonate) and its calcination through the use of alternative fuels (including biomass) is very important worldwide.

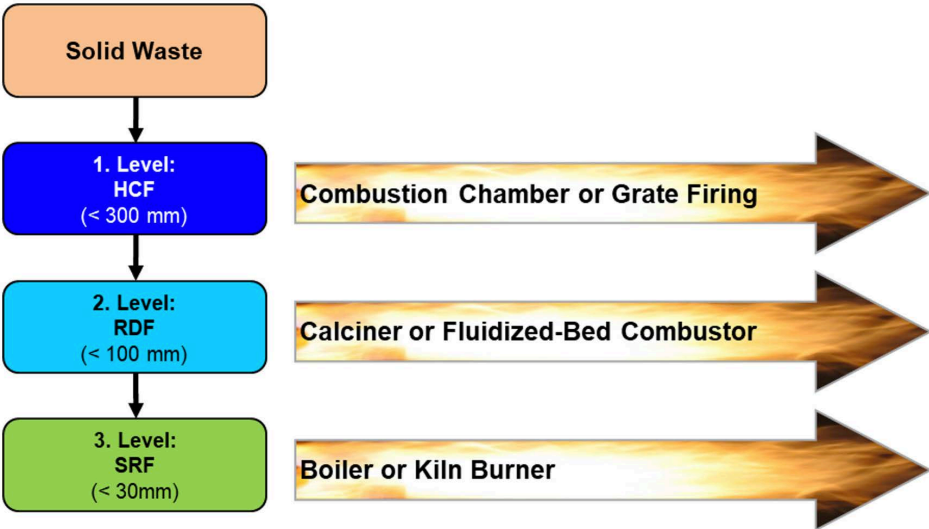


FIGURE 9: Application of alternative fuel according to quality. Source: WLTP, 2012.



FIGURE 10: Holcim Cement Plant in Beckum, Germany. Source: Author's personal collection, 2023.

The clinker burning process requires a lot of energy, which is why it is a constant search for cost-effective alternatives. The plant requires around 3,600 tons of raw material and 600 tons of fuel per day to produce around 2,800 tons of clinker. Lignite dust is stored in four silos, two smaller storages are used for the continuous substitution of the coal dust with alternative fuels. The kiln is of normal length but without a pre-calciner. Via the kiln inlet there is the possibility of giving up small quantities of lower quality (but with a high gate-fee). At the kiln end, the long flame of the sinter zone burner (of almost 20 meters) generates the required high clinkerization temperature which can reach 2,000-2,500°C gas temperature (the clinker formation temperature applies approx. at 1,400°C), which requires a SRF quality similar to lignite.

The bypass technology, which will become necessary, when the suppliers are not willed or in the position to control and reduce that process disturbing chlorine entrance during preprocessing. Cold air is used to rapidly cool the gases just leaving the kiln (at a temperature of 1,100-1,200°C), where chlorine is condensed and removed by combination with alkaline elements. The installation of such a quenching system and the cooling with additional air costs a lot of energy and money, respectively, which has to be compensated by the gate fee to be paid by the supplier in case of delivery of insufficient AF qualities.

The emission limits in Germany are adapted from the municipal waste incinerators and extremely strict and are directly related to the facility's operating permit. Combustion-relevant data are tracked in real time by the authorities. Therefore, the measurements are made for fourteen different chemical elements every 30 minutes (half-hour average of emissions), currently per moment average) and per day (daily average). It is mandatory by law to measure once a year such as dioxin and furane which cannot be detected constantly and elaborate a report. In the case of any irregularities the facility is notified by the authorities to stop its operation in order to correct any deviations. It is also possible to be fined and/or lose the permission in case of non-compliance with the law and failure in presenting an appropri-

ate justification to the environmental inspection agency. The adequacy of technologies in cement plants in the country occurs aligned with the updated legislation, especially in the implementation of new emission limits in 2013.

One of the biggest problems faced by cement plants is the large amount of CO₂ emissions related to the calcination of raw material to produce clinker, whose process aims to reduce CaO content in the clinker. Much has been discussed in the sector regarding new strategies to reduce emissions through the optimization and/or adoption of new technologies, such as the application of hydrogen in the clinker formation process, oxy-fuel combustion and the use of calcined clay to replace clinker. Such adaptations in the chemical composition of lime are complex and require caution, since any change in this process, however small, implies major impacts on the burner, the clinker mineralization and also on the fuels. For this reason, chlorides are a major challenge in the industry, since they present several combination/fixation mechanisms for the salts' formation, whose stability can be influenced by the presence of acid aggregates or mineral additions, causing a decrease in pH and consequently, a reduction in the cement resistance (Saciloto, 2005).

Another future possibility to mitigate CO₂ emissions in the sector, although it is in early studies, is a solution called 'End-of-Pipe Technology', whose carbon capture happens from the removal of CO₂ at the end of the process. Although the cement manufacturing process has existed for many years and has been constantly optimized and new technologies developed to maximize energy efficiency, such as pre-calciner, calciner expansion, low-NOx burners, SNCR, energy recovery by modern clinker cooler technology, waste heat recovery, etc., it is at the edge of a new technology, for example, hydrogen utilization and CO₂ capture and use (CCSU). Ultimately, infrastructure measures (such as roads, schools, hospitals, streets, sewers, etc.) must remain affordable.

Finally, it was observed that the main barrier to the use of pre-processed waste as an alternative fuel in cement plants is to be able to adjust it in the production of clink-

er within the limits required for the final quality, in order to maintain control of the entire process, and also to ensure the best price per ton of clinker in harmony with all costs and risks from the sector. The cement market is very dynamic and has a constant volatilization, such as price oscillations in the carbon market, the recent authorization regarding the biomass use (saw dust, animal meal etc.) and the mandatory emissions trade certificate for incinerators, which is expected to be effective in 2024, referring to payment for each CO₂ ton that has been emitted - estimated at 35 euros per ton of waste-derived fuel. Thus, it is extremely important to align the industry to such updates, especially the regulatory ones.

5. CONCLUSIONS

This present case report aimed to analyze the waste sector in Germany in order to study the main measures that led to the good performance of the entire chain, as well as to identify the best waste treatment and energy recovery practices in the country. It was found that this successful performance was achieved in a long-term evolutionary process based on technological development, stricter legislation, educational actions, and cooperation between industries and society. The prohibition of untreated waste disposal in landfills, the implementation of the circular economy concept, the establishment of the polluter-pay principle, the consolidation of mechanical-biological treatment and the thermal treatment, recycling activities, as well as, household waste separation and selective collection, all played an important role that led to a successful management.

It was also possible to observe the pre-processed waste application for alternative fuel purposes, whose monitoring and specification of performance parameters are aligned with environmental and regulatory requirements, in order to obey emission limits and ensure the quality standards.

Thus, it was shown that is very important to have the correct differentiation about the denomination of alternative fuels in terms of their final quality, since this is directly related to the type of equipment and technology to be applied. The lighter fractions of pre-processed waste are the most desirable for the high grade SRF purposes in cement kilns, because they have less potential for contamination and can achieve higher calorific value, allowing a better burning performance in suspension after leaving the kiln end burner or vessel burners. While the heavier fractions are used as RDF with lower burning performance, eventually becoming ash.

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For those who believe in science.

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PHOTOVOLTAIC PANEL RECYCLING: TESTS IN THE PILOT PLANT IN VENICE AND SILICON APPLICATIONS FOR FERROALLOY PRODUCTION

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ABSTRACT

In 2020 9-Tech developed one of the most advanced technologies in Europe for recycling PV panels. In detail, the proposed technology is able to recover all the valuable materials contained in PV panels (aluminum, glass, copper ribbons, silver, silicon PV cells). The pilot plant, authorized by Veneto region in 2022, has been operating in 2023 showing good performances. The recovered PV cells contain mainly silicon that is a Critical and Strategic Raw Material, but the presence of aluminum paste and silicon nitride makes challenging its reuse. Therefore, several uses for silicon are being investigated under the PARSIVAL project, specifically in the photovoltaic sector, battery anode manufacture, and ferroalloy production use. In this work, the results of the tests in the pilot plant with the 9-Tech technology and the main findings regarding the possible use of the PV cells as reagent in STILLMETAL process for the production of ferrosilicon alloys are presented.

1. INTRODUCTION

The technological methods used to produce energy and in particular the possibility of employing renewable energies play a key role in the way through reaching carbon neutrality. (Li et al., 2023; Li and Li. 2018). Among renewable energy source, photovoltaic (PV) technology is one of the most promising and greenest. The global solar PV modules reached an installed capacity of 1 TW at the end of 2022 (IRENA, 2023) and according to the International Renewable Energy Agency this value is predicted to increase to 5 TW by 2050 (Weckend et al., 2016). Although this expansion is considered as necessary for the transition to a low-carbon energy future, it also introduces new issues, primarily linked to the economic and environmental sustainability of the whole PV supply chain, particularly the End-of-Life (EoL) management of PV modules. The usual lifetime of a PV panel is roughly 25-30 years, and since the first installations date back to the 2000's, a significant rise in the yearly flow of EoL PV panels may be expected in the early 2030's (Wang et al., 2022; Weckend et al., 2016). Considering the rapid increase of the PV waste generation, as can be observed in the graph reported in Figure 1, the recovery of precious materials is an urgent and critical issue that requires both cost-effective and sustainable solutions.

The majority of installed PV technology is based on crystalline silicon modules containing glass, aluminum, silver, copper and amounts of tin and lead.

The amounts of raw materials recovered from PV recycling should be reintroduced into the market to produce new PV panels or other items, hence increasing the security of future PV supply or other products that rely on raw materials used in PV modules. In particular, metallic silicon, a key component in PV modules, has been classified as a Critical Raw Material (CRM) by the European Commission because of its importance in the electronics and solar industries, its potential to improve Lithium-Ion battery manufacturing for increased energy storage, and the risk associated with its supply, which is largely controlled by China. Furthermore, the creation of crystalline silicon is a high-energy process, therefore, silicon recycling becomes crucial (Cerchier et al., 2022).

Nevertheless, PV modules are made to last 30 years outdoors but are not intended to be recycled. In reality, disassembling the module sandwich to retrieve the various elements included is a significant difficulty due to the employment of strong encapsulation polymers that glue and preserve solar cells. As a result, mechanical shredding is the only commercial method available today to achieve

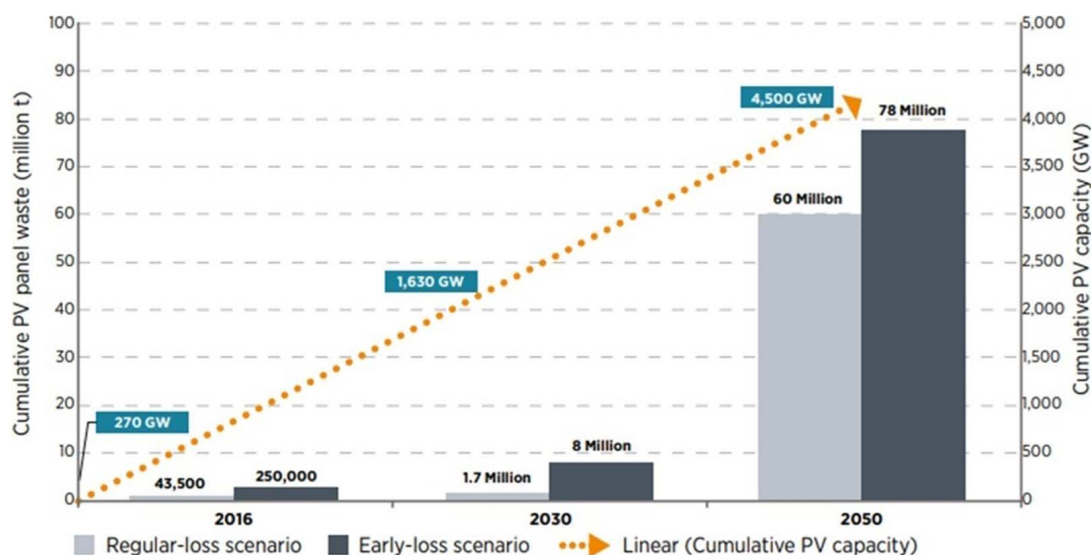


FIGURE 1: Estimated PV waste in the next year, based on PV panels capacity installed (Weckend et al., 2016).

the EU WEEE Directive's mandatory recovery rate (80% by weight) for End of life PV panels. However, different recycling processes have been studied and proposed in order to overcome the problems related to mechanical shredding and recover all the materials contained in End of Life panels. These methods can be classified as mechanical, thermal, and chemical (Cerchier et al., 2022; Wang et al., 2022).

Mechanical methods seem to be in higher number in comparison to thermal and chemical ones, however all the currently available techniques resulted quite widely studied in literature from the point of view of the technical issues, economical advantages and disadvantages and waste management issues (Shahariar Chowdhury et al., 2020; Preet and Smith, 2024; Vinayagamoorathi et al., 2024; Sah et al., 2024).

From 2020 a new process, which comprises a thermo-mechanical treatment was developed and tested at pilot scale. Specifically, the technology can correctly separate glass, characterized by high purity level, copper ribbons and pieces of silicon cells. These cells can further be treated by hydrometallurgy processes to recover the silver. It would be important to exploit such technology in many regions facing the issue of PV waste. In particular, Apulia in Italy and Extremadura in Spain are RIS regions (i.e. countries with moderate or modest innovation scores as defined by the European Innovation Scoreboard) characterized by an intensive installation of photovoltaic (PV) panels and in the next 15 years a large amount of PV waste is expected.

In details, Apulia (South of Italy) is the second region in Italy for installed PV power capacity, 3.3 GW (GSE, 2023) 10.9% of the national installed capacity, and it is a Regional Innovation Scheme (RIS) region where the matter of End of Life PV panels is particularly relevant: ~300k tons of PV waste will be collected in the next 15 years. The case of Extremadura, with 3.9 GW of installed capacity (23.4% of the PV solar energy of the country), is similar. In both

these regions there are still no recycling plants and it is extremely urgent the realization of a structured value chain for the recovery and reuse of valuable materials contained in PV.

PARSIVAL is a RIS (Regional Innovation Scheme) capacity building project financed by EIT RawMaterials that aims to solve this issue by proposing this new technology for recycling End of Life PVs and promoting a circular value chain for PV in these regions. The aim of PARSIVAL project is also to study the most profitable routes to valorize silicon recovered from the recycling of PV panels and three different applications for the recovered silicon: Li-ion batteries, ferroalloy, and PV manufacturing. The project, aims to build up a solid network able to support PV panels recycling, exploit results in Extremadura and increase the skills of students in Apulia on the themes of recycling, recovery, reuse and substitution of CRM. The global idea of the project can be found summarized in Figure 2.

In the present work there will be briefly described the performance that the pilot plant in Venice shown during the tests conducted and the first results of PARSIVAL project in terms of reusing the recovered silicon in the production of Si-ferroalloy. Both these results are novel considering that both the technology for photovoltaic panels recycling and the silicon application in the production of Si-ferroalloys are new. Furthermore, for the first time these two processes are being tested on a pilot scale to verify whether the silicon is suitable for the addressed application. Positive results can pave the way to industrialization of efficient recycling process and to the reuse of metallic silicon in the huge steel industry market.

2. MATERIALS AND METHODS

2.1 Test in the pilot plant in Venice

Thank to financing of EIT Raw materials, the largest consortium in the raw materials sector worldwide (EIT 2024) since 2020, 9-Tech developed a technology that allows to recover all valuable materials contained in the PV

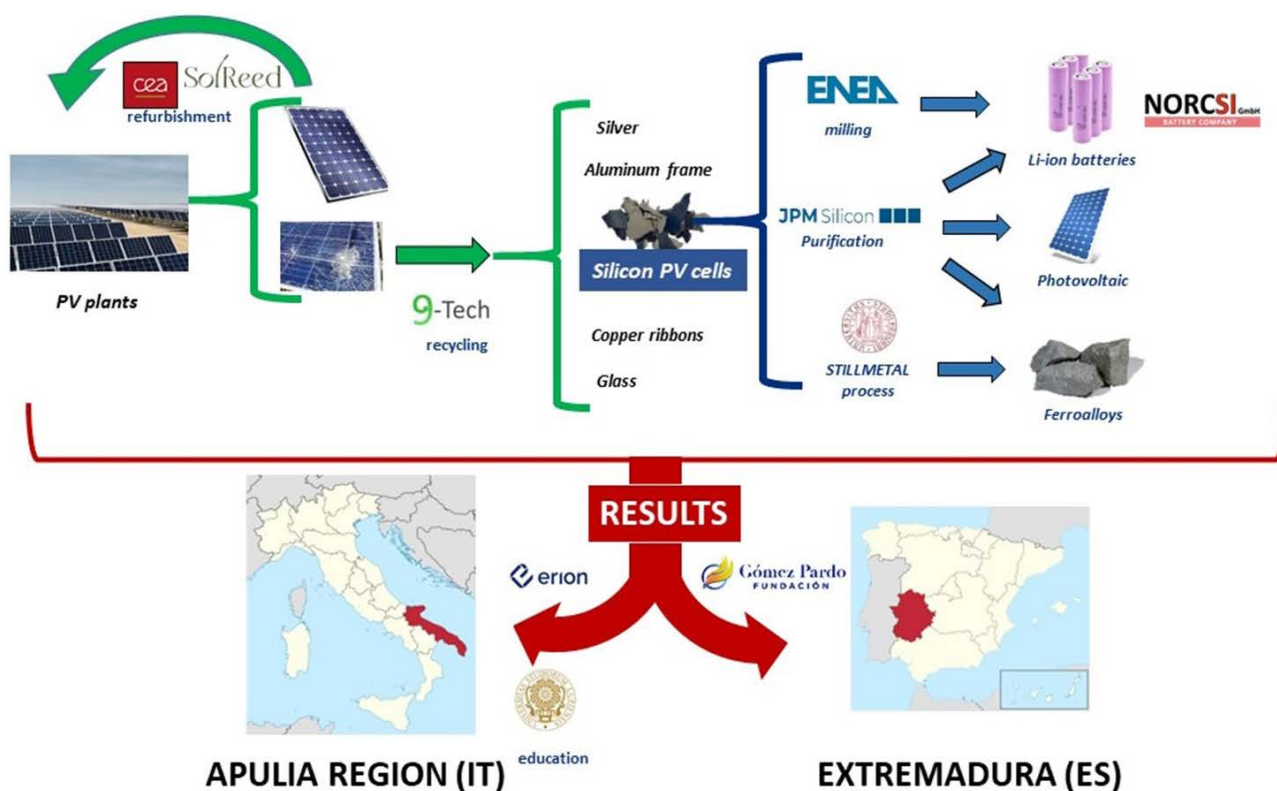


FIGURE 2: Scheme of PARSIVAL project.

panel with high yield and quality in comparison to other methods currently available.

In Figure 3a is reported the process flowchart of 9-Tech technology. The process begins with the removal of the aluminium frame and junction box. Then, after cutting the panel, a thermal treatment removes the polymeric fraction and delaminate the PV panel. After this step, glass, copper ribbons, and silicon solar cells are mechanically separated. The silicon cells, after mechanical separation, can be subjected to a hydrometallurgical treatment to extract the silver.

The advantage of this approach is the reduction of the inter-contamination of the recovered materials, that can compromise their reuse, in particular for glass. On the other hand, the possible disadvantages of thermal approach are due to energy consumption and fumes treatment (Cerchier et al., 2021).

Nevertheless, with the objective of demonstrating that the energy consumption can be contained and fumes treated without critical issues, since 24 October 2022 a pilot plant performing the first part of the process (thermal and mechanical treatment) has been operating in Porto Marghera, Venice. This plant, whose results are hereafter presented, is shown in Figure 3b. The pilot plant comprises two containers that occupy respectively 15m² for the dismantling and waste stocking area and 17.5 m² to host the furnace and separation machine. There are also 14m² for the air compressor, PLG tanks, heat exchanger and filters for the fumes.

The main equipment of the process line is the patented continuous furnace that was designed and realized specifically for this application and to maximize energy efficien-

cy. The separation is performed by two rollers that remove the copper ribbons, by a linear sieve that eliminates thin powder and finally by a customized and patented sieve that separate PV cells and glass pieces.

The pilot plant, authorized by Veneto region (lg. 183 of 4/7/2022), showed a maximum treatment capacity of approximately 30 kg/h of PV panels.

During the first year of experimental phase, approximately 900 kg of end-of-life PV modules have been treated and the performance of the plant were assessed in terms of energy consumption, recovery yield, purity of recovered material and environmental impact. In particular, the materials were analysed to verify the possibility to reuse them. Furthermore, PV cells were analysed with SEM-EDS using Cambridge Stereoscan 440 Microscope equipped with a Philips PV9800 EDS to check composition for the re-use test in ferroalloy production. Finally, the fumes produced during the thermal treatment were analysed before and after the abatement system according to UNI EN specific regulations, typically used in such plants.

2.2 Re-use of the silicon

STILLMETAL (upscaling project funded by EIT RawMaterials 2022-2024) aims to solve the problem of the "white" slag generated from steel making or cast iron foundries through a process, sustainable from both technical and economical point of view, able to transform a waste into two valuable materials. As matter of fact, these types of slag are considered waste because, in comparison with other slag types, cannot be reused and must be disposed of in landfill.

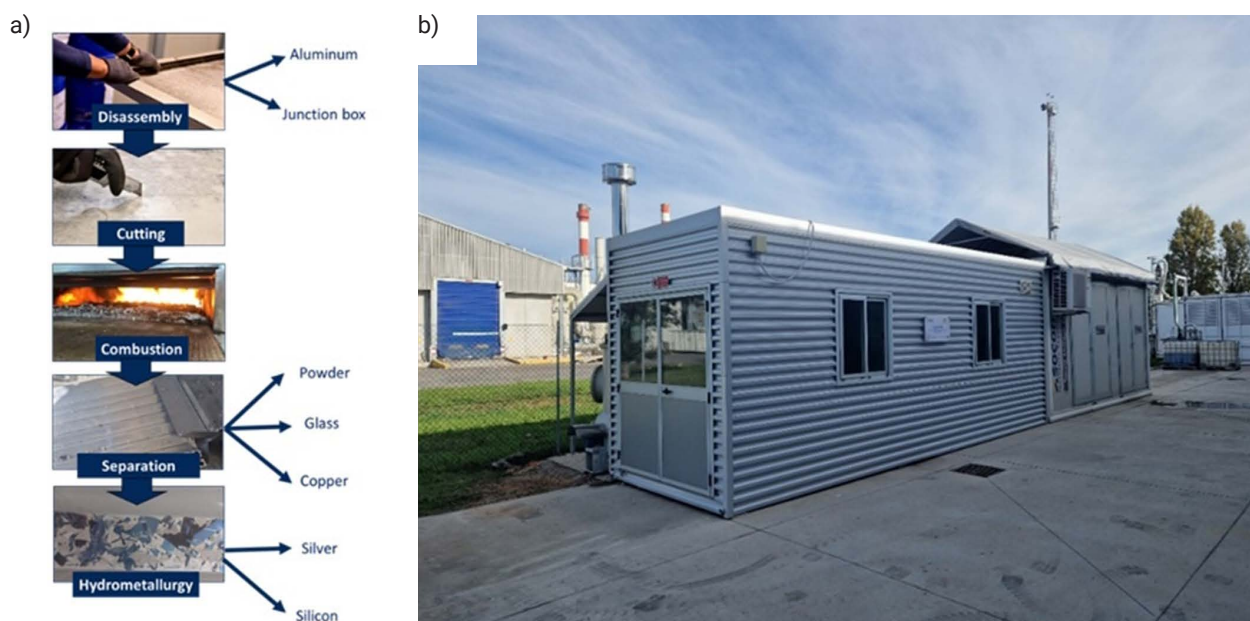


FIGURE 3: Flowchart of 9-Tech technology (a) and pilot plant installed in Porto Marghera (a).

The treatment involves the reaction at high temperatures, in the molten state, between the slag and reagents (specific added materials) to give a metal and a residual oxide slag. The obtained metal has a composition of a Ferroalloy, a high value material that finds employment in the steel and cast iron industry, whereas the residual oxide slag has a right composition to have a market value for the ceramic and/or cement industry. The project involves the construction of TRL9 STILLMETAL plant.

Ferroalloys are namely composed by one or more alloying elements with iron, and not a single steel grade is produced without ferroalloys. In detail, in the STILLMETAL process Ferrosilicon (FeSi) are obtained and this type of ferroalloy can be used as deoxidizer and as alloying

element. The value of FeSi ferroalloys increases with the amount of Si which can vary from 10 to 90% by weight.

University of Padova (UNIPD) had studied the use of PV cells, recovered from 9-Tech pilot plant, in STILLMETAL process (Figure 4).

Scanning electron microscope (SEM) analysis of the PV cells recovered from the 9-Tech pilot plant revealed that Si and Al paste were the major component of the cells, namely, 90%wt Si and 10%wt Al (silicon nitride was considered negligible for this application). PV cells were treated in order to remove silver before UNIPD employed them in the STILLMETAL process. Numerical simulations with Thermodynamic Software Fact Sage 8.2, using FSS- tel, FTmisc and FToxid databases, were carried out to de-



FIGURE 4: Scheme of STILLMETAL process.

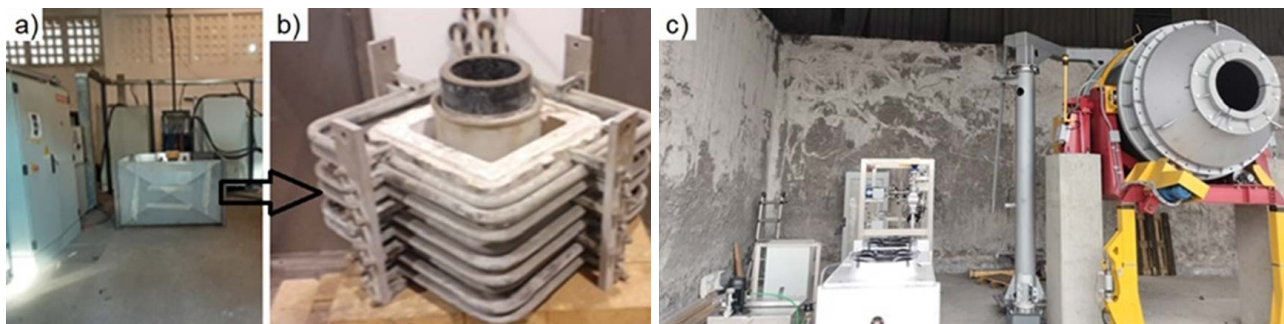


FIGURE 5: a) Image of TRL 5 pilot plant; b) conductive coils; c) Image of TRL9 STILLMETAL pilot plant.

termine the amount of white slag and the type and amount of reagents that have to be employed to obtain a FeSi and a residual oxide slag with an adequate composition and low melting temperature. In detail, objective of the simulation was to optimize the process to obtain a ferroalloy with amount of Si higher than 40% and a residual oxide slag with melting point temperature lower than 1400°C.

Starting from the simulation of the process, the use of photovoltaic cells was found to allow the production of a FeSi ferroalloy with higher amount of Si (Si higher than 45%) than the one produced with conventional STILLMETAL process (about 25% Si). Furthermore, it was found that the presence of Al paste does not influence the process. Therefore, the use of PV cells in STILLMETAL process could represent an advantage since they allow obtaining a ferroalloy richer in Si with lower quantity of reagents, making the process more advantageous from an economic and environmental point of view. Also it is demonstrated that it is possible to valorize the Si without the need for further PV cell purification.

Ferrosilicon production is also an energy-intensive industry with a high consumption of electricity and coke with high emission of CO₂.

The reduction in CO₂ emissions from ferroalloy production using the STILLMETAL process is expected to be 70÷85% when using photovoltaic cells in place of quartz.

After the simulation the results were experimentally validated at TRL5 with an induction furnace with stirring. This furnace is shown in Figure 5, where it is possible to see the power supply converter generators, the cooling system and the conductive coils. This system is a unique prototype and all the parts have been custom made (Figure 5a, b). The test with the TRL9 plant of STILLMETAL project are scheduled for October 2024. The plant with a capacity 400 kg for batch (80% of slag, 20% of reagents), will be able to treat 8960 kg of slag/day, about 2000 ton/year (Figure 5c). It has been installed at the first customer site and they are waiting for the final authorization to start work.

The materials (white slag, PV cell and the other reagents), shown in Figure 6, were weighted, mixed inside a bucket and then put inside a crucible of graphite with the external part of ceramic. The materials obtained at the end of tests were characterized with a Cambridge Stereoscan 440 electron microscope equipped with a Philips PV9800 EDS.

3. RESULTS AND DISCUSSION

3.1 Test in the pilot plant in Venice

The pilot plant showed a high recovery yield of 87%. Specifically, from 898kg of PV panels, the pilot plant was able to recover: 581 kg of glass, 146 kg of aluminium, 14 kg of junction boxes (mainly composed of plastic and copper), 26 kg of PV cells (mainly composed of silicon), 6 kg of copper ribbons, 0.45 kg of silver and 9 kg of powder (mainly composed of glass and PV cells). Examples of materials recovered with 9-Tech technology are reported in Figure 7.

The analysis showed that purity of recovered material was high enough to be suitable for other applications. In fact, the recovered glass, aluminium, and copper showed to have the specifications for end-of-waste and can be thus reused by other companies to produce new items.

Regarding the photovoltaic cells, the material comes in the shape of thin sheets (thickness 2÷300 µm) broke into pieces of irregular size, ranging from 2 mm to 5 cm. The base material of these fragments is silicon, with aluminum paste on the rear side and thin silver contacts on the front one. Furthermore, there is a thin layer of silicon nitride, present on the front side of the cell. Finally, on the surface



FIGURE 6: Materials and crucible used for STILLMETAL tests.



FIGURE 7: Materials recovered in the pilot plant performing thermomechanical recycling.

of the recovered cells titanium oxide powder was detected due to contamination from backsheet combustion.

The SEM-EDS analysis confirmed the presence of aluminum (Figure 8a) and silver on the photovoltaic cells. (Figure 8b).

The process showed electricity consumption of 105 kWh per ton of treated PV waste, which is comparable to that of mechanical treatment plants, and PLG consumption is also negligible. The fumes, after treatment, always respected the limits prescribed by the authorisation and the

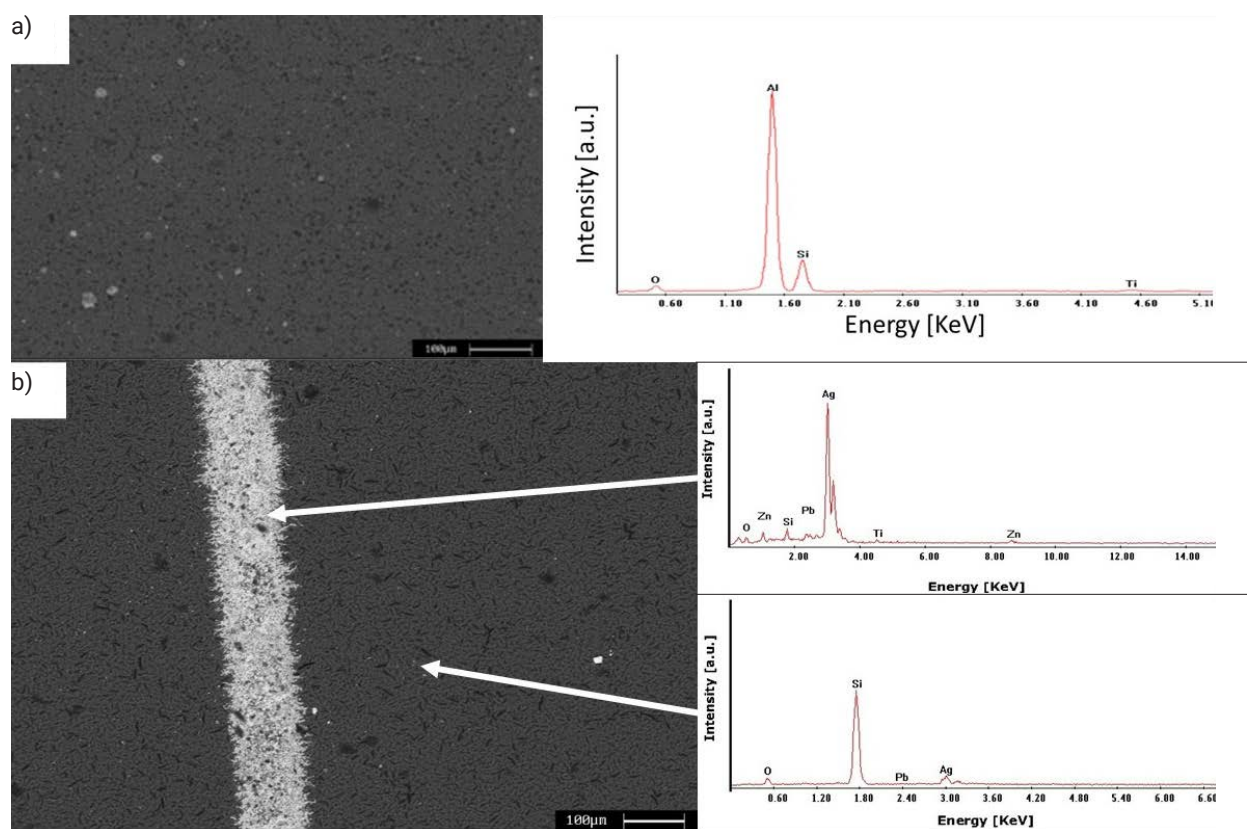


FIGURE 8: SEM-QBSD picture and SEM-EDS analysis of a) cell rear side containing aluminum, b) cell front side: silver contact and silicon substrate.

LCA of the process showed an extremely positive overall environmental impact saving up to 3850 kg of CO₂ equivalent per ton of PV waste treated. Finally, it must be noted that both energy consumption and environmental impact can be reduced when upscaling the process.

3.2 Re-use of the silicon

Different tests were performed at TRL5, varying the amount of the reagents, the temperature and the time of the process.

The crucibles, the day after the test, were broken to recover the solidified materials inside. It was possible to observe in all the samples two distinct phases: on the top the oxide phase (residual oxide slag), and on the bottom the metallic phase (ferroalloy) (Figure 9).

In the first tests, the metallic phase resulted a ferroalloy rich in Si but the amounts of Si, varying from 32 to 42% depending on the starting amount of reagents, temperature and time of process, resulted lower than the one expected with the simulation. Moreover, the microstructure and amount of Si were not homogenous along the height of the sample. This was attributed to a not adequate stir-

ring during the reaction, to a low reaction temperature and to an inadequate amount of charge in the crucible. The last tests were carried out trying to solve the problems highlighted in the previous ones and therefore the time of stirring, the temperature and the starting charge were increased.

From the SEM and EDS analysis of the ferroalloy obtained in the last tests, the microstructure and the composition resulted the same along the whole height of the ingot, with an average value of Si of about 44%, still slightly lower than the expected one of 47.9% but considered acceptable for the purpose of this project. In Figure 10 the cross section of the ferroalloy and the SEM-BSE images at different height are shown. The microstructure of the ferroalloy resulted constituted by 2 main phases characterized by different amount of Si: the dark one 53.3% Si and light one 37.3% Si.

3.3 Broad impacts

The proposed process represents a significant advancement in technology to treat PV waste and has the potential to positively impact various areas.

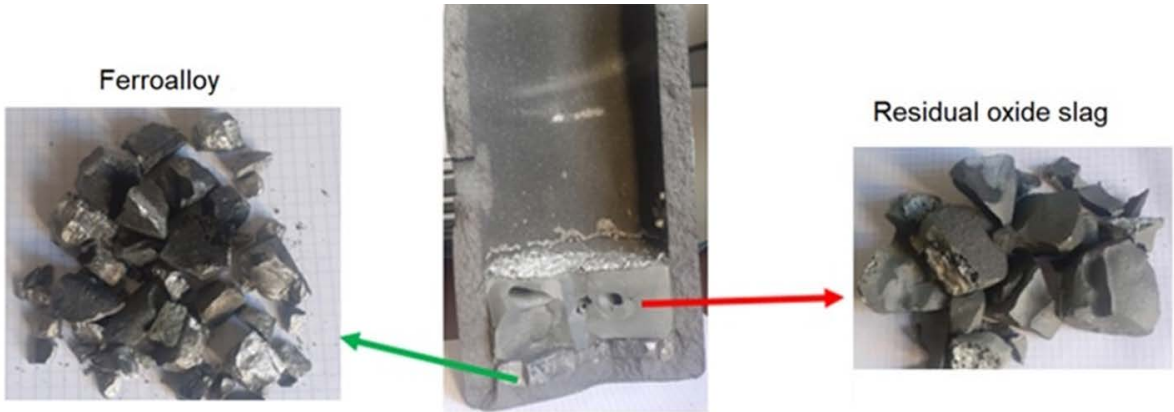


FIGURE 9: Cross-section of the crucible with ferroalloy and slag.

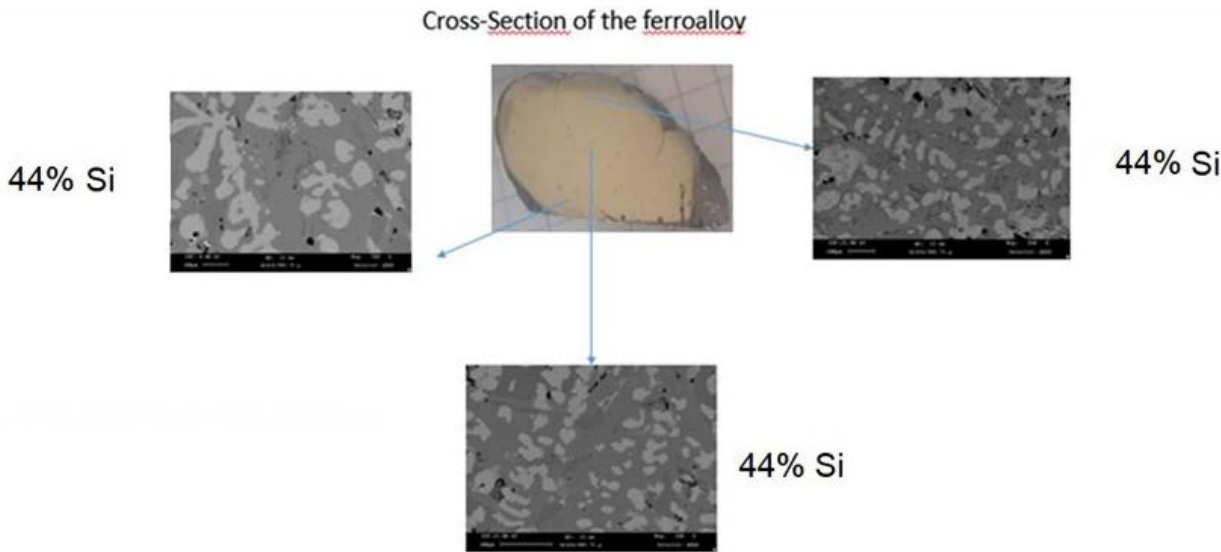


FIGURE 10: Cross-section of the ferroalloy and SEM backscattered electrons image and EDS analysis.

From a social perspective, PV recycling promotes sustainable practices by encouraging recycling and conservation efforts within communities. It also contributes to the creation of green jobs, which can support socio-economic development in regions involved in photovoltaic recycling.

Economically, the proposed recycling technology can recover over 95% of the raw materials' economic value, including high-quality materials such as silicon and silver. This reduces dependence on virgin material extraction and creates opportunities for secondary raw material markets. The adoption of secondary raw materials can also lower production costs in some sectors, such as the steel industry presented, supporting economic growth and improving competitiveness in a sustainable and circular way. From an environmental perspective, since the recycling process reduces the need for extraction of virgin resources, it helps reduce environmental degradation and greenhouse gas emissions. Its potential to reduce carbon footprints by approximately 3850 kg of CO₂ equivalent per ton of PV waste processed (Mazzi et al., 2024) indicates a significant contribution to climate change mitigation.

The process not only complies with the EU WEEE Directive 2012/19/EU, meeting the mandatory collection and recycling thresholds for PV panels, but also supports the supply chain of strategic materials such as silver, a critical component in the solar industry, and silicon, a critical raw material. By ensuring a resilient supply of these materials, the technology supports the transition to green energy and supports global efforts towards sustainable and efficient energy solutions. The reuse of recovered silicon for the production of Si ferroalloys will enable the application of this material in a huge market, such as the steel industry, which has a high demand for Si ferroalloys.

The advantage of using recovered silicon instead of virgin silicon is significant, because ferrosilicon production is an energy-intensive industry with high consumption of electricity and coke and a consequent high production of carbon dioxide. The production of ferroalloys from slag with photovoltaic cells (in the presented STILLMETAL process) can reduce CO₂ emissions by about 85% compared to the production of FeSi45 from quartz.

4. CONCLUSIONS

A pilot plant for the recovery of raw materials (glass, aluminum, copper ribbons and silicon PV cells) from end-of-life PV panels has been designed and tested by 9-Tech in Porto Marghera (VE). The results, are very promising with a recovery yield of around 90% and high purity of raw materials, and they will be exploited through the construction of 3000 ton/year plant, planned in 2025. The process demonstrates a highly efficient and environmentally friendly approach to PV waste treatment. The significant positive environmental impact of this process is driven by the effective recovery of valuable secondary raw materials that can contribute to a circular economy by minimizing waste and maximizing resource recovery. The electricity consumption of 105 kWh per tonne, comparable to traditional mechanical treatment methods, the minimal PLG consumption and

the adherence to emission standards further contribute to the sustainability of the process.

While glass, copper and aluminum reuse are easy, with end-of-waste status, the re-use of silicon is still uncertain. PARSIVAL project is currently studying possible reuse of the recovered silicon in order to valorize all the recovered raw materials and maximize process profitability. Among the applications studied in the framework of the project, University of Padova investigated the use of PV cells for Si-Ferroalloy production in STILLMETAL process at TRL5 and showed that is possible to obtain a ferroalloy with a higher amount of Si (44%) in comparison with conventional STILLMETAL process. This application can effectively recycle silicon with economic and environmental benefit for PV waste treatment. Next steps will be the upscaling at TRL9 both of the recycling technology and of the Si-Ferroalloy production.

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ENVIRONMENTAL ASSESSMENT OF GLASS FIBER REINFORCED POLYMER (GFRP) VALORISATION PROCESSES FROM THE DECOMMISSIONING OF WIND TURBINES

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ABSTRACT

The production of wind energy is one of the most mature technologies in the field of renewable energy, and the rapid technological advances bring out the need to replace the old models of wind turbines with more efficient versions. Operators must plan the management of a large amount of resource flows from the end-of-life wind farm disposal. The use of composite materials, in particular epoxy resins reinforced with glass fibers (Glass Fiber Reinforced Polymer - GFRP), most used in the wind blades, makes the material separation and recovery operations quite complex. The purpose of the study is to compare the environmental performance of the life cycle of two treatment processes (mechanical recycling and co-processing in cement kiln), identified by literature as possible technologies for the treatment of GFRP, including also the environmental benefits obtainable through the valorisation of both the waste material itself (used as fuel) and the secondary material obtained at the end of the treatment process. The LCA study provides some general considerations about the possible recycling and recovery of GFRP material contained in wind turbine blades. In general terms, it can be stated that co-processing in cement kilns is a highly efficient process because it allows energy to be recovered from the waste material and part of it to be integrated into the clinker production process.

1. INTRODUCTION

Since wind energy is one of the most promising sources of renewable energy and that the installation of wind plants has grown exponentially in recent years (IEA, 2023), it is essential to address the economic and environmental issues of the management and end-of-life treatment of composite waste (glass or carbon fibre reinforced polymers) resulting from the decommissioning of wind power plants and, in particular, from wind turbine blades. Currently, 2.5 million tons of composite material are in use in the wind energy industry globally and it is estimated that by the end of 2023 about 14,000 blades could be decommissioned, corresponding to 40,000-60,000 tonnes (WindEurope - Cefic - EuCIA, 2020).

However, so far, the amount of composite waste generated by wind turbines accounts for only 10% of total thermoset composite waste (WindEurope - Cefic - EuCIA, 2020). Since these volumes are relatively low, a recycling activity with only this waste stream is rather difficult and expensive from both a technological and economic point of view. Moreover, from a normative point of view, national and European norms are currently limited and not uniform

in the regulation of the end-of-life of composite materials. In general, these have the objective of limiting the disposal of composite materials in landfills.

Nevertheless, different techniques of recycling composite materials are currently available, but only some of them are applied at industrial scale, due to their different technology readiness, high investment and operating costs (WindEurope, 2020). Composite materials, consisting of epoxy resins reinforced with carbon fibers (CFRP - Carbon Fiber Reinforced Polymer) or, most commonly used in wind turbines, with glass fibers (GFRP - Glass Fiber Reinforced Polymer) can be recycled or recovered by mechanical grinding, thermal (pyrolysis, fluidized bed), thermochemical (solvolysis) or electrochemical (high voltage fragmentation) processes or by a combination of them.

The aim of this study is to assess and compare, through the standardised Life Cycle Assessment method (ISO 2021a, ISO 2021b), the environmental impacts of the life cycle of two GFRP waste material treatment processes resulting from the decommissioning of wind turbines. The LCA studies and their results are presented in the following paragraphs.

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2. MATERIALS AND METHODS

2.1 Goal and scope of the study

2.1.1 Goal of the study

The LCA method was applied to the following end-of-life treatment processes for GFRP waste, in order to evaluate and compare their environmental performances:

- Mechanical recycling (Scenario 1);
- Co-processing (also known as “the cement kiln route”), which is used for recycling GFRP waste because it is technologically and economically more advantageous than mechanical recycling (Gonçalves et al., 2022) (Krauklis et al., 2021) (Scenario 2). This technology is already used on a commercial scale because it is considered, at the moment, one of the most solid options for the end-of-life management of wind turbines since it provides several advantages:
 - Reduction of the amount of fossil fuel (coal) used in the clinker production process;
 - Reduction of both the use of raw material (sand) and the emissions of its production to provide silica to the kiln, because silica is replaced by glass fibres contained in the input composite material;
 - No residue generated, because the mineral content in the composite material is incorporated into clinker.

For both technologies the environmental benefits from the GFRP waste valorization as fuel and/or as secondary material from the treatment process have been considered.

An attributional approach was used for the study, reproducing the existing supply chain and using background processes representative of the average market consumption mix (European Commission - Joint Research Centre, 2010).

2.1.2 Function of the system, functional unit and reference flow

The function of the system is the management and treatment of GFRP waste material resulting from the disposal of a wind farm located in Italy which consists of 6 wind turbines at the end of their useful life.

The functional unit is 1 MW of power, and the reference flow is the amount of waste material GFRP collected from the disposal of the 6 wind turbines needed to obtain 1 MW of power.

2.1.3 Description of the recycling processes

Most of the wind turbine blades on the market are made of GFRP. However, there are other blade types partially (hybrid) or totally made of Carbon Fiber Reinforced Polymer (CFTP). This study considers wind turbine blades are consisting only of GFRP. In this case, the two main materials of the blades are glass fiber and polymer resin. Other support components, made of metal materials and lower than 5% in weight in most of the wind turbine design were not considered in the study because it was assumed that they are removed during the first phase of dismantling (Liu et al., 2022).

In the wind turbine model analyzed in the study, the GFRP material is contained in the blades, the nose cone and the housing of the nacelle; the amount of GFRP contained in a turbine is about 13 tons (Vestas, Wind Systems, 2024).

The LCA study considered two of the commonly used, due to their technological and economic advantages and high maturity level, end-of-life treatment technologies for GFRP waste material resulting from the decommissioning of wind turbines (Gonçalves et al., 2022) (Krauklis et al., 2021):

- Mechanical Recycling: this process involves a first phase of material dimensional reduction by fragmentation and grinding, and any subsequent sifting to separate and classify the crushed material in different sizes. The machines used to dimensionally reduce the material can be different: the hammer mill can better process the waste material and obtain a ready-to-use material for new composites. The material obtained at the end of the recycling process has the form of granules (rich in glass fibers) or powders (rich in polymer resin) and can be used as a filler in the building industry or as a reinforcement in new composite products (Gonçalves et al., 2022) (Oliveux et al., 2015). The characteristics of the obtained material largely depend on the type of input material and the grinding technique used. In general, its value is poorly competitive if compared to virgin materials. Moreover, only a part of the recycled product is suitable to be used for the production of new composites and in some cases further processing (i.e., grinding and screening) have to be added for a better usability, in other applications. When used as a filler, an average particle size lower than 10 µm is required. In this case, the larger particles will have to be ground again. Materials leaving the recycling process can be generally classified into two categories (Palmer, 2009):
 - “Coarse” quality materials consisting of larger particles, typically bundles of glass fibers held together by the resin: they are too large to be used as filling materials but can be used as reinforcement in new composites or can be broken into smaller pieces;
 - “Fine” quality materials, ground into very fine particles, to be used as fillers.
- Co-processing in a cement kiln (Jacob, 2011) (Krauklis et al., 2021) (Yazdanbakhsh et al., 2014): as a result of the dismantling of the rotor, the blades are transported to the co-processing, which consists of a furnace for the production of clinker, the constituent material of cement. The process uses the polymer matrix of the waste material GFRP as fuel in the process itself (thus replacing the coal normally used in the combustion process), while the inert fibrous component of GFRP, based on alumina borosilicate, constitutes the mineral raw material that can be used to produce clinker (the basic component, made from clay and limestone, for cement production). It can therefore be considered that the process uses 100% of the input GFRP waste material (even if a part of it it's just energetically valorized), because the whole waste material is used for energy recovery, since it is a fuel for combustion in the furnace,

or for material recovery, because the the secondary material leaving the furnace can be used for a subsequent valorisation.

2.1.4 System boundaries

The system boundaries are “from gate to grave”, including only the phases of GFRP waste material transport from the wind farm to the treatment plant and the waste treatment with mechanical recycling or co-processing in a cement kiln.

Since both treatment process leads to a multifunctionality, a system expansion approach was used, similarly to Karuppannan Gopalraj et al., (2021) and according to ISO 14040-44, in order to include the additional function of the analyzed systems. In this way, the substitution of coal (as fuel for combustion) and inert material (sand) for clinker production was considered by modeling specific avoided impacts (Figure 1). According to Karuppannan Gopalraj et al. (2021), a “zero burden” approach was applied for GFRP waste material, which did not have any environmental impact due to its production. Therefore, the LCA study does not consider the environmental impacts due to the waste production phases (which are equal in Scenarios 1 and 2), because the purpose of the study is to compare the environmental performances of the two different systems for recycling and recovery the GFRP waste material.

2.1.5 Data quality and Life Cycle Impact Assessment method

For the LCA study, secondary data from different literature sources and related to the specific treatment processes for the management of GFRP waste material were used, the specific characteristics of the GFRP material itself and the technical characteristics of the secondary material obtained were used. More details about the used literature sources are included in the Inventory paragraph.

The study was carried out using SimaPro 9.5 software and Ecoinvent 3.9.1 database (Wernet et al., 2016) for the modeling of background data. From the geographical, time and technological representativeness point of view, the more recent data about European or global average technologies available in Ecoinvent 3.9.1 were used.

The impact assessment method used is EF 3.1 (Zampori et al., 2019), applying only the phases of characterization and normalization, as they are considered more scientifically robust. In regards to the geographical, temporal and technological representativeness of the background data used, reference was made to European average technologies when available, otherwise global, and, as last option, to the data available in Ecoinvent 3.9.1 (Wernet et al., 2016).

2.2 Inventory analysis

For both scenarios, the total amount of GFRP waste material contained in the wind turbines, rotor cone nose and nacelle housing were considered equal to 12,838.62 kg (amount referred to 1 MW of power of the entire wind farm).

In both scenarios an average road transport distance (by a Euro 6 truck with an average load capacity of 32 tonnes) of 200 km was considered from the wind farm to the treatment plants (Karuppannan Gopalraj et al., 2021).

In addition, an alternative distance (AD) of 400 km from the plant to the treatment site was considered for both scenarios in order to assess the impact of transport on the two processes. Furthermore, for the co-processing scenario (Scenario 2), an alternative option has been studied where the transport of the material over the 400 km distance is carried out by vehicles with a lower capacity (a Euro 6 lorry with an average load capacity of 5.7 tonnes), on the assumption that the crushing phase of the material takes place at the dismantling site (Option A).

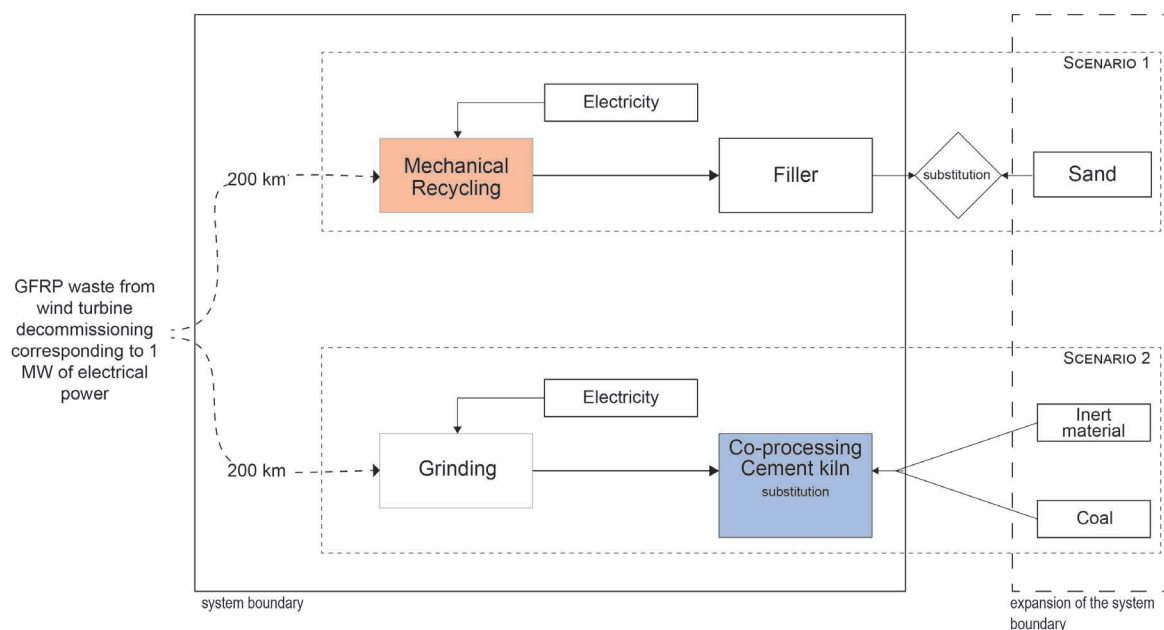


FIGURE 1: System boundary.

This option was considered because roads may not be suitable for large vehicle transport and it may be necessary to reduce the size of the material before it is transported to the cement plant so that it can be handled by smaller vehicles.

In Scenario 1 (mechanical recycling), the energy consumption for the mechanical treatment of GFRP waste, i.e. to cut the blades of the dismantled blades into pieces, to chop and grind the waste material into powder and into fiber sections of about 10 mm was considered. According to Kuruppannan Gopalraj (2021), energy consumption for this process was assumed to be electricity, using the Italian medium voltage mix and equal to 0,27 MJ/kg of treated GFRP waste material, in agreement with Liu et al. (2019) and Howarth et al. (2014),

To calculate the amount of exploitable secondary material, the GFRP waste material was considered to consist of the following components (Jacob, 2011):

- 67% in weight was assumed to be a mineral part made of fiberglass, rich in mineral components such as silica, calcium carbonate, alumina;
- 33% in weight was assumed to be an organic polymer part.

In order to quantify the benefits deriving from the use of the secondary material obtained at the end of the mechanical treatment (recycled material), in agreement with Liu et al. (2019), it was assumed that this benefit is propor-

tional to the tensile strength of the fibrous component of the GFRP waste material, because tensile strength is one of the most important properties of wind turbine materials.

In detail, it was considered that the fibrous component of the secondary material has a tensile strength of 78% of the tensile strength of the fibrous component of virgin GFRP material (Liu et al., 2019) and that the yield rate of the fibrous component at the end of the mechanical recycling process is equal to 55% (Liu et al., 2019) (Palmer, 2009); similarly, it was considered that the polymer component of the secondary material has a technical performance equal to 50% of the performance of the polymer component of the virgin GFRP material and that the yield rate of the polymer component at the end of mechanical recycling is 100% (Liu et al., 2019).

Table 1 summarizes the data and parameters used in Scenario 1 for mechanical recycling. By multiplying these parameters and yield rates by the quantities of the fibrous component and polymer component of the GFRP waste material, the quantities of recycled fibre and recycled polymer released from the mechanical treatment were finally calculated. These quantities were then added together to obtain the total amount of secondary material obtained, assuming that it is not further separated and constitutes a homogeneous mix of fibrous and polymer components.

In order to identify the product replaced by the secondary material stemming from the mechanical recycling

TABLE 1: Inventory data and parameters used for modelling mechanical recycling and co-processing processes in cement kilns.

Process	Data/parameter	Unit	Values Scenario 1	Values Scenario 2	Reference
Mechanical recycling (scenario 1)					
	Total amount of GFRP waste material contained in wind turbines, rotor cone nose and nacelle housing, referring to 1 MW of the entire wind farm	kg	12,838.62		Authors calculation based on Vestas data (2015)
	Energy consumption for mechanical treatment, with reference to 1 kg of treated GFRP material	MJ/kg	0.27	-	Howarth et al. (2014); Liu et al. (2019)
	Tensile strength of the fibrous component of the secondary material compared to the tensile strength of the fibrous component of the virgin GFRP material	%	78	-	Liu et al. (2019); Liu et al. (2022)
	Final yield rate of fibrous component of GFRP waste material	%	55	-	Liu et al. (2019); Palmer (2009)
	Technical performance of the polymer component of the secondary material, compared to the technical performance of the polymer component of the virgin GFRP material	%	50	-	Liu et al. (2019)
	Final rate of return of polymer component of GFRP waste material	%	100	-	Liu et al. (2019); Keith et al. (2017)
	Distance from wind farm to treatment plant	km	200	-	Karuppannan Gopalraj et al. (2021)
	Alternative distance (AD) from wind farm to treatment plant	km	400	-	Authors calculation
Co-processing in cement kiln (scenario 2)					
	Total amount of GFRP waste material contained in wind turbines, rotor cone nose and nacelle housing, referring to 1 MW of the entire wind farm	kg		12,838.62	Authors calculation based on Vestas data (2015)
	Energy consumption for grinding pre-treatment, referring to 1 kg of treated GFRP material	MJ		0.27	Howarth et al. (2014) Liu et al. (2019)
	Superior calorific value of GFRP waste material	MJ/kg	-	33.5	Karuppannan Gopalraj et al. (2021)
	Higher calorific value of coal	MJ/kg	-	28	Karuppannan Gopalraj et al. (2021)
	Distance from wind farm to treatment plant	km	-	200	Karuppannan Gopalraj et al. (2021)
	Alternative distance (AD) from wind farm to treatment plant	km	-	400	Authors calculation

of the waste material GFRP (avoided product), several papers on its possible uses were analysed. The approach of Yazdanbakhsh et al. (2014) was followed: according to the authors, mechanically recycled GFRP waste has a potential application as a filler in Portland cement mortar and concrete.

Further possible uses of the recycled material include:

- its transformation into a compound, made of ground GFRP (obtained from mechanical recycling), mixed with virgin or recycled plastic resins (polypropylene, polyethylene, acrylonitrile butadiene styrene) for injection moulding (Gees Recycling, 2024);
- the production of recycled fibreglass panels made of recycled fibre-reinforced composites (glass fibre, kevlar, carbon fibre, etc.) that are finely ground and re-glued at high pressure with minimum percentages of adhesives (3D Wall Panels Italia, 2024).

Nevertheless, in the LCA study these uses could not be accounted for, due to the lack of appropriate datasets in the Ecoinvent database, which are necessary to identify and model the replaced material. The study therefore assumed that the substituted product is sand, used as a filler, using a suitable Ecoinvent dataset to model its production, considered an avoided impact. Finally, the amount of replaced sand was considered to be equal to the total amount of secondary material obtained at the end of the recycling process, calculated as specified in the previous

paragraph. Table 2 shows the input and output data used to model Scenario 1, and the Ecoinvent datasets used.

As regards the co-processing in a cement kiln (Scenario 2), GFRP material waste is first ground to reduce its size, and then fed into the kiln where combustion takes place: the polymer organic component (33% of the total amount of GFRP waste material) burns, providing energy, and the mineral fiber component (67% of the total amount of GFRP waste material) (Jacob, 2011), can be used as the main component of Portland cement, providing the raw material for the production of cement clinker (Yazdanbakhsh et al., 2014).

The study therefore accounted for the electricity consumption to grind GFRP waste material before it is inserted into the furnace. Similarly to mechanical recycling, it was assumed that this electrical consumption, modeled through the use of the Italian medium voltage electric mix, was equal to 0.27 MJ/kg of treated waste (Liu et al., 2019), because the grinding process is conceptually the same, from the technological point of view, to the grinding of mechanical recycling. Karupannan Gopalraj et al. (2021) has a value of 0.09 MJ/kg of treated waste for grinding that takes place before being placed in the oven. However, in this study a conservative approach was used, adopting the same energy consumption of mechanical recycling.

Moreover, it was considered that the polymer organic component of the GFRP waste material, because it is a fuel, is used to replace fossil fuels for the combustion process inside the furnace (energy recovery), thus avoiding

TABLE 2: Mechanical recycling and Co-processing in cement kiln input and output data, referring to 1 MW of power of the entire wind farm.

	Unit	Values	Dataset used by Ecoinvent
Mechanical recycling (scenario 1)			
Input			
Energy for grinding	MJ	3,466.43	Electricity, medium voltage {IT} market for electricity, medium voltage Cut-off, U
Distance from wind farm to treatment plant (200 km)	kg*km	2,567,723.08	Transport, freight, lorry >32 metric ton, EURO6 {RER} market for transport, freight, lorry >32 metric ton, EURO6 Cut-off, U
Alternative distance (AD) from wind farm to treatment plant (400 km)	kg*km	5,135,446.154	Transport, freight, lorry >32 metric ton, EURO6 {RER} market for transport, freight, lorry >32 metric ton, EURO6 Cut-off, U
Output			
Power of the entire wind farm	MW	1	
Replaced inert material (sand)*	kg	5,808.57	Sand {RoW} market for sand Cut-off, U
Co-processing in cement kiln (scenario 2)			
Input			
Energy for grinding	MJ	3,466.43	Electricity, medium voltage {IT} market for electricity, medium voltage Cut-off, U
Distance from wind farm to treatment plant (200 km)	kg*km	2,567,723.08	Transport, freight, lorry >32 metric ton, EURO6 {RER} market for transport, freight, lorry >32 metric ton, EURO6 Cut-off, U
Alternative distance (AD) from wind farm to treatment plant (400 km)	kg*km	5,135,446.154	Transport, freight, lorry >32 metric ton, EURO6 {RER} market for transport, freight, lorry >32 metric ton, EURO6 Cut-off, U
Alternative distance (AD) from wind farm to treatment plant (400 km with a smaller truck) (Option A)	kg*km	5,135,446.154	Transport, freight, lorry 3.5-7.5 metric ton, EURO6 {RER} market for transport, freight, lorry 3.5-7.5 metric ton, EURO6 Cut-off, U
Output			
Power of the entire wind farm	MW	1	
Replaced carbon*	kg	5,068.96	Hard coal briquettes {RER} hard coal briquettes production Cut-off, U
Replaced inert material (sand)*	kg	5,763.25	Sand {RoW} market for sand Cut-off, U

* Such material is modelled in the system as avoided product

the use of coal for the combustion process (avoided product). In this way, it was taken into account that the amount of polymer organic component totally replaces the amount of coal necessary for the combustion process, also assuming that the furnace, for its operation, does not need any other fuel.

In order to calculate the amount of coal replaced, the energy content of both the GFRP waste material and the coal was considered: in detail, the higher calorific values were assumed to be 33.5 MJ/kg for the former and 28 MJ/kg for the latter (Karuppannan Gopalraj et al., 2021). In this way, the quantity of coal replaced by 1 kg of GFRP waste material is equal to 1,196 kg.

This environmental benefit was then included in the LCA model by using a specific Ecoinvent dataset for the production of coal, considering it as avoided impact.

Finally, it was assumed, similarly to mechanical recycling, that the fibrous mineral component of the GFRP waste material can be used as a filler for clinker production, and the substituted product is sand, because it is a silica-rich material (avoided product). It was therefore considered that the substitution factor is 1:1, i.e. 1 kg of GFRP waste material, which contains 0.67 kg of fibrous mineral component, rich in components such as silica, replaces 0.67 kg of sand, a material rich in silica.

Table 1 summarizes the data and parameters used in Scenario 1 and 2. Table 2 shows the input and output data used to model Scenario 1 and 2, and the Ecoinvent datasets used.

3. ENVIRONMENTAL IMPACT ASSESSMENT

3.1 Characterisation and normalisation results

Table 3 shows the results of characterization and normalization related to the treatment of GFRP waste material,

referring to 1 MW of the entire wind farm, for the Scenario 1 (mechanical recycling) and for the Scenario 2 (co-processing in cement kiln).

4. RESULTS AND DISCUSSION

The analysis of normalization results shows that the most relevant impact categories (i.e. those categories that contribute cumulatively to 80% of the total normalisation results) are, for Scenario 1, "Resource use, fossils" (28% of the total normalisation phase), "Climate change" (15%), "Eutrophication, freshwater" (9%), "Ecotoxicity, freshwater", "Particulate matter", Photochemical ozone formation (8% each), and "Human toxicity cancer" (6%) (Table 3, Figure 2).

For the same relevant impact categories, an increase in impacts ranging from 44% to 111% is observed when considering the transport of the material over an alternative distance of 400 km (AD). From this it can be concluded that an increase in the distance of transport in this scenario has a significant environmental impact.

The most significant impact categories for Scenario 2 are "Eutrophication, freshwater" (-46% of the total normalisation phase), "Resource use, fossils" (-26%), "Ecotoxicity, freshwater" (-6%), "Photochemical ozone formation" (-4%), "Acidification" and "Eutrophication, marine" (-3% each) (Table 3, Figure 3).

In Scenario 2, an increase in transport distance (AD) causes a non-significant decrease in avoided impacts for the main impact categories. Using a lower capacity means of transport to transport the material over the same distance (Option A) contributes significantly to the reduction in avoided impacts, but is still negligible given the high contribution of avoided impacts in the process.

As regards to characterisation results, the comparison between the two scenarios will be presented in the follow-

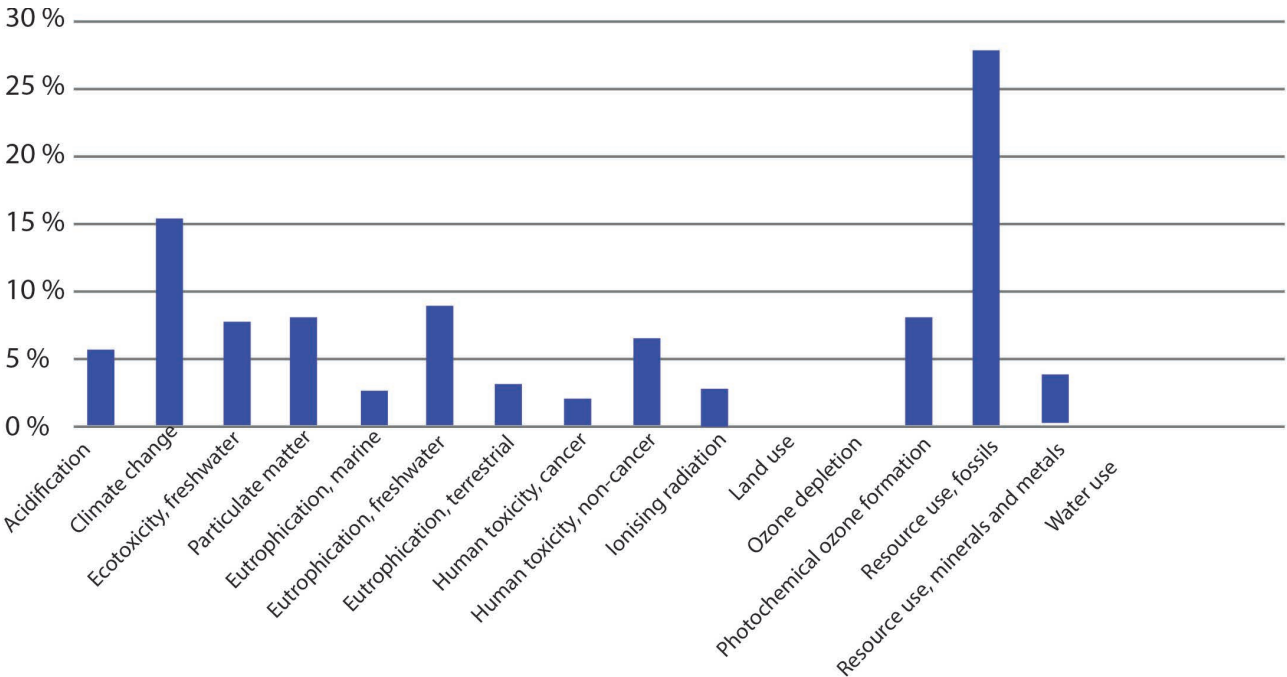


FIGURE 2: Normalisation results for Scenario 1 (mechanical recycling).

TABLE 3: Characterization and Normalization results related to GFRP waste material treatment, referring to 1 MW of the entire wind farm: Scenario 1 (mechanical recycling) and Scenario 2 (co-processing in cement kiln).

Impact category	Unit	Scenario 1		Scenario 2	
Characterization		200 km	400 km (AD)	200 km	400 km (AD)
Acidification	mol H ⁺ eq	1.50E+00	2.14E+00	-1.33E+01	-1.26E+01
Climate change	kg CO ₂ eq	5.69E+02	8.26E+02	-1.37E+03	-1.12E+03
Climate change-Biogenic	kg CO ₂ eq	4.57E+00	4.65E+00	3.55E+00	3.63E+00
Climate change-Fossil	kg CO ₂ eq	5.64E+02	8.22E+02	-1.38E+03	-1.12E+03
Climate change-LU and LU change	kg CO ₂ eq	1.33E-01	2.58E-01	-1.29E+00	-1.17E+00
Ecotoxicity. freshwater-part 1	CTUe	1.50E+03	2.88E+03	-2.19E+04	-2.05E+04
Ecotoxicity. freshwater-part 2	CTUe	6.29E+02	1.13E+03	-3.42E+03	-2.93E+03
Ecotoxicity. freshwater-inorganics	CTUe	1.94E+03	3.75E+03	-2.50E+04	-2.32E+04
Ecotoxicity. freshwater-organics-p.1	CTUe	3.97E+01	5.47E+01	-9.15E+01	-7.66E+01
Ecotoxicity. freshwater-organics-p.2	CTUe	1.45E+02	2.03E+02	-2.31E+02	-1.73E+02
Particulate matter	disease inc.	2.30E-05	4.85E-05	-7.70E-05	-5.15E-05
Eutrophication. marine	kg N eq	2.31E-01	4.04E-01	-5.03E+00	-4.86E+00
Eutrophication. freshwater	kg P eq	6.91E-02	8.81E-02	-5.87E+00	-5.86E+00
Eutrophication. terrestrial	mol N eq	2.51E+00	4.29E+00	-4.63E+01	-4.45E+01
Human toxicity. cancer	CTUh	1.62E-07	2.76E-07	-1.01E-06	-8.91E-07
Human toxicity. cancer-inorganics	CTUh	7.97E-08	1.36E-07	-7.91E-07	-7.35E-07
Human toxicity. cancer-organics	CTUh	8.22E-08	1.41E-07	-2.14E-07	-1.56E-07
Human toxicity. non-cancer	CTUh	3.97E-06	6.76E-06	-4.22E-05	-3.94E-05
Human toxicity. non-cancer-inorg.	CTUh	3.77E-06	6.41E-06	-4.07E-05	-3.81E-05
Human toxicity. non-cancer-org.	CTUh	2.04E-07	3.46E-07	-1.53E-06	-1.39E-06
Ionising radiation	kBq U-235 eq	5.22E+01	5.72E+01	-3.88E+01	-3.39E+01
Land use	Pt	4.64E+01	4.01E+03	-1.74E+04	-1.34E+04
Ozone depletion	kg CFC11 eq	1.33E-05	1.92E-05	-2.35E-05	-1.77E-05
Photochemical ozone formation	kg NMVOC eq	1.58E+00	2.62E+00	-1.29E+01	-1.18E+01
Resource use. fossils	MJ	8.91E+03	1.28E+04	-1.32E+05	-1.28E+05
Resource use. minerals and metals	kg Sb eq	1.11E-03	1.83E-03	-6.40E-04	7.92E-05
Water use	m3 depriv.	-6.03E+00	1.26E+01	-1.36E+02	-1.17E+02
Normalization					
Acidification	-	2.70E-02	3.85E-02	-2.39E-01	-2.27E-01
Climate change	-	7.53E-02	1.09E-01	-1.82E-01	-1.48E-01
Ecotoxicity. freshwater	-	3.76E-02	7.06E-02	-4.46E-01	-4.13E-01
Particulate matter	-	3.86E-02	8.14E-02	-1.29E-01	-8.66E-02
Eutrophication. marine	-	1.18E-02	2.07E-02	-2.57E-01	-2.49E-01
Eutrophication. freshwater	-	4.30E-02	5.48E-02	-3.66E+00	-3.64E+00
Eutrophication. terrestrial	-	1.42E-02	2.43E-02	-2.62E-01	-2.52E-01
Human toxicity. cancer	-	9.38E-03	1.60E-02	-5.83E-02	-5.16E-02
Human toxicity. non-cancer	-	3.08E-02	5.25E-02	-3.28E-01	-3.06E-01
Ionising radiation	-	1.24E-02	1.35E-02	-9.20E-03	-8.03E-03
Land use	-	5.66E-05	4.89E-03	-2.12E-02	-1.64E-02
Ozone depletion	-	2.55E-04	3.66E-04	-4.49E-04	-3.37E-04
Photochemical ozone formation	-	3.86E-02	6.40E-02	-3.15E-01	-2.90E-01
Resource use. fossils	-	1.37E-01	1.97E-01	-2.03E+00	-1.97E+00
Resource use. minerals and metals	-	1.74E-02	2.87E-02	-1.01E-02	1.24E-03
Water use	-	-5.26E-04	1.10E-03	-1.19E-02	-1.02E-02

ing paragraphs, focusing only on the impact categories relevant for both treatment processes (i.e. “Resource use, fossils”, Eutrophication, freshwater”, “Ecotoxicity freshwater” and “Particulate matter”). Moreover, “Climate change” category is added to the discussion, due to its global relevance.

The results of “Climate change” and “Resource use, fossils” for Scenario 2 (co-processing in a furnace for the production of cement), equal to $-1.37\text{E}+03$ kg CO₂eq. and $-1.32\text{E}+05$ MJ respectively”, highlight that its avoided impacts are much higher than the results of Scenario 1 (mechanical recycling), where “Climate Change” is equal to $5.69\text{E}+02$ kg CO₂eq., and “Resource use, fossils” is equal to $8.91\text{E}+03$ MJ (Table 3, Figure 4, Figure 5). Scenario 2 is therefore more environmentally sustainable and this is due to the higher avoided impacts, which are affected by coal substitution in the co-processing.

As regards to “Eutrophication freshwater”, the characterisation result for Scenario 1 is $6.91\text{E}-02$ kgPeq. (positive value), whereas it is $-5.87\text{E}+00$ kgPeq. (negative value) for Scenario 2: the latter therefore is an avoided impact. More in detail (Figure 6), the two processes that most affect the results of this impact category are the electricity consumption to grind the GFRP waste material in Scenario 1 ($6.07\text{E}-02$ kg P eq.) and the avoided impacts from coal substitution in Scenario 2 ($-5.94\text{E}+00$ kg P eq.). On the contrary, the contributions of transport and the use of recycled material to replace sand are negligible.

The characterisation results of “Ecotoxicity freshwater” and “Photochemical ozone formation” show that, similarly

to the other impact categories, the impact differences for the two treatment processes are strongly affected by coal substitution in the co-processing. Figure 7 and Figure 8 show that the avoided impacts of the replaced coal in Scenario 2 are equal to $-2.74\text{E}+04$ CTUe in “Ecotoxicity freshwater” and equal to $-1.45\text{E}+01$ kg NMVOC eq. in “Photochemical ozone formation”, whereas the avoided impacts from the substitution of the sand in Scenario 1 are equal to $-4.93\text{E}+02$ CTUe in “Ecotoxicity freshwater” and equal to $-6.12\text{E}-01$ kg NMVOC eq. in “Photochemical ozone formation”. The avoided impacts of sand substitution in Scenario 1 have therefore the same order of magnitude of the avoided impacts of sand substitution in Scenario 2, where they are equal to $-4.89\text{E}+02$ CTUe in “Ecotoxicity freshwater” and $-6.07\text{E}-01$ kg NMVOC eq. in “Photochemical ozone formation”.

5. CONCLUSIONS

The LCA study of the two scenarios for the treatment of GFRP waste material contained in wind turbines shows that mechanical recycling (Scenario 1) has significantly higher environmental impacts than co-processing in cement kiln (Scenario 2), because in the latter the avoided impacts have a much higher value. This is mainly due to the fact that in the co-processing, in addition to the valorisation of the recycled secondary material at the end of the treatment process, the energy needed for the process is provided by the polymer organic component of the GFRP waste mate-

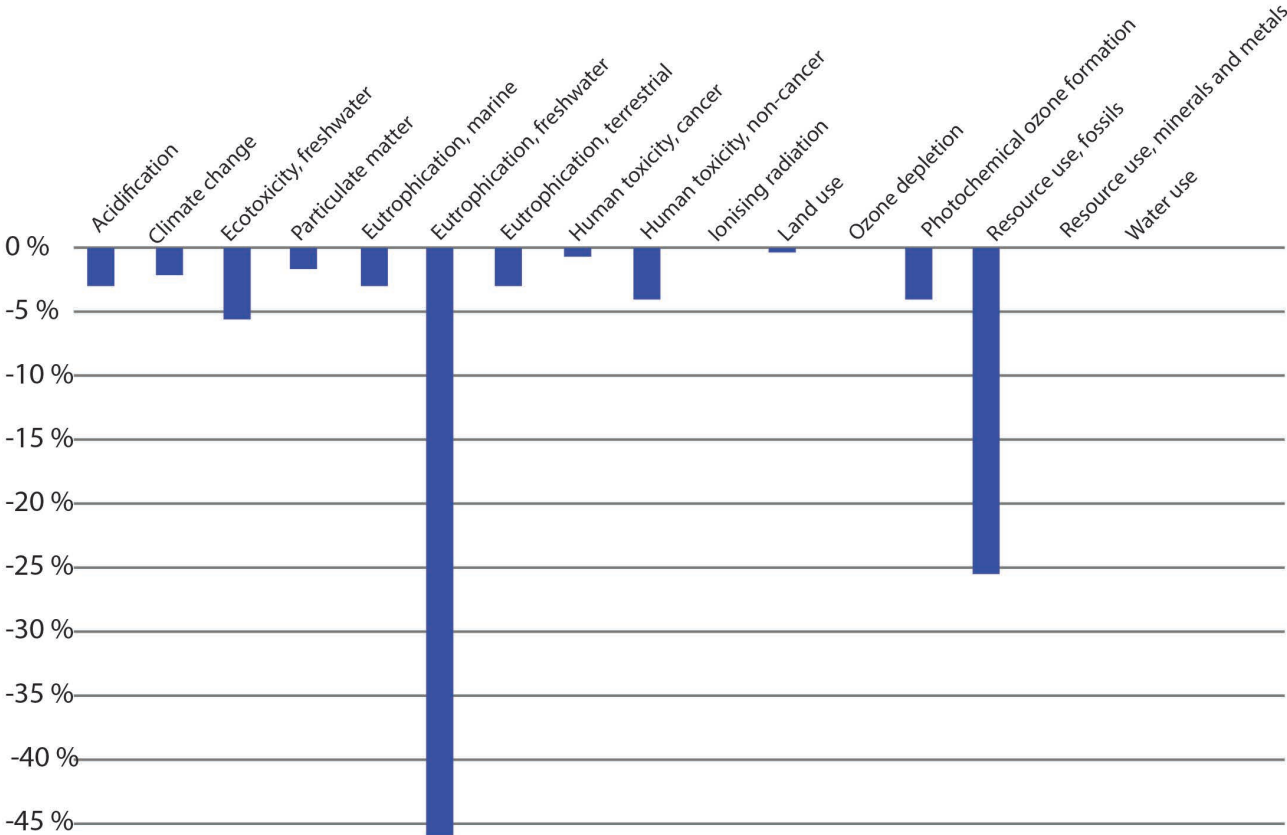


FIGURE 3: Normalization results for Scenario 2 (co-processing in a cement kiln).

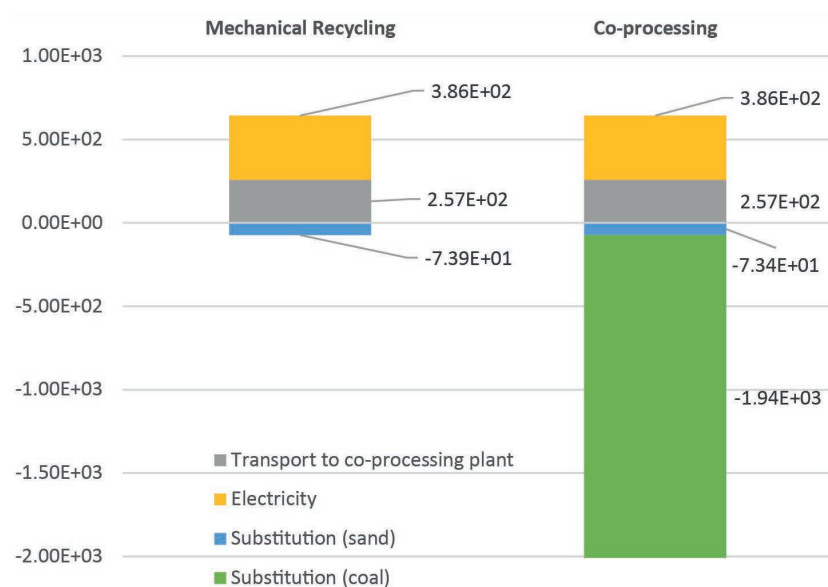


FIGURE 4: Climate change characterisation results (kg CO₂ eq) for both scenarios. Process contribution of the GFRP waste material treatment.

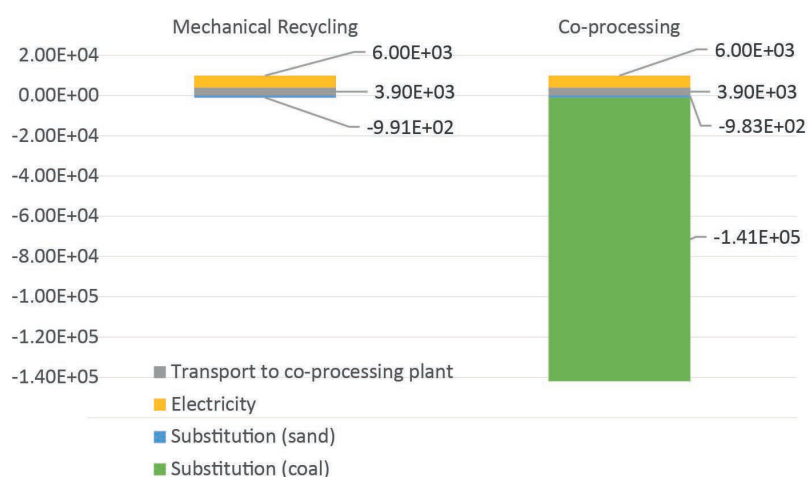


FIGURE 5: Characterisation results of the category Resource use, fossils (MJ eq) for both scenarios GFRP - Process contributions of the GFRP waste material treatment.

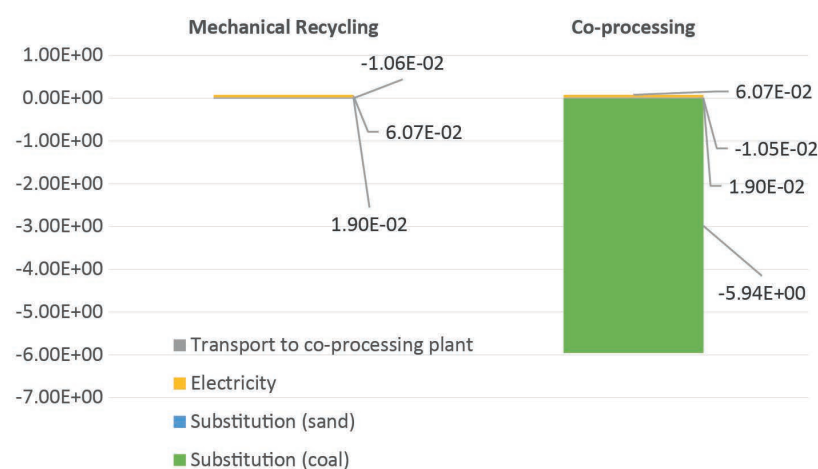


FIGURE 6: Characterisation results of the category Eutrophication freshwater (kg P eq) for both scenarios – Process contributions of the GFRP waste material treatment.

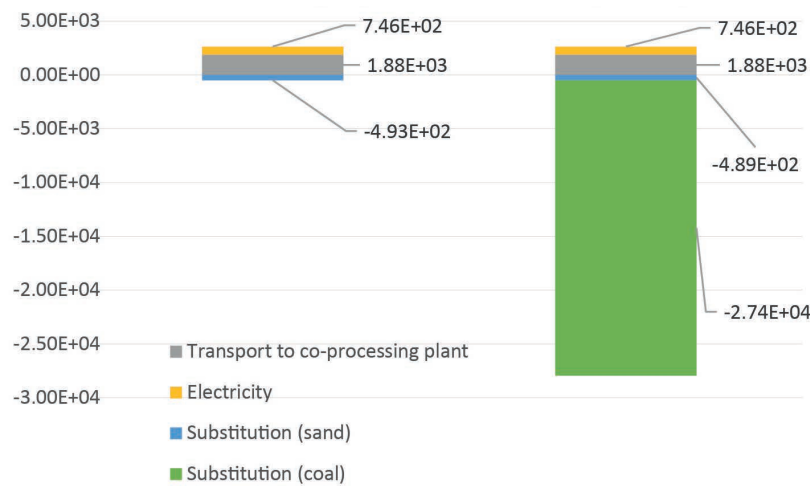


FIGURE 7: Characterisation results of the Ecotoxicity category freshwater (CTUe) for both scenarios - Process contributions of the GFRP waste material treatment.

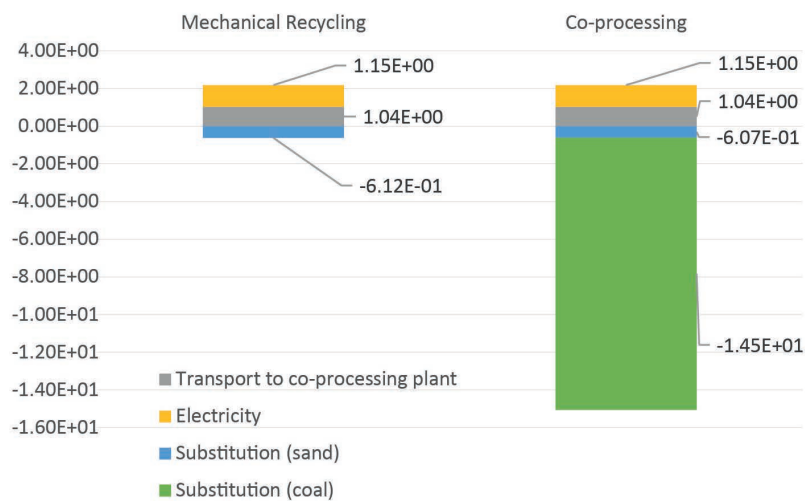


FIGURE 8: Characterisation results of the Photochemical ozone formation (kg NMVOC eq) category for both scenarios - Process contributions of the GFRP waste material treatment.

rial, thus avoiding the production of a similar quantity of coal, which would be necessary for the combustion inside the furnace. In fact, the GFRP waste material, due to its polymer component, has a higher calorific value than the replaced fuel. In this way, the co-processing is more environmentally friendly than mechanical recycling, especially in "Resource use, fossils" (Scenario 1: 8.91E+03 MJ; Scenario 2: -1.32E+05 MJ) and "Climate change" (Scenario 1: 5.69E+02 kg CO₂eq. Scenario2: 1.37E+03 kg CO₂eq.) impact categories.

The analysis of characterisation results shows that also the other relevant impact categories results are affected by the valorisation of GFRP waste material: this is evident in Scenario 2, where coal to feed the process is replaced by the same waste material. In fact, the avoided impacts of "Eutrophication freshwater", "Ecotoxicity freshwater" and "Photochemical ozone formation" in Scenario 2 are higher than in Scenario 1, where only sand is replaced by the GFRP waste material at the end of the mechanical treatment.

Finally, the literature analysis (Yazdanbakhsh et al., 2014) highlights that mechanical recycling has a low energy consumption but, on the other hand since it does not separate the polymer organic component and the fibrous mineral component, it produces a recycled material with poor mechanical properties which is suitable for limited use as a filler or reinforcement in construction. Therefore, this type of treatment produces a barely valuable material if compared to the physical properties and functions of the virgin GFRP material. In order to increase its valorisation potential, the material must undergo further processing, with a greater expenditure of energy, to finally obtain a "fine" grade material, usable as filler for the production of new composites, for example as a secondary material in the production of thermoplastics.

On the contrary, co-processing in cement kilns is a highly efficient process because it can recover energy from the waste material and can integrate part of it into the clinker production process. Nevertheless, the fibrous mineral component of the GFRP waste material lose its physical

form during the process, and this makes its recovery difficult for more valuable applications, such as the production of new composite materials.

In conclusion, the results of this LCA study provide some general considerations about the possible recycling and recovery of GFRP material contained in wind turbine blades, including also an assessment of the environmental benefits obtainable through the secondary material valorisation. These outcomes could therefore be used for future in-depth analysis, by means of the realisation of further LCA studies on recycling and recovery technologies based on primary data. Finally, they could support the definition of ecodesign strategies also useful for improving the environmental performance of wind farms life cycle, taking also into account the dismantling and waste treatment phases, in order to increase recyclability of its components and materials and so the overall circularity of the system.

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BACTERIOLOGICAL EVALUATION IN EXPERIMENTAL LANDFILL CELLS: CO-DISPOSAL OF HOUSEHOLD SOLID WASTE AND HEALTHCARE SOLID WASTE

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ABSTRACT

This work aimed to study the effects of co-disposal of household solid waste (HHSW) and healthcare solid waste (HCW) on the microbiota of the leachate generated in experimental landfill cells. To this end, three waste containment units were built, differentiated by the percentage of HHSW and HCW: HHSW cell (100% household solid waste), HCW cell (100% healthcare solid waste) and COD cell (cell simulating co-disposal containing 98% HHSW and 2% HCW). Volume, temperature, pH and evaluation of the leachate's microbiota from the cells were monitored over 810 days of confinement. Culture-dependent methods (quantitative analysis of environmental contamination indicators and investigation in enterobacteria and *Staphylococcus aureus*), antimicrobial susceptibility patterns, and inhibitory activity were evaluated for microbiota monitoring. The microbiological characterization of the leachates from the cells showed no statistically significant differences in total coliforms, *Escherichia coli*, enterococci and *Pseudomonas aeruginosa* densities during the period evaluated. Enterobacteria of medical importance were identified in the leachates from the three cells: *E. coli*, *Klebsiella pneumoniae*, *Enterobacter* sp and *Proteus mirabilis*. After 810 days of confinement, the leachates showed no antimicrobial activity against *S. aureus*, *P. aeruginosa* and *Salmonella enterica* serovar Choleraesuis. Statistical analysis of antimicrobial resistance patterns revealed significant similarities among the three cells. The results suggest no significant differences in the behavior of experimental landfills containing HHSW and HCW concerning the presence of pathogenic microbiota.

1. INTRODUCTION

All waste produced by medical facilities, labs, and research facilities is referred to as healthcare waste (HCW). Most HCW is non-hazardous, and 85% in weight is similar to household waste; the remaining 15%, which may be radioactive, toxic, or pathogenic, is considered dangerous (Aziz et al., 2022). The HCW management includes the stages of minimization, generation, segregation, packaging, identification, collection, storage, transportation, treatment and environmentally appropriate final disposal (Santos et al., 2022).

There are increasing concerns about the negative impacts of healthcare waste generated in hospitals,

especially in low- and middle-income countries. According to Mol et al. (2022), median values indicated that HCW generation was the highest in North and South America (4.42 and 1.64 kg HCW/bed/day, respectively); Africa was the lowest HCW producer (0.19 kg HCW/bed/day).

Several types of hazardous waste include infectious, sharp, chemical, pharmaceutical, and non-hazardous. Therefore, proper methods for handling HCW must be considered, and segregation must be implemented (Lee et al., 2024).

The treatment of HCW in Brazil is an important public and environmental health issue regulated by various standards and legislation. The main regulation is Resolution



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222 from the National Health Surveillance Agency (ANVISA, 2018), which provides for managing waste from healthcare services. The resolution classifies the HCW into different groups according to its risk potential (Table S1 – Supplementary Material). In Brazil, some state or municipal regulations require the thermal treatment (autoclaving) of part of Group A waste (subgroup A1), such as cultures and stocks of microorganisms, waste from manufacturing biological products, culture media and instruments, and genetic manipulation laboratories. They are treated directly at the source, usually by autoclaving, and are treated again (incineration or autoclave processes) after leaving the healthcare facility. After the second treatment, they become considered common waste. In some cities, the same occurs with some biological waste (subgroup A3, for example, human anatomical parts) that, by law, does not need to be treated. This causes a burden for municipalities, especially low-income ones, as thermal treatment systems are much more expensive than sending them to landfills.

Some researchers consider HHSW a potential agent for spreading pathogenic microorganisms to humans and other animals and a vehicle for spreading antimicrobial drug resistance markers (Nascimento et al., 2009; Chagas et al., 2011; Zhang et al., 2012). On the contrary, studies on the bacterial content of hospital waste indicate that it is not a very good culture medium for pathogens and disease transmission (Mühlich et al., 2003).

While it is true that the treatment of HCW is mandatory in the United States and Europe and similarly required in Brazil, it is important to consider the economic realities developing countries face. In many regions, the high cost of advanced waste treatment technologies presents a significant barrier to implementation. Advanced technologies, such as incineration, autoclaving and microwave ovens, may be inaccessible or limited in scope due to a lack of technical support or investment (Blenkharn, 2006). Around 891 tonnes of solid healthcare waste in Brazil are collected daily (Theophilo, 2018). Although regulations mandate healthcare waste treatment in Brazil, the financial burden associated with these treatments is often prohibitive for many municipalities. These costs are not always sustainable for cities in developing countries, where resources are limited and competing priorities are numerous. Therefore, it is crucial to explore alternative solutions that are both effective and economically feasible for ensuring public health and environmental standards are met without imposing undue financial strain on developing regions.

In this context, the ability to release microorganisms and substances that compromise the environment is not inherent to waste originating from healthcare establishments. Waste generated at home, in commerce and industry, as well as effluents from water treatment plants, are known for their contribution to the load of antimicrobial drugs and sanitizers in wastewater and microorganisms from different sources of contamination, such as diapers, condoms, sanitary pads and paper (Threedeach et al., 2012; Theophilo et al., 2018), as well as piercing and scarifying materials (Cussiol, 2005; Theophilo et al., 2017). In contrast, not all waste classified

as HCW results from procedures applied to patients with infectious diseases, but from plastic surgery, victims of various accidents (car accidents, falls, cuts) and diseases non-infectious, like heart disease. Recently, SARS-CoV-2 was detected on HHSW fresh truck leachate in a study to investigate the potential of SARS-CoV-2 surveillance in these samples (Lanzarini et al., 2023).

The co-disposal method can be used to treat HCW. However, this method is only allowed during emergencies in some countries (Lee et al., 2024).

Because of the disposal of solid waste in landfills, leachate is currently considered the most impactful emission from an environmental point of view. Exposure of the environment to leachate can occur in different ways: through excessive rainfall, uncontrolled overflow from equalization ponds, infiltration or failures in treating leachate discharged into water bodies. The impact of landfill leachate on microflora and microfauna is determined by factors such as high concentrations of organic matter and ammonia nitrogen, as well as other contaminants carried by the mass of waste (Liu et al., 2011; Costa et al., 2019). In this sense, the landfill is a complex heterogeneous system in terms of its physical, chemical and biological characteristics, where different microorganisms coexist and interact (Szyłak-Szydłowski and Korniłowicz-Kowalska, 2011).

The characteristics of the leachate generated from the disposal of solid healthcare waste in landfills can be similar to those of hospital wastewater (Novo and Manaia, 2010). The entry of hospital-origin bacteria into an environment can represent a significant source of the spread of antimicrobial resistance (Prado et al., 2008). In particular, the occurrence of antibacterial agents in the environment raises concerns about environmental risks and the spread of antimicrobial resistance among microorganisms (Chagas et al., 2011). Whether products with antimicrobial activity are transported into the environment from human or animal sources, select antimicrobial-resistant strains of bacteria, or cause deleterious effects on water quality are controversial but unresolved issues (Chang et al., 2010).

The confinement of waste in landfills can provide microbial communities with the right conditions for their interactions and successions, making leachate a vehicle for multidrug-resistant microorganisms and producers of pathogenicity mechanisms (Liu et al., 2019).

This study aimed to investigate whether healthcare solid waste and household solid waste differ significantly in terms of their microbiota and their pathogenic potential when confined in a landfill. For this purpose, the presence and the diversity of pathogenic microorganisms and phenotypic characteristics were analyzed using experimental landfills for the confinement of household solid waste and healthcare solid waste. The novelty of this work lies in comparing the microbiota and pathogenic potential of healthcare solid waste and household solid waste when confined in a landfill. This study investigated potential differences by analyzing the diversity of pathogenic microorganisms and phenotypic characteristics in experimental landfills dedicated to each waste type.

2. MATERIALS AND METHODS

2.1 Landfill experimental cells

The study was carried out by monitoring leachates generated from three landfill experimental cells. The site chosen to construct the experimental cells was the Recycling and Composting Plant of Jacarepaguá, which belongs to COMLURB (Municipal Urban Cleaning Company – Rio de Janeiro Municipality, Brazil). Three experimental cells were built, arranged side by side, separated by a distance of 2.0 m each other (top of slope) in a total plane area of 129 m² model as shown in Figures S1a, b and c (Supplementary Material).

Implementation steps followed the essential criteria for the construction of landfills (Brazilian Association of Technical Standards- ABNT NBR 15849, 2010), such as liner system, waste compaction, drainage of the leachate, rainwater and gases, input control waste (weighing), solid waste disposal, coatings with a layer of soil and environmental monitoring.

The cells were identified as the composition of their waste. The HHSW cell comprised 100% household solid waste and 43.350 tonnes of mass. The HCW cell comprised 100% infectious healthcare solid waste and a mass of 13.680 tonnes. The third cell, the COD cell comprised 98% of HHSW and 2% of HCW, with a mass of 43.207 tonnes. This composition corresponds to the mass percentage of urban solid waste sent to the final disposal units of the public system of the city of Rio de Janeiro (PMGIRS, 2024).

HCW was deposited in the cell as they were collected from the generating units and belonged to groups A, D and E (Table S1, Supplementary Material). According to Brazilian legislation (ANVISA, 2018), subgroup A1 waste cannot leave the generating unit without prior treatment. This waste is autoclaved but still leaves the units belonging to subgroup A1. The other groups, for which no prior treatment was required, were disposed of as they were collected at the generating unit. HCW from Group E was disposed of in cardboard boxes at the generating unit and received no treatment before final disposal.

Each cell had the following dimensions: 4.5 m lower base, 6.0 m higher base, 5.0 m high and 6.0 m side length, totaling 157.5 m³ volume capacity. For forming embankments and bases of cells (berms), construction wastes were used. The base and sides of the cells were lined with a high-density polyethylene (HDPE) geomembrane liner, with 1.0 mm thickness. The drainage gutters were excavated manually, and the center of the cell had a depth of 30 cm wide and 4.0 m long. After HDPE (high-density polyethylene) geomembrane, drainage trenches are filled with a layer of geotextile (Bidim 400.5 x 2.5 m). In sequence, the drainage system was installed, consisting of a tube canaflex type, connected to a PVC tube receiver coupled to a 50 mm PVC-registration union 50 mm and a tap with a 90° curve exit for collection samples. The tubes containing the trenches were filled with crushed n°. 3, starting at a height of 10 cm (back of the cell) and deepening 30 cm (front cell). The Bidim was then folded over the gravel bed, wrapping it completely. Besides filtering the effluent, this device facilitates its flow, avoiding clogging. Vertical drains

were built in a circular section with PVC pipe DN 100 mm perforated (3/8 mm holes), 2.3 m long, and a wire mesh of 1 m wide and 2 m long. The system extended from the base of the waste mass to 20 cm above the surface layer of clay. The tube was disposed of at the center of the cell, on the drainage leachate. Then, the wire screen is arranged around the pipe, 10 cm, and its side was stitched with wire to wrap around it. The space between the tube and the screen was filled with crushed stone n° 3. After filling the cells and surface waterproofing, galvanized iron burners were attached to the PVC pipe because there was no intention to collect gas generated during the experiment.

2.2 Gravimetric composition of waste

The gravimetric composition consisted of determining the percentage fractions of different types of waste obtained by sampling the collections made in the following neighborhoods in the municipality of Rio de Janeiro. One 240L container of waste was taken from the contents of each truck. The waste was homogenized using a mechanical shovel. HCW was collected on 2 consecutive days from the route of various healthcare establishments.

Samples were collected using the quartering method (ASTM, 2003). After homogenization, the waste was divided into four equal parts. From each part, two samples were taken from diametrically opposite positions. Each sample was equivalent to a 200L container. The total amount sampled at this stage was approximately 1600L. The quartering procedure was repeated thrice on the split samples, reducing the total sampled to 800L, 400L and 200L (final sample). From the 200L sample, the materials were separated and differentiated into 11 components: Paper, Cardboard, Plastic (polyethylene, polypropylene, polystyrene), Glass (colorless/colored), Metal (ferrous/non-ferrous), Textiles, Putrescible Organic Matter, Wood, Electronics, Inert materials and Others.

The components separated by the methodology proposed in this project to determine the gravimetric composition of household waste were assessed for the presence of hazardous waste. They were characterized by previously determined elements, as shown in Table 1.

2.3 Leachate Collection

Leachate were drained daily through the tap until the cell was empty. The samples were preserved by packaging at a temperature below 4°C for later carrying out the physicochemical and microbiological analyses.

2.3.1 Physicochemical analyses

The pH and temperature were measured. For temperature measure, a mercury glass thermometer with a scale of -10 to 100°C was inserted into the volume of liquid used to measure the temperature. The temperature in the external environment was also measured.

2.3.2 Microbiota analyses- culture-dependent methods

- Total coliforms, *Escherichia coli* and enterococci
The analyses were carried out following the APHA (2012). The densities of total coliforms, *Escherichia coli* and enterococci were expressed as the Most Probable

TABLE 1: Potentially hazardous components of HHSW.

Component	Potentially infectious or chemical constituents
Paper	Toilet paper; Paper towel
Plastic	Disposable syringe; Shaving razor; Flexible rods; Serum bottles; Absorbent; Disposable diaper; Hair dye gloves; Vaginal applicator; Bandages; Chemical product bottles
Glass	Shards; Ampoule; Lamp; Medicine bottles
Putrescible organic matter	Animal feces, Carcasses, animal parts
Metal	Needle; Pin; Open can; Nail, screw, tack; Staple; Cutlery; Razor blades; Rebar; Chemical bottles; Pill bottle
Rubber	Procedure gloves
Textile material	Bandage; gauze; cotton; adhesive plaster; burlap
Other	Cat sand; ceramic shards, wood splinters

Numbers per 100 milliliters of sample (MPN 100 mL⁻¹) and determined using the Multiple Tube Technique.

- *Pseudomonas aeruginosa*

The multiple tube technique determined the analysis according to ISO 16266 (2006). The method consisted of inoculating serial dilutions in asparagine broth (5 tubes per dilution) followed by incubation at 36°C ± 0.5/24-48h.

- Total aerobic mesophilic heterotrophic bacteria

The deep plating technique was used with Tryptone Glucose Yeast Extract Agar ("Plate Count Agar"), starting with dilutions 10⁻¹, 10⁻², 10⁻³ and 10⁻⁴. The samples were analyzed in duplicate for each dilution and incubated at 36°C ± 0.5/24-48h. Heterotrophic bacterial populations were expressed as Colony Forming Units per milliliter sample (CFU mL⁻¹).

- Enterobacteria testing

For the isolation of enterobacteria, an inoculum taken from the leachate sample was transferred to eosin methylene blue agar (EMB agar), and plated in duplicate, followed by incubation at 35-37°C for 24 hours.

Suspicious colonies were selected, and the identification of the bacterial species was based on the phenotypic characteristics provided by conventional biochemical tests following the ANVISA (2013) for the detection and identification of bacteria of medical importance, which indicates the following tests as the main ones for identifying enterobacteria: oxidase, lactose fermentation, glucose fermentation, L-lysine decarboxylation, gas production, use of citrate as the sole carbon source, indole production, hydrogen sulfide (H₂S) gas production, urease production, phenylalanine deaminase production and motility. The complementary tests were used for the beta-galactosidase (ONPG), ornithine decarboxylation, arabinose fermentation, and tryptophan deamination tests.

Confirmation of the identification was made using the API (Analytical Profile Index) System program for detecting similarities between species.

Salmonella testing was based on ISO 6579 (2017). The test consisted of the selective enrichment of 1 mL of the crude sample in 10 mL of Kauffmann's tetrathionate broth, added to an aqueous iodine solution and a 0.1% brilliant green solution. After incubation, an inoculum of the broth was transferred to plates with the selective-indicator media Salmonella-Shigella agar (SS agar)

and xylose lysine deoxycholate agar (XLD), followed by incubation at 35-37°C for 24 hours. Slide agglutination tests using polyvalent serum "O" (Probac do Brasil Produtos Bacteriológicos Ltda.) confirmed colonies indicative of Salmonella.

- *Staphylococcus aureus* testing

The occurrence of *Staphylococcus aureus* was recorded as presence or absence. As a selective enrichment method, 10 mL of the raw leachate was transferred to salt mannitol broth, followed by incubation at 36°C ± 0.5 / 24 hours. After this period, the culture was sown on salt mannitol agar with egg yolk (20g/L) in triplicate, followed by incubation at 36°C ± 0.5 for 24 hours (McFaddin, 1977). Colonies that were translucent yellow surrounded by a precipitation halo were subjected to the catalase and coagulase tests (Nascimento et al., 2009). *Staphylococcus aureus* (ATCC 6538) strains were used as a positive control, and *Enterococcus faecalis* (ATCC 11700) as a negative control.

- Susceptibility of the isolated strains to antimicrobials

According to the CLSI (2023), the agar diffusion method was used for this analysis. Pure colonies of recently cultured bacteria (18-24 hours) isolated from non-selective culture medium were suspended in sterile saline solution (0.85% NaCl) until turbidity compatible with 0.5 on the Mac Farland scale (106 CFU/mL) was obtained. A sterile swab was soaked in the bacterial suspension and then sown gently in all directions on the plate (five directions), trying to cover the entire surface. The disks were deposited on the surface of the inoculated medium. The plates with the disks were incubated in a bacteriological oven at 36°C for 18 to 24 hours.

Based on the diameters of the inhibition halos, the samples were defined as susceptible or resistant to each of the antimicrobials tested, using standards established by CLSI (2023). Standard strains of *E. coli* (ATCC 25922), *P. aeruginosa* (ATCC 27953) and *S. aureus* (ATCC 25923) were used as a quality control. In this study, the intermediate susceptibility profile was considered resistant. For the strains of enterobacteria isolated in the leachate, the following antimicrobials were selected according to CLSI (2013): Ampicillin (10µg); Amoxicillin-clavulanic acid (20/10µg); Cefoxitin (30µg); Cefazolin (30µg); Cefotaxime (30µg); Ceftazidime (30µg); Cefepime (30µg); Amikacin (30µg);

Gentamicin (10µg); Meropenem (10µg); Imipenem (10µg); Ciprofloxacin (5µg); Aztreonam (30µg) and Sulfamethoxazole-Trimethoprim (25µg). For the enterococci strains isolated in the leachate, the following antimicrobials were selected according to CLSI (2013): Ampicillin (10µg); Oxacillin (10µg); Gentamicin (10µg); Gentamicin (120µg); Streptomycin (10µg); Streptomycin (300µg); Cefoxitin (30µg); Ciprofloxacin (5µg); Teicoplanin (10µg); Sulfamethoxazole-Trimethoprim (25µg) and Vancomycin (30µg). For the *Pseudomonas aeruginosa* strains isolated in the leachate, the following antimicrobials were selected according to CLSI (2013): Amikacin (30µg); Gentamicin (10µg); Meropenem (10 µg); Imipenem (10µg); Ciprofloxacin (5µg); Aztreonam (30µg); Cefotaxime (30µg); Ceftazidime (30µg); Cefepime (30µg); Piperacillin/Tazobactam (100/10µg).

- Search for ESBL-producing enterobacteria (extended-spectrum beta-lactamase enzymes)

The procedure recommended by CLSI (2023) included an initial screening test using disk diffusion by analyzing the inhibition halos obtained using disks of third-generation cephalosporins and aztreonam. The presence of inhibition halos smaller than 22 mm for the ceftazidime disk (30 µg), < 27 mm for the cefotaxime disk (30µg), < 25 mm for the ceftriaxone disk (30µg) or < 27 mm for the aztreonam disk (30µg) should raise the suspicion of ESBL production (CLSI, 2013). To confirm the phenotype, the disk approximation test proposed by Jarlier et al. (1988) was used (Zemelman et al., 2002; Dalmarco et al., 2006). This method placed a disc containing amoxicillin and clavulanic acid in the center of a Mueller-Hinton agar plate. Around this, the marker antimicrobials (Aztreonam, Cefepime, Ceftazidime and Cefotaxime) were positioned at 20 mm from the amoxicillin-clavulanate disk. The appearance of a "ghost zone" or the widening of the cephalosporin inhibition halo confirms the production of ESBL. The standard strain *Klebsiella pneumoniae* (ATCC 700603) was used as a positive control for the test. In contrast, *Escherichia coli* (ATCC 25922) was used as a negative control.

2.4 Statistical analysis

The data obtained from determining the volume of leachate and rainfall at the site was analyzed using Pearson's correlation coefficient. Pearson correlation is a statistical technique used to measure the degree of relationship between two variables. Pearson correlation coefficient takes values between -1 and 1, where 1, 0, and -1 indicate a perfect match, no correlation, and perfect negative correlation, respectively (Berman, 2016).

The data obtained from determining microbiological parameters was applied to Box Plot graphs and subjected to statistical analysis using non-parametric tests.

The Kruskal-Wallis test was used to compare the medians (the null hypothesis in Kruskal-Wallis testing was H_0) of the population densities of total coliforms, *E. coli*, *Enterococci* and *P. aeruginosa* and pH, with a confidence level of 95%. When the value of the Kruskal-Wallis test statistic indicated that at least one pair of sample groups

was different from the others, the multiple comparisons test should be applied, as suggested by Siegal and Castellan (1988).

Fisher's exact probability test was applied to check the similarities between antimicrobial susceptibility profiles. This test requires the samples to be random and independent, where the classes are mutually exclusive, and the measurement levels are on a nominal scale (Siegal and Castellan, 2008). An advantage of this test is that there are no restrictions on the sample size of each cell.

Statistical tests were performed on Statistica version 7.

3. RESULTS AND DISCUSSION

3.1 Cell content and waste composition

The mass of waste needed to fill the HHSW and COD cells was approximately 3 times greater than that of HCW needed to fill HCW. The differences in the specific weight of HCW and HHSW explain this fact. This characteristic is related to the compression or compaction properties of the waste. The HHSW deposited in the HHSW and COD cells had a specific weight of 320 kg/m³, while the waste disposed of in the HCW cell had a specific mass of 101 kg/m³.

Figure 1 (a and b) shows the gravimetric composition of the HHSW destined for the cells. The highest percentage found among the components was the putrescible organic matter, represented by the remains of processed food, expired food, fruit and vegetable peelings and feces. As for the recyclable components of HHSW, plastic had the highest percentage among potentially recyclable materials (19%w/w), followed by paper (16% w/w). The waste composition that filled the cells (HHSW and COD) is similar to the mean composition of Brazilian solid waste (SNIR, 2024). Figure 1 (c and d) show the percentages of the components identified as HCW. Of the total recyclable materials, plastic products accounted for the most considerable fraction (38% w/w), followed by "Other" components (31% w/w). In contrast to HHSW, the "organic matter" component in HCW, represented by anatomical parts, blood, viscera, and food scraps, is found in a lower percentage concerning recyclable materials, contributing 13%.

Common solid waste was also found with the HCW, such as leftover food, yogurt pots, cookie wrappers and soda bottles, revealing the poor management of this waste in hospital environments, mainly due to the lack of preparation of healthcare professionals.

In the context of this study, it was essential to highlight the presence of materials with infectious piercing or scarifying potential originating in the home. The infectious portion of HHSW was characterized by 859 kg of waste collected from 22 residential districts in Rio de Janeiro municipality. Voudrias et al. (2012) determined the composition and production rates of pharmaceutical and chemical waste produced by a Hospital in Greece and found that the total production of chemical waste comprised 1.8% w/w of the total hazardous medical waste produced by the hospital.

Around 17% of the mass of HHSW collected corresponded to hazardous waste. Plastic was

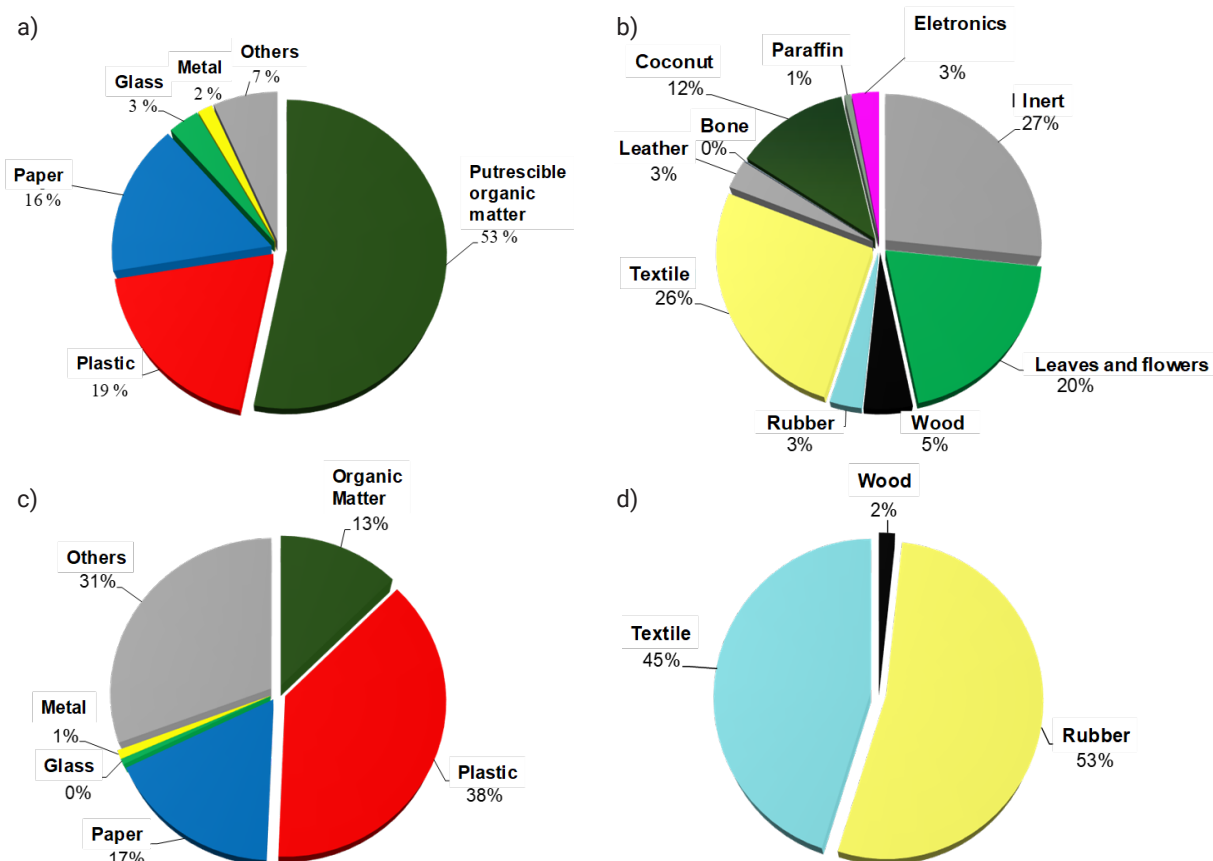


FIGURE 1: Gravimetric composition of the (a) HHSW destined to fill the HHSW and COD Cells; (b) fraction "Other"; (c) HCW destined to fill the HCW and COD Cells; and (d) fraction "Other".

predominantly used as a potentially infectious or chemical material. Figures S2, S3 and S4 (Supplementary Material) show the identification percentages of these materials according to the main component. Around 25% of the sample's paper/cardboard and plastic components were identified as potentially infectious (hazardous waste) (Figures S2 and S3 – Supplementary Material). Regarding the organic fraction, around 1.53% showed infectious characteristics (Figure S4 – Supplementary Material).

Materials such as needles, razors, vaginal applicators, disposable syringes and gloves, flexible rods, dye gloves, ceramic shards and condoms were detected. However, their percentage was less than 0.01% in HHSW. Based on the information from the analysis of the composition of HHSW and HCW, both had in their composition medicines, drugs, used diapers, sanitary napkins, condoms, disposable clothing, food waste, sanitary paper, disinfectant bottles, and other chemical substances such as cresols and ammonia, solvents, pigments, batteries and sharp materials (hypodermic needles) and scarifying materials (razor blades).

3.2 The volume of leachate produced

In this study, leachate generation was constant, and the efficiency of the drainage and collection system installed in the cells was proved. The mean volumes of leachate generated in the cells each month and the rainfall in mm measured at the pluviometry station are shown in Table 2.

When comparing the mean volumes of leachate generated during the month with the pluviometry using Pearson correlation analysis, a perfect positive association was found between the values (>0.8). A Pearson coefficient greater than 0.7 indicates a strong correlation between two variables (Berman, 2016). Therefore, the results indicate a positive and strong correlation between the rainfall and leachate generation.

During the driest season (winter), a lower volume was generated in the systems of the 3 cells (maximum volume values of 28.3, 47 and 55.2 L for HHSW, HCW and COD cells, respectively). During the rainiest periods of the year (summer) in the municipality of Rio de Janeiro, the volumes reached maximum values (254, 327.5 and 450 L for HHSW, HCW and COD cells, respectively). The presence of water in the waste due to the initial humidity and the rainfall that infiltrates are important mainly for the first step of anaerobic degradation (hydrolysis), promoting the dilution of inhibitory agents and facilitating the distribution of microorganisms and nutrients in the MSW mass.

The volume of percolate produced in sanitary landfills depends on the following factors: precipitation and evapotranspiration in the landfill area; surface run-off; underground infiltration; natural humidity of the waste mass; degree of compaction of the waste; moisture retention capacity of the waste mass (Costa et al., 2019). In addition, the quality of a landfill's operation affects the amount of leachate produced. Operational care with adequate cover

TABLE 2: Average values of leachate generation and rainfall per monitoring month. HHSW - Household Solid Waste; HCW - Healthcare Waste and COD - Co-disposal of HHSW and HCW.

Season/ Year	Average leachate volume (L) generated in each cell per month of monitoring			Accumulated rainfall per month of monitoring (mm)
	HSW	HCW	COD	
Spring – Year 1	115	233	159.5	179.2
Summer - Year 1	152.5	327.5	398	82.8
	26	84	80.5	45.2
	3	6.2	5.8	123.6
Autumm - Year 1	44.7	80.4	139.9	154.6
	27.4	58.9	88.6	113.6
	8.9	21.6	20.4	34
Winter - Year 1	17	28.6	55.2	49.4
	7.4	11.6	14.3	51.6
	28.3	47	27.8	32
Spring - Year 2	20.7	100.9	140.5	160.4
	26	26.9	19	88.2
	40.2	35.9	39.6	204
Summer - Year 2	48.9	48.9	87	155.4
	16.5	23.9	24.3	48.6
	6	8.1	9.9	80.6
Autumm - Year 2	5.1	4.9	6	119.4
	113	109.2	167	148.8
	117.4	100.5	162.5	156
Winter - Year 2	15.8	22	24.3	54.6
	19.9	27.2	36.7	16.8
	22	11.2	30.3	88.4
Spring - Year 3	39.6	31.3	47.8	44.4
	77	61.5	141	94.2
	45	65.5	88.9	28.4
Summer - Year 3	126	91	245	125
	254	225	450	128.6

can reduce the amount of leachate generated, even in conditions of heavy rainfall (Gomes and Silva, 2005).

In this study, some factors were equivalents among the cells, such as environmental conditions (local rainfall, external temperature, and wind regime), constant cover layer material, absence of a vegetation cover layer and construction aspects (occupied area, slope inclination, drainage system and base waterproofing). Other factors were different, such as the percentage of recyclable and organic material in each cell and the degree of compaction of the waste and the cover layer.

3.3 pH and temperature

It was expected that during this first decomposition stage, there would be an increase in temperature because the oxygen present in the spaces between the newly buried waste is quickly consumed by aerobic microorganisms, producing carbon dioxide, water and an increase in temperature. However, the values indicate no variations in the 342 temperature measurements taken directly on the liquids leached from the cells (low standard deviation).

Among the cells, the mean values were not statistically different (HSW cell=29.1±0.8°C; HCW cell=28.4±0.5°C; COD cell=28.6±0.6°C). The mean leachate temperature was in the mesophilic range (between 25 and 45°C), which is suitable for the growth of most pathogenic microorganisms, as well as indicators of environmental contamination.

The pH monitoring results are shown in Figure S5 (Supplementary Material). The pH values showed statistically significant differences. The average values and ranges for the HHSW, HCW and COD cells were, respectively, 7.0 (5.8-7.7), 6.8 (5.6-7.4) and 7.2 (6.7 - 7.7). Only one observation with a pH of 5.8 was obtained in the HHSW cell after 15 days of confinement and one with a pH of 5.6 in the HCW cell after 45 days of confinement, which can be explained by the acidic characteristic of the leachate in the initial stages of biological degradation of the organic matter present in the solid waste (Kjeldsen et al., 2002; Costa et al., 2019). The fact that the pH of the COD cell does not reduce to weak acid at the beginning of confinement can be explained by the heterogeneous nature of the waste. Alkaline materials, such as construction

waste, can increase the leachate pH. However, the pH results of the leachate generated in the 3 cells point to the optimal range for bacterial growth, between 6.5 and 7.5 (Kjeldsen et al., 2002) in all the observations after 45 days of confinement.

Other physicochemical parameters and ecotoxicity (organisms *Danio rerio* and *Vibrio fischeri*) of the HHSW and HCW cells at different monitoring times were statistically evaluated by Costa et al. (2019b). This study concluded that leachate from healthcare solid waste presented less or equal polluting potential compared with household solid waste: Values for organic matter parameters (COD-Chemical Oxygen Demand, TOC-Total Organic Carbon, BOD-Biochemical Oxygen Demand and Absorbance at 254nm), color, hardness, conductivity, sulfate, alkalinity, chloride, total solids were significantly higher from HHSW cell than HCW cell. Phosphorus, ammonia nitrogen and metals showed results statistically similar for both cells. Comparing EC50-50% Effective Concentration (for *V. fischeri*) and LC50-50% Lethal Concentration (for *Danio rerio*), the results showed that the HCW cell generated leachate that presented less or equal toxicity than the HHSW cell.

3.4 Microbiota analyses- culture-dependent methods

Figure 2 (a, b, c and d) shows the boxplot of the log of for the total coliform, *E. coli*, Enterococci and *P. aeruginosa* densities in the 3 cells. Table S2 (Supplementary Material) shows the mean values and variations for these measures.

The comparative evaluation of the cells showed similarities between the values observed, highlighting the biochemical versatility of this microorganism and its ability to survive in different environments. The bacterial populations of total coliforms, *E. coli* and *P. aeruginosa* in the three cells showed no statistically significant difference ($p>0.05$) over 810 days of monitoring (54 observations) when subjected to the Kruskal-Wallis test. Enterococci density from HCW presented a significant difference compared to other cells presenting lower values.

By dividing the monitoring time into two phases, the first year and the second year, the results were different: the statistical evaluation showed different results for each year of analysis: 25 observations referring to the first year of monitoring and 29 observations referring to the second year of monitoring. In the first period evaluated, the bacterial populations of total coliforms, *E. coli*, enterococci and *P. aeruginosa* in the three cells showed no statistically significant difference ($p>0.05$) when subjected to the Kruskal-Wallis test. However, in the second year evaluated, the data show that only for the *P. aeruginosa* population, the values are not statistically different for the three cells. The leachates from the cells showed different values for the total coliforms, *E. coli* and enterococci densities. There were no differences between the values of the HHSW and COD cells for any bacterial populations evaluated. However, when comparing HHSW and HCW cells, there were differences in the densities of these bacterial populations. Analysis of the average of the 29 observations in each cell indicated that the density of

total coliforms and *E. coli* was higher in the HHSW cell than in the HCW cell. When comparing the HCW and COD cells, there were differences in the values obtained for the total coliform and enterococcus populations, with the lowest value for the HCW cell, while for the *E. coli* parameter, the values were significantly the same.

It was not possible to establish a correlation between the values observed and the time of confinement. Populations of total coliforms and *E. coli* were detected at 15 days of confinement (first observation) and 810 days of confinement (last observation) in the 3 cells. The results point to understanding the solid waste landfill as a dynamic biological system in which different microbial growth phases occur simultaneously.

Smaller enterococci density was obtained in the HCW cell in most observations. This result may be associated with the absence of fresh feces in the HCW, as opposed to the HHSW, whose gravimetric composition includes human feces and, above all, the mammals and birds feces, as well as materials from domestic animals (sand and absorbents for hygiene). The increase in the population of animals in households has been considered a public health problem due to the presence of pathogenic microorganisms and because they are reservoirs of antimicrobial-resistant bacteria, such as vancomycin-resistant enterococci (VRE) (Cinquepalmi et al., 2013).

Another point to note is the presence of *Escherichia coli* in most of the observations, which has the highest population density among the microorganisms in the coliform group. This result indicates that the solid waste confined in landfills provides the conditions for *E. coli* to survive and develop, even outside the intestinal tract. The intestinal tract is the primary habitat of this microorganism. For some authors, *Escherichia coli* did not demonstrate the ability to survive well in the environment, i.e., in secondary habitats such as surface water, sediments and soil (Ishii et al., 2007). However, it has been shown that *E. coli* can survive in these environments for long periods and grow, even in temperate climates (Ibekwe et al., 2011).

As a strict aerobic microorganism, detecting *P. aeruginosa* throughout the experiment indicates that the landfill has aerobic points among the various accumulated materials. The oxygen dissolved in the rainwater supplies its growth needs. This strengthens the hypothesis that such specialized degraders are favored in the lack of easily degradable organic matter and the initiation of oxygen penetration (Remmas et al., 2017). Song et al. (2015), using 454 pyrosequencing, showed that *Pseudomonas* dominated in landfill leachate in China and in the solid phase of landfill refuse (2017).

Saini et al. (2004) confirmed that non-hospital waste contains more pathogens than HCW and that the microbiota present in HCW and non-hospital waste are similar. For Souza (2003), with a few exceptions, pathogenic microorganisms do not resist environmental conditions external to the human body and other animals, which makes environmental contamination in the medium- and long-term equivalent to that of HCW.

Comparisons were made between the average bacterial population densities of the waste that filled the

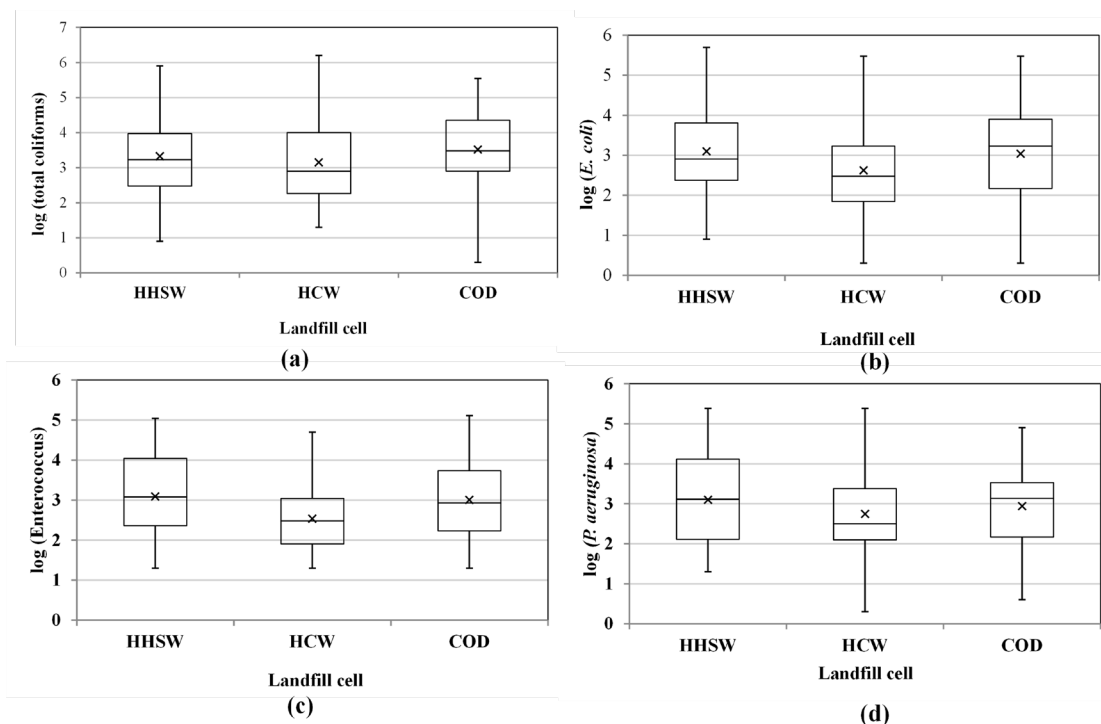


FIGURE 2: Boxplot of results of microorganism densities. (a) total coliforms (b) *E. coli* (c) Enterococci (d) *P. aeruginosa*. Results 810 days of confinement (n=54). All results (y-axis) are reported as the logarithm of the most probable number (MPN per 100mL).

cells and the average values observed in the last 90 days of monitoring. Before confinement, samples were taken of the accumulated liquid from the basins of the trucks that transported the waste. Figure 3 shows that confining the waste reduced the bacterial density in HHSW and HCW cells. As described by Mangimbulude et al. (2009), it is possible to observe a decrease in the absolute load of parameters such as microorganisms, ammonia and heavy metals in landfill leachate.

Based on the data obtained from the microbiota characterization using cultivation-dependent methods, the leachate generated in the HCW cell showed lower values for total coliforms, *Escherichia coli* and enterococci compared to the other cells. The values for the same parameters were significantly similar for HHSW and COD. To this end, the multiple comparison test was used, suggested by Siegal and Castellan (1988). Analysis of the median indicated that the population of enterococci was lower in the RSS Cell than in the others.

Table 3 shows the densities of heterotrophic bacteria detected in the leachate from the three landfill cells. These bacteria were detected in all observations and had compatible average densities considering the standard deviations. The maximum values were observed in the 3 cells after 165 days of confinement, the minimum values after 300 days for the HHSW cell, and 480 days of confinement for the others.

The diversity and concentration of heterotrophic bacteria determined by an analytical method will depend on the variables of that method, which include the composition of the medium, the incubation time and temperature, and the inoculation technique. Only a tiny

proportion of the metabolically active microorganisms present in the sample will grow and be detected depending on these variables. These results can also vary significantly among different sites, periods of year and even between consecutive samples taken at the same site (Gómez et al., 2012).

Figure 4 shows the number of strains isolated in the cell leachate in 54 observations collected over 810 days of monitoring, highlighting the prevalence of 3 strains: *E. coli*, *K. pneumoniae* and *Enterobacter* sp. According to Adeyeba and Akimbo (2002), landfill leachate is generally considered free of pathogenic organisms due to its inactivation under adverse conditions during waste degradation. However, in the present study, 192 strains of enterobacteria of medical importance were isolated in the leachate samples from the 3 cells.

In order to evaluate the profiles of bacteria identified in the cells in association with the time of confinement, isolation patterns were generated in two periods: the first year and the second year of confinement, as seen in Table S3 (Supplementary Material). The patterns observed in the second year showed greater diversity than in the previous year for the 3 cells. *Enterobacter* sp, *K. pneumoniae*, *E. coli*, and *P. mirabilis* were identified in all samples. The detection of strains of *Salmonella* spp, *Providencia rettgeri*, *K. oxytoca*, *C. freundii* and *Pantoea* sp occurred in the second monitoring year in the HHSW and COD cells and of *Serratia* sp and *K. oxytoca* in the HCW cell.

The occurrence of pathogenic microorganisms in HCW landfill leachate was reported by Nascimento et al. (2010), who identified strains of *Citrobacter* sp, *Providencia* sp, *Klebsiella* sp, *Proteus* sp, *E. coli*, *Hafnia* sp, *Morganella* sp,

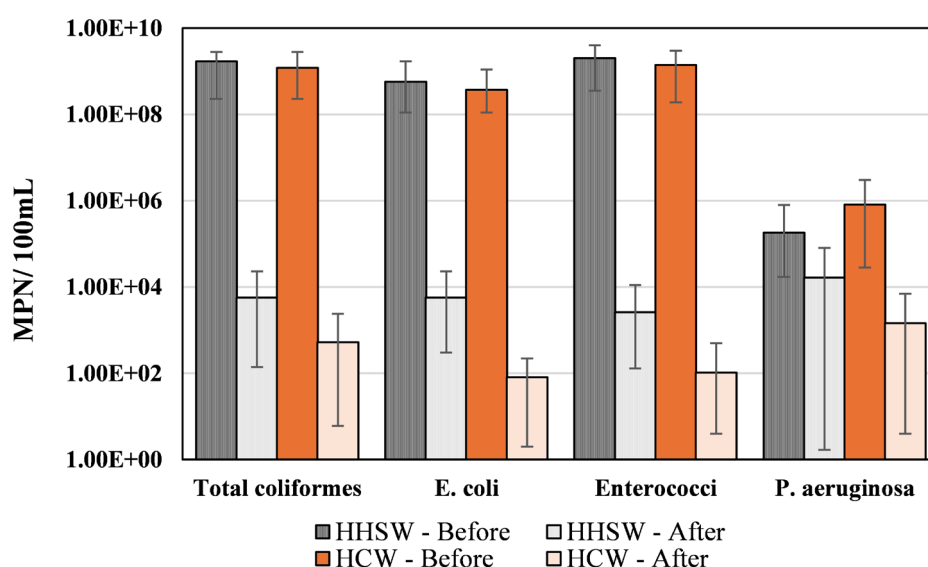


FIGURE 3: Results of microorganisms' density before and after (last 90 days) confinement for HHSW and HCW cells.

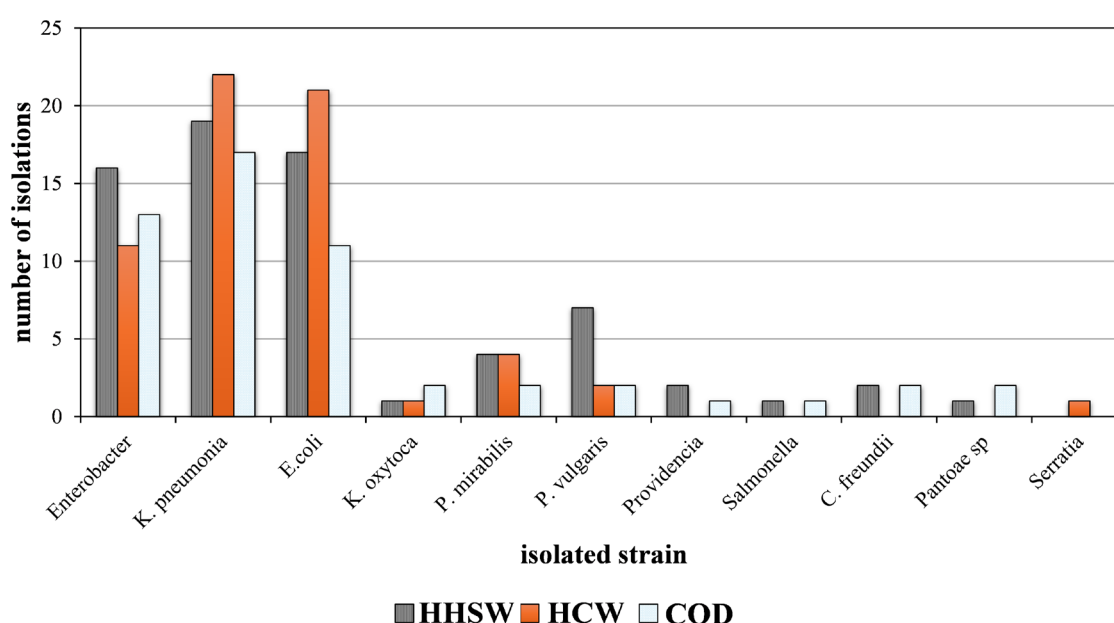


FIGURE 4: Number of enterobacteria strains isolated in cell leachate over 810 days of monitoring.

Salmonella sp, *Shigella* sp and *Serratia* sp.

3.4.1 Search for *Staphylococcus aureus*

No *Staphylococcus aureus* was isolated from the 54 leachate samples generated in each cell. This result suggests that it is difficult for this bacterial strain to recover or survive in the leachate due to adverse factors such as

microbial competition, metabolism products, the antibiotic activity of the microbiota, pH, conductivity, heavy metals, medicines, disinfectants and other sanitizers, paints and solvents. This result is consistent with that Grisey et al. (2010) obtained when assessing the transit and survival of pathogenic bacterial populations in an aquifer under the Etueffont landfill site in France. Cussiol (2005) described landfill leachate as not offering adequate and sufficient nutrients for developing *S. aureus*.

Therefore, it can be considered, by the culture-dependent methods, that the co-disposal of hospital waste and household waste does not differ from the disposal of only household waste and that the fraction of hospital waste collected in the municipality of Rio de Janeiro does not interfere with the microbiota of landfill leachate.

TABLE 3: Mean, maximum and minimum (in brackets) density of heterotrophic bacteria of leachate generated in waste disposal cells (n=43).

Microorganism/ Landfill Cell	HHSW	HCW	COD
Heterotrophic bacteria (CFU/ mL)	1.3×10 ⁶ (20-2.6×10 ⁷)	1.0×10 ⁶ (30-2.5×10 ⁷)	9.6×10 ⁵ (50-2.8×10 ⁷)

3.4.2 Susceptibility of the isolated strains to antimicrobials

Table 4 shows the overall results of the susceptibility of the isolated strain to antimicrobials. Regarding to enterobacteria strains isolated in the HHSW cells, all samples exhibited sensitivity to gentamicin (GEN) and ciprofloxacin (CIP). In contrast, in the HCW cell, sensitivity was observed to amikacin (AMI), aztreonam (ATM), GEN, cefepime (CPM), and CIP. Conversely, complete sensitivity was observed for AMI, CPM, and CIP in the COD cell. Across all cells, the antimicrobial cefazolin (CFZ) displayed the highest resistance percentage.

Furthermore, among the 192 enterobacterial isolates, various phenotypes, including sensitive (S), resistance to a single antimicrobial (R1), resistance to two antimicrobials (R2), and multidrug resistance (MDR), were identified in the three cells. The prevalence of multidrug resistance was notably high in all cells, with the highest frequency observed. Additionally, strains sensitive to the tested antimicrobials were lower in the COD cell.

Concerning enterococci, this group of microorganisms demonstrates diverse resistance mechanisms to antimicrobials, some intrinsic to the group. The resistance profile of *Enterococcus* sp. strains to various antimicrobials was compared among the three cells. It was found that 100% of the strains isolated in the 3 cells were sensitive to the following antimicrobials: CIP, TEC and VAN. In the HHSW cell, 9.4% of the strains showed a resistance profile to AMP; 7.1% of the strains isolated in the HCW cell were resistant to this antimicrobial, and 7.9% in the COD cell. Murray (1997) states that vancomycin resistance in enterococci (VRE) associated with ampicillin resistance is a combination of devastating therapeutic consequences. Enterococci have been recognized as a reservoir of antimicrobial resistance genes with the potential to transfer them to other bacteria in food-containing environments (Aslam et al., 2012). Enterococci with resistance to high concentrations of aminoglycosides (HLAR) have been described in strains isolated from humans and animals (Han et al., 2011). No HLAR-producing strains were detected in cell leachates.

The percentage of phenotypes identified for *P. aeruginosa* (Figure S6) showed the absence of multidrug resistance and a prevalent sensitivity to all tested antimicrobials for the three cells. Strains isolated from hospital wastewater displayed resistance to carbapenem antimicrobials (Fuentefria et al., 2011), as well as in the leachates of the 3 cells in this study, emphasizing the importance of monitoring antimicrobial resistance in *P. aeruginosa* within environmental settings.

3.4.3 Detection of ESBL-producing enterobacteria

Screening for the ESBL phenotype (Table S4) indicated that 32% of the Enterobacteria isolated in the HHSW cell were classified as potential ESBL producers as per CLSI (2023) criteria, 30% in the HCW cell, and 44% in the COD cell. The disk approximation method substantiated the suspected presence of ESBL-producing Enterobacteria in 15.6% of the strains from HHSW, 10.7% from HCW, and 13.6% from COD.

Figure S7 (Supplementary Material) illustrates the positive outcomes of the disk approximation method, demonstrating an increase in the zone of inhibition for oxyimino-cephalosporins (CTX and CAZ) and monobactam (ATM) due to synergy with clavulanic acid present in the amoxicillin-clavulanic acid conjugate disk.

The emergence of extended-spectrum beta-lactamase enzymes represents a critical resistance mechanism against third-generation cephalosporins within the Enterobacteriaceae family, particularly noteworthy in *Klebsiella pneumoniae* (Vespero et al., 2007). Although ESBL-producing strains were identified in all three cells, 81% of the 68 suspected Enterobacteria strains tested were negative. In a related study, Chagas et al. (2011) investigated antimicrobial resistance patterns in gram-negative bacteria isolated from hospital sewage treatment facilities, identifying 44% of the isolated bacteria as ESBL producers, with *K. pneumoniae* and *Enterobacter cloacae* prominently featured.

3.4.4 Detection of carbapenemase-producing enterobacteria

Screening for carbapenemase production revealed that 22% of the isolated bacteria were probable carbapenemase producers in the HHSW cell, 23% in the HCW cell and 51% in the COD Cell. The modified Hodge test detected the carbapenemase-producing strains (Table S5-Supplementary Material).

3.4.5 Detection of AmpC-producing enterobacteria (ESBLs), AMBLER Class C

To detect the production of these enzymes in other bacteria that do not show this phenomenon constitutively, as is the case with *K. pneumoniae*, the best way is to visualize resistance to cephamycin, such as cefoxitin (CFO), as this is the best inducer of group C beta-lactamase production. At the phenotypic level, this phenomenon was observed in *K. pneumoniae* strains that screened positive for ESBL using the modified disk approximation methodology. Figure S8 (Supplementary Material) shows the positive result for the presumption of Ambler class C ESBLs. In 15% of the *K. pneumoniae* strains isolated from the leachate of the HHSW cell, in 7.5% of the HCW cell and in 10% of the COD cell, a junction zone appeared between CFO and ATM (Aztreonam, a beta-lactam drug), to which the bacteria were initially sensitive, demonstrating the decrease in sensitivity caused by the bacteria's exposure to the inducing antibiotic (CFO).

4. CONCLUSIONS

This study simulated three landfill cells for disposal of HHSW and HCW and co-disposal of both wastes. The results suggest no significant differences in the behavior of experimental landfills containing HHSW and HCW concerning the presence of pathogenic microbiota. This finding indicates that pathogenic microbiota patterns are similar in HCW and HHSW. It does not influence the leachate characteristics in these landfill models under the conditions tested.

TABLE 4: Overall results of antimicrobial activity in cell leachates.

Isolated	Total number	Experimental landfill cells	Antibiotic resistance pattern	Antibiotic resistance results	
<i>E. coli</i>	17	HSW	82%S 18%R1	AMC - 11.7% CFZ - 11.7% ESBL - 0% Carbapenemase - 0% ESBL Amp - 0%	
	22	HCW	86.5%S 9%R1 4.5% R2	SUT - 9% CFZ - 4.5%; CTX - 4.5% 4.5% ESBL Carbapenemase - 0% ESBL Amp - 0%	
	14	COD	86%S 7%R1 7% R2	AMC - 14% CFZ - 29% IPM - 7% ESBL - 21% Carbapenemase - 0% ESBL Amp - 0%	
<i>Enterobacter sp</i>	21	HSW	10%S 52%MDR 14%R1 24%R2	AMP - 62% CAZ - 4.7% COM - 9.5% CFO - 87.5% Carbapenemase-0%	AMC - 67%0% CFZ - 76% IMP - 4.7% ESBL - 14% ESBL Amp
	10	HCW	0%S 80%MDR 0%R1 20%R2	AMC - 80% CFO - 90% CFZ - 100% ESBL - 20% Carbapenemase - 0% ESBL Amp - 0%	
	12	COD	8.3%S 50%MDR 33.3%R1 8.3%R2	AMP - 8% AMC - 50% SUT - 8% CAZ - 33% Carbapene- mase - 0%	COM - 8% CFO - 50% CFZ - 67% ESBL - 16.7% ESBL Amp - 0%
<i>Klebsiella sp</i>	21	HSW	14%S 48%MDR 29%R1 9%R2	AMP - 71% AMI - 4.7% AMC - 33% ATM - 14% SUT - 14% CAZ - 14% Carbapenemase - 19.4%	CPM - 4.7% CFO - 33% CFZ - 52% CTX - 10% MER - 4.7% ESBL - 28.6% ESBL Amp - 15%
	16	HCW	12.5%S 50%MDR 0%R1 12.5%R2	AMP - 44% AMC - 50% SUT - 12.5% CFO - 81.2% Carbapenemase - 23.2%	CFZ - 81.2% IMP - 18.75% ESBL18.75% ESBL Amp - 7.5%
	20	COD	18%S 53%MDR 29%R1 0%R2	AMP - 47% AMC - 47% CFO - 41% CFZ - 65% ESBL - 17.6% Carbapenemase - 27.1% ESBL Amp - 10%	
<i>Proteus sp</i>	11	HSW	9%S 46%MDR 27%R1 18%R2	AMP - 11% CFO - 11% CFZ - 11% ESBL - 0% Carbapenemase - 0% ESBL Amp - 0%	
	9	HCW	67%S 22%MDR 11%R1 0%R2	AMP - 11% CFO - 11% CFZ - 11% ESBL-33% Carbapenemase - 0% ESBL Amp - 0%	
	6	COD	25%S 50%MDR 25%R1 0%R2	AMP - 73% AMI - 9% CFZ - 91% CTX - 36% ESBL-0% Carbapene- mase-0%	ATM - 18% CAZ - 9% CPM - 18% CFO - 27% ESBL Amp - 0%

AMP- Ampicilin; AMI – Amikacin; AMC – Amoxicillin+ clavulanic acid; ATM – Aztreonam; GEN – Gentamicin; SUT – Sulfametoazol-trimetoprim; CAZ – Cef-tazidime; CPM – Cefepime; CFO – Cefoxitine; CFZ – Cefazolin; CTX – Cefotaxime; CIP – Ciprofloxacin; IMP – Imipenem; MER – Meropenem

The following considerations stand out:

- The cells constructed proved to be a good study model for the biological parameters of landfill leachate.
- The values found for the environmental contamination indicators of the 3 cells suggest that pathogens persist in the leachate after 810 days of confinement.
- The landfill system reduced the population densities of microorganisms that are indicators of fecal pollution from the start of confinement.
- The leached liquids from the HHSW and COD experimental cells showed no significant difference in microbiota throughout the study. In contrast, results from the HCW cell showed lower values for total coliforms, *Escherichia coli* and enterococci than the other cells.
- The co-disposal of HHSW and HCW did not differ from the exclusive disposal of HHSW regarding antimicrobial resistance.
- Microorganisms with a hospital multidrug resistance profile were similarly isolated in the 3 cells.

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A PRELIMINARY STUDY ON REAL-TIME ASBESTOS RECOGNITION IN TUNNEL SPOIL OF GREEN STONES

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ABSTRACT

This study focuses on the analysis of serpentinite rocks to detect asbestos minerals, with a particular emphasis at discriminating chrysotile from other minerals and matrices (antigorite, lizardite) in serpentinites. The primary objective of this exploratory study is to lay the groundwork for the development of an innovative system for real-time identification of asbestos on the surface of materials and rocks. In more detail, the study emphasizes the advantages of utilization of rapid and reliable portable instrumentation, aided by machine learning (ML) techniques. Serpentinite rock debris from tunnel excavation activities (i.e. tunnel spoil) were collected at the Cravasco railway tunnelling site (Genoa - Italy) of the TEN-T Rhine-Alps Core Corridor. On-site analysis was conducted using the ASD FieldSpec 4 Standard-Res spectrophotometer (350-2500 nm). A two-class classifier was developed based on acquired reflectance spectra, to discriminate between the "Presence of asbestiform fibers" and the "Absence of asbestiform fibers". The classification model was calibrated through stereomicroscopic inspection and electron microscopy carried on selected sample fractions.

1. INTRODUCTION

The economic growth is leading to an increasingly interconnected world, with the development of new infrastructures. However, the alteration of natural ecosystems may cause significant environmental and health risks, that must be carefully assessed and monitored. In the context of the circular economy paradigm, a sustainable management is needed, aimed at preventing the production of waste and the misuse of resources. To this end, during earth/rocks excavation (e.g., tunneling works) it could be very useful to carry out a fast characterization of spoils, that could allow the real-time sorting of materials. This can be a complex task, considering the high heterogeneity of natural matrices.

In the case of Green Stones (GS), the presence of Naturally Occurring Asbestos (NOA) minerals constitutes a serious health risk. Therefore, to ascertain the presence of asbestos fibers is crucial to distinguish between by-product (i.e., inert) and hazardous waste; the latter requiring specific and expensive environmental and safety measures (Voit et al., 2020). In European Countries, if the excavated

material contains more than 0.1% of asbestos fibers, it is compulsory classified as hazardous waste and landfilled (Paglietti et al., 2016).

The GS are a group of mafic and ultramafic metamorphic rocks of magmatic origin made up of different lithotypes, including serpentinites, mainly used as inert materials or ornamental stones (Carmignano et al., 2019). Asbestos is a common constituent, but the mineralogical composition depends on regional-local geological conditions. In Italy, chrysotile and tremolite asbestos occur often within serpentinites, the former being the most common one (Bloise et al., 2014). Since it has the same chemical composition as antigorite and lizardite (i.e., not-fibrous serpentines), the analysis is complex and is usually carried out in laboratories with expensive instruments (mainly electron microscopes) and procedures that require many hours for sample preparation and analysis to get reliable results (Lee et al., 2008). During spectroscopic analysis, many minerals such as talc, chlorite, balangeroite and carlosturanite can interfere and bias the results (Olori et al., 2017). In this context, new tools and instruments can be tested, which are more and more available for material



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analyses, including Machine Learning (ML) techniques applied on sensor-generated data (Bonifazi et al., 2019a; Neo et al., 2022).

We describe a preliminary study carried on with a promising real-time technique based on reflectance spectroscopy, calibrated through standard analytical methodologies, that has previously showed to allow an affordable automatic recognition of asbestos inside anthropic materials (Bonifazi et al., 2019b). The study focuses on serpentinite-based debris coming from tunneling excavations in Cravasco, near Genoa (Liguria Region-Italy), where two sections of the Genoa - Milan high-speed railway line are under construction. The section is called “Unico Terzo Valico di Giovi - Nodo di Genova” and is a portion of the Rhine-Alps Core Corridor (one of the main European railway lines). The tunneling activities began in 2014 and should be completed by 2035 (Webuild-RFI, 2021). As chrysotile was found, the executive projects were specifically adapted, including strict safety procedures (IMT, 2024). At first, the excavated material is placed inside a three-stage confinement (internal storage), then it is moved to a further static confinement with greater capacity (external storage), from which the waste is progressively sent to landfills, while the uncontaminated sub-products are deposited in an adjacent disused quarry.

2. MATERIALS AND METHODS

2.1 Sampling

In September 2023, twenty-four samples were taken from the Cravasco tunneling site, specifically from the following areas/lots: 1) quarry front, 2) internal contaminated debris deposit, 3) internal uncontaminated debris deposit, 4) external contaminated storage. The sample's number and location were chosen to obtain composed samples that are representative of the four investigated areas. Two different approaches were followed for sampling from the piles:

- at the internal storages and the excavation front, due to complex operating conditions (static and dynamically confined environment, instability of the front, difficulty in transporting large quantities of sample) an incremental approach was used, taking at least 5 aliquots in different points for each sample;
- at the external storage, considering the larger volume stored, the lower safety risks and the proximity to site facilities, a greater number of samples was preferred, each consisting of a single increment, taken in different points of the storage, both on the horizontal and vertical plane.

The materials present different grain sizes (from decimetric blocks to silt) and are mainly composed by serpentinites. Within individual lithoid blocks, a compact matrix is recognizable, made by either dark-grained or bright green, or bluish minerals, interconnected with friable veins and aggregates with light shade-whitish colors (Figure 1a). The humidity is extremely variable, and few samples are composed mainly by mud. Sometimes, anthropic materials used in the work were present, such as cement and artifi-

cial vitreous fibers, especially in the quarry front. In detail, the 24 samples are reported in Table 1.

2.2 On-site reflectance spectra acquisition

On-site, 50 to 100 reflectance spectra were acquired for each sample using the ASD FieldSpec 4 Standard-Res portable spectrophotoradiometer, that operates in the Vis-SWIR regions (350-2500 nm) with a spectral resolution of 3 nm at 700 nm and 10 nm at 1400/2100 nm. The spectra were acquired in different positions on the sample's surface, to collect most of the radiometric (i.e., compositional) variations. To avoid cross and hazardous contamination, a borate glass was placed between the sample and the probe. Data acquisition and calibration procedure were performed using RS3 software (Ver. 6.2).

2.3 Stereomicroscopy

In laboratory, representative fractions (sub-samples) were manually taken from each sample, via homogenization and coning – quartering procedures. To carry out a fast qualitative analysis by stereomicroscopy, a sub-sample set were inserted inside borosilicate glass Petri dishes (Figure 1b). The capsules were sealed with plasticine, parafilm and/or adhesive tape, for safety reasons. The instrument used was a Leica M205C equipped with an Axio-cam208 digital camera, that allows to determine the main morphological parameters of the elongated particles, such as aspect ratio, color and presence of fibrils diverging from a bundle, thus resulting in a screening for asbestos fibers.

2.4 Analysis of the fine fraction

Another sub-sample set was separated by considering the sampling area (e.g., the quarry front) to get fast qualitative results via state-of-art Scanning Electron Microscopy with Energy-Dispersion Spectroscopy (SEM-EDS). The unified sub-samples were dried at 110°C, then sieved with a sieve shaker (Retsch GmbH AS200) to separate the < 106 µm size particles - that was assumed to contain most of the eventual asbestos. SEM-EDS was performed with a Carl Zeiss GmbH Evo MA15 operating at 20 kV in high vacuum and mounting a X-Max model X-ray detector (Oxford Instruments) with a resolution of 127 eV at 5.9 KeV. To separate and analyze the particles, dispersions at 1% concentration were prepared with distilled water, then aliquots containing 0.1 mg for each sample were filtered onto 0.2 µm pore-size polycarbonate membranes (IMH, 1994). After drying at room temperature in a bell jar glass desiccator, filter portions were finally attached to aluminum stubs and sputtered with gold.

2.5 Analysis of reflectance spectra

Data files acquired on-site with the portable spectrophotoradiometer were converted into ASCII text files using ViewSpec Pro, then imported and analyzed in MATLAB (Ver. 9.12; The Mathworks, Inc., Natick, Massachusetts, United States) environment with PLS_toolbox (Ver. 8.2; Eigenvector Research, Inc Manson, Washington, United States). At first, a 2nd Derivative algorithm (Savitzky-Golay (S-G) filter) and a Mean Center algorithm were applied on

TABLE 1: Samples list and information.

Sample ID	Sampling area	Info
CRA01	External deposit	Block with ~20 cm size and smaller fragments
CRA02	External deposit	Fragment with ~10 cm size and smaller fragments
CRA03	External deposit	Mud, fragment with ~5 cm size and smaller fragments
CRA04	External deposit	Block with ~25 cm size and smaller fragments
CRA05	External deposit	Fragments with ~5 cm size and smaller fragments
CRA06	External deposit	Fragment with ~8 cm size and smaller fragments
CRA07	External deposit	Fragment with ~10 cm size
CRA08	External deposit	Fragment with ~3 cm size
CRA09	External deposit	Fragment with ~3 cm size
CRA10	External deposit	Fragment with ~10 cm size and smaller fragments
CRA11	External deposit	Fragment with ~5 cm size and smaller fragments
CRA12	External deposit	Fragment with < 3 cm size
CRA13	External deposit	Mud, fragment with ~5 cm size and smaller fragments
CRA14	External deposit	Mud, fragment with ~5 cm size and smaller fragments
CRA15	Quarry front	Concrete with wire and fragments < 5 cm
CRA16	Quarry front	Mud with wire and fragments < 5 cm
CRA17	Quarry front	Fragment with ~15 cm size and smaller fragments
CRA18	Quarry front	Fragment with ~8 cm size and smaller fragments
CRA19	Quarry front	Fragment with ~10 cm size and smaller fragments
CRA20	Quarry front	Fragments with < 3 cm size, cement and reinforcing glass fibers
CRA21	Internal deposit with asbestos	Mud with fragments with < 3 cm size
CRA22	Internal deposit without asbestos	Fragments with < 5 cm size and cement
CRA23	Internal deposit with asbestos	Fragments with < 7 cm size
CRA24	Quarry front	Reinforcing glass fibers bar

the averaged sample spectra. After pre-processing, a Principal Component Analysis (PCA) was carried out to relate the variance between reflectance spectra to qualitative data acquired with microscopy (Wold et al., 1987). Then, a Partial Least Squares-Discriminant Analysis (PLS-DA) model was built to predict the asbestos presence in unknown samples (Bonifazi et al., 2019b). To this aim, the da-

taset was splitted into two parts using the Kennard-Stone (K-S) algorithm, keeping the replicates together: 70% of the spectra (i.e., 70% of the samples) was used in calibration, while the remaining part (30%) was used as the validation set. To assess the performance, the right number of Latent Variables (LVs) and generalizability of a predictive model, Venetian blinds (VBs) was adopted as cross-validation

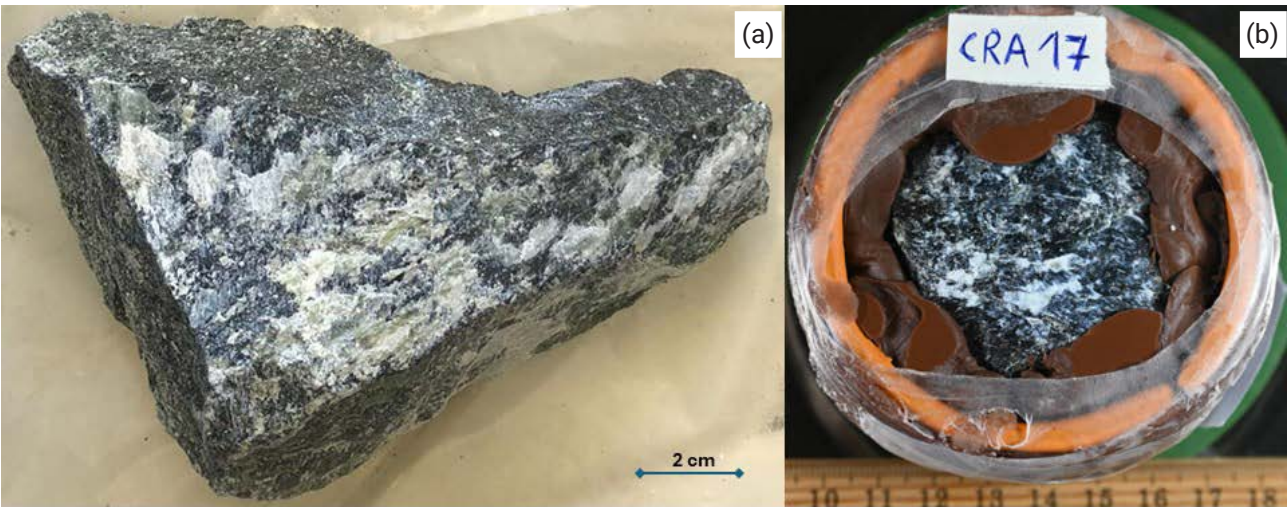


FIGURE 1: Sample named CRA_17 acquired on field (a) and in laboratory (b).

algorithm across different subsets of data. The goal of cross-validation is to ensure that the model will perform well on unseen data, rather than just on the data it was trained on.

The classifier performance was evaluated using the parameters calculated from the confusion matrix, that compare the True Positives (TP), the False Positives (FP), the True Negatives (TN) and the False Negatives (FN) results. The resulting parameters are:

- Precision = $TP/(TP+FP)$
- Sensitivity = $TP/(TP+FN)$
- Specificity = $TN/(TN+FP)$
- Accuracy = $(TP+TN)/(TP+FP+TN+FN)$

3. RESULTS AND DISCUSSION

3.1 Conventional techniques

3.1.1 Stereomicroscopy

As expected, the rock fragments present colors ranging from light up to dark green and they often show translucency and present strongly altered surfaces, partially covered by white aggregates, and isolated translucent fragments (Figure 2a). The latter frequently show straight edges and shapes, that are different from those typically exhibited by chrysotile bundles (Figure 2b).

Also focusing on the aggregates, translucent elongated minerals are noticed, embedded in a white matrix, the latter composed by fine granules that resemble talc (Figure 2c). In most cases these fibers are poorly resolvable and difficult to classify. However, they frequently show morphological characteristics like the isolated fragments. Also,

Man-Made Vitreous (MMV) fibers are frequently found, especially in samples collected at the excavation face, where fiberglass bars were used to reinforce it. MMV fibers are easily distinguishable, because the diameters vary from 30 to 50 μm and the shapes are quite always regulars (Figure 2d).

At high magnifications (around 100X), few Asbestiform Fibers (AFs) can be found, which are clearly referable to chrysotile (Figure 3). The number of AFs per each sample is reported in Table 2. The values range from 0 to 25 (CRA_17) with an average equal to 3 fibers per sample. They are present in 16 of 22 samples, in most cases within the whitish aggregates. As expected, no asbestiform fibers have been found in the uncontaminated internal storage.

3.1.2 Analysis of the fine fraction

Results concerning only the external storage are here presented. Chrysotile fibers and bundles have been found in the composed sample. A SEM image and the EDS spectrum of a typical one are reported in Figure 4. In the same figure, granular fragments are notable, which show the same elemental composition as chrysotile. These fragments belong to not-fibrous serpentines (e.g., lizardite, antigorite) and have been frequently found in the samples (Figure 5). Since the latter show often strongly elongated shapes, to conduct an accurate morphological analysis is crucial to obtain precise and reliable results. It seems, however, that using SEM-EDS alone to classify chrysotile and other serpentines can cause misclassification errors, when the particle dimensions fall below approximately 0.2 μm , as regards thickness, and 10 μm for the lengths (see the small, elongated particles in Figures 4 and 5). Indeed, the

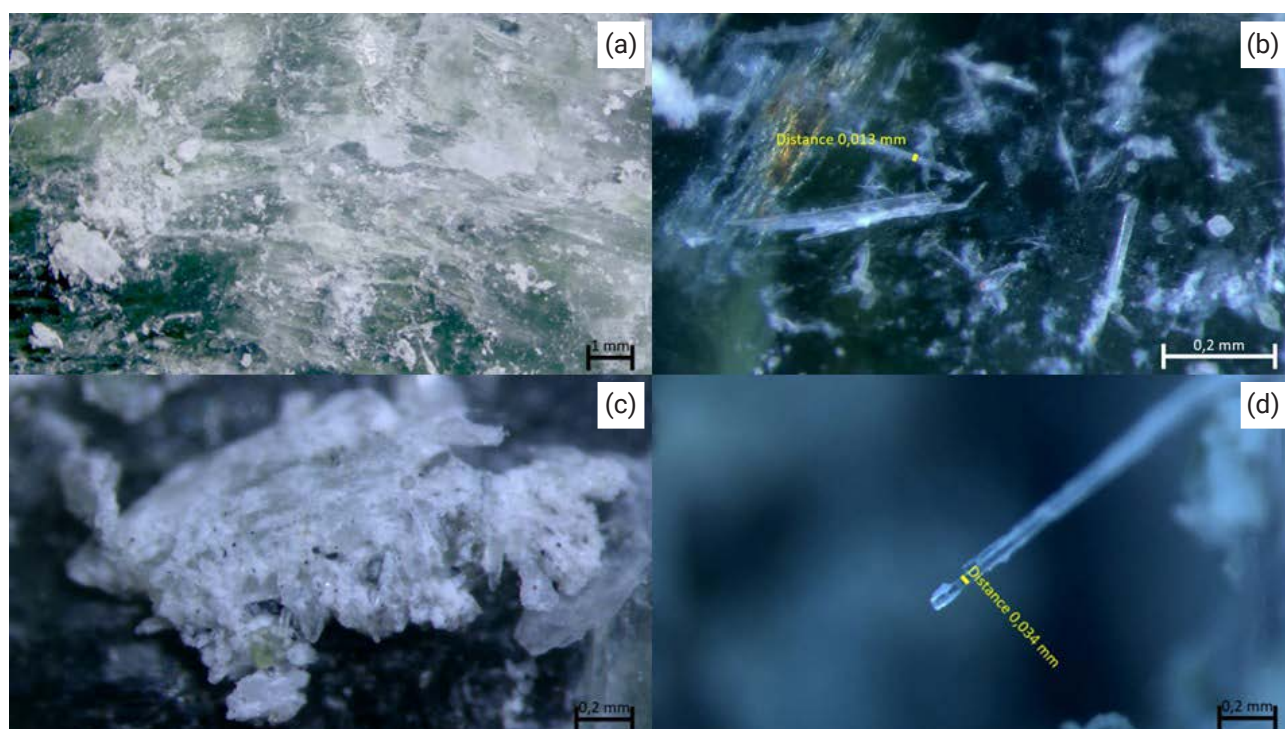


FIGURE 2: Typical surface exhibited by a serpentinite at 7.8X (a); elongated (b) and aggregated (c) mineral particles, and Man-Made Vitreous Fiber at 50X (d).

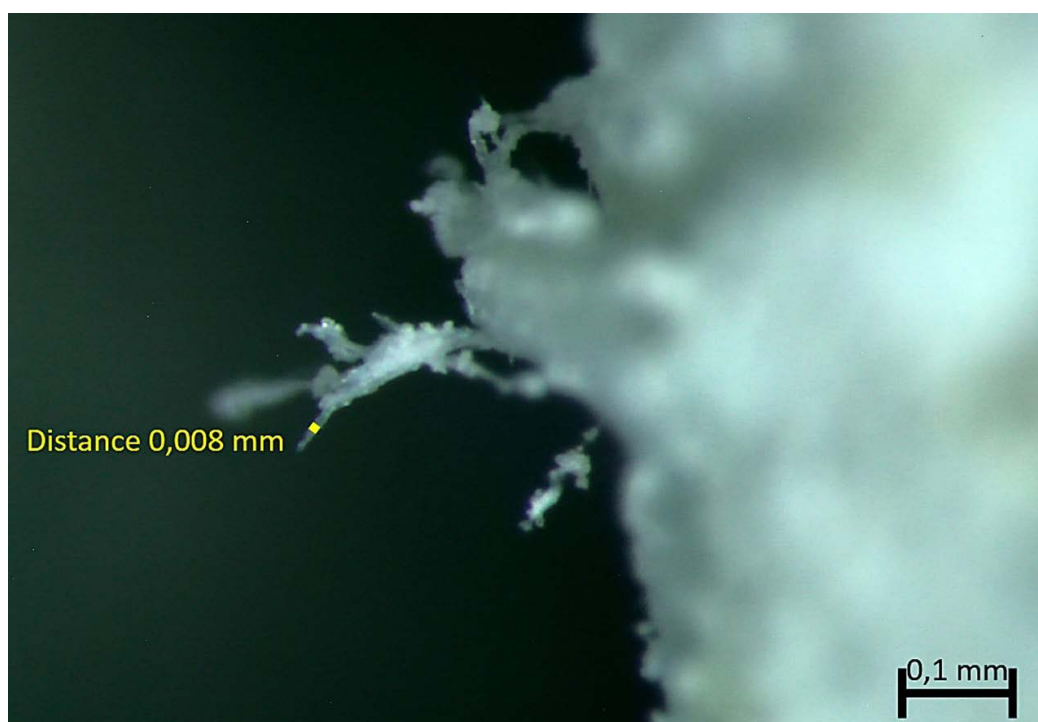


FIGURE 3: Chrysotile fibers covered by white fine granules at 100X.

TABLE 2: Number of asbestiform fibers for each sample.

Sample ID	Sampling area	Chrysotile fibers (N)
CRA01	External deposit	1
CRA02	External deposit	2
CRA03	External deposit	0
CRA04	External deposit	4
CRA05	External deposit	2
CRA06	External deposit	5
CRA07	External deposit	1
CRA08	External deposit	N.A.
CRA09	External deposit	1
CRA10	External deposit	0
CRA11	External deposit	2
CRA12	External deposit	0
CRA13	External deposit	0
CRA14	External deposit	1
CRA15	Quarry front	1
CRA16	Quarry front	0
CRA17	Quarry front	25
CRA18	Quarry front	4
CRA19	Quarry front	4
CRA20	Quarry front	4
CRA21	Internal deposit with asbestos	3
CRA22	Internal deposit without asbestos	0
CRA23	Internal deposit with asbestos	6
CRA24	Quarry front	N.A.

state-of-art analytical technique is the Transmission Electron Microscopy (TEM), but it is even more time-consuming and complex than SEM. The reported results are consistent with the scientific and normative literature (Malinconico et al., 2022; Militello et al., 2019).

3.2 Reflectance spectroscopy analysis

3.2.1 Pre-processing

The average spectra of the samples acquired on-site are shown in Figure 6. The same figure also shows the spectra averaged by the presence/absence of AFs, as resulting from stereomicroscopy analysis. Figure 7, instead, shows the pre-processed spectra of the two classes ("Presence of AFs" and "Absence of AFs").

3.2.2 Principal Component Analysis

The scores plot of the PCA model conducted with two principal components is showed in Figure 8. The model captured the 89% of the total variance. The uncontaminated samples are grouped mainly in the first quadrant (i.e., negative PC1 and positive PC2 spaces), due to reflectance variations within the 1400-1420 nm and 2345-2365 nm ranges (see the loadings plot in Figure 9). Such spectral ranges include both: i) weak absorption features related to the common peaks of silicates and ii) one strong absorption caused by carbonates near 2350 nm (Clark et al., 1990). As expected, the scores of the contaminated samples are more spread and difficult to interpret. In this case, some samples are well separated from the uncontaminated ones by the strong absorptions that almost all silicates exhibit within the 1380-1400 nm range and around 1900 nm (Clark et al., 1990). The influence of mineral-specific

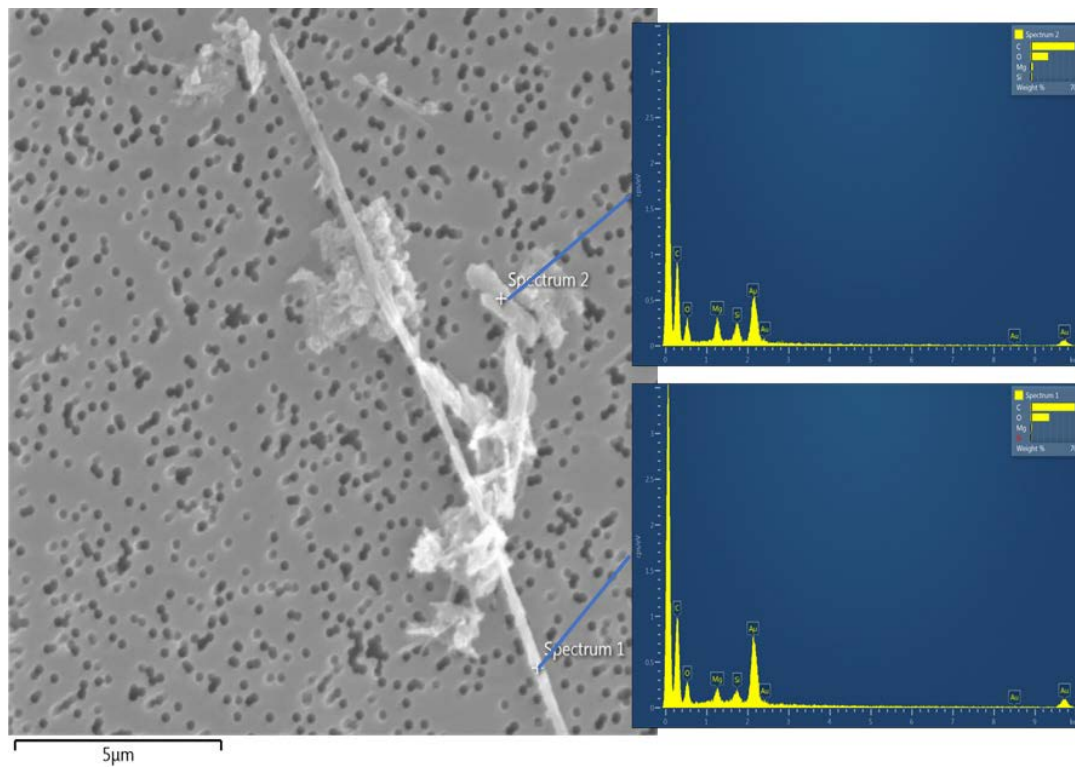


FIGURE 4: SEM image and EDS spectra of a chrysotile fiber (1) and not-fibrous serpentine fragments (2). The carbon signal comes from the underlying adhesive tab.

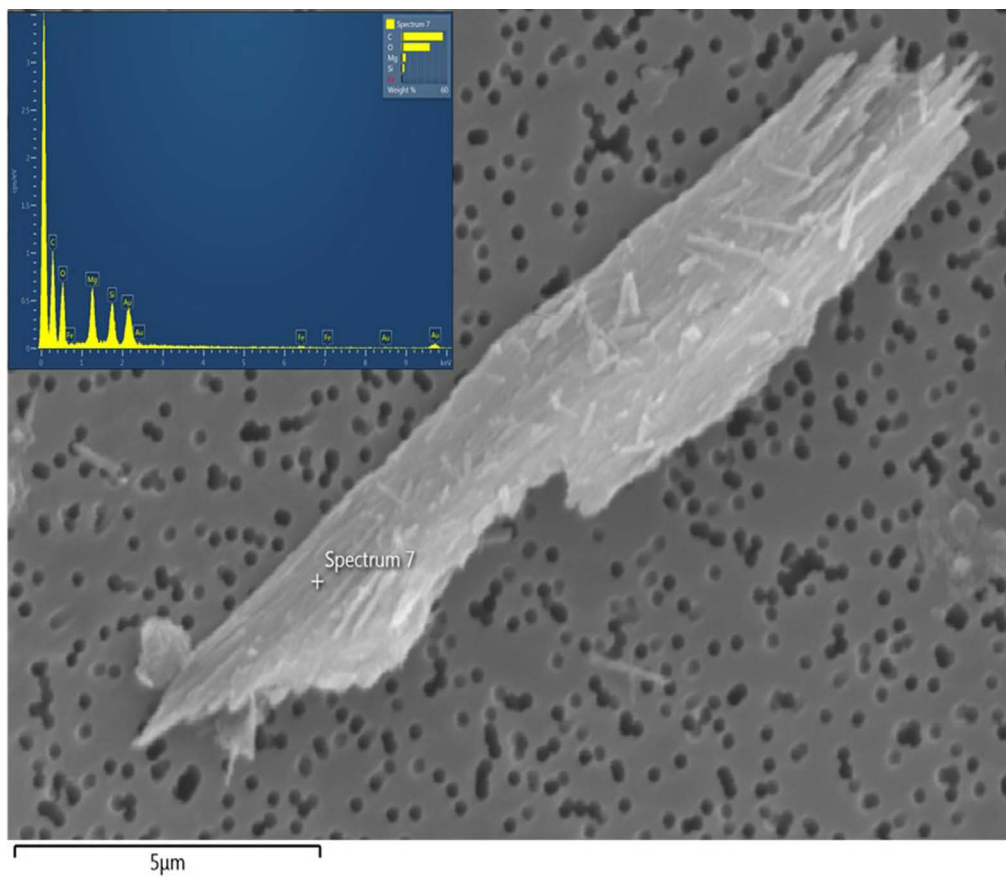


FIGURE 5: SEM image and EDS spectrum of a not-fibrous serpentine fragment and several unclassified small particles. The carbon signal comes from the underlying adhesive tab.

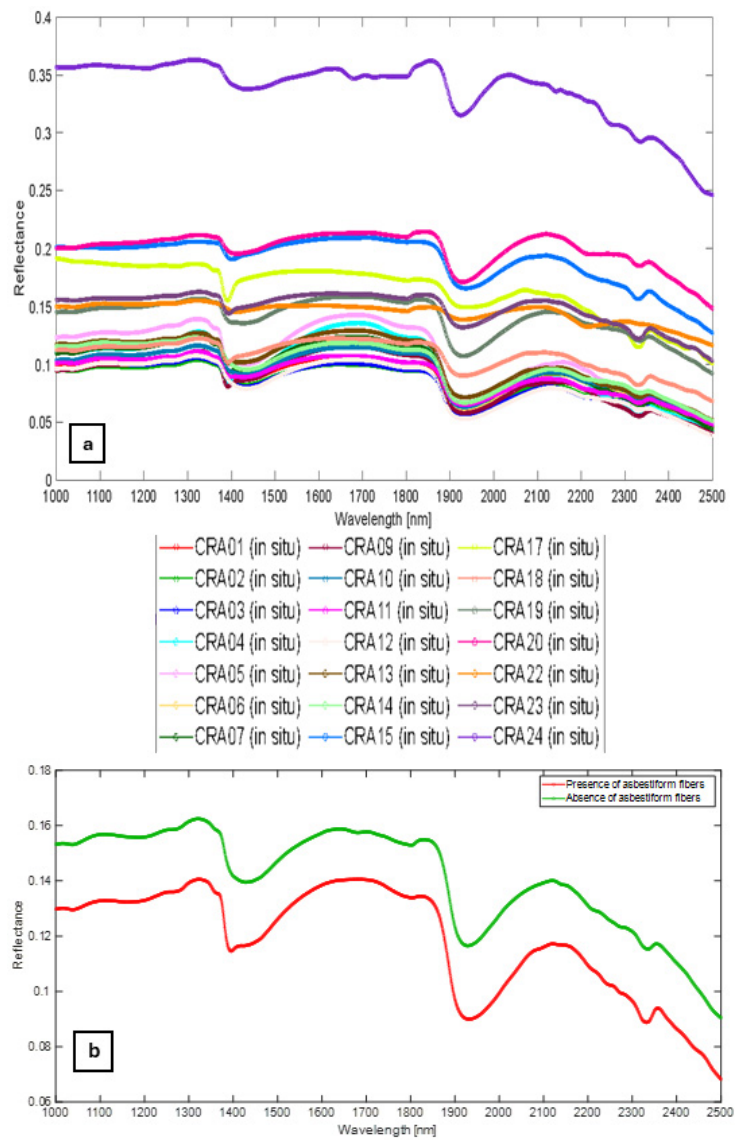


FIGURE 6: Reflectance spectra (1000-2500nm) of the samples acquired in situ averaged for: a) identification of the samples, b) presence/absence of AFs detected with stereomicroscopy.

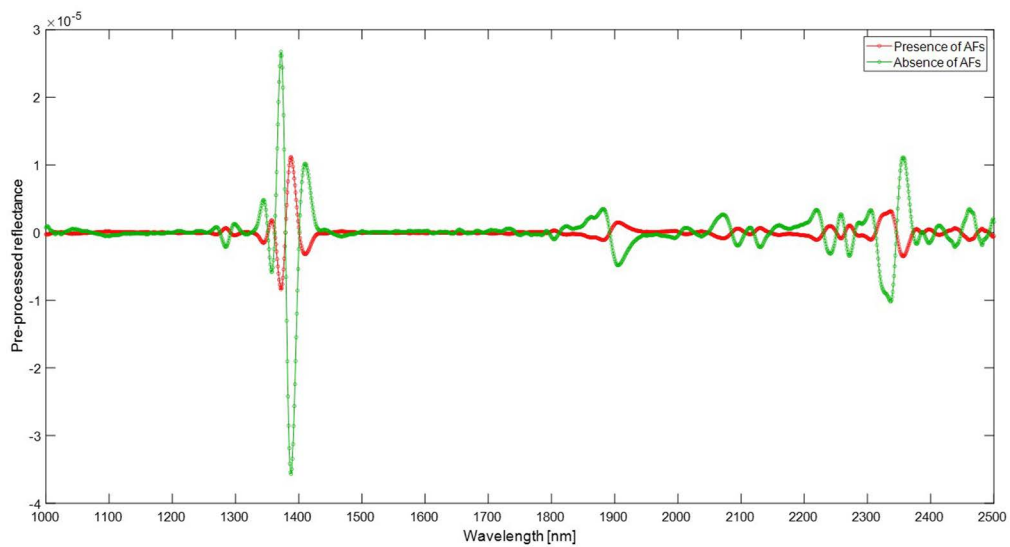


FIGURE 7: Pre-processed spectra averaged by presence/absence of AFs.

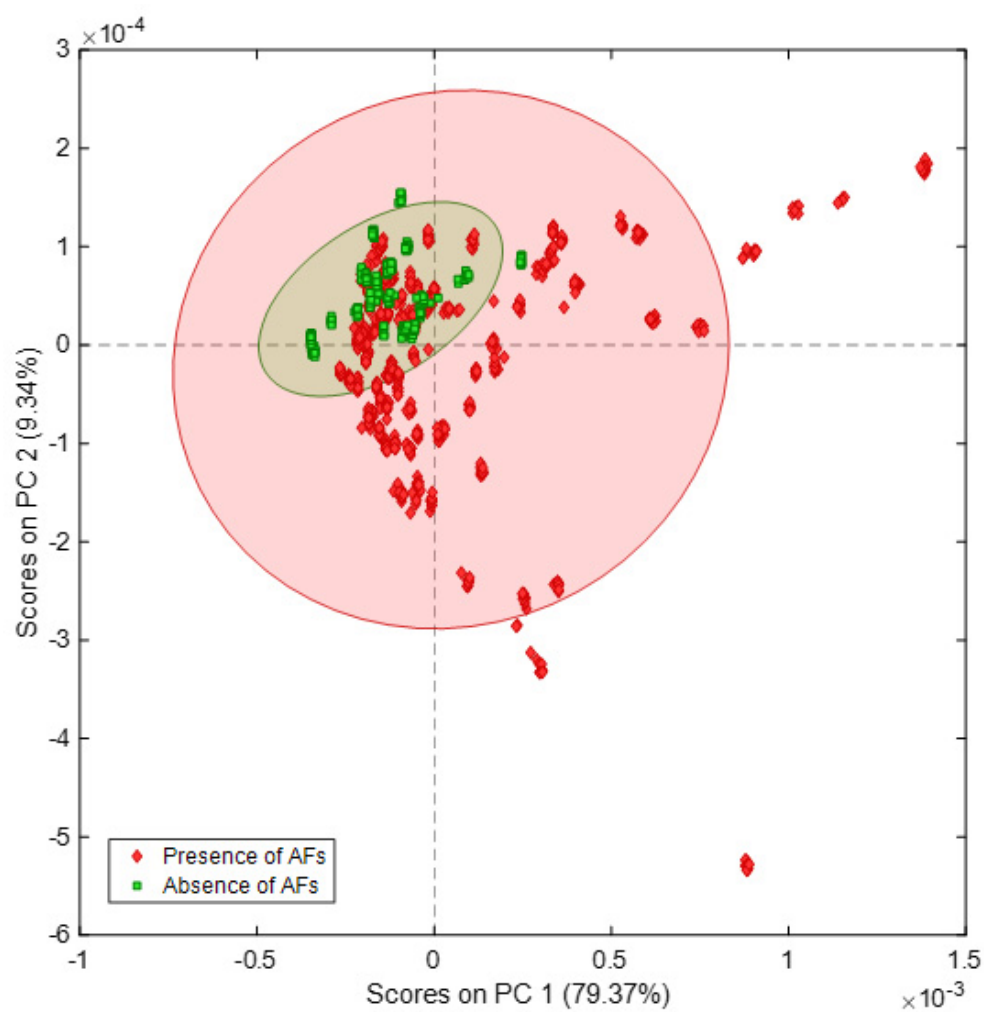


FIGURE 8: PCA Scores plot for spectra acquired on-site and grouped by presence/absence of AFs.

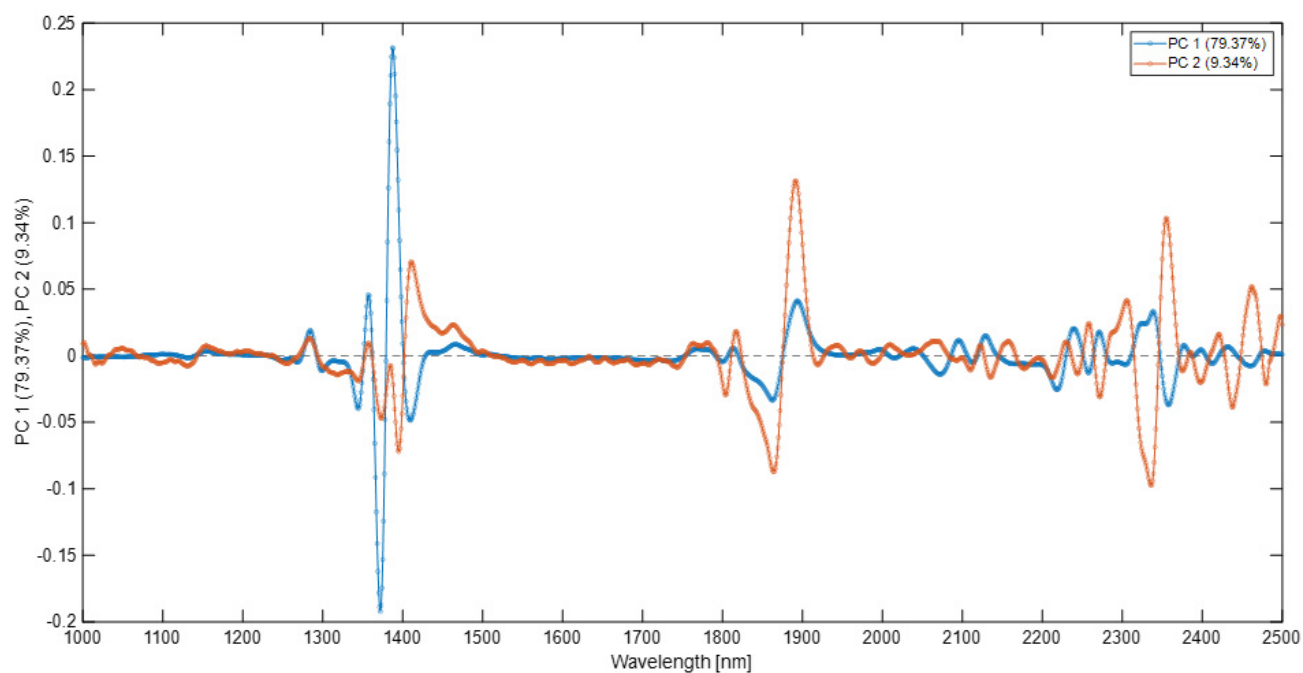


FIGURE 9: Loadings plot of the first two principal components of the on-site dataset.

TABLE 3: Confusion matrix of the PLS-DA model applied to the spectral dataset acquired on-site.

Model phase	Class	Sensitivity	Specificity	Precision	Accuracy	Error	N. of spectra
Calibration	Presence of asbestiform fibers	0.948	0.970	0.995	0.951	0.049	650
	Absence of asbestiform fibers	0.970	0.948	0.740	0.951	0.049	100
Cross-validation	Presence of asbestiform fibers	0.946	0.960	0.994	0.948	0.052	650
	Absence of asbestiform fibers	0.960	0.946	0.733	0.948	0.052	100
Validation	Presence of asbestiform fibers	0.633	0.173	0.434	0.403	0.597	150
	Absence of asbestiform fibers	0.173	0.633	0.321	0.403	0.597	150

absorptions in the variance should be better assessed in future studies.

3.2.3 PLS-DA model

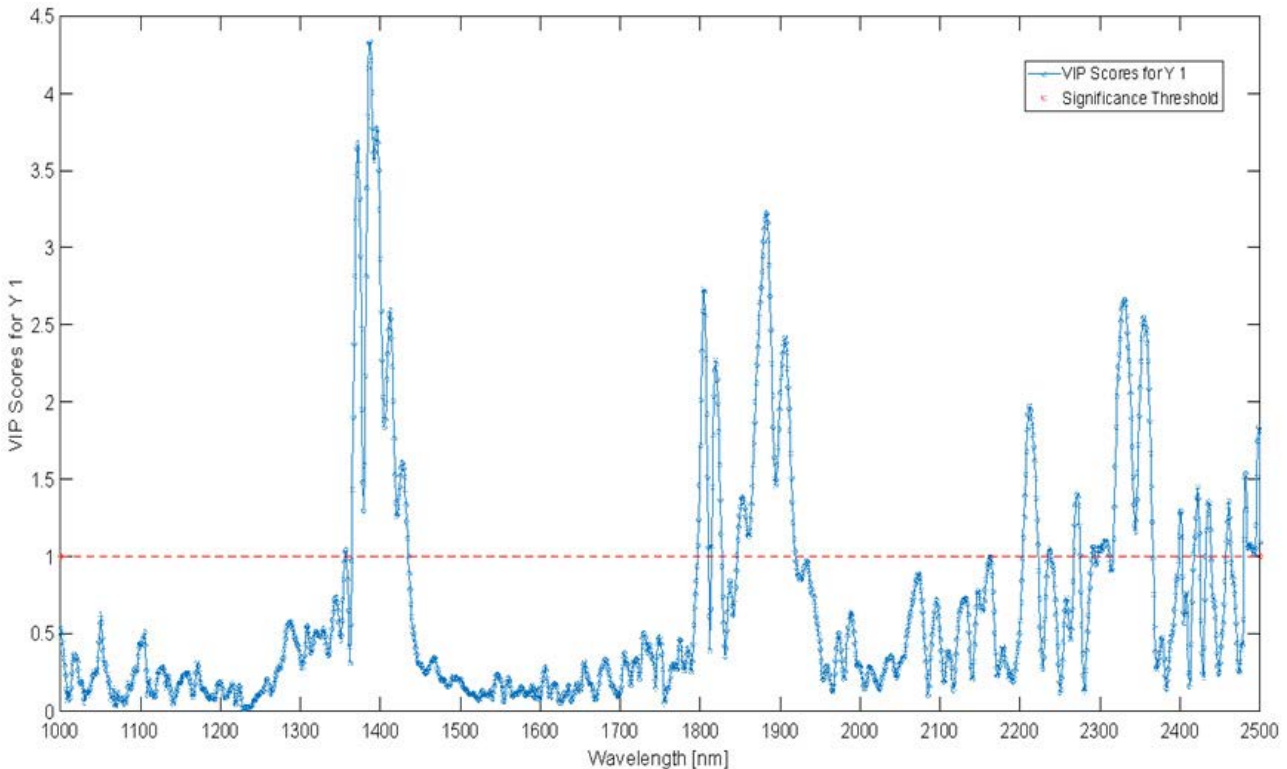
It can be seen in Table 3 (confusion matrix) that the two-class model works very well in both calibration and cross-validation. Instead, the asbestos-free samples CRA10, CRA13 and CRA03, and the sample CRA14 (only one AF found) are not correctly classified in the prediction phase.

Finally, the Variable Important in Prediction (VIP) scores plot are reported in Figure 10. Also in the classification model, the most important wavelengths are related to absorptions that are typically showed by silicates (near 1380 nm, 1880 nm and 2330 nm) and carbonates (2350 nm) (Clark et al., 1990).

4. CONCLUSIONS

This document reports the preliminary results of a study conducted to automatically classify the asbestos-contaminated tunnel spoils produced during the excavation of a new high-speed and great capacity railway from Genoa to Milan.

Twenty-four samples were collected and analyzed with an innovative approach that exploits on-site spectroscopy coupled with chemometrics and calibrated through microscopy techniques. In more detail, the classification model was trained with qualitative data obtained by means of stereomicroscopy and partially supported by electron microscopy. At this phase of the project, the results are promising because both specificity and sensitivity values in the model cross-validation are around 95%. Hence, on-site reflectance spectroscopy coupled with chemometrics

**FIGURE 10:** VIP scores of the PLS-DA model as a function of wavelengths.

show to us a good potential for real-time discrimination of asbestos presence in GS tunnel spoils. There is a need for in-depth research to demonstrate, on a more substantial analytical and statistical basis, the validity of this innovative sensor-based methodology and, specifically, to achieve an estimation of the asbestos concentration within the samples.

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IMPACT OF CHEMICAL COMPOSITION ON HOT TEARING IN SECONDARY WROUGHT ALUMINUM ALLOYS

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ABSTRACT

Hot tearing represents one of the most detrimental defects in metal castings and it is a key parameter for evaluating alloy castability. Hot tearing in aluminum alloys has gained attention from several research groups over the past century. To date, there are several theories that aim to understand the mechanisms and identify the parameters affecting the formation of hot cracks. This study aims to evaluate the influence of chemical composition on the hot cracking index of eight commercial wrought alloys of 6xxx series (Al-Mg-Si alloys), produced from recycled scrap. First, a thermodynamic calculation software was used to determine the solidification path of the alloys under non-equilibrium conditions; subsequently, a pure theoretical criterion, the Kou model, was adopted to predict the susceptibility of hot tearing. The results reveal that variations in chemical composition affect the cracking index, and the alloy 6082 shows the higher vulnerability to hot cracking. Furthermore, minimizing the element content yields a minimum value of the hot cracking susceptibility index.

1. INTRODUCTION

Although aluminum (Al) is one of the most used non-ferrous metals (Padamata, Yasinskiy and Polyakov, 2021) its production from primary ores (e.g. bauxite) involves high energy consumption and large CO₂ emissions. Many governments have adopted regulations to encourage the production of aluminum from recycled scrap; in fact, secondary Al production demonstrates notable economic and environmental advantages compared to primary production (D'Errico and Mattavelli, 2017) such as an energy saving estimated at approximately 90%, a reduction in CO₂ emission close to 80% and a lower amount of solid waste (Cui and Roven, 2010).

Among the aluminum alloys, wrought ones are the primary choice to replace steel components due to their high strength-to-weight ratio, low density, and recyclability. The wrought alloys of 6XXX series (Al-Mg-Si alloys) offer an optimal combination of properties, exhibiting strength comparable to the 2XXX (Al-Cu) or 7XXX (Al-Zn) series alloys but higher workability, weldability, and corrosion resistance (Graf, 2021). However, wrought alloys exhibit poor tolerance to impurities and, as a result, their production from the recycled scrap is significantly limited as only a smaller share of the collected scrap is used to produce the alloys. Moreover, it is often necessary to adjust the chemical composition by diluting with large quantities of primary aluminum ingots (Van den Eynde, et al., 2022).

Another drawback for wrought alloys is their susceptibility to hot tearing, a macroscopic defect related to the formation of macroscopic cracks during the final solidification stage, i.e. when the solid fraction reaches values between 85 and 95% (Li and Apelian, 2011).

Since the 1950s, several researchers have attempted to understand the mechanisms of hot tearing and develop predictive mathematical models (Eskin, Suyitno, and Katgerman, 2004). Clyne and Davies (Clyne and Davies, 1975), for example, proposed a criterion where the susceptibility to hot tearing is determined by the ratio of vulnerable time to recovery time. The "vulnerable time" is defined as the period during which the alloy transitions from 90% to 99% solid fraction. During this phase, liquid feeding is either impossible or highly restricted, preventing the filling of any developing cracks by the remaining liquid. Conversely, the "recovery time" is the time required for the alloy to transition from 60% to 90% solid fraction. According to Nyama (Niyama, 1977) and Feurer (Feurer, 1977), the formation of hot cracks occurs exactly when the solid dendritic network acts as a barrier for the remaining liquid phase, preventing it from flowing and healing the defects. Later, Rappaz-Drezet and Gremaud (Rappaz, Drezet, and Gremaud, 1999) developed a criterion deriving from a two-phase model that considers both the flow of the liquid and the deformation occurring during the solidification. Such models require a detailed experimental approach to study the solidifica-



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tion dynamics and mechanical stresses generated during component cooling. Finally, Kou (Kou, 2015) proposed a theoretical model without the use of experimental data. His method is based on Campbell's assumption that hot tearing is a uniaxial failure mechanism and the accumulated stress from the pressure drop must exceed a maximum value typical of the material. To evaluate the susceptibility to hot tearing, Kou has defined the following index:

$$HTS = \left| \frac{dT}{d(f_s^{\frac{1}{2}})} \right| \quad (1)$$

Where ds_f is the chosen solid fraction range and dT is the corresponding temperature interval. This criterion provides preliminary assessments of the tendency to hot cracking starting from thermodynamic simulations.

The trend in the coming years will be to increasingly reduce the use of primary alloys in favour of utilizing already available aluminum. Since the properties of aluminum alloys are influenced by the compounds present in their structure and, consequently, by the level of impurities, this study aims to use Kou's predictive criterion to evaluate the impact of chemical composition on the hot tearing resistance of 6XXX alloys. Special attention has been given to the effect of iron, as it is considered the major impurity in recycled aluminum alloys.

2. EXPERIMENTAL PROCEDURE

The chemical compositions of 8 commercial wrought alloys of 6xxx series were collected over two-year production period (2022-2023) and the average values with standard deviations are reported in Table 1. The Thermo-Calc simulation software was used to obtain curves of solid fraction vs temperature under non-equilibrium conditions by using the Scheil-Gulliver equation. For all the simulations, the initial temperature was set to 750°C with a step of 1°C; Figure 1 illustrates the resulting solidification curve of the 6060 alloy.

According to the Kou's criterion, the solid fraction range for hot tearing index was from 0.87 to 0.94. and the temperatures corresponding to the selected range were determined directly from the solidification curves. A regression analysis was conducted using Minitab, employing response surface regression to derive an equation that directly correlates the susceptibility index value with the alloying element content.

To assess the impact of iron on the hot tearing index, simulations were performed on the three alloys of greatest industrial significance – 6060, 6082, and 6005 – with varying iron contents. The specific levels examined were 0.4 wt%, 0.8 wt%, 1.0 wt%, and 1.5 wt% Fe.

3. RESULTS AND DISCUSSION

Considering the Kou indexes calculated for the average chemical composition (Figure 2), 6082 alloy exhibits the highest susceptibility to hot tearing, whereas the lowest index is observed for 6060 alloy. As it results from their solidification curves (Figure 3), 6082 alloy shows a larger temperature difference compared to alloy 6060 in the selected solid fraction range, confirming its greater tendency to hot tearing.

This result is consistent with practical outcomes from industry, where components manufactured with 6082 alloy commonly exhibit hot tearing cracks (Möller, Daswa and Govender, 2015).

The Kou indexes for the alloys with the upper and lower limits of the compositional range are shown in Figure 4. The values differ slightly from the indexes obtained for the average chemical composition. Considering the compositions of the upper limits, an increase of the hot tearing index is observed for most of the materials, while the indexes assume lower values in almost all cases for the alloys with minimum chemical composition.

The cases where the trend is not followed pertain to the 6061 and 6082 alloys. This deviation may be associated with reaching and surpassing the peak of maximum susceptibility. It is well-known that hot tearing susceptibility in many alloy systems follows the so-called lambda curves (Easton, St. John and Sweet, 2010; Campbell, 2015; Li and Apelian, 2011). Specifically, these curves illustrate how increasing alloying element content leads to a rise in hot tearing susceptibility up to a peak, after which susceptibility decreases. Figure 4 demonstrates that, for the 6061 alloy, increasing alloying element content corresponds to a progressive decrease in the index value. In contrast, for the 6082 alloy, an initial increase in the index is observed when transitioning from the minimum to the average composition. However, as the alloying content continues to rise, a subsequent decrease in hot tearing susceptibility is noted.

However, 6082 and 6061 alloys consistently remain the materials with the greatest susceptibility to hot tearing. Ex-

TABLE 1: Average chemical compositions of Al-Mg-Si alloys with standard deviations.

Alloy	Si %wt	Fe %wt	Mn %wt	Mg %wt	Cu %wt	Ti %wt	Zn %wt	Cr %wt	Al %wt
6060	0.449 ± 0.032	0.235 ± 0.018	0.045 ± 0.007	0.426 ± 0.038	0.030 ± 0.006	0.014 ± 0.003	0.040 ± 0.007	0.009 ± 0.003	Bal.
6061	0.667 ± 0.086	0.241 ± 0.019	0.081 ± 0.030	0.896 ± 0.064	0.209 ± 0.037	0.043 ± 0.018	0.042 ± 0.007	0.111 ± 0.037	Bal.
6082	0.922 ± 0.100	0.252 ± 0.019	0.505 ± 0.084	0.740 ± 0.080	0.048 ± 0.013	0.040 ± 0.013	0.046 ± 0.010	0.038 ± 0.039	Bal.
6005	0.638 ± 0.028	0.239 ± 0.022	0.072 ± 0.041	0.536 ± 0.037	0.053 ± 0.039	0.024 ± 0.009	0.042 ± 0.009	0.009 ± 0.002	Bal.
6005-A	0.631 ± 0.060	0.238 ± 0.020	0.198 ± 0.040	0.512 ± 0.039	0.055 ± 0.027	0.026 ± 0.011	0.041 ± 0.007	0.045 ± 0.030	Bal.
6063	0.517 ± 0.034	0.239 ± 0.018	0.046 ± 0.011	0.522 ± 0.041	0.032 ± 0.007	0.017 ± 0.006	0.042 ± 0.005	0.009 ± 0.002	Bal.
6101	0.445 ± 0.119	0.271 ± 0.082	0.040 ± 0.014	0.443 ± 0.128	0.035 ± 0.035	0.013 ± 0.006	0.036 ± 0.007	0.014 ± 0.022	Bal.
6101-B	0.447 ± 0.028	0.254 ± 0.022	0.047 ± 0.002	0.499 ± 0.022	0.033 ± 0.006	0.025 ± 0.002	0.044 ± 0.007	0.010 ± 0.002	Bal.

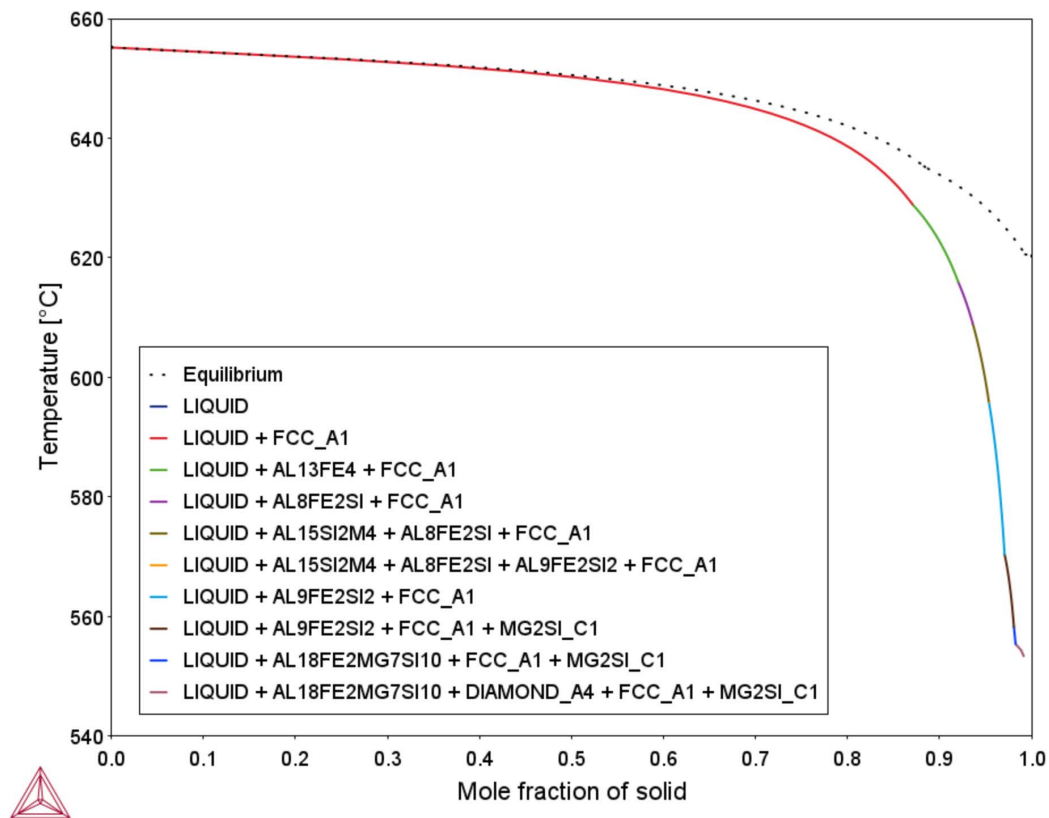


FIGURE 1: Solidification path of a 6060(average chemical composition) according to Scheil model.

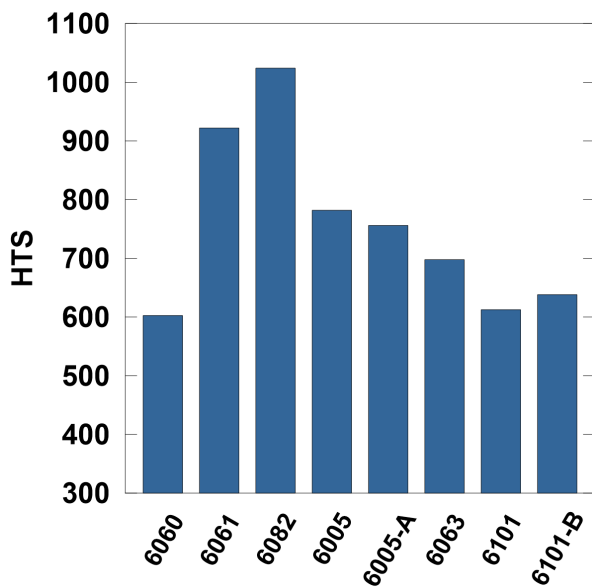


FIGURE 2: Kou's cracking index for different alloys.

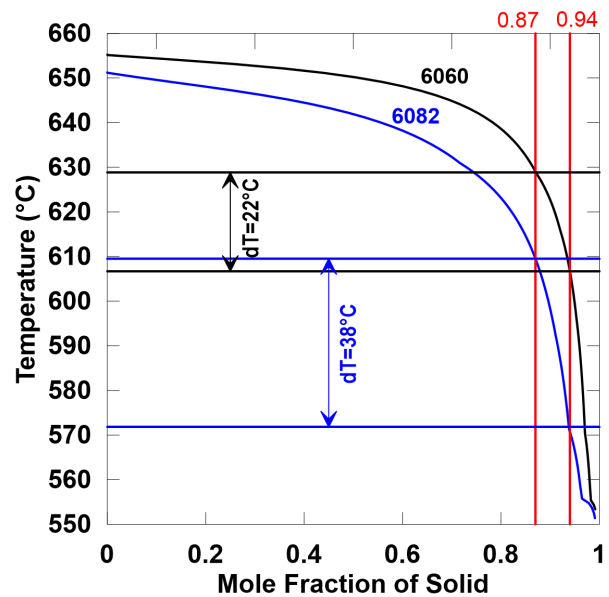


FIGURE 3: Solidification curves with the chosen solid fraction range and the respective temperature variations.

aming the chemical composition of the alloy, it can be observed that high concentrations of silicon, magnesium, and manganese could play a key role in influencing the behavior of the two materials.

Previous studies on Al-Mg-Si alloy systems (Eskin, Suyitno and Katgerman, 2004; Easton, et al., 2012) have experimentally demonstrated that the susceptibility to hot

tearing reaches a maximum at approximately 0.25 wt% silicon and around 0.2 wt% magnesium. Additionally, research conducted by Sweet's group (Sweet, Taylor, Easton, Couper and Parson, 2012) indicates that increasing the magnesium content to nearly 1 wt% almost completely mitigates the issue of hot cracking. In contrast, Clyne and Davies (Clyne and Davies, 1975) analyzed the effect of magnesi-

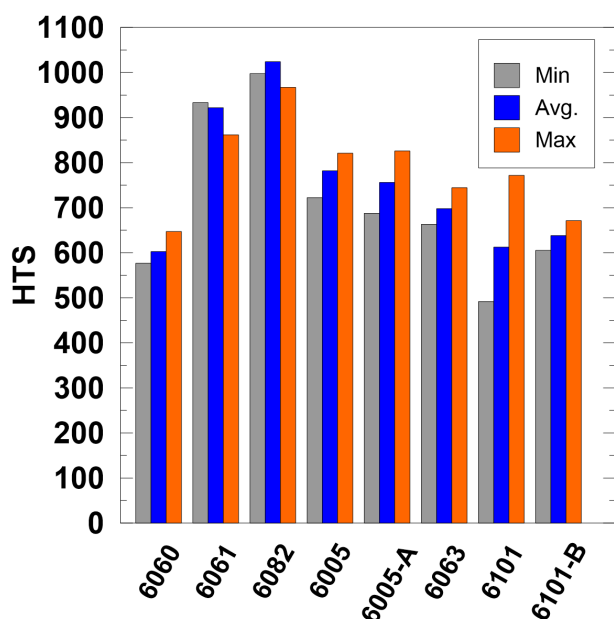


FIGURE 4: Kou's HTS index for minimum, average and maximum chemical composition.

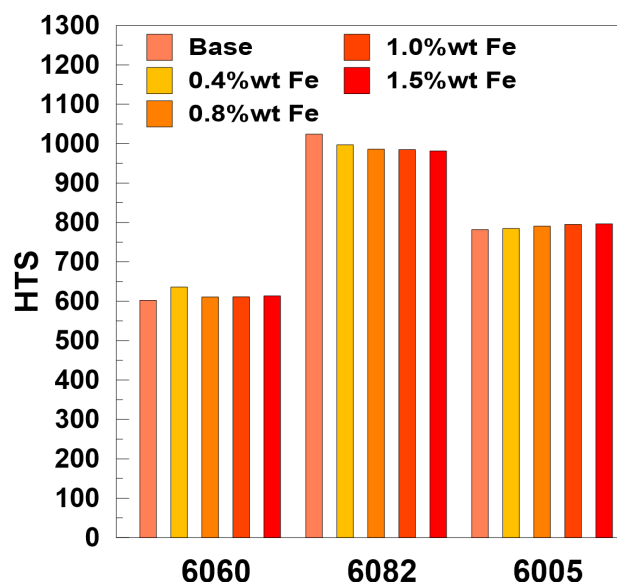


FIGURE 5: Change in cracking indices with varying iron content.

um in aluminum alloys and observed that concentrations close to 1 wt% increase the susceptibility to hot tearing, consistent with observations for 6082 alloys with magnesium concentrations around 0.8 wt%. Li et al. (Li, et al., 2020) found that in magnesium-rich alloys without silicon, hot tearing susceptibility increases, while the addition of silicon appears to prevent the onset of cracking. All these studies agree that increasing the eutectic content of Al + Mg₂Si + Si tends to reduce hot tearing susceptibility.

Moreover, Kou (Kou, 2015) compared the vulnerability of A206 and A356 alloys to hot tearing and reported that the A206 alloy, with low silicon content (0.04%) and higher manganese content, suffers more from hot tearing.

The discrepancies between our results and the experimental data reported in the literature may be attributed to the range of solid fractions selected for calculating the Kou index, as there is no precise guideline for choosing this interval.

By inputting all the simulation data for industrial alloys into the statistical software, the following equation was derived through regression analysis:

$$HTS = 437 - 1216 * Si - 1345 * Fe + 3858 * Mn + 1108 * Mg + 2366 * Si^2 - 13367 * Fe^2 + 3288 * Mn^2 + 3946 * Mg^2 + 19218 * SiFe - 9782 * SiMg - 20893 * FeMn \quad (2)$$

The model was obtained after removing all non-significant interactions ($\alpha > 0.1$) from the analysis. Notably, the equation exhibits an R-squared value of 99.36%, despite having large residuals. Nevertheless, this equation enables the immediate qualitative assessment of the impact of increasing the content of an element on the hot tearing index without the need for further simulations.

Interesting results emerge from the examination of the Kou indices calculated for alloys with incremental iron content: the histogram in Figure 5 reveals that all three alloys show no changes of the hot crackability index by increasing the iron concentration.

This trend is associated to the solidification curves (Figure 6) calculated for different iron contents: the curves differ exclusively in the initial solidification stage, while they are overlapped in the final phase, especially in the solid fractions range considered for the calculation of the Kou index (0.87-0.94%).

This results seems to be confirmed by Sweet et al. (Sweet, et al., 2013) and Benoit et al. (Benoit, et al., 2023) who argued that the susceptibility to hot tearing is correlated to the iron content, reaching the maximum when the iron content is less than 0.15%. Moreover, the studies also reported that exceeding 0.22% of iron causes an increase of the intermetallic phase, the so-called β -Al₅FeSi, favoring the interconnection of the grain boundaries and the eutectic feeding the mushy-zone.

4. CONCLUSIONS

This study aimed to analyze the susceptibility of recycled 6XXX series alloys, with a particular focus on the influence of chemical composition and iron content. The main conclusion drawn from the thermodynamic analysis are as follows:

1. Both 6061 and 6082 alloys exhibit the highest susceptibility to hot cracking, regardless of variations in chemical composition.
2. The higher the levels of magnesium, the greater the hot tearing susceptibility.
3. The reduction in alloying elements content leads to a decrease in hot tearing susceptibility compared to alloys with an average commercial chemical composition.
4. Iron does not significantly influence the hot tearing tendency of 6XXX series alloys. This suggests favorable prospects for scrap handling and recycling in alloy production processes.

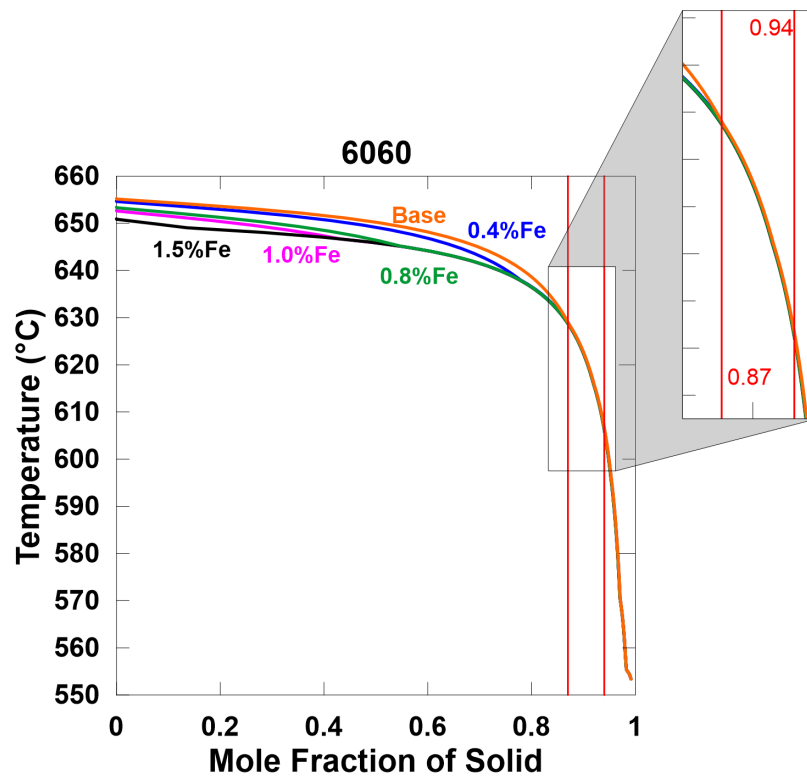


FIGURE 6: Solidification curves with varying Fe content. The overlap of the curves is highlighted.

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TOWARDS EVALUATING THE CIRCULAR ECONOMY IN THE BUILT ENVIRONMENT: INSIGHTS FROM THE CIRCULARITY ASSESSMENT PROTOCOL (CAP)

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ABSTRACT

The built environment requires extraction and consumption of enormous quantities of raw materials, water, and energy. While these materials remain in use for several years or decades, growing global populations and aging infrastructure are driving widespread generation of one of the largest and most challenging waste streams to manage. There is growing interest from communities in integrating circular economy (CE) strategies in the context of construction & demolition (C&D) material management. Many approaches for doing so focus on small-scale CE applications like individual products, materials, or projects. However, greater understanding is needed at the city-scale given communities' complex position at the frontlines of local development, resource consumption, and waste management. This study summarizes the development of an evaluative framework for community-based C&D circularity at a city or regional level. The framework expands upon a mixed methods approach called the Circularity Assessment Protocol (CAP), which integrates aspects of urban metabolism, geospatial analysis, and qualitative research methods to examine plastic waste management in communities. To advance convergent CE research, here, we aim to adapt the CAP framework to C&D. We describe our adaptation of the CAP to C&D through a conceptual review describing research, methods, and strategies related to seven elements of a local CE context: C&D Analytics, Building Material and Design, Community, Use, Collection, End-of-Cycle, and C&D Emissions. This work describes a novel yet preliminary conceptualization for developing a baseline understanding of circular C&D material management and a holistic examination of barriers, affordances, and opportunities for improving city-wide circularity.

1. INTRODUCTION

Integration of diverse expertise, disciplines, and research methodologies will be crucial for addressing increasingly complex environmental and sociotechnical issues the world faces. Though convergence research is not new, recent efforts have begun calling for such integration of scientific disciplines and researchers to develop strategies for tackling compelling topics ranging from natural hazards and disasters (Peek et al., 2020), to health-care (Sharp & Hockfield, 2017), and nanotechnology (Roco & Bainbridge, 2013). With at least two billion metric tons of municipal solid waste (MSW) generated globally every year, research in sustainable waste management similarly needs integrated expertise and research methodologies given the economic, social, governmental, and technical aspects of managing materials. While all waste streams are salient, much attention has been given to plastic waste

in the last decade, with elevated public visibility derived from the estimated 8 – 15 million metric tons entering the ocean annually (Forrest et al., 2019; Jambeck et al., 2015), primarily from land-based sources (Jambeck et al., 2020; Law, 2017). As a lightweight material, plastics comprise a small portion of construction & demolition (C&D) debris by mass (a typical metric for reporting waste quantities) relative to higher-density materials such as concrete, metal, lumber, etc. However, substantially greater quantities of global waste are from the C&D context, with nearly twice as much C&D waste generated (1.68 kg per capita) compared to municipal solid waste (0.74 kg per capita per day; (Kaza et al., 2018)). As a result, material from the built environment (BE) in which plastics are managed may be a ripe context for convergent research to develop complex interventions. In particular, the circular economy (CE) has been one of the leading concepts in both plastics and BE



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material management, aiming to shift traditional linear (take-make-waste) systems into looping systems that efficiently maximize material use and minimize, if not completely remove, waste.

Given the integration of plastics and C&D materials, this paper reports on the progress of adapting a holistic, community-based assessment of plastic material management, called the Circularity Assessment Protocol (CAP), to the wider context of C&D materials to uncover ways to leverage community knowledge, research methods, and varying expertise related to plastics to better understand how to sustainably measure materials regardless of context or waste stream. We define C&D materials to include the materials during all the phases of the BE, from raw material extraction to production, construction, use, deconstruction, demolition, and reuse. Through integrating CE concepts in the BE, C&D waste management practices, and the CAP (herein, we refer to CAP as the original CAP developed for plastics), we aimed to develop a convergent approach for C&D that incorporates aspects of building design and planning, community interactions with the BE, C&D waste collection and treatment, and losses of material to the environment. We incorporate aspects of the original CAP that include a synthesis of the role of local government, economics, and technology. To this end, this work aims to answer the primary research question: How can the Plastics CAP be adapted and expanded for the C&D sector?

1.1 Abbreviations

MSW:	Municipal solid waste
C&D:	Construction & Demolition
CE:	Circular Economy
BE:	Built Environment
CAP:	Circularity Assessment Protocol
EOC:	End-of-Cycle
NGO:	Non-Governmental Organization
LCA:	Life Cycle Assessment
LCC:	Life Cycle Costing
CBA:	Cost-Benefit Analysis
MFA:	Material Flow Analysis
MSA:	Material Stock Analysis
BIM:	Building Information Modeling
DT:	Digital Twin
PET:	Polyethylene terephthalate
PP:	Polypropylene
NSI:	National Structure Inventory
CBPR:	Community-Based Participatory Research
PESTEL:	Politics, Economy, Social, Technology, Ecology, and Law
EPA:	Environmental Protection Agency
UN:	United Nations

2. BACKGROUND & LITERATURE REVIEW

2.1 Summary of the CAP

The CAP is a standardized assessment tool used to aid decision-makers through collaboratively collecting community-level data on material usage and management of plastics. The CAP consists of seven spokes: Input, Community, Material and Product Design, Use, Collection, End-

of-Cycle (EOC), and Leakage. At the center, the system is driven by policy, economics, and governance with key influencers including non-governmental organizations, industry, and government. International or national fieldwork for CAP is conducted by compensated local implementation partners, who have input on the context-sensitive design of CAP, are trained virtually on the methods of data collection, and then contribute to the analysis and dissemination of the data. For a CAP focused on the material of plastic packaging, quantitative data collected through the CAP process includes documenting the types of materials available for the packaging of fast-moving consumer goods (i.e. tobacco products, chips, candy, and grab-and-go beverages) in stores and restaurants; articulating common alternatives to plastic packaging (such as organic and compostable materials); quantifying the distance of the parent companies and manufacturers of these products; documenting the broader context of frequency and availability of various disposal methods throughout the community; and quantifying the amount of litter found in random sample sites throughout the area using the free and open data app developed at the University of Georgia, Marine Debris Tracker (herein referred to as "Debris Tracker") (Jambeck & Johnsen, 2015). Interviews with key stakeholders, including government officials, plastic manufacturers, non-governmental organizations (NGOs), grassroots groups, activists, and waste management experts are conducted. These interviews are thematically analyzed to provide a complementary qualitative analysis of the quantitative data collected.

To date, CAPs focused on plastic packaging have been conducted or are underway in 56 cities in 16 countries. Successful applications of the CAP process and results include use in the Urban Ocean program with multiple partners and many stakeholders in 11 cities around the world. An example of an intervention from the first six cities includes the creation of a waste sorting facility based on community input in Semarang, Indonesia after engaging with CAP data (150 jobs created; (Maddalene et al., 2023). Similarly, the CAP in Metro Manila revealed prominent patterns of plastic material attributes in the city, such as a high concentration of products sold in single-use multilayer film format that offers low value for recycling, which could be connected to the high prevalence of food packaging litter documented by the study (Jambeck et al., 2024). Plastics are interwoven with and are the backbone of many materials and products. Thus, we explored, in this paper, how to expand beyond plastics. We focused on construction and demolition based on the findings from our report (Bilec et al., 2020). We anticipate that this newly developed C&D CAP will continue to evolve as we pilot in regions around the world, along with exploring additional materials such as textiles and electric vehicle batteries.

2.2 Circular economy in the construction & demolition context

The concept of CE is wide-ranging, which leads to a variety of ways in which the CE is defined, applied, and measured over time (Kirchherr et al., 2018). Broadly, the CE has been defined as an economic system that shifts away from

a traditional and extractive linear economy toward a looping system that ultimately reduces or eliminates the concept of waste through systems that return products and materials back into production and distribution for re-consumption. Historically, the CE has largely focused on manufactured goods and products with relatively short lifespans (Singh & Ordoñez, 2016), while CE in the built environment remains relatively underdeveloped, yet emerging, due to the individuality of construction projects, long lifespans, and unique life cycles of buildings, which themselves comprise a multitude of composite materials or products assembled into components (Pomponi & Moncaster, 2017). Unlike plastic packaging which has an average lifetime of one year, construction materials and products are designed for decades of useful life, emphasizing the substantial lag between production and manufacturing of building components and their end-of-life collection and management (Aktas & Bilec, 2012; Geyer et al., 2017). These innate phases mean that assessing or applying CE strategies must also encompass a range of dimensions such as early feasibility, planning, and design efforts, material manufacturing, construction activities, building operation, and ultimately end-of-cycle management (Çimen, 2021), which all occur over decades and potentially experience periodic changes due to building renovations, refurbishments, and even ownership.

In translating CE from an abstract concept to an applied framework, the CE can be assessed at the micro- (products, companies, consumers), meso- (eco-industrial parks), and macro-scales (city, region, nation) (Kirchherr et al., 2018; Pomponi & Moncaster, 2017). Within the built environment, there are also generally five dimensions of circularity including governmental, economic, technological, environmental, societal, and behavioral (Cruz Rios et al., 2021; Ossio et al., 2023; Pomponi & Moncaster, 2017). Interventions within these dimensions can further be oriented as top-down approaches led by national or regional entities such as market incentives for salvaged materials, extend-

ed producer responsibility policies and regulations, and green building rating systems, or bottom-up approaches such as marketing initiatives, material tracking, and prefabrication and modularization that might be led by individual businesses or grassroots organizations (Cruz Rios & Grau, 2020; Pomponi & Moncaster, 2017). Ossio et al. (2023) also identified five clusters of conceptual approaches to assessing circularity in the BE, which included focusing on C&D waste management systems, the “R” framework (“reduce, reuse, recycle”), life cycle assessment, building design approaches, and/or business models or networks that can be applied across the three scales (Figure 1). Further, ARUP & EMF (2024) developed four key principles on circularity specific to the built environment which focus on closing the loop for building materials including 1) Build nothing, 2) Build for long-term value, 3) Build efficiently, and 4) Build with the right materials. Among each of these, they have developed indicators and metrics that can be used to target specific interventions and measure impacts. This framework is adaptable across scale (i.e., micro, meso, etc.) as well as specific BE components (e.g., lighting systems, timber structures, etc.) and digital solutions (e.g., material passports), emphasizing the diversity of how CE can be evaluated in the C&D context.

2.3 Evaluating construction & demolition circularity at the city scale

There are several frameworks for evaluating the CE in the built environment at scales such as circularity of buildings (Bozeman et al., 2023; Honarvar et al., 2022; Khadim et al., 2022), construction projects (Tokazhanov et al., 2022), and building materials and products (Dräger et al., 2022). Common tools for assessing CE in construction and other sectors include life cycle assessment (LCA), life cycle costing (LCC), cost-benefit analysis (CBA), material flow analysis (MFA), or a combination of these, but historically they have focused on waste management, building design,

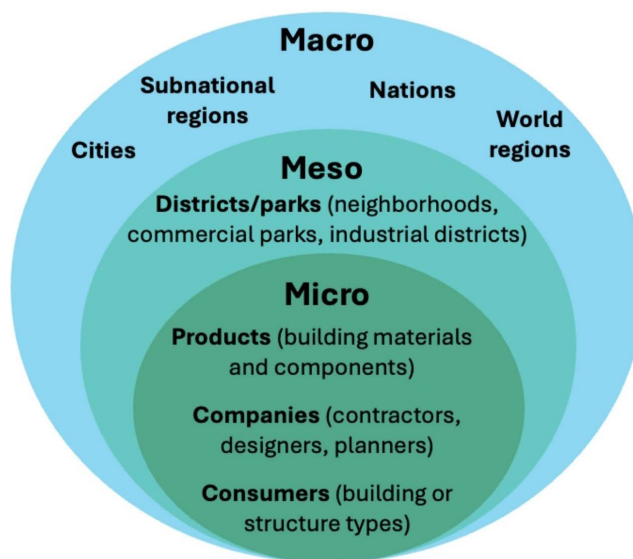


FIGURE 1: Evaluation scales for Circular Economy in the Built Environment and Construction and Demolition contexts (adapted from Kirchherr et al. (2018) and Çimen (2021)).

or product design, with lagging attention on environmental impacts (López Ruiz et al., 2020; Lovrenčić Butković et al., 2023). At the city scale, CE applications have been published in recent years, though few incorporate the full life cycle of materials and products to account for losses of C&D materials to the environment (Table 1). In mapping recent city-scale CE assessments across the CAP spokes, some studies incorporate multiple aspects of material life-cycles, but these tend to be concentrated in waste applications. Additionally, few studies integrate community-based information into their material management assessments, and only one recent CE assessment accounted for losses of material (Zhu et al., 2022).

While focusing broadly on circular material management, the CAP aims to elevate aspects of the community in identifying specific interventions, filling a key gap in CE assessments in the C&D context. Place-based assessments are useful for supporting the just transition to and feasibility of local C&D CE. For example, Yung et al. (2014) focused on community-led research examining a bottom-up model of adaptive building reuse in a Shanghai historic district, finding that residents experienced an overall improvement to quality of life in connection with the preservation of historic structures and heritage, highlighting the value of interviews in retaining knowledge within communities. Other city and community-focused frameworks review completed technical assessments of the building and land use within a city as well as reviewing community efforts for sustainability (Sharifi & Murayama, 2014; Yigitcanlar et al., 2015; Zheng et al., 2017). Similarly, Doughty and Hammond (2004) reviewed “linear metabolisms of cities compared to “circular metabolisms of cities,” illustrating the importance of sustainable city planning to reverse the current linear

use of materials. While these approaches provide utility within a specific locale and geographic scale, aspects of waste management, such as collection and emissions of debris to the environment, were not considered.

3. ADAPTING THE CAP TO CONSTRUCTION & DEMOLITION

We identified gaps in key areas to advance C&D circularity – authentic engagement with community, pragmatic solutions derived from quantitative and qualitative methods, and holistic elements from policy, governance, and economics. Thus, we adapted the proven CAP for C&D. The development of the C&D CAP involved six iterative steps (Figure 2). Step 1 included developing a deep understanding of the CAP and building familiarity with previous applications through publications and working with the original developers. Steps 2 and 3 (the scope of this paper), included a literature review assessing circularity in the C&D context (summarized in Section 2), and the conceptual development of the C&D CAP (this section). Future publications focus on Steps 4 through 6.

While balancing our adaptation of the existing CAP, we conceptualized our translation of each CAP spoke to the C&D context through Patil et al.’s (2023) four key phases of circularity assessments including 1) goal setting and scoping based on stakeholder interests, 2) data acquisition from reliable sources, 3) circularity measurement against reference points and benchmarks, and 4) assessment of circularity impacts on system and relative stakeholders under investigation. Since the CAP and C&D CAP are intended to be conducted concurrently, we aimed to retain the underlying essence of the CAP spokes (i.e., Input, Material

TABLE 1: Examples of recent construction & demolition (C&D) circularity assessments in cities mapped across Circularity Assessment Protocol (CAP) spokes (IN = Input; CM = Community; PD = Product design; CL = Collection; EOC = End of cycle; LK = Leakage).

Reference	City, Country	CAP Spokes						
		IN	CM	PD	USE	CL	EOC	LK
Barbaro et al. (2022)	Sicilian municipalities, Italy	X			X			
De Medici et al. (2018)	Ortigia, Italy	X						
Corral et al. (2022)	Almócita, Spain	X	X					
Bao and Lu (2020)	Shenzhen, China		X				X	
Bao et al. (2019)	Suzhou, China		X					
Lederer et al. (2020)	Vienna, Austria			X	X	X	X	
Mohammadizazi et al. (2021)	Pittsburgh, USA			X				
Arora et al. (2019)	Singapore			X				
Pearlmutter et al. (2019) ^a	Negev, Israel Rende, Italy Malmo, Sweden Freiburg, Germany			X				
Lynch (2022)	Vancouver, Canada				X	X	X	
Yu et al. (2022)	Nanjing, China Shanghai, China Hangzhou, China					X	X	
Oliveira et al. (2021)	Manaus, Brazil					X	X	
Zhu et al. (2022)	26 cities, China							X

^a Evaluation conducted at building project scale

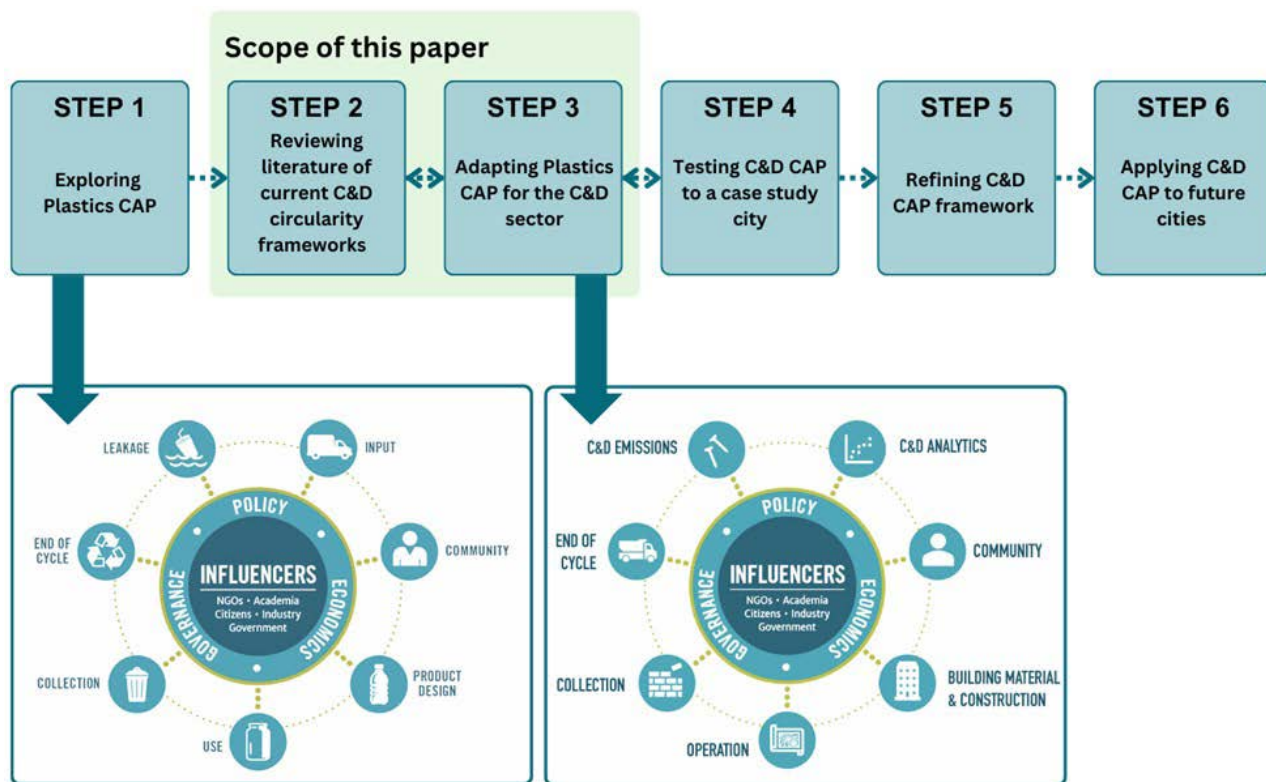


FIGURE 2: Process of adapting the Circularity Assessment Protocol (CAP) on the bottom left to the newly developed construction & demolition CAP on the bottom right. The scope of this paper is focused on Steps 2 and 3. Table 2 details changes.

& Product Design, Leakage, etc.), but adapt them to make practical sense in our sectoral application. Due to the inherent structural variation between the plastic and C&D contexts, we modified some spoke names, methods, and approaches where applicable to reflect characteristics most relevant to C&D materials in this initial version (Table 2). For example, we retained attributes such as rates of land-filling and recycling from the original CAP as criteria for the End-of-Cycle (EOC) spoke, but we modified the original Input spoke, which encompasses field surveys at retailers throughout the city under investigation to generate a profile

of common plastic consumer packaging. In contrast, for the C&D CAP, we translated the Input spoke to C&D Analytics which aims to reflect attributes such as construction rates relative to city or community growth and common materials used in local construction projects and applications (i.e., residential, commercial, etc.). Finally, as we developed modifications of the CAP, we explored the availability of relevant data (including global, national, and local datasets), published scientific and grey literature related to circular C&D material management city, and sought regular discussions with the original CAP developers.

TABLE 2: Summary of conversion of the Circularity Assessment Protocol (CAP) to the construction & demolition CAP.

Spoke *	Plastic CAP approach ^{a,b,c}	C&D CAP approach
C&D Analytics (Input)	Identifying local vs. non-local producers of fast-moving consumer goods through retail sampling	Rates of new construction, demolition, and deconstruction, geographic trends, key sectors
Community	Thematic analysis of semi-structured interviews with relevant stakeholders (Local knowledge, education and awareness campaigns, etc.)	
Building materials & construction (Product design)	Attributes of common retail or takeaway consumer goods (e.g., weigh, material, brand, product type, etc.)	Common building materials, local regulations for green building design, local building examples
Operation (Use)	Identification of plastic alternatives or product delivery systems for common retail or food goods	Land use, building and property utilization rates, rate of renovations
Collection	Municipal solid waste collection practices, regulations, and infrastructure; Surveys of waste bins and collection status	Construction & demolition waste collection practices, regulations, and infrastructure; Deconstruction activities
End-of-Cycle	Plastic waste generation, composition, treatment methods	C&D waste generation, composition, treatment methods
C&D Debris Emissions (Leakage)	Litter density and composition via Debris Tracker ^d surveys, pollution abatement and prevention measures	Locations and issues related to illegal dumping and C&D littering via Debris Tracker ^d surveys; pollution abatement and prevention measures

* Original CAP spoke names in *italics*; ^a Maddalene et al. (2023); ^b Jambeck et al. (2024); ^c Youngblood et al. (2022); ^d Jambeck and Johnsen (2015)

4. DESCRIPTION AND DEVELOPMENT OF EACH C&D CAP SPOKE

Because of the variability in methodologies, study contexts, and research objectives, we found some similarities and distinctions that helped inform the C&D CAP spokes criteria in relation to the CAP. We compiled examples of previously reported measures of circularity relevant to C&D and collated them based on the CAP spokes. We found that metrics related to 'hard' infrastructure were commonly represented in previous literature, which supported our metrics applied in the Operation, Collection, and EOC spokes such as reported waste treatment rates, composition studies, and identification of existing or needed local waste management facilities to draw CE conclusions. For example, many C&D CE studies have focused on adaptive reuse (Sharifi & Murayama, 2014; Yung et al., 2014; Zheng et al., 2017). However, few studies reported direct connections to environmental losses of C&D material despite ubiquitous recognition of the role the CE may play in preventing emissions of waste debris into the environment. Few studies incorporated mismanaged debris as criteria informing the overall assessment of the circularity of a city, district, building, or material. As such, we have adapted each CAP spoke to leverage various approaches for data collection and analysis as described in detail in the following sections. We first summarize the overarching goals, approaches, and examples from previous plastic-based applications of the CAP. We then make recommendations for the C&D CAP structured around Patil et al.'s (2023) key phases of circularity assessments by 1) defining applicable goals and/or scope of each spoke, 2) identifying outlets for relevant data acquisition and relevant reference points or benchmarking for measurement, and 3) considering potential impacts to the community and relevant stakeholders.

4.1 C&D analytics

CAP Input summary: The original CAP Input spoke aims to explore what products are sold in the community under investigation and where they originated. To understand local plastic production and input of plastic goods, the CAP involves field surveys of products sold in common retail environments (e.g., grocery stores, convenience stores, restaurants, and food vendors) and desktop research to identify local producers of plastic products. For example, Youngblood et al. (2022a) surveyed plastic-packaged fast-moving consumer goods across retailers in communities in India and Bangladesh, with a focus on snacks, candy, beverage, and tobacco items, ultimately creating a distribution map of producers and manufacturers and determining that many brand decisions were made in high-income locations away from where goods are sold, consumed, and impact participating communities.

Defining C&D Analytics goals and scope: In the C&D context, individual products and materials are difficult to survey as they are often in composite formats, difficult to access, or manufacturing information is unavailable or unfeasible to gather at the city-scale. There are two primary goals of this C&D spoke. The first goal is to take a 'pulse' of a city's C&D activity, exploring how population, geogra-

phy and topography, and industries play into trends related to urban planning, a key component to circularity in cities (Petit-Boix & Leipold, 2018). The second goal is to explore the relationship between population trends and construction activity; thus, we refer to this spoke as 'C&D analytics' to reflect our focus on the city's growth/degrowth and urban planning trends used to understand the general needs and flow of building materials, which includes collecting available data on C&D trends. The geographical scope of this spoke is the city or community under consideration.

Data acquisition and benchmarking: Avoiding new construction and building for long-term value are key aspects of the EMF framework for evaluating CE in the built environment (ARUP & EMF, 2024), which uses metrics such as percent of reused floor area and building occupancy to determine if buildings are reaching their maximum utility. At the city scale, some recent studies have examined the interplay of urban development and planning issues with CE, as the volume or rate of construction compared with the rate of population change may be an indicator of circularity. Some qualitative case studies have been conducted such as De Medici et al. (2018), who documented the role of CE in the historic urban landscape of Ortigia, Italy by evaluating how knowledge and decision-making among educational, industrial, and governing institutions reflected in the broader cultural and historical development. Corral et al. (2022) similarly conducted an analysis in Almócita, Spain, a rural community dealing with depopulation, to examine how the CE has been implemented across the PESTEL elements: Politics, Economy, Social, Technology, Ecology, and Law (PESTEL). In another study, an analysis of social and economic circularity in Sicily, Italy, Barbaro et al. (2022) used the ratio of land consumption rate to the population growth rate as an indicator to explore the balance between development, population, and building use, ultimately finding that a surplus of underutilized buildings was failing to meet residential needs, potentially leading to building deterioration, decay and reduction in urban services, and threatened community identity.

Building on the work of Barbaro et al. (2022), we recommend assessing rates of construction relative to population change. By comparing construction rates and population, larger systemic issues can be revealed regarding building material management within a community. Depopulation within cities can also cause social, economic, and environmental challenges (Sutradhar et al., 2024). For example, if the population is decreasing within a community while new construction is increasing there may be an excess of raw material extraction, unnecessary site clearing and development, and ultimately construction waste, particularly if there are existing available and developed properties, structures, and buildings that are otherwise unoccupied or underutilized due to population shrinkage or migration out of the community.

Thus, we recommend informing the C&D CAP Analytics spoke with publicly available population datasets. For historical assessments, as well as consistency between CAP cases, population data can be sourced from national or international population datasets (e.g., US Census Bureau (2023) or the United Nations Statistics Division), or

global geospatial datasets for estimating (e.g., Oak Ridge National Laboratory LandScan population data (Sims et al., 2023)). Many cities also collect data on building, demolition, and renovation permitting. In the US, for example, the US Census compiles residential building permit data through the Building Permit Survey, which reports national, state, and municipal building quantities and values part of its annual economic statistics (US Census Bureau, 2024). Lastly, depending on the level of detail or data availability, construction permits or value of construction put in place on a more granular spatial scale (e.g., neighborhood or zip code) and displaying this information geospatially can be used to elucidate potential material stocks, storage, and disparities in where construction is occurring.

Impact and stakeholders: Understanding a community's C&D 'pulse' can effectively highlight systemic planning issues that can help or hinder advancing local circularity. As mentioned, depopulation within cities can cause social, economic, and environmental challenges (Sutradhar et al., 2024). Key stakeholders relevant to the C&D Analytics spoke can encompass a wide range of professionals, government representatives, and the public. In particular, government representatives such as legal authorities, local development agencies, and civic institutions can raise awareness around circularity, encourage circular business practices, and use fiscal or regulatory tools to encourage sustainable development (Munaro & Tavares, 2023).

4.2 Community

CAP Community summary: The original CAP Community spoke aims to explore what conversations are happening around plastic consumer goods and what are stakeholders' attitudes and perceptions. Rooted in mixed methodology design, the CAP leverages knowledge from the community to better understand local perceptions of operations, affordances, and challenges to complement typically quantitative and technical data collected in other spokes. In previous CAP applications, researchers and local implementation partners have led interviews with key stakeholders and 'influencers' on the management of plastic consumer goods such as retail and food business owners, plastic and waste industry representatives, local government and regulatory officials, NGO operations, community leaders, informal waste workers, and members in local academia (Jambeck et al., 2024; Maddalene et al., 2023). Some CAP applications have also included social media analysis and workshops, aiming to uncover important community beliefs, attitudes, and values related to consumption and waste management behaviours (Jambeck et al., 2024).

Defining C&D Community goals and scope: The goal of the C&D community spoke, in alignment with the original CAP, is to work with the community to ensure the community's voice is heard and represented using a Community-Based Participatory Research (CBPR) framework, which aims to build trust and foster collaboration between the community. CBPR focuses on producing actionable outcomes that benefit the community directly, sharing power and resources fairly, and developing local capabilities. A crucial element of CBPR is jointly identi-

fying problems and solutions with the community, rather than simply generating knowledge for academic purposes (Rickenbacker et al., 2020). The CBPR framework aims to not only support the community but also provide the community with technical expertise to co-create authentic solutions that are long-lasting through shared knowledge and action.

Data acquisition and benchmarking: There are several approaches to working with the community ranging from community action teams to conducting interviews (Shackleton et al., 2021). Interviews are regularly used in the C&D waste management context for developing holistic assessments of communities that are in complex social-ecological system. Several studies have used semi-structured interviews to explore the adoption of low-waste technologies at various stages of construction projects in Hong Kong (Zhang et al., 2012), determine the effect of a national green building strategy on C&D waste (Bao et al., 2020), and examine barriers and countermeasures experienced by Chinese C&D recycling enterprises (Ding et al., 2023). Similarly, Ottosen et al. (2021) used semi-structured interviews to build understanding related to scaling CE principles in Danish C&D contexts, citing the need for robust terminology in the sector, methods for documenting progress toward CE, incorporating technology such as digitalization and building passports, and supporting CE transition with improved value chains. Nghiem et al. (2020) similarly complemented their quantification of C&D waste in Vietnam with interviews investigating demolition contractors' awareness of and challenges associated with policies regulating C&D waste.

For adapting the CAP to C&D, we recommend utilizing the CBPR framework. As a first step, we suggest leveraging semi-structured interviews guided by a protocol designed to elicit participant responses related to their role and experience in their C&D context and local processes related to C&D debris management (Table S1). Like previous literature, interview participants could be selected based on input from the local implementation partners and snowball sampling (Parker et al., 2019). Similar to the CAP, C&D CAP participants should include representatives from a range of roles in academia, government, non-profits, and local businesses, and interviews should be designed to gain insight from community members to understand what efforts have already been made, what has worked, and what has not been attempted, ultimately strengthening a holistic co-understanding of gaps and opportunities. In future publications, we will consider broadening the scope to explore additional strategies for CBPR consideration.

Impact and stakeholders: The focus of the aforementioned surveys was pragmatic and expansive, yet they may have not considered authentic community outcomes. Engaging with societal complexities is needed for evaluating and advancing CE strategies in the BE (Cruz Rios et al., 2021). Including the public, in particular, can increase trust and visibility of the CE, while also encouraging greater participation in the CE agenda and local projects (Munaro & Tavares, 2023). Given the participatory nature of CBPR, CAP Community stakeholders can encompass a wide range of professionals, government representatives, and

the public. In particular, community representatives, academic researchers, environmental authorities improve local CE awareness and lead CE-related projects that can advance more mindful consumption behaviors and attitudes (Munaro & Tavares, 2023).

4.3 Building materials & construction

CAP Product design summary: The original CAP Product design spoke aims to explore what materials, formats, and innovations are found in products, particularly packaging. Materials and design are critical aspects of the circular economy as upstream systemic changes in the design phase can reduce material dependencies, reduce waste, and attempt to harness it as valuable material inputs for circularity. To explore product materials and design, the CAP product samples collected during the retail and food business surveys are used to identify types of plastic packaging material and attributes such as mass, material type, and brand. Analysis of these products illuminates common formats of plastic products as well as prevalent materials. For example, Maddalene et al. (2023) found that polypropylene (PP), a low-value plastic in terms of recyclability (Moss, 2017), was prevalent among to-go packaging, representing the need for appropriate waste management strategies such as education around how to properly segregate PP items to avoid contaminating the recycling waste stream.

Defining C&D Building materials & construction goals and scope: In the C&D context, procuring and evaluating products and materials this way is less practical, as the BE comprises a range of materials, components, and products that provide specific, and often large-scale, utility such as structures, roofing, flooring, waterproofing, etc. Further, in-stock material quantities in the C&D context can be difficult to measure as the primary components of C&D materials are particularly heavy (e.g., concrete, asphalt, masonry, etc.), which can eclipse other materials like lightweight plastics used for interior and exterior finishes, waterproofing, etc. (Mohammadizazi & Bilec, 2022). This overshadowing is evident in US estimates of C&D material consumption with concrete making up 68% of C&D debris by mass in 2018, followed by asphalt concrete (18%) and wood products (6.8%) (US EPA, 2019). As such, the goal of this spoke is to examine the existing building material stock, and the scope is within a defined time period in the geographical region of the city.

Data acquisition and benchmarking: For our adaptation to develop the C&D CAP, we explored proxy outlets for estimating common building materials and design norms in the community depending on data availability. To account for all building materials used for construction in communities, we recommend three major considerations: (1) existing buildings, (2) material origins, (3) examination of construction sites, and (4) adoption of green building standards in the community.

For determining the material composition of existing buildings, a multi-level system based on readily available data and information is recommended, largely focusing on the core and shell of the buildings (Figure 3). The first and most basic level of assessment is a visual inspection

of buildings. We harness the random stratification approach used by the CAP to determine sampling locations throughout the community (See for example, (Maddalene et al., 2023); Youngblood et al. (2022)). Additionally, visual inspections should span various building types and occupancy classifications such as commercial businesses, education, industrial, residential, institutional, and utility buildings. This approach fosters active data collection and detailed observations of buildings, however, depending on the city size, the sample may not be representative.

The second level utilizes the publicly available National Structure Inventory (NSI) generated by the US Army Corps of Engineers, which at this time is limited to locations in the US. The NSI database was developed for hazard planning and includes a wide range of attributional data for buildings such as occupancy type, structure value, construction material types, foundation type and height, and square footage and number of stories (US Army Corps of Engineers, 2022). For the C&D CAP, the NSI provides spatialized data including categories for material types used per building that can be extracted based on community administrative boundaries in ArcGIS Pro. While this approach provides a spatially broader sample for analysis for the city compared to visual inspections the accuracy is limited to the material categories offered by the NSI (e.g., concrete, masonry, steel, wood, and manufactured).

Regarding levels one and two, a tiered approach should be considered. First, the top five local "heavyweight" materials (e.g., concrete and brick) should be identified through semi-structured interviews and/or searches of local construction commodities as typically heavy-weight material is more local. For example, concrete aggregate, bricks, and metals may have a more local footprint. Second, regional specifications should be identified through online sources such as governmental building codes, and if possible, obtaining specifications on specific projects is recommended to garner additional levels of depth. The material type and manufacturers should be noted and reviewed to show where materials are being manufactured and imported.

The third level is urban building material stock analysis (MSA), which is a stock-driven tool for quantifying the total available stock of material within a system (Augiseau & Barles, 2017), typically within a defined boundary such as building type, building components, geospatial, and/or the temporal scale (Mohammadizazi & Bilec, 2022). Similarly, material flow analysis (MFA) focuses on the circulation of material through a system, that is, the sources, pathways, and sinks of material (Brunner & Rechberger, 2016), while MSA focuses on the material quantities. One way that can be utilized for C&D CAP is the MSA process developed by Mohammadizazi and Bilec (2023), which estimated quantities of building materials used for existing buildings and renovations using geospatial and remote sensing data (e.g., LiDAR scans). A detailed analysis can provide a more in-depth overview of in-stock building material. For example, this model can account for interior flooring that cannot be seen during visual inspections.

The fourth level incorporates modeling technology, such as Building Information Modeling (BIM) or digital twins (DT) at the city scale. BIM is a process that generates

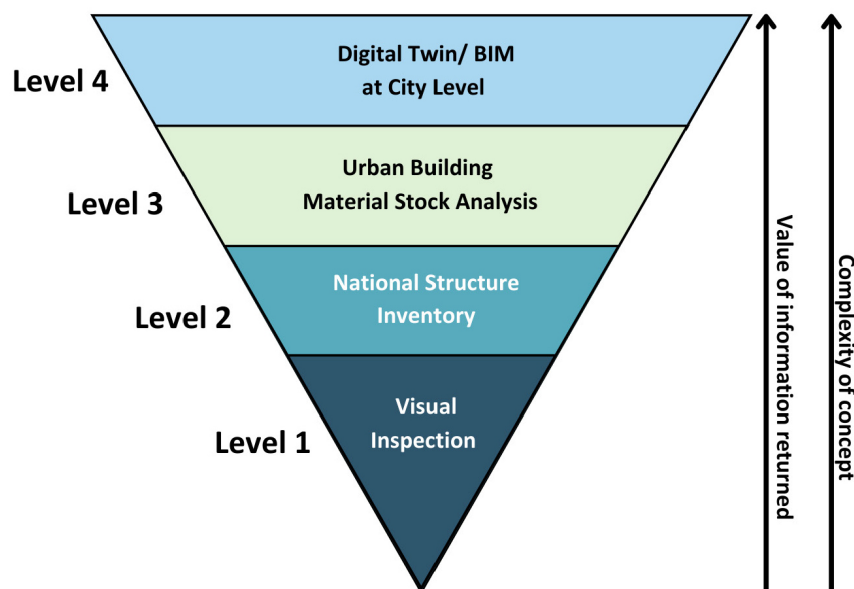


FIGURE 3: Levels of implementation for the building materials & construction spoke.

a computer-simulated building model that can be used to manage the planning, design, construction, or operations (Azhar et al., 2012). These models can be useful for estimating material stocks as they can aggregate a bill of materials, specifically listing the material type and quantity for all building components, including interior and exterior components (Zima, 2017). Similarly, a DT is a virtual representation of a physical entity that exchanges data in a real-time (Singh et al., 2021). DTs can be developed from different data streams, including but not limited to laser scanning and BIM, and “can make BIM a living instrument” (Lee et al., 2021), allowing for predictive modeling related to deconstruction and renovations (Kineber et al., 2023). While BIM has become an industry standard in recent years, DTs are a rapidly developing technology and have yet to become fully operationalized in the field of urban metabolism applications (Geremicca & Bilec, 2024). While BIMs and DTs offer promise, communities may not have the technical expertise, data, and digital infrastructure to support these models.

Impact and stakeholders: Implementing circular designs in the built environment can have many positive impacts not only on building systems in use but also in terms of sustainable materials management, for example, through uptake of bio-composite materials, prioritizing nature-based solutions in design, and selecting green building sites (Pearlmutter et al., 2019). By exploring existing building materials and design culture in CAP cities, we can identify outlets for improving the adoption of green building designs and advanced building stock management to aid in material recovery at the end of buildings’ useful life. Relevant stakeholders include design, building, and construction professionals, as well as waste management subcontractors, who can integrate low-waste building techniques, specify construction waste audits, advance a culture of sustainability materials management in the local building industry, create integrated information sys-

tems through BIM or DTs, and implement design criteria to better manage C&D debris generation (Munaro & Tavares, 2023).

4.4 Operation

CAP Use summary: In a review of macro-scale circularity assessments, Harris et al. (2021) found indicators from the use phase has largely been lacking from existing literature, with a major focus on raw materials, waste generation, and recycling and recovering indicators in evaluation frameworks. Filling this gap, the original CAP Use spoke aims to explore community trends around use and reuse of various product types. The Use spoke assesses how products are procured and consumed throughout a community, including identification of broad efforts to engage the public to use alternatives to plastic or participate in plastic reduction schemes such as refill stations or bulk buying. For example, previous CAPs have surveyed if and how retailers and food businesses offer packaging alternatives such as reusable bags or implement reduction strategies such as product bans or fees. Maddalene et al. (2023) documented examples of non-plastic alternatives for to-go packaging, which may have resulted from city responses to national bans on certain plastic items.

Defining C&D Operation goals and scope: Translating this spoke to the C&D context, the goal of the Operation spoke is to understand what is occurring in the collective “use” phase of the community’s building stock. In the EMF Circular Buildings Toolkit reuse, renovations, and repurposing an existing asset are key to avoiding new construction and building efficiently. Previous studies have explored land consumption and unused buildings as a critical feature of circularity. For example, along with examining land consumption rates relative to population changes in Sicilian municipalities in Italy, Barbaro et al. (2022) also evaluated unused building stock based on the Italian census data. For the C&D CAP, we suggesting using indicators such as

use and rate of land development, building and property utilization rates, and rate of renovations.

Data acquisition and benchmarking: Determining vacant land locations can be useful for cities because it provides the potential for reuse or bringing community members together, for example, by creating a community garden. The next aspect to review is the building and property utilization rates. The goal is to determine what already exists within a city, what is needed, and what is excess. If available, buildings will be determined to be in use or vacant. Determining the building's occupancy status can better inform cities about what type of construction or deconstruction is necessary for growth. For example, if there is an excess of vacant homes or apartments, cities could take advantage of existing but under utilized building materials to construct more housing units. Lastly, the rate of renovation is considered. Commercial and residential renovation permits will be examined to see the frequency of renovation compared to new construction within a city. If renovation is being utilized, it can indicate cities are reducing their environmental impact by limiting new construction.

Impact and stakeholders: Strategic land usage can impact the community's heat island effect, promote biodiversity, and help a city withstand the impacts caused by climate change (Colding, 2007; Coseo & Larsen, 2014; Sachs et al., 2019). If cities are physically expanding while also experiencing land vacancies and abandoned structures, it may indicate that they are experiencing urban sprawl and missing sustainable growth opportunities. Governmental authorities and organizations can play several important roles relevant to the Operation spoke. For example, government stakeholders can improve awareness among construction professionals, waste management servicers, and the public, provide financial aid, and manage fiscal and regulatory actions to advance circularity of local C&D materials (Munaro & Tavares, 2023).

4.5 Collection

CAP Collection summary: While the upstream aspects of the CAP diverge for the two material streams (i.e., plastic consumer goods versus C&D materials), properties of material management begin to converge in downstream phases such as waste collection, treatment, and emissions. The CAP Collection spoke aims to explore how much and what types of waste are generated in communities, as well as how much waste is collected and by what means. When relevant, the approach also gathers information related to informal waste collection processes that contribute to a community's waste management system. For example, Jambeck et al. (2024) identified collection challenges associated with maintaining public waste receptacles throughout Manila as well as household waste segregation that led to inefficiencies in recovering potentially recyclable waste.

Defining C&D Collection goals and scope: We define the goal of the C&D Collection spoke to encompass documentation of quantities and types of waste from C&D activities and corresponding collection practices. C&D waste is historically challenging to quantify due to several factors such as inconsistent C&D material definitions, whether there is

infrastructure in place to measure and report quantities, and how contractors operate relative to the other waste streams (i.e., MSW) (Clark et al., 2006; Kaza et al., 2018). Further, waste generation and respective collection activities can vary widely due to local economic conditions, location seasonality, and occurrence of disasters or manmade events. Additionally, C&D waste stream composition can vary from place to place due to climate-specific needs, local availability of materials, building costs, and aesthetics (Townsend & Anshassi, 2023).

Beyond quantifying waste generation and composition, identifying local systems, regulations, and organizations that contribute to collection of waste can uncover gaps or inefficiencies throughout the C&D waste management system. In the MSW context, waste is commonly collected door-to-door at a pre-determined frequency or deposited by households in central containers or locations for pick up and transport for disposal (Kaza et al., 2018). In the C&D context, debris generated on-site, including unused or scrap material, as well as packaging, can be managed through a range of options based on the phase of construction and is primarily overseen by building contractors and/or waste management servicers. Common approaches for waste collection during construction include placement of material-specific containers on site that are regularly hauled to a disposal facility, hiring of a job-site clean-up service that stages and removes debris periodically, direct utilization of primary contractor crew and equipment, and requiring subcontractors to manage their own waste staging, processing, and transport for disposal (e.g., requiring an electrical subcontractor to manage their debris). Demolition waste differs from construction waste in that it includes whole components and/or composite building materials, and often these materials are in various conditions due to use and aging.

In the context of CE, there are several demolition methods ranging from unselective demolition such as mechanical and implosive demolition, as well as hand demolition, selective salvaging, deconstruction, and soft stripping, which follow strict methods for abatement of hazardous materials that are managed separately (Townsend & Anshassi, 2023). The adoption of circular C&D waste collection methods can vary substantially, with local governance often driving uptake through incentives and ordinances as well as strategic funding or partnerships that encourage the development of local markets for salvaged materials. For example, deconstruction, which encompasses planned building disassembly to maximize recovery of reusable and recyclable materials, is required in only a few cities across the US (CROWD, 2023), highlighting the lack of widespread integration of this important circular materials management strategy.

Data acquisition and benchmarking: Despite these challenges, there are some established and widely adopted approaches for estimating C&D waste generation and composition to help approximate demolition job costs, inform governance and policy approaches, and anticipate community planning needs (Table 3 summarizes US-based estimation methods). Such approaches include relying on historically reported data when available, taking physical

measurements and sorting, correlative statistics using building and economic statistics, and MFA (Townsend & Anshassi, 2023). For the case of estimating waste generation and composition at the city or regional scale, mathematical modelling approaches may be better suited and are commonly used in extant literature. For example, Lederer et al. (2020) conducted a city-scale MFA of construction mineral wastes in Vienna, Austria, finding that the city consumed 4.4 metric tons of concrete, masonry, asphalt, and aggregate in 2014, which they ultimately used to explore various scenarios advancing local CE efforts.

Impact and stakeholders: The C&D CAP will document how C&D waste generation and collection are managed and monitored in local cities to ultimately inform steps to create more sustainable outlets that are appropriate for the community. By identifying quantities and types of C&D waste generated in a community, the CAP can help identify targeted waste reduction interventions. Similarly, CAP C&D findings can inform strategies for segregation and collection operations such as regulating or incentivizing on-site waste segregation to maximize material recovery. Relevant collection stakeholders include construction professionals such as designers, contractors, and builders, as well as waste management suppliers such as haulers and service providers (Munaro & Tavares, 2023). Together, these stakeholder groups lead jobsite management decisions, the development of guidelines related to on-site waste generation and segregation, and hauling and processing operations that are key to efficient C&D debris management and material recovery. Additionally, government stakeholders play an important role in incentivizing deconstruction activities that contribute to advancing CE at the city scale.

4.6 End-of-cycle

CAP EOC Summary: The original CAP EOC spoke aims to explore how waste is disposed in (or near) the community. While waste reduction is a high priority, effectively treating waste that is generated is critical for recovering reusable or recyclable goods and preventing losses to the environment. To understand the CAP community's solid waste management system as part of the local CE, the CAP maps the local treatment infrastructure related to consumer plastic goods, such as landfills, material recovery facilities, or composters. Additionally, this spoke can uncover the role of alternative treatment streams and local systems. For example, in addition to identifying both formal and informal waste collection outlets in Manila, Jam-

beck et al. (2024) also uncovered the role that informal junk shops played in recovering rigid plastics and clean PET which could be sold for recycling.

Defining C&D EOC goals and scope: In alignment with the original CAP developed for plastics, we similarly define the goal of the C&D EOC spoke to explore how C&D waste is managed and treated in the community under investigation. Broadly, non-hazardous C&D waste is often collected and treated in conjunction with MSW, however, some municipalities and countries require separate collection, sorting, and treatment. EOC treatment for C&D debris includes similar treatment outlets as plastic goods, such as recycling, landfill and permanent storage, and energy recovery. When C&D debris is not recovered for salvage, reuse, and refurbishment, there are several treatment methods that are driven by material type as well as local levels of infrastructure and economic development. In the US, 52% of C&D debris is managed via conversion to aggregate, followed by landfill (24%), and conversion to manufactured products (22%). A small fraction (<1%) is managed via conversion to fuel, soil amendments (excluding composting), and compost and mulch (US EPA, 2020). Like other phases of C&D material life cycle, the treatment of C&D waste can vary drastically from place to place. For example, in the US, regulations of C&D waste are largely under the purview of each state, resulting in a wide range of policies targeting the management of C&D waste ranging from how it is defined, how C&D landfills are designed, requirements for C&D facility permitting, and whether proximal groundwater monitoring is required (Clark et al., 2006). Additionally, recycling is heavily dependent on local drivers such as costs of recovering and market conditions, the presence of green building programs, regulatory programs or requirements, and local desire to work toward environmental or sustainability goals (Townsend & Anshassi, 2023). Taken together, understanding how various types of waste are managed can help to inform reduction targets and alternate treatment options.

Data acquisition and benchmarking: The modeling approaches to estimate C&D waste generation quantities and composition as described in the previous section (Collection) can also inform waste treatment targets. As such, we recommend using common data sources for the Collection and EOC spokes, with additional EOC-specific input from stakeholder interviews, inventories of local infrastructure, potential salvage or resale markets, and waste treatment facilities to determine where C&D waste is disposed of and

TABLE 3: Examples of US-based approaches for estimating quantities of construction & demolition waste generation, composition, and end-of-cycle management (Adapted from Townsend and Anshassi 2023).

Approach	Source	General method description
Construction statistics	US EPA, Franklin Associates (1998)	Estimates building-related debris using correlation of production statistics for six construction & demolition sectors: residential and non-residential construction, demolition, and renovation.
Consumption statistics-based material flow analysis	Cochran and Townsend (2010)	Estimates total construction & demolition debris generation based on type of construction project (e.g., new vs. renovation) with consideration of material input
Disposal statistics-based material flow analysis ("CDDPath")	Townsend et al. (2019)	Estimates construction & demolition debris generation and incorporates end-of-life management pathways (e.g., landfill, recycling, etc.) and secondary feedback and markets for 12 material types

identify outlets for landfill diversion and improved recycling and reuse. Like the Collection spoke, waste quantities can be used as benchmarking metrics to help examine how various EOC interventions are impacting waste treatment over time.

Impact and stakeholders: The EOC spoke can help identify gaps in the local waste management system, as well as identify potential outlets for material recovery. Improving EOC waste treatment encourages efficient use of material resources and prevents C&D debris emissions to the environment. For example, Blengini (2009) showed that recycling of building materials can be environmentally beneficial by offering reductions in energy consumption and greenhouse gas emissions. Further, improving the capture and treatment of C&D debris that cannot be recycled or reused can lead to reduced environmental impacts such as land degradation and pollution. Relevant EOC stakeholders include government authorities and establishments, who might advance integrated CE processes, guidelines, and funding into the local waste management system, as well as waste management suppliers such as haulers, processors, and service providers (Munaro & Tavares, 2023), who may lead decision-making around waste operations, including data collection and reporting.

4.7 C&D Emissions

CAP Leakage summary: The CAP Leakage spoke aims to explore what materials end up in the environment and relevant leakage pathways. To do so, the CAP documents litter via field transect surveys and observations, establishing a litter density and composition which can then be connected with findings from interviews with local stakeholders and literature to develop strategies for intervention. Both Maddalene et al. (2023) and Youngblood et al. (2022a) found variation of litter densities across sampled cities and population densities in south and southeast Asian communities, finding that higher populations did not necessarily

correspond to higher litter densities. Similarly, the CAP revealed that the proportion of plastic in litter found in Manila, Philippines (Jambeck et al., 2024), was less than that found in cities with relatively developed economies located in the United States (UN Environment, 2021).

C&D Emissions goals and scope: In the case of C&D, we similarly define the scope of the C&D Emissions spoke to explore what and how C&D materials end up in the environment. First, C&D debris emissions can encompass a wide range of materials including such as concrete, roofing, lumber and wood products, gypsum board, steel, finishes and components such as carpeting, paint, appliances and electronics, insulation, ducting, and inert waste such as aggregate, dirt, and sand (Townsend & Anshassi, 2023). Additionally, there are a range of emission pathways for C&D debris throughout the building life cycle (Figure 4), such as littering and illegal dumping/fly-tipping (Du et al., 2021), natural disasters (Dubey et al., 2007), and building wear and tear (Müller et al., 2020). Active C&D job sites may be an important source of lost materials (Zadjelevic et al., 2023), though most construction pollution research is focused on managing particulate emissions (Cheriyen & Choi, 2020) rather than solid waste debris losses, which may often be presumed to be collected and managed. As such the scope of the C&D Emissions scope must be expanded to include various leakage pathways relevant to the C&D life cycle.

Data acquisition and benchmarking: The C&D CAP can leverage similar approaches as the CAP to document debris losses (i.e., transect surveys), but due to the variation in seasonality and building and construction lifetimes, other supplemental methods might include surveys of C&D jobsites and illegal dumping hotspots. Few methods have been established to specifically target C&D macro-debris losses, potentially due to unclear definitions delineating C&D emissions from other sources and sectors. However, few studies exist that specifically document jobsite losses,

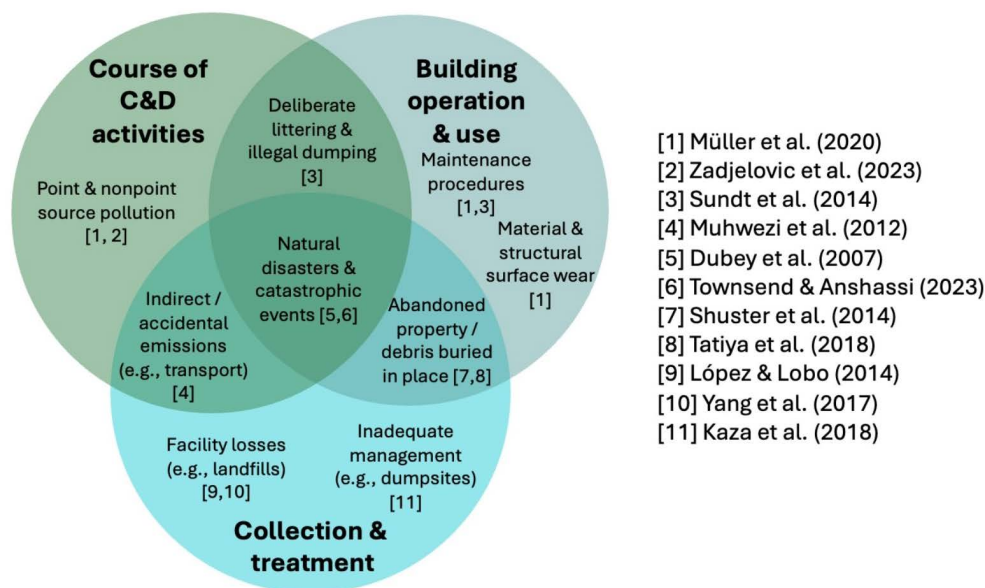


FIGURE 4: Sources of C&D debris emissions and exemplary references.

highlighting a need for establishing survey procedures. Zadjelovic et al. (2023) conducted field surveys proximal to a construction site in Chile to document losses of expanded polystyrene material used in sidewalk expansion joints, while Järllskog et al. (2021) sampled stormwater inlets, street swept debris, and road dust to examine pollutants lost from a reconstruction project in Gothenburg, Sweden. The C&D CAP could similarly track losses adjacent to active construction areas, in addition to documenting other potential pathways for construction losses to the environment. Findings from the C&D CAP can be referenced against previous surveys that may have been conducted by the community, local nonprofits, or academic researchers. Additionally, C&D CAP cases can be compared based on leakage density or product and material composition. In cases that have not yet established litter surveys, the C&D CAP will act as a benchmark, providing critical knowledge around C&D material losses such as materials or products that can be targeted, hotspots or community areas that can be mitigated via enforcement or continued monitoring activities, etc.

Impact and stakeholders: Ultimately, knowing the profile and locations of C&D debris losses can help inform upstream efforts to curb them such as monitoring, enforcement of litter and dumping policies, and education efforts that are adapted to the local community. As one of the most impactful issues of material management, reducing impacts from pollution can bring important change to communities. Losses may be prevented by integrating CE-based strategies into construction and building policies, with one recent study finding that CE policies may reduce construction-related air pollution by 2.92% (Zhu et al., 2022). Further, connecting upstream strategies to leaked materials could aid in offsetting costs for cleaning up illegal dumpsites (Du et al., 2021). Key stakeholders for this spoke include community members who may be impacted by inadequately managed C&D waste, local contractors or businesses who may contribute to illegal dumping, organizations that conduct cleanups and education campaigns, and policymakers who decide on local ordinances that encourage and regulate sound waste management systems and prevent leaked materials from reaching the environment in the first place (Santos et al., 2019).

5. RECOMMENDATIONS & CONCLUSIONS

Importantly, we propose an iterative process to allow for community empowerment, progressive assessment, and reflection. CE assessments offer an essential utility by generating baseline measurements that can be used as reference points to monitor impacts and make informed decisions (Patil et al., 2023). Therefore, we recommend the following: 1) compilation of initial findings/assessment; 2) review of the initial findings with the community and then refinement of the assessment; and 3) reporting of findings and identification of strategies (Figure 5). The initial assessment is the first phase of the C&D CAP providing a general overview of construction and demolition materials being used and discarded within a city. Information gathered from the initial assessment is presented to stakeholders within the community. Based on the objectives determined

by groups, the C&D CAP will be reassessed within the city. Based on the results from the focused C&D CAP, opportunities are provided and implemented by stakeholders.

We note that in its current form, the C&D CAP has been conceptualized based on exploring data outlets and accessibility with a US-based and urban perspective given the authors' positioning. As a result, the initial concept of the C&D CAP presented here is based upon assumptions of data availability and access that may need to be adjusted in locations where reporting of construction, demolition, or general waste management data are not common or consistent. To address this limitation, future iterations of the C&D CAP will aim to explore ways to integrate the use of proxy representations or maximize the use of standardized, global datasets when needed.

Further, the CAP comprises a wide range of topics, knowledge, and methods. Here, we have generated a brief overview of potential evaluation tools relevant to each spoke, with the anticipation that future work will further investigate theory, methods, and relevant applications to fulfill C&D CAP needs in a robust manner, including application and testing of methods in C&D CAP case cities. Additionally, a key facet of the original CAP method is the co-creation of community knowledge with local implementation partners. As we examine outlets for generating understanding of C&D debris management at the city-scale, we anticipate identifying areas for inclusive and participatory ways for community members to be involved in the work beyond stakeholder engagement. For example, trained community members could lead visual inspections as part of the Building materials & construction spoke or could contribute to documenting C&D litter and illegal dumping locations as part of the C&D Emissions spoke.

We note that the approach used by the CAP aims to generate transferable, rather than generalizable findings, such that practical interventions are case-specific but can offer insights to other locales in the form of lessons learned or best practices. The nature of case studies means that the trajectory of evaluation is not always systematic or linear, which highlights the importance of considering data reliability, issues that the methodology has been criticized for in the past. However, these issues can be addressed throughout the case study research process by clearly defining the research goal and design, establishing a robust data collection and case analysis protocol, and elaborating on methods in reporting (Quintão et al., 2020). Additionally, validity techniques such as triangulation and member checking can help support validating study findings, while researcher reflexivity can help identify prior assumptions or biases that could influence qualitative data collection and analysis (Mays & Pope, 2000; Torrance, 2012). Lastly, relying on public datasets, such as those generated by municipalities or local contractors, could lead to inconsistencies in methods, reporting, units, and definitions that can make assessing solid waste challenging (Kaza et al., 2018). One recent project adapting climate change methods to the issue of plastic pollution called Project Drawdown created a rating system for various data sources used in their case evaluations that could be similarly useful in the application of the C&D CAP (Royle et al., 2022).



FIGURE 5: Iterative process of C&D CAP assessment.

By assessing the full life cycle of C&D materials at the macroscale (e.g., cities, communities), we can identify opportunities for aligning various parts of the local CE. For example, by understanding community needs, we can identify outlets for workforce development, capacity building, and potential issues of safety and well-being in relation to the C&D activities. Further, the nature of the CAP as a community-based framework portends that outputs and deliverables of the CAP are both specific to each case as well as accessible to a wide audience in order to foster open dialogue and collaboration. Supporting cities' transitions to the CE can be instrumental for advancing global circularity. The methodological concept presented here aims to strengthen implementation of such local transitions by looking across the full life cycle of C&D, while also paving a pathway for convergence among multiple sectors.

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SUSTAINABLE SOIL STABILIZATION OF ROAD PAVEMENT LAYERS USING WASTE MATERIALS: A MINI-REVIEW

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ABSTRACT

The escalating generation of waste necessitates sustainable management strategies. This review explores the potential of waste materials as sustainable stabilizers for soil used in road construction. Conventional stabilizers like cement, while effective, comes with environmental drawbacks that should be addressed. Alternatively, diverse waste-based materials are being investigated as potential substitutes. Their usage in this regard contributes to lessen adverse ecological impact and improve soil properties at potentially low costs. Prior to the application of a specific soil stabilised with a specific waste material for a specific purpose, the potential risk of local environmental and health impacts from release and spreading of harmful substances due to leaching or escape of fugitive dust must be assessed and appropriate action (precautions, mitigating measures, changes of the soil/stabilizer mixture) must be taken when necessary to prevent unacceptable impacts. If properly managed, the utilization of waste materials for soil stabilization aligns with several United Nations Sustainable Development Goals (SDGs). By diverting waste from landfills and promoting their use in construction, this approach presents a win-win scenario for both economic and environmental sustainability. Further research and development efforts are crucial to optimize waste-based soil stabilization techniques, ensuring durable and eco-friendly pavements. These wastes ultimately improve strength of the target soil, however, careful consideration should be given to determine the performance of these wastes as stabilizers and to know what aspects would require mitigation to maximize the potential of these waste materials to be used for good, instead of causing more harm to the ecosystem.

1. INTRODUCTION

The pace of waste generation is skyrocketing globally. As per a World Bank assessment executed in 2022, the world produced solid wastes (SW) which amounted to 2.24 billion tons in 2020, estimating 0.79 kg of SW generated each day per individual (The World Bank, 2022). Dense population expanse and industrialization are forecasted to cause the yearly waste accumulation to climb by 73%, from 2020 to over 3.88 billion tons by 2050. Conforming to the United States Environmental Protection Agency (USEPA) in 2022, nearly every single person, business and overall human activities produces wastes of some kind, which include sewage sludge, hazardous and non-hazardous industrial waste, medical waste, nuclear waste, demolition and construction debris, distillation and mining waste, wastes from producing oil and gas, residues and ashes from thermal treatment of industrial, agricultural and domestic waste and from energy production. Inefficient handling of waste has a detrimental effect on inhabitants

of world economies. Hence, waste management should be properly investigated. Sustainable waste management is a component of the new 'green' economy, which strives to encourage economic expansion while safeguarding the environment.

The generation, collection, storage, processing (recovery), transportation, and disposal phases are the six segments that make up the typical division of the solid waste management procedures. Inadequate administration/regulation and oversight of these activities may have undesirable effects on health and our ecosystem. The cycle of negative impacts that plagues improper management of wastes would continue to go on if they are not checked.

Waste management strategies like recycling and reuse are the most typical in the building sector, where waste materials have been recycled for use as or in conjunction with other materials to yield potentially sustainable and cost-effective building materials (Ofuyatan et al., 2020). Barbuta et al. (2015) stated that employing disposal resources while also lowering environmental pollution are both possi-



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ble with the new generation construction materials that are blended with various wastes. The assessment of waste usage in construction has been known to be broad especially in road pavement construction where recycled materials have been used for asphalt concrete and bitumen (Rahman et al., 2020); and also, for subsoil and embankment stabilization (Vukićević et al., 2019) so as to strengthen the geotechnical attributes of the soils used in pavement layer design. In this review paper, a detailed examination of the use of wastes for sustainable soil treatment is presented. These materials are investigated to determine their potential to improve soil properties in pavement applications. This eco-friendly and cost-effective approach could mitigate the harmful effects of these wastes on human health and the environment.

2. SOIL TYPES AND STABILIZATION TECHNIQUES IN ROAD CONSTRUCTION: GEOTECHNICAL PROPERTIES, ENVIRONMENTAL CONSIDERATIONS, AND APPLICABILITY

One of the most crucial and fundamental media for any building project is soil. Any structure's durability and strength are contingent on the soil's strength capabilities. The complex blend of mineral substances, matter that is organic, gaseous substances, fluids, and innumerable biological organisms that makes up soil is what preserves existence on the Earth. Defective soils are viewed as those that do not satisfy all or part of the requirements necessary for them to function competently in geotechnical building work. Poor load-bearing capacity, severe shrink-swell potential, rapid water drainage, and pollution are frequent characteristics of these deficient soils. These are likely to be seen in base courses for roadways, embankments for roads or dams, subsurface bases for foundations, clay

liners for leachate control, and fills for retaining structures (Alhassan & Mustapha, 2015). Stabilization would eventually be necessary to achieve an improvement in the geotechnical characteristics of these weak and defective soils. The process of modifying a natural soil to enhance one or more of its characteristics by adding specialized soil, cement, or other chemicals is known as "soil stabilization" (Habiba, 2017). For overall soil improvement, a variety of techniques have been established and used, as shown in Figure 1. These techniques start with the more fundamental ones, like soil replacement, excavation, and drainage, while moving on to more sophisticated ones, like chemical-based treatment, the reinforcement approach, variable modification (hydraulic, electrical, thermal mechanical), and so on (Gomes Correia et al., 2016; Md Zahri & Zainor-abidin, 2019; Scope et al., 2021). The methods are suited for deep, medium, and shallow soil modification, and are commonly classified into two groups based on the site details and geotechnical requirements: a) mechanical techniques including soil replacement or displacement, phased construction, the preloading phase, the use of stone columns, embankment supports, and applications of synthetic reinforcing (Bordoloi et al., 2017); and b) chemical methods, such as stabilization of surfaces and deep integration/mixing in place with the use of admixtures including slag from blast furnaces, cement, gypsum, fly ash, bentonite, silica fume etc. (Celik & Nalbantoglu, 2017; Sharma & Sivapullai-ah, 2016; Yadu & Tripathi, 2013).

Conventional stabilizers, such as cement, have been widely used to stabilize soil over time. Cementitious reactions act to link particles, enhancing stiffness and strength while decreasing strain and enhancing permeability. Its calcium-rich nature is essential to its setting, development of strength, and general effectiveness. However, the production of cement is associated with substantial energy

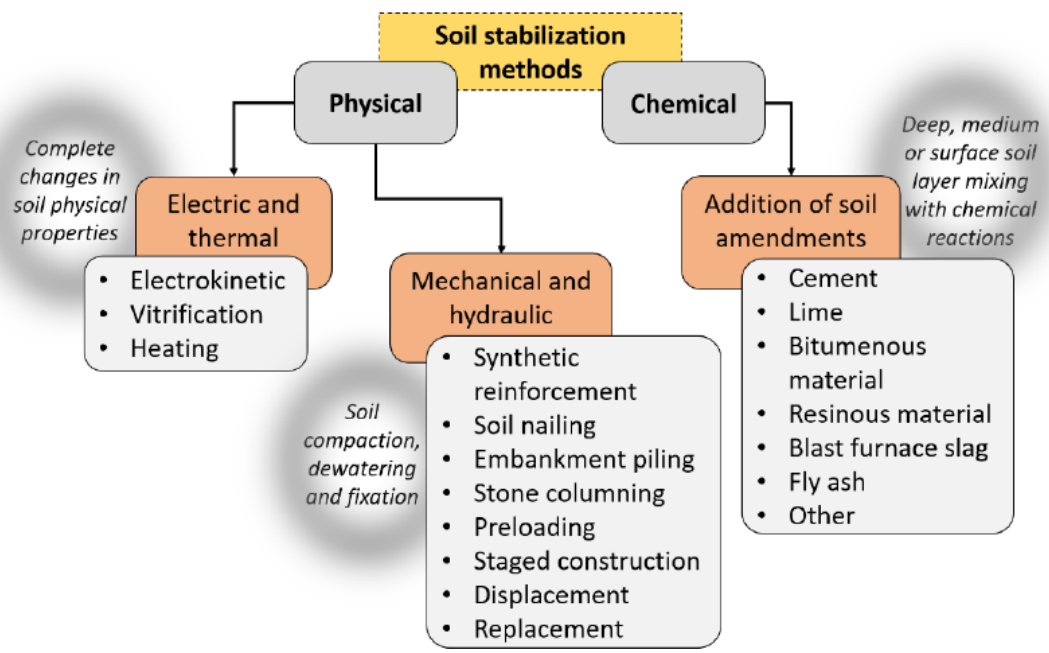


FIGURE 1: Methods of Soil Stabilization (Gomes Correia et al., 2016).

and non-renewable resource consumptions as well as a significant global carbon dioxide (CO₂) generating process (Akinwumi et al., 2023), soil cracking and high cost. The production of cement is thought to produce significant discharges of greenhouse gases, which contribute significantly to global warming and climate change (Tesanasin et al., 2022). This is considered very alarming. Hence, alternative materials to cement are a current necessity (Pelissaro et al., 2023). As a substitute to cement/lime, the development of innovative sustainable binding agents is crucial to promoting greener manufacturing and lowering carbon emission (Dahal et al., 2019).

To improve soil for the construction of road pavements, researchers have been investigating this and are continuously coming up with ways to use waste materials like fly ash, rice husk, calcium carbide residue, bagasse ash, etc., to create environmentally friendly substitute binders. The development of these cutting-edge low-carbon stabilization products and procedures is continually progressing and forging a new path for the field of civil engineering (Yu et al., 2024). These substitute admixtures are taken into consideration because of how well they enhance soil characteristics. Reclaimed asphalt pavements (RAP), for instance, have a better capacity to release stresses conveyed to subgrade, reduce permeability, and have excellent durability against imposed loads, all of which can improve the mechanical performance of the pavement layers (Hashemian et al., 2023). Furthermore, when employed as stabilization agents, waste ash from materials like plantain peels, rice husk, bagasse, etc., exhibits pozzolanic properties. The primary cause of stabilization efficacy is the pozzolanic interaction that occurs between calcium-based compounds and soil. Calcium hydroxide is transported by water and combines with alumina and silica in aggregates to generate calcium aluminate hydrate (C-A-H) and calcium silicate hydrate (C-S-H), which are the products of the pozzolanic reaction, which is often slow and lengthy (Komaie et al., 2023). The C-S-H gel contributes to the adhesion of the particles, strengthening the mixtures (Juveria et al., 2023). By virtue of pozzolanic processes, clay soil that has chemical additions becomes stronger with time. Upon reaction with the aluminates and silicates in the soil to form cement gels, the calcium hydroxide (CaOH) creates a pozzolanic reaction that reinforces the stabilized soil. Stronger cementation results from higher particle binding as the amount of cement gels created rises (Sosahab et al., 2023). These waste products have the effect of obstructing pores. As a result of their low coefficient of permeability, capillary pores that are blocked cause permeability reduction, acting as a hydraulic barrier (Karami et al., 2021), which subsequently supports improved strength and expansion (Mansour et al., 2022).

In general, alternative binders with strong mechanical qualities and energy conservation have become very popular recently. This is because using them to treat soil potentially keeps them out of landfills, which may help in reducing several environmental difficulties and land waste issues (Li et al., 2024). Due to their low specific gravity, waste ashes can pollute land, air, and water, while leakage into groundwater can result from their disposal in open pits. However,

these issues can be potentially remedied by mixing them with soil, which when wet, reacts to produce cementitious materials. Considering these cementitious products are eco-friendly, using their waste materials for road pavement stabilization turns out to be an environmentally suitable way to manage waste (Khandelwal et al., 2023). Before utilizing these materials in this regard, environmental and health that may arise from their usage should be assessed and take steps should be taken prevent them.

Ultimately, the future of soil stabilization lies in embracing beneficial alternatives. By utilizing waste materials as stabilizers, we can achieve strong, durable construction while minimizing environmental impact. Research into novel low-carbon binders and processes is crucial for promoting cleaner production and a lower carbon footprint in the civil engineering industry. This shift towards sustainable soil stabilization presents a win-win scenario for both construction and the environment.

3. WASTE MATERIALS USED FOR SOIL STABILIZATION: OPTIMIZING GEOTECHNICAL PROPERTIES AND MINIMIZING ENVIRONMENTAL IMPACTS

Through mix proportions and introducing the appropriate additives or stabilizers, soil stabilization seeks to augment the integrity or bearing capability of the soil involved (Díaz-López et al., 2023). The use of conventional stabilizers like cement and lime has been employed over the years for soil improvement and has been effective irrespective of the expensive nature (Anburuvel et al., 2023). Nevertheless, most highway agencies are not inclined to commit money on these typical stabilizers for soil restoration because of budgetary constraints (Zimar et al., 2022) and also due to their unfavorable impact on the ecosystem as at a global level, the processing of cement generates around 7% of greenhouse gas emissions (Chen et al., 2020). As a result, road authorities and researchers seek industrial byproducts with pozzolanic characteristics that are potentially sustainable and more affordable. Some of these waste materials that have been investigated for use as soil stabilizers are reviewed in this paper. They include reclaimed asphalt pavement (RAP), recycled concrete aggregates (RCA), ground granulated blast-furnace slag (GGBS), steel slag (SS), rice husk ash (RHA), coal fly ash (CFA), granite dust (GD), plantain peel ash (PPA), calcium carbide residue (CCR) and bagasse ash (BA).

3.1 Reclaimed asphalt pavement (RAP)

Maintenance of asphalt roads typically involves milling the pavement, sometimes down to its full depth, followed by applying a fresh layer of pavement. The material removed during milling, known as reclaimed asphalt pavement (RAP), consists of granular fragments that, when reused, enhance an asphalt mixture's resistance to moisture and deformation (Spreadbury et al., 2021; Aravind & Das, 2007; Zaubanis & Mallick, 2014).

Reusing RAP offers significant financial savings and ecological benefits such as reduced greenhouse gas emis-

sions, landfill diversion, and lower transportation costs (Foye, 2011; Nassar & Nassar, 2006; Huang et al., 2009; Celauro et al., 2015; Yang et al., 2015). RAP has become increasingly popular for soil stabilization due to its environmental advantages, conserving natural resources and reducing environmental pollution (Avirneni et al., 2016). It contains bitumen and aggregates that improve soil strength and reduce water permeability, making it beneficial for road construction (Alhaji & Alhassan, 2018). For instance, it has been successfully used as a granular subbase (GSB) material, enhancing structural integrity and stability (Pradhan & Biswal, 2022; Lima et al., 2023).

Despite these benefits, challenges exist. "Old" asphalt based on tar has a much higher content of potentially hazardous polycyclic aromatic hydrocarbons (PAH) than "newer" asphalt based on bitumen (IARC, 2010), which should be considered. Preferably, utilization of tar-based asphalt should be avoided. Also, variability in RAP composition, including potential contaminants like heavy metals or PAHs, requires careful testing and mix design optimization to ensure consistent performance (Edeh et al., 2011; Legret et al., 2005). Strategies such as determining optimal RAP proportions and evaluating specific RAP properties are crucial for effective soil stabilization (Ahmed et al., 2019). Prior to using RAP-stabilized soil, assessing and mitigating risks of harmful substance release through leaching or dust to prevent adverse environmental and health effects is vital. Overall, RAP offers a beneficial solution for soil stabilization, reducing the global demand for virgin aggregates and energy consumption while diverting waste from landfills (Milad et al., 2020; Ochepo, 2014). Careful consideration and management are essential to maximize its benefits and mitigate potential drawbacks in construction projects.

3.2 Recycled concrete aggregates (RCA)

The disposal of construction and demolition (C&D) debris, particularly concrete waste, is escalating worldwide due to the proliferation of structures in developed and developing nations. In Europe alone, over 800 million tons of C&D waste are generated annually, with concrete constituting up to 75% of this waste by weight (Wang et al., 2023). Recycling and reusing these concrete wastes for pavement layers have gained global attention as an environmentally sustainable approach to construction (Lu et al., 2021; Poltue et al., 2020; Saberian et al., 2018; Yaghoubi et al., 2018). Recycled concrete aggregates (RCA) predominantly constitute C&D waste, ranging from 65% to 80% natural aggregates (NA) and 20% to 35% cement mortar, typically sized between 20 and 30 mm (Saberian et al., 2020; Al-Obaydi et al., 2021; Tavakol et al., 2020; Sohrabpour & Long, 2021; Arulrajah et al., 2022). RCA finds applications in various construction contexts such as backfills around culverts, railroad capping layers, stabilized road pavements, and geobag walls (Sohrabpour & Long, 2021; Mohammadinia et al., 2020; Vegas et al., 2011; Arulrajah et al., 2014; Wu et al., 2020).

Recent research highlights the superior cementitious properties of sand-sized RCA (less than 5 mm), which contain higher cement paste compared to gravel-sized RCA, making them suitable as supplementary cementitious materials (Evangelista et al., 2015; Moreno-Juez et

al., 2021; Mikos et al., 2021). In soil stabilization, RCA has proven effective in enhancing engineering properties such as reduced plasticity, improved compaction efficiency, and increased bearing capacity. For instance, it has been successfully used to stabilize clay soils and improve the bearing potential of soils like black cotton soil (Kianimehr et al., 2019; Pazare et al., 2023). The addition of RCA to soils enhances their strength and stability, making them suitable for various construction applications including subgrade or subbase of road pavements. RCA usage in pavement construction may also be characterized by reduced costs. This is because as CBR rises, it is possible to decrease pavement thickness, thereby lowering construction costs (Datta & Mofiz, 2021).

However, the application of RCA is not without challenges. Variability in RCA properties due to differences in source concrete composition and processing methods necessitates rigorous testing and optimized mix designs (Datta & Mofiz, 2021). The angular and rough texture of RCA can affect workability, requiring adjustments in mixing techniques or additives (Shourijeh et al., 2022). Moreover, residual contaminants like heavy metals from the original concrete, such as chromium, lead, and arsenic, pose potential risks of leaching into the soil under certain conditions (Galvín et al., 2013). Careful engineering design, thorough testing of RCA and soil mixes, and selecting low-contamination risk sources should be taken into consideration (Tavakol et al., 2019; Kianimehr et al., 2019). Conducting a thorough evaluation of potential environmental and health risks from leaching or dust in RCA-stabilized soil and taking steps to mitigate these impacts before application is important. In conclusion, while RCA offers potentially significant benefits in terms of cost savings, resource conservation, and environmental sustainability in construction, careful consideration of its properties and likely drawbacks is crucial for its effective application in soil stabilization and other construction activities.

3.3 Ground granulated blast-furnace slag (GGBS)

Increasing the use of supplemental cementing components like ground granulated blast-furnace slag (GGBS), which are often discarded in lagoons and landfills, is essential for reducing construction costs and greenhouse gas emissions from traditional stabilizers like cement and lime, such as carbon dioxide (CO₂), nitrous oxide (N₂O), and methane (CH₄) (Sharma & Sivapullaiah, 2016). GGBS is a byproduct of pig iron production, ground into a fine, non-crystalline form after rapid cooling. Its reuse in engineering applications is considered environmentally beneficial, frequently serving as a binding agent in concrete and mortar (Pal et al., 2003; Jin et al., 2015; Zhao et al., 2023). With increasing availability, GGBS has also been applied in soil treatment and stabilization.

Rasool and Kapoor (2017) explored GGBS as a substitute for cement or lime to support increased traffic stresses on foundation structures. They added GGBS to clayey sand in proportions of 0%, 6%, 12%, 18%, and 24%, finding that 18% GGBS maximized the CBR value. These results suggest that the effectiveness of GGBS can be enhanced with an activator to break its glassy phase, as GGBS is a la-

tent hydraulic material (Mumtaz & Bhatia, 2022). Yadu and Tripathi (2013) also assessed GGBS for soft soil stabilization, combining soil with 3%, 6%, 9%, and 12% GGBS. They observed improved soil strength and physical properties, reduced OMC, and increased MDD, attributed to GGBS's higher specific gravity (2.89) compared to soil (2.56). Additionally, they reported a significant reduction in soil swelling and an increase in UCS and CBR values, with 9% GGBS being optimal. Further studies indicate that GGBS can improve various soil types. Patel et al. (2020) found a 15% inclusion optimal for black cotton soil, while for soft clay with poor shear strength, 40% was most effective. Research consistently supports GGBS as a beneficial and efficient replacement for traditional stabilizers, enhancing soil strength and stability while addressing environmental concerns. Optimizing GGBS activation methods could yield even greater benefits (Pathak et al., 2014).

However, caution is necessary due to potential drawbacks. GGBS effectiveness can vary with its chemical composition, influencing its reaction with soil components (Al-khafaji et al., 2017; Sharma & Sivapullaiah, 2016; Rizki Abdila et al., 2020). In certain soils or environmental conditions, elements in GGBS may leach into the environment (Müllauer et al., 2015). Before using soil stabilized with GGBS, assessing the risk of environmental and health impacts from harmful substances leaching or dust escaping, and taking necessary precautions to prevent negative effects is vital.

3.4 Steel slag (SS)

Urban expansion has led to a significant rise in steel slag (SS), an alkaline waste product from iron and steel smelting processes. Annually, China alone generates over 100 million tons of SS (Yu et al., 2024). While SS is used in various applications such as iron-ore processing, road fillers, and railroad ballast, its utilization rate remains low, below 25% (Pang et al., 2015). This underutilization exacerbates environmental concerns due to SS disposal.

Efforts to harness SS more effectively have focused on enhancing soil properties, particularly for soil stabilization purposes. Studies have shown promising results in improving geotechnical characteristics when incorporating fine steel slag aggregate (FSSA). For instance, Aldeeky and Hattamleh (2017) demonstrated that adding 20% FSSA to high-plasticity subgrade soil reduces plasticity index and free swell by 26.3% and 58.3%, respectively. Moreover, it increases unconfined compressive strength (UCS), maximum dry unit weight, and California Bearing Ratio (CBR) by 100%, 6.9%, and 154%, respectively. Similar benefits were observed in lateritic soils and expansive soils, where SS additions improved workability and reduced swelling potential (Jaja & Tamunoemi, 2022; Kabeta & Lemma, 2023).

Optimal SS content varies by soil type but generally ranges between 8% to 25%. Above this range, diminishing returns and increased leaching risks of heavy metals become concerns (Navarro et al., 2010; Piatak et al., 2019). Several types of steel slag exist such as basic oxygen furnace (BOF) slag, Electric Arc Furnace (EAF) slag, Ladle Furnace Slag (LFS), and others. BOF slag, the most common type, is produced during oxygen injection to remove impu-

rities (Chen et al., 2023). Also, EAF slag is formed during electric arc melting of scrap steel (Skaf et al., 2017), while LFS is produced during refining stages in a ladle after BOF or EAF (Najm et al., 2021) and has similar composition to BOF slag but finer particles. These various steel slag types differ in both functional and leaching properties, and the most problematic substances with respect to leaching and environmental impact are generally Barium (Ba), Vanadium (V), Molybdenum (Mo), Chromium (Cr), and Fluorine (F) (Riley & Mayes, 2015; Spanka et al., 2017). The leaching properties may change with time after application due to changes in pH (carbonation of the generally alkaline slags) and changes in redox potential (Van Zomeren et al., 2011). The potential of SS to undergo Alkali-Silica Reaction (ASR) with silica-rich soils, leading to expansion and cracking is also a concern (Choi & Yang, 2020). Before applying soil stabilized with SS, identifying and addressing risks of harmful substances leaching or dust escaping to safeguard the environment and health is a necessary step to take. In conclusion, while SS offers a cost-effective solution for improving soil properties and mitigating environmental impact from steel production (Akinwumi, 2014), careful consideration of its application and potential risks is crucial to ensure beneficial use and effective soil stabilization practices.

3.5 Rice husk ash (RHA)

Rice husks, a byproduct of paddy dehusking, vary in weight from 20% (Mehta, 1986) to 23% (Della et al., 2002) of the total paddy weight. Traditionally regarded as waste, they are often incinerated in boilers used for processing paddy, with about 20% of their weight converting to ash (Mehta, 1986). Rice husk ash (RHA) predominantly consists of silica, its amorphous nature influenced by burning methods (Nair et al., 2006), making it a pozzolanic material (Mehta, 1986). Annually, India alone generates over 4 million tons of RHA from its 100-million-ton paddy production (Kumar & Gupta, 2016), while China produces approximately 40 million tons (Yan et al., 2022), and Nigeria contributes an estimated 2 million tons (Oyetola & Abdullahi, 2006). As rice milling is ongoing, the accumulation of rice husks in the environment continues to rise, prompting efforts to find beneficial solutions like soil stabilization.

Using RHA for soil enhancement can lead to cost-effective construction methods and environmentally friendly waste disposal, reducing cement use, energy consumption, and greenhouse gas emissions (Kumar & Gupta, 2016). Due to its pozzolanic properties, RHA can effectively stabilize soil as an alternative to conventional binders like lime and cement (Basha et al., 2005). Studies have shown that incorporating RHA can enhance soil strength, as observed in various soil types such as A-7-6 lateritic soil (Alhassan, 2008; Paul & Sarkar, 2023) and black cotton soil (Ghutke et al., 2018). Optimal results typically occur at specific RHA concentrations, such as 6-8% for lateritic soil (Alhassan, 2008; Okafor, 2009) and 12% for black cotton soil (Ghutke et al., 2018), underscoring the importance of precise mix proportions.

While RHA offers advantages in terms of cost-effectiveness and environmental benefits (Behak, 2017; Pushpakumara & Mendis, 2022; Reis et al., 2022), its efficacy as

a stabilizer can vary due to factors like inconsistent burning temperatures during production (Tuhin et al., 2020). High alkalinity in RHA can also pose risks of element leaching upon contact with water (Reis et al., 2022). Assessing and mitigating potential environmental and health risks from harmful substances in soil stabilized with RHA before use should be considered to prevent adverse effects from leaching or dust. In conclusion, RHA presents a promising option for soil stabilization, offering both technical and environmental advantages, provided proper production and application protocols are observed.

3.6 Coal fly ash (CFA)

Geotechnical and transportation engineers are keen on improving materials for highway pavement layers, as deficient pavement layers significantly raise construction costs. Stabilizing the base, sub-base, and subgrade layers is crucial when naturally occurring materials are inadequate. Coal fly ash (CFA), a byproduct of coal combustion in thermal power plants, is increasingly used in soil stabilization due to its low cost, regional availability, and environmental benefits.

CFA, known for its pozzolanic properties, enhances the qualities of pavement layers by reducing the swell potential of highly plastic clays and improving the strength and compressibility of weak soils. Studies have shown that adding CFA to soils can significantly improve their engineering properties. For example, CFA addition to black cotton soil reduced Atterberg limits and increased CBR and UCS values. Optimal CFA content for various soils varies, with 10-20% often recommended (Gireesh Kumar & Harika, 2021; Andavan & Hassaan, 2018). The use of CFA in soil stabilization not only improves pavement performance but also addresses environmental concerns by reducing landfill waste from coal-fired power plants. It provides a cost-effective and sustainable solution for enhancing weak pavement layer materials (Renjith et al., 2021).

However, CFA properties can vary depending on the source coal, affecting its stabilization effectiveness. Additionally, some CFA contains trace heavy metals, posing environmental and health risks if not properly managed (Wang et al., 2022). For alkaline CFA, the major potentially critical substances with respect to leaching and environmental impact are the oxyanions formed by chromium (Cr), arsenic (As), selenium (Se), vanadium (V), and molybdenum (Mo) (Cornelis et al., 2008), while for acidic CFA metals like cadmium (Cd), zinc (Zn), and copper (Cu) are of greater relevance (Liu et al., 2005). Evaluating environmental and health risks from leaching or dust in CFA-stabilized soil and implementing necessary precautions before application to avoid harmful impacts is key. In conclusion, CFA is a valuable tool for highway and geotechnical engineers, offering benefits such as improved soil strength and stability, beneficial waste management, and reduced construction costs. With ongoing research and optimization, CFA's role in sustainable pavement construction is likely to grow.

3.7 Granite dust (GD)

Granite dust (GD) results from cutting and grinding granite, forming colloidal waste when mixed with water

and becoming a non-biodegradable dry pile in landfills (Singh et al., 2016; Taji et al., 2019). Industrial processes on granite yield about 65% waste, leading to environmental concerns like water pollution, air contamination, and land infertility (Singh et al., 2016; Danish et al., 2021). To mitigate these issues, GD has been increasingly used as a supplementary cementitious material (SCM), replacing cement to reduce environmental impact, CO₂ emissions, energy consumption, and resource depletion (Arel, 2016; Danish et al., 2019). Researchers have shown interest in using GD to create cementitious materials, benefiting the construction industry by reducing disposal costs, promoting beneficial use, and enhancing composite properties (Li et al., 2019; Aydin & Arel, 2019; Sadek et al., 2016).

GD is also useful in soil stabilization. Eltwati et al. (2020) investigated GD's impact on clayey soil, finding that adding 8% GD significantly improved soil density, CBR, and shear strength, enhancing engineering properties to be about 2.8 times better than untreated soil. Babu and Nagaraju (2022) observed similar results with black cotton soil, noting GD's cost-effectiveness and environmental benefits. Amulya et al. (2021) found GD to be a low-carbon-intensity component for geotechnical and concrete activities; reducing CO₂ emissions and improving soil properties through enhanced inter-particle bonding and new mineral formation (Abdelkader et al., 2022).

Despite its benefits, GD's effectiveness depends on the optimal content and soil type (Eltwati et al., 2020). Thorough laboratory tests are essential to determine the best GD content for specific soils, ensuring effective stabilization. Proper gradation of GD particles can improve the packing density and strength of stabilized soil. Additionally, conducting a risk assessment for harmful substance release from soil stabilized with GD is necessary. It is also important to ensure preventive measures are in place to mitigate environmental and health impacts before use. In summary, GD, once a waste product, now offers sustainable solutions in construction and soil stabilization, supporting a more economically and ecologically beneficial future.

3.8 Plantain peel ash (PPA)

The global agro-industry generates significant agricultural waste annually, causing environmental issues (Vilamizar et al., 2012). Experts have explored alternatives like soil stabilization to limit waste, with plantain peels gaining attention, especially in developing nations where they are abundant. Nigeria, a top plantain producer, consumes most of its yield domestically, leading to considerable waste (Amaya, 2018; Usman et al., 2018). The disposal of plantain peels produces harmful gases like ammonia (NH₃) and hydrogen sulfide (H₂S) (Ifetayo & Pretorius, 2018).

Researchers have investigated using plantain peel ash (PPA) for soil stabilization. Ishola et al. (2019) studied its impact on tropical red soil strength. Plantain peels were dried, burned at 500°C, and sieved to produce ash. Tests showed varying specific gravity, reduced plasticity index, liquid and plastic limits, and moisture content, with increased maximum dry unit weight. Unconfined compressive strength (UCS) peaked at 10% PPA, and California Bearing Ratio (CBR) improved at 4% PPA. PPA, combined

with lime or cement, was suggested for road pavement layers to reduce construction costs. Akinwumi et al. (2023) found PPA-treated soil suitable for rural road embankments and fills due to pozzolanic reactions forming calcium silicate hydrate (C-S-H), which densifies the soil matrix. This process causes soil particles to agglomerate, enhancing strength over time. PPA is seen as a beneficial, cost-effective soil stabilizer, particularly in developing nations with high plantain production.

However, PPA might not always meet high-strength requirements for all applications, and its effectiveness depends on soil type and ash amount (Ishola et al., 2019). There is also a risk of heavy metals leaching (Olabanji et al., 2012). Prior to using PPA-stabilized soil, assessing and mitigating risks of harmful substance release through leaching or dust to prevent adverse environmental and health effects are necessary steps to take.

3.9 Calcium carbide residue (CCR)

Certain waste materials rich in Ca(OH)_2 , such as calcium carbide residue (CCR), a byproduct of acetylene manufacturing, can be repurposed to create cementitious materials, benefiting the environment and economy (Horpibulsuk et al., 2013). CCR, primarily a slurry with high alkalinity and water content, often pollutes water sources and contributes to environmental degradation when stockpiled (Krammart & Tangtermsirikul, 2004; Sharma & Reddy, 2004; Du et al., 2011). China alone produces around 28 million tons of CCR annually, most of which ends up in landfills (Li et al., 2024).

A low-carbon, energy-efficient way to mitigate CCR stockpiles is to use it for soil stabilization, which also alleviates environmental concerns (Horpibulsuk et al., 2012). CCR has been effective in pavement and geotechnical applications. Field tests by Du et al. (2016) demonstrated that CCR is a cost-effective, environmentally friendly binder for stabilizing soft subgrade soils, with significant engineering, economic, and environmental benefits. Chindapasirt et al. (2020) confirmed that CCR-stabilized laterite's engineering properties improve due to pozzolanic reactions. Higher CCR concentrations enhance soil strength and durability, making it a viable alternative to traditional stabilizers like lime and cement (Ayodele et al., 2020). Research shows a direct proportionality between UCS, CBR, and CCR contents, enhancing soil workability and strength (Akinwumi et al., 2018). An optimal 4% CCR application is recommended for stabilizing sand in road construction, offering a low-cost, eco-friendly method, especially for developing countries. Using CCR reduces pollution from stockpiled waste, lowers construction costs, and produces strong, durable soil for various construction uses.

However, CCR usage has potential drawbacks. Strength may decrease after repeated wetting and drying cycles, necessitating further durability studies (Julphunthong et al., 2024). CCR may also contain trace heavy metals, risking soil and groundwater contamination if not properly managed (Hassan et al., 2019). Combining CCR with other pozzolanic stabilizers can further improve long-term strength. Furthermore, ensuring a risk assessment for environmental and health impacts from harmful substance release in

soil stabilized with CCR is important, as well as applying necessary precautions to avoid unacceptable outcomes. In summary, CCR offers a sustainable solution for waste management while enhancing soil properties, providing a viable and cost-effective alternative to traditional soil stabilizers.

3.10 Bagasse ash (BA)

Bagasse, the fibrous residue from crushed sugarcane, and its ash (BA), produced when bagasse is burned, pose environmental risks if improperly disposed of. Inhaling bagasse dust can lead to "bagassosis," a rare but serious respiratory condition (Choudhari et al., 2020). Proper disposal and utilization of BA are crucial, particularly in engineering applications.

With high silica content, BA is a valuable pozzolanic material suitable for stabilizing road subgrades (Hasan et al., 2016). It offers an eco-friendly and cost-effective alternative for soil improvement, essential given the significant sugarcane production in countries like Brazil, India, China, and Thailand (Shahbandeh, 2024). The use of BA in construction can support sustainable development by recycling industrial waste into building materials (Dang et al., 2021). Research indicates BA's effectiveness in stabilizing wet, unstable soils, reducing soil swell, and mitigating settlement issues (Khandelwal et al., 2023; Gandhi, 2012; Srinivasa et al., 2017). It also improves low to high plastic clays and can stabilize expansive soils and peat, supporting road embankments in agricultural or rural areas (Khan, 2019; Surjandari et al., 2017; Abu Talib & Noriyuki, 2017).

While BA is cost-effective and improves soil strength, it may not match cement's strength for high-traffic applications (Osinubi et al., 2009). Variations in BA composition due to different sugarcane varieties and burning conditions necessitate project-specific testing to determine optimal dosages (Xu et al., 2018). Combining BA with other pozzolanic stabilizers and sourcing from reliable suppliers can enhance its performance (Awadalseed et al., 2023). Before using soil stabilized with BA, the risk of environmental and health impacts from harmful substances leaching or dust escaping should be assessed, and necessary precautions to prevent negative effects should be taken.

4. SUSTAINABILITY IN UTILIZING ALTERNATIVE STABILIZERS

It is no surprise that numerous researchers have been looking into various waste materials aimed at boosting earth mineral content by originating avenues to incorporate them as environmentally suitable equivalent resources for developing and upholding roads (Oyebisi et al., 2020; Bamigboye et al., 2021). Studies like these verify that utilizing such wastes, notably for improving soil, assists in transforming wastes in a beneficial manner (Ewa et al., 2022). Careful attention must also be given when utilizing these wastes for soil stabilization as their usage may be characterized by some potential drawbacks as summarized in Table 1. Ultimately, it is necessary to ascertain the geotechnical properties (strength, leaching, workability, etc.)

of the soil and stabilizer combination and perform relevant risk assessments. This would help to determine the performance of these wastes as stabilizers and also to know what aspects would require mitigation and/or precautions to maximize the potential of these waste materials to be used for good, instead of causing more harm to the ecosystem. This points to the fact that prior to the application of a specific soil stabilised with a specific waste material for a specific purpose, the potential risk of local environmental and health impacts from release and spreading of harmful substances due to leaching or escape of fugitive dust must be assessed and appropriate action (precautions, mitigating measures, changes of the soil/stabilizer mixture) must be taken when necessary to prevent unacceptable impacts.

As a criterion for choosing stabilizers for soil stabilization, sustainability must be the principal focus. One of the major aims of seeking for alternative soil stabilizers in contrast to the traditional method used (cement-soil and lime-soil stabilization) is to protect the environment from the environmental hazard associated with these conventional methods; as CO₂ emission is characterized with their production (Tesanasin et al., 2022). CO₂, a greenhouse gas, is characterized by its increasing concentration in the atmosphere which contributes to increasing temperatures on Earth. This is a source of concern in our ecosystem. Thus, selecting alternative soil stabilizers should be of beneficial use in terms of cost-effectiveness and environmental safety. Stabilization using waste ash and waste slag is an economical way of reducing these pollutants and inducing considerable soil strength. However, on the long run, they may induce weak shear planes in the soil when applied beyond optimal points, thus impeding further strength gain (Kassa et al., 2020). This can be a major problem during freeze-thaw cycles and increased traffic loads which can adversely affect the road pavement. On the other hand, waste materials like GD, CCR and RCA have been known to produce better results compared to the aforementioned, which may be attributed to having some cementitious properties comparable to traditional stabilizers (Danish et al., 2019; Gencel et al., 2020). They yield appreciable strength for sub-grade while providing sustainable disposal strategies for waste mitigation. Therefore, waste materials to be used for soil stabilization should be chosen based on the criteria that its usage would not pose any danger to the environment; and that its financial feasibility and has potentially beneficial use. Also, when utilizing waste ash for soil stabilization, careful attention should be given to the combustion process. To optimize beneficial use, the combustion process producing the ash should take place under properly controlled conditions, including cleaning of the stack gases and proper management of quench water so as to mitigate emissions of chemicals that are harmful to human health and the environment and also to regulate the entire combustion system for process efficiency and control.

Mekonnen et al. (2020) concluded that conventional stabilizers, like cement/lime, are costly; In some places, the cost can even triple that of alternative additives, and this cost increases even further when the bulky materials need

to be transported over lengthy distances to low-volume road construction destinations. Ultimately, utilizing waste materials for improving the qualities of road pavement layer materials reduces construction cost since the waste materials are often readily available and far cheaper, or even free, compared to conventional stabilizers like cement or lime. This translates to direct savings on material procurement. Using waste materials for soil improvement means that they are essentially being diverted from landfills. This eliminates tipping fees associated with waste disposal, leading to additional cost savings. Also, using these waste materials as alternative soil stabilizers contributes to several Sustainable Development Goals (SDGs) by the United Nations (2015), as seen in Figure 2. It supports SDG 6 – Clean Water and Sanitation. Improper waste disposal can contaminate water sources. By diverting waste from landfills and putting it to good use, we can help protect water quality. In essence, using waste as a soil stabilizer offers a potentially beneficial and environmentally friendly approach to construction compared to traditional methods. Furthermore, utilizing these alternative materials encourages SDG 11 – Sustainable Cities and Communities. This goal aims for sustainable development in cities and communities. Reusing waste materials for construction purposes reduces reliance on virgin materials and minimizes landfill waste, contributing to a more sustainable built environment. It also fosters SDG 12 – Responsible Consumption and Production. This goal promotes responsible production and consumption patterns. By finding beneficial uses for waste products, we lessen the environmental impact of production processes and reduce the overall amount of waste generated. SDG 15 – Life on Land is also promoted. This objective is to safeguard, replenish, and encourage beneficial use of our terrestrial ecosystems. Stabilizing weak soils with waste materials can help prevent erosion and improve land quality, promoting healthy ecosystems.

Utilizing waste materials for soil stabilization presents a compelling solution. It not only offers a cost-effective alternative to traditional methods stemming from local production and minimal transport, but also promotes environmental sustainability by reducing CO₂ emissions, diverting waste from landfills, and contributing to several UN Sustainable Development Goals. It is also important to note that the degree at which the release of potentially harmful substances to the environment will be reduced as regards the utilisation of a waste material as a soil stabiliser depends on the quality and operation of the landfill and the leaching properties of the stabilised soil as well as on the conditions of use, i.e. the utilization scenario (and the climatic conditions). Before using waste-stabilized soil, assessing and mitigating risks of harmful substance release through leaching or dust to prevent adverse environmental and health effects are key steps to take to ensure optimal usage. As we move forward, continued research and development in this area can further optimize waste-based soil stabilization techniques, ensuring strong, potentially beneficial, and eco-friendly construction practices.

TABLE 1: The advantages and potential drawbacks of utilizing waste materials for soil stabilization.

Waste Materials	Advantages in stabilizing soil	Drawbacks
Reclaimed asphalt pavement (RAP)	Reduces demand for virgin aggregates, lowers energy consumption during production of new materials and diverts waste from landfills (Ochepo, 2014; Milad et al., 2020).	“Old” asphalt based on tar has a much higher content of potentially hazardous polycyclic aromatic hydrocarbons (PAH) than “newer” asphalt based on bitumen (IARC, 2010). RAP composition can vary depending on the source and age of the asphalt pavement, requiring careful testing and mix design optimization for consistent performance (Edeh et al., 2011). RAP may contain trace amounts of heavy metals that could leach into the surrounding environment if not properly managed (Legret et al., 2005).
Recycled concrete aggregated (RCA)	Strength enhancing qualities, cost savings in construction projects, and resource conservation (Tavakol et al., 2019; Datta & Mofiz, 2021).	RCA properties can vary depending on the source concrete’s composition and processing methods. This inconsistency requires careful testing and mix design optimization (Datta & Mofiz, 2021). The angular shape and rough texture of RCA can make mixtures less workable compared to smooth, rounded natural aggregates (Shourijeh et al., 2022). Heavy metals from the original concrete e.g. chromium, lead, arsenic etc., can leach into the soil under certain conditions (Galvin et al., 2013).
Ground granulated blast-furnace slag (GGFS)	Improve various engineering properties of soil in a cost effective and environmentally friendly way (Pathak et al., 2014).	The effectiveness of GGFS can vary depending on its chemical composition, which can influence its reaction with soil components (Al-khafaji et al., 2017; Sharma & Sivapullaiah, 2016; Rizki Abdila et al., 2020). There’s a risk of some elements in GGFS leaching into the surrounding environment (Müllauer et al., 2015).
Steel slag (SS)	Proffers a cost effective and sustainable way of improving soil properties (Akinwumi, 2014).	Steel slag can potentially leach heavy metals like barium, vanadium, molybdenum, chromium, and fluorine into the surrounding environment if not properly characterized and managed (Riley & Mayes, 2015; Spanka et al., 2017). Steel slag with high alkalinity can react with silica-rich soils leading to expansion and cracking within the stabilized soil (Choi & Yang, 2020).
Rice husk ash (RHA)	Cost-effective, environmentally friendly and ability to boost soil strength (Behak, 2017; Pushpakumara & Mendis, 2022; Reis et al., 2022).	Inconsistent burning temperatures can affect its effectiveness as a stabilizer (Tuhin et al., 2020). RHA with high alkalinity can lead to leaching of certain elements upon exposure to water (Reis et al., 2022).
Coal fly ash (CFA)	Offers several advantages such as improving strength and stability of soil, sustainable waste management and reduction of construction costs (Ahmad et al., 2024)	CFA properties can vary depending on the source coal, which can affect its effectiveness in stabilization (Bhatt et al., 2019). Some CFA contains trace amounts of heavy metals. For alkaline CFA, the major potentially critical substances with respect to leaching and environmental impact are the oxyanions formed by chromium, arsenic, selenium, vanadium and molybdenum (Cornelis et al., 2008), while for acidic CFA metals like cadmium, zinc, and copper are of greater relevance (Liu et al., 2005). Improper use can lead to leaching into groundwater, posing environmental and health risks (Wang et al., 2022).
Granite dust (GD)	Strength inducing capabilities in soil, cost effective and eco-friendly (Zorluer & Gücek, 2017; Babu & Nagaraju, 2022).	The optimum content of GD is crucial as this significantly impacts the results. The effectiveness of GD can vary depending on the soil type (Eltwati et al., 2020).
Plantain peel ash (PPA)	Moderately improves soil strength.	It might not always meet the high-strength requirements for all construction applications. The effectiveness of PPA is dependent on the specific soil type and the amount of ash used (Ishola et al., 2019). There is also a likelihood of trace amounts of heavy metals leaching into the soil from PPA (Olabanji et al., 2012).
Calcium carbide residue (CCR)	Offers a sustainable solution for managing waste while improving the qualities of soil.	While CCR initially improves strength, there is a likelihood of a potential decrease in strength after repeated wetting and drying cycles (Julphunthong et al., 2024). CCR can also contain trace amounts of heavy metals (Hassan et al., 2019), raising concerns about potential soil and groundwater contamination if not properly managed.
Bagasse ash BA)	Offers a cheaper alternative to conventional stabilizers in improving workability of soil.	While BA improves soil strength, it might not achieve the same level of strength as cement, particularly for high-traffic applications (Osinubi et al., 2009). The composition of BA can vary depending on sugarcane variety, burning conditions, and other factors (Xu et al., 2018).

5. CONCLUSIONS AND RECOMMENDATIONS

An overview of sustainable soil stabilization techniques for road pavement layers using waste materials is presented in this study. It highlights the growing concern of the global emergence of wastes and its detrimental impact on

health and the ecosystem. Through examination of various waste materials (reclaimed asphalt pavement, recycled concrete aggregates, ground granulated blast-furnace slag, steel slag, rice husk ash, coal fly ash, granite dust, plantain peel ash, calcium carbide residue and bagasse ash), this review highlights the potential of these materials in enhancing soil properties for road construction while



FIGURE 2: SDGs promoted through the utilization of wastes as soil stabilizers.

mitigating environmental impacts. Since traditional stabilizers like cement and lime are linked to significant carbon emissions and environmental risks, the study highlights the significance of finding beneficial substitutes. Not only can constructing expenses be minimized but landfill contamination can be prevented by using waste materials as stabilizers. Utilizing waste materials also supports the Sustainable Development Goals, which include protecting life on land, fostering sustainable cities and communities, clean water and sanitation, and responsible consumption and production. Additionally, the need for careful consideration in selecting stabilizers to ensure beneficial use in terms of cost-effectiveness and environmental safety remains key for sustainable waste management. While waste materials offer promising solutions, their optimal usage should be determined to prevent potential drawbacks such as weak shear planes and impeded strength gain. Furthermore, continued research and development in waste-based soil stabilization techniques are essential to optimize their effectiveness and promote eco-friendly construction practices. Also, conducting field trials and case studies to validate the effectiveness of waste-based soil stabilization techniques in real-world construction projects is recommended. These trials can provide valuable insights into the practical challenges and benefits associated with using waste materials for road pavement layers. In general, the stabilization potential of more waste materials should also be investigated. An in-depth investigation of these wastes having cementitious properties should be looked into on how they can be used conjointly to stabilize soils of different kinds so as to rid the environment of these waste pollution in a cost-effective and potentially beneficial manner. These wastes ultimately improve strength of the target soil; how-

ever, careful consideration should be given to determine the performance of these wastes as stabilizers. The geotechnical properties (strength, leaching, workability, etc.) of the soil and stabilizer combination as well as relevant risk assessments should be conducted to know what aspects would require mitigation to maximize the potential of these waste materials to be used for good, instead of causing more harm to the ecosystem. This simply suggests that prior to the application of a specific soil stabilised with a specific waste material for a specific purpose, the potential risk of local environmental and health impacts from release and spreading of harmful substances due to leaching or escape of fugitive dust must be assessed and appropriate action (precautions, mitigating measures, changes of the soil/stabilizer mixture) must be taken when necessary to prevent unacceptable impacts.

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ECONOMIC INSTRUMENTS FOR THE CIRCULAR ECONOMY TRANSITION: THE (NEGLECTED) ROLE OF VIRGIN MATERIALS AND LANDFILL TAXATION IN ITALY

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ABSTRACT

The use of economic instruments in policies supporting a circular economy has attracted an extensive and growing interest in the international and European debate in recent decades. Environmental taxes and levies, subsidies and other market-based instruments have been seen as useful policy tools for enhancing environmental protection, getting the price right and creating market-based incentives – price and cost signals – to spur environmentally-friendly behaviors. From this standpoint, taxes and other economic instruments acquire, primarily or at least partially, a regulatory and preventive role: they address the market failure of environmental externalities, aligning private and social costs and, consequently, stimulating more socially desirable outcomes. The paper mainly focuses on the sector of buildings and construction, i.e. the sector responsible for the largest share of extraction of raw material and of the generation of non-municipal wastes, presenting an analysis of the two main price instruments implemented in Italy at the beginning and at the end of the flow of materials. It aims to bring to the fore the main characteristics of the two instruments in the Italian multilevel government, and to propound new paradigms for future developments. The value added of the work is to prove, through a coherent and updated policy framework, the extent to which the regulatory role of these subnational environmental levies is largely forsaken, thereby jeopardizing the possibility to use them in a more effective and consistent way.

1. INTRODUCTION

Policies to reduce raw material consumption constitute a salient concern in the sustainable development policy agenda of several countries. The extraction, processing, and disposal of raw materials, in fact, give rise to numerous negative environmental impacts, including greenhouse gas emissions and air pollution, noise hazards, loss of ecosystem services and landscape alteration (Auci et al., 2012; OECD, 2024; Soderholm, 2011). Furthermore, in an intertemporal perspective, the difficulty in establishing property rights and/or the implementation of too high discount rates may imply that resources are exploited at a faster rate than it is socially efficient (McKenzie, 2006; Soderholm, 2011). These consequences appeal towards the needs for a circular economy, in which the flow of virgin materials is narrowed, and the value retention of material resources is preserved, hence fostering resource efficiency and contributing to the reduction of environmental externalities.

Italy's performance in this field presents mixed results and possible favourable developments for the future.

On the one hand, Italy ranks highly for many indicators related to resource productivity and circularity (Circular Economy Network, 2022). For example, the country's material productivity of 4.6 USD/kg is well above the EU and OECD average, while domestic material consumption (DMC¹), at 8.6 tonnes per capita, is one of the lowest in Europe (OECD, 2024). Non-metallic minerals account for the largest part (68.2%) of domestic material extraction, followed by biomass (29.7%) and fossil materials (2.1%) while the percentage of the extraction of metal ores is negligible (ISTAT, 2024). The net import of materials is largely positive for biomass and fossils, while for aggregates the overall effect of trans-border flows is substantially balanced. On the other hand, the country has yet to decouple resource extraction and waste generation definitively from economic activity, especially when non-municipal waste is concerned, thus revealing a percentage of waste disposed into landfill still higher than that found in EU best practices. Construction and demolition (C&D) waste is a key component of the non-municipal (special) waste stream: in 2021 its total production has been estimated at 78.7 mil.



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tonnes, corresponding to nearly 50% of the total amount of special waste produced in Italy (ISPRA, 2023b). 77% are wastes produced by the building sector (demolition and construction phases), while 23% are wastes and scraps produced in the virgin materials extraction phase. No absolute nor relative decoupling from economic growth (GDP) has taken place in the last decade: the total amount of C&D waste, in fact, increased by 36.4% from 2010 to 2021, and its ratio to the sectoral GDP increased by 50.8% in the same period (Figure 1).

Moreover, even if the total rate of recycling/recovery of C&D waste (80.1% in 2021) has largely met the minimum objective (70%) set on the EU level by the Directive 2008/98 and the Decision 2011/753 for the year 2020, a low level of 'internal' circularity can still be observed (Fondazione per lo Sviluppo Sostenibile, 2023). As emphasised by the National Waste Management Plan, in fact, the bulk of C&D waste is mainly recovered in other low-value chains, because selection and treatment plants are technologically unsophisticated, and materials obtained often do not respect technical standards adopted on the European as well as on the national level (ibidem, 2023). The Italian construction sector indicates a clear preference for virgin materials and still maintains a largely sceptical attitude towards recycled materials, which are used mainly for secondary purposes (e.g. substrates, road asphalt concrete, filling, etc.) or prepared for recycling (EEA, 2008; Fondazione per lo Sviluppo Sostenibile, 2023; Legambiente, 2021). These trends are expected to continue to prevail and become of utmost importance due to measures and projects financed by the National Recovery and Resilience Plan (NRRP), as well as by the 'Ecobonus' and 'Superbonus' tax deductions² which are strongly boosting the overall sectoral demand.

Through the adoption of the National Circular Economy Strategy in June 2022, Italy has explicitly placed the circular economy transition and the construction sector at the centre of its environmental policy agenda, regarding this commitment also as a key lever in implementing the 2030 UN Sustainable Development Agenda. The Strategy identifies objectives, actions, and measures, hinging its

implementation upon a multi-level and multi-instrument approach. From one perspective, it requires the vertical and horizontal coordination amongst all the governments involved, with full recognition of the decisive role of sub-national governments in enacting the main contents of the Strategy. Nonetheless, it entails the combination of several kinds of instruments: regulatory instruments, economic instruments, voluntary approaches, information-based instruments, research-promoting measures, communication, and awareness-raising campaigns.

Under this framework, economic instruments (taxes and charges, subsidies and tax reliefs, tradeable quotas and compliance certificates, extended producer responsibility, green public procurement) may represent important tools to align existing incentives and behaviours with environmental goals and objectives. Correct price signals, achieved via the internalisation of costs associated with activities that contribute to environmental harm, foster practices that are more sustainable, maintain resources in circulation and penalise those that are detrimental to the environment.

Economic policy instruments can act at different stages of the value chain, discouraging the extraction of raw materials, improving product design in line with principles of circularity, supporting the use of recycled materials, influencing consumer behaviour in sales and during product utilisation, modifying waste disposal practices and, at the end of the circle, incentivising recycling by operators.

The current Italian policy framework already includes several economic and quasi-economic instruments that may stimulate the transition to a circular economy (Table 1). They represent relevant opportunities to induce behavioural changes in a flexible way, avoiding excessive economic impacts and providing a continuous incentive for innovation and for the improvement of environmental performance above the regulatory standards. At the same time, many technical and political obstacles can hinder their implementation; they therefore necessitate well-designed execution strategies and implementation paths.

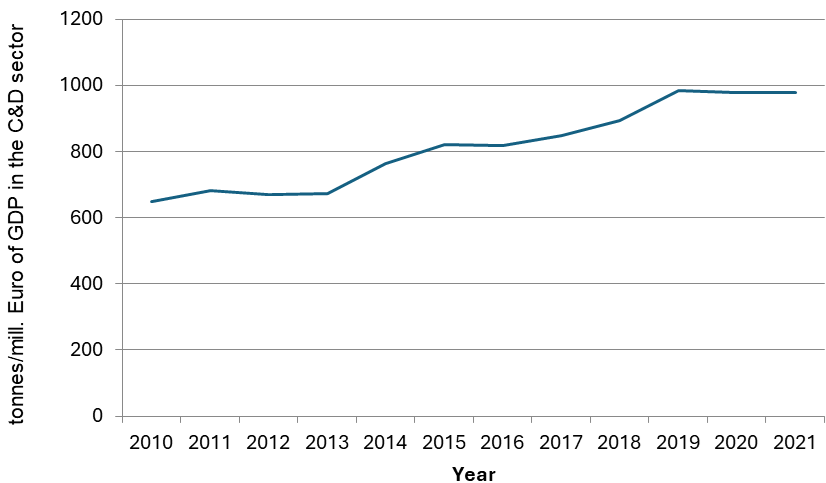


FIGURE 1: C&D wastes and Gross domestic product (GDP) in the construction sector (Italy, 2010-2021) - Source: our calculations on data from ISPRA (2023b).

TABLE 1: Potentially hazardous components of HHSW - Source: elaborations on OECD (2024).

Instrument	National level	Sub-national level
Explicit economic instruments		
Fees on extractive activities	National legislation only covers general matters in extractive sectors (e.g. environmental standards, workers' health and safety, disposal of waste from extractive activities) .	Regions detain main competencies on quarrying (barring energy-related minerals), including the planning and authorisation of extractive activities, environmental impact assessments, regulation of fee coverage and rates, recovery of waste materials, mine/quarry closure and rehabilitation requirements, and the collection of statistical data. Municipalities: collection and use of revenues (exceptionally by Regions).
Landfill tax	National legislation: general criteria and guidelines on tax scope (tax rate upper and minimum levels), destination and earmarking of revenues, bonus/malus mechanism for separated collection.	Regions hold main competencies, including tax scope and tax rate (including reduced tax rates), collection of revenues (e.g. exact share going to municipalities, including with bonus/malus systems), use of revenues. Municipalities: may receive a share of revenues.
Plastic tax	National legislation (paused)	NA
Waste charge (including Pay as you throw - PAYT)	National legislation: guidelines for fee setting. ARERA is the regulatory body.	Regions: promotion or mandating of adoption of PAYT systems. Municipalities: in charge of planning for local waste management (incl. the financing model), fee setting (including granting tax reductions in line with national guidelines) and collection of revenue.
Quasi-economic instruments		
Extended Producer Responsibility (EPR)	State: introduction of EPR schemes, certification of Producer Responsibility Organisations/Consortia (PROs). In most cases, PROs organise and execute EPR obligations nationally. PROs charge Advanced disposal fees at the point of sale.	Municipalities are responsible of waste management and obtain payments from PROs for the different waste flows collected, also according to their quality.
Green Public Procurement (GPP)	National legislation: introduction of public procurement approaches, and monitoring of application of GPP criteria. MASE (Ministry of Environment and Energy Security): definition and regular update of GPP criteria, as well as of the National Action Plan on Green Public Procurement. Consp: National Central Purchasing Body implementing framework agreements including GPP criteria. National Anti-Corruption Authority (ANAC): monitoring of application of GPP criteria.	Regional aggregators implementing framework agreements including GPP criteria and trainings on GPP. All sub-national public entities: implementation (uptake of GPP criteria in tenders). Regions participate in the planning phase.

In what follows, after a brief outline of the vertical distribution of environmental and tax competencies, we will focus on two price instruments implemented in Italy at the beginning and at the end of the material stream - i.e. the aggregates levy and the landfill tax - with the aim of emphasising the main characteristics of the two instruments and suggesting some prospective developments. Even if single topics have been already scrutinized and discussed, at least partially, in other works, this is the first time, to our knowledge, that all these elements are critically reviewed for the Italian experience in a coherent and updated framework.

2. ENVIRONMENTAL AND TAX COMPETENCES IN THE ITALIAN MULTILEVEL SYSTEM

Although environmental protection is listed as among the exclusive competences of the national government (art. 117, 2 co, Lett. S, Constitution), the Italian Constitutional Court has repeatedly affirmed that the environment is a 'horizontal value', for the safeguard/realisation of which the State and the territorial governments take part with shared responsibilities. In this regard, the national

competence does not exclude the possibility for Regions to pursue and enhance environmental protection and promotion, even by dint of the enactment of competences that are intrinsically related to the environment such as health, transport, energy, territorial and land management, etc.

This complex set of connected and potentially overlapping responsibilities is governed by the principle of prevalence, according to which national rules prevail over regional legal acts when they are aimed towards environmental protection. The principle has been further developed in two directions of interest for the purpose of this analysis.

Firstly, the Constitutional Court has stated in several cases ³ that interventions regarding environmental protection, made by Regions employing their 'legislative' prerogatives ⁴, must be carried out in compliance with the minimum levels of environmental protection set by the central government and, potentially, only increasing them (unidirectionality principle). For example, in the Judgement 86/2014, the Court has stated that 'the only direction allowed by the legal system to the regional legislator is that of raising the level of environmental protection established by the State'.

At the same time, the Court has acknowledged the possibility for the state to define also maximum standards and

fully binding rules for regional governments to find an equilibrium amongst possibly conflicting interests (e.g. local interests as opposed to national energy security). This general statement turns out to be particularly relevant when sites and environmental issues have strong interjurisdictional and cross-border implications, to avoid that stringent local limits and/or disproportionately high environmental taxes set at the decentralized level become tools in the service of the NIMBY syndrome (Zatti, 2023).

According to the discipline of fiscal federalism adopted in Italy, Regions can introduce new taxes for their own or for local governments but can only do so respecting the general principles set by the national government in terms of fiscal coordination. In particular, they are allowed to introduce their own taxes in their territories only if they do not duplicate state taxes (pre-emption principle), and regarding matters that are not under the exclusive competence of the State. Accordingly, there are different opportunities for subnational governments to influence the management of environmental matters through fiscal disincentives (Verigni, 2013; Zatti, 2020):

- Regions will be allowed to introduce autonomous new taxes on matters indirectly related to the environment (tourism, energy, land management) that are devolved to the shared legislative competence between the central State and Regions, or to the exclusive competence of the Regions, but only if the link with this matter prevails over the environmental one (prevalence principle).
- Regions and local governments have the power to discipline pre-existing environmentally related taxes introduced by the State, within the range set by the national law. The more or less detailed prescriptions included in the laws directly influence the degree of autonomy conceded to the local government.
- Regions and local government have the power to establish and fix non-tax levies, including user charges and special fees, even in the absence of an explicit national rule. For the management and design of the main characteristics of these hybrid instruments (rates, rebates, exemptions, etc.), it is in fact widely recognised that a larger room of manoeuvre for local governments is generally granted.

All things considered, there are several reasons to enhance the role of subnational governments in environmental management, even acting through the disincentive effects of fiscal instruments. However, their design and administration require a well-coordinated approach to valorise synergies and co-benefits and prevent overlaps and strategic behaviours. The cases of the extractive activities' levy and the landfill tax show how relevant steps are still to be done in this direction.

3. FEES ON EXTRACTIVE ACTIVITIES

A tax on natural resources extraction can be implemented for several reasons (EEA, 2008; Ettlinger, 2016; HM Treasury, 2020; Soderholm, 2011; Zatti, 2023). It can slow down the rate of resource depletion in cases where property rights are not well defined and the intertemporal rate

of consumption underestimates the needs of future generations; it helps to reduce the environmental impact of quarries and pits by decreasing extraction of materials, while stimulating the market for using secondary sources, such as construction waste; it also provides financial resources to cover administrative costs, and to mitigate local environmental impacts and ensure site restoration. Positive aspects also include a limited influence on competitiveness, simple structure (for rates) with an easily identifiable tax base, and relatively little cross-border impact due to high transportation costs (EEA, 2008; Ettlinger, 2016; HM Treasury, 2020).

At the international level, according to the OECD database, there are currently 88 taxes applied on extractive activities on non-metallic minerals, 16% are applied on market value, whereas the remaining on weight or volume (OECD, 2023). Several European countries apply extraction taxes – i.e. taxes on gravel, rock, limestone, ornamental stones, etc. – to reduce both the reliance on virgin materials and to limit the amount of construction and demolition wastes landfilled (Soderholm, 2011). Often, revenues are earmarked for environmental or compensation purposes: this is the case for the UK 'Aggregate Levy', the proceeds of which are utilised for environmental rehabilitation and improvement, and for improving the quality of secondary materials (HM Treasury, 2020).

In cases where some form of policy evaluation exists (Ettlinger, 2016; Soderholm, 2011), they indicate that taxes have had effects on the use of aggregates and on material substitution. At the same time, evidence from international case studies suggests that virgin material taxes on construction minerals are most effective when they are combined with additional policies aimed at stimulating the demand for and the supply of recycled materials (e.g., landfill taxes, minimum recycled content requirements, support for selective demolition and improved quality standards for CDW, end-of-waste criteria for construction and demolition waste and classification of industrial by-products), and with more effective controls on waste management, especially on illegal disposal.

In Italy, internal resource extraction has considerably declined since 1990, but the trend has stabilized in the last decade (Figure 2). Among other sectors, the construction industry plays a crucial role, since it is one of the largest consumers of non-metallic raw materials and significantly contributes to waste generation (Auci et al., 2013; EEA, 2008; Mazzanti, Zoboli, 2013). Recently, the sector's output is experiencing a new growth and OECD projections indicate that both output and material intensity of the construction sector are expected to increase significantly by 2050 (OECD, 2024). In the same study it is reported that in 2021 184 Mt of materials were extracted from quarrying sites, a fifth more than in 2017. This baseline scenario raises concerns in terms of environmental externalities and security of supply, underscoring the need to focus on reducing waste generation, boosting recycling, and enhancing secondary raw materials use.

In Italy, mining activities⁵ have been progressively delegated to the regional competence with both the Presidential Decree N.616/77 and the Legislative Decree N. 112/98,

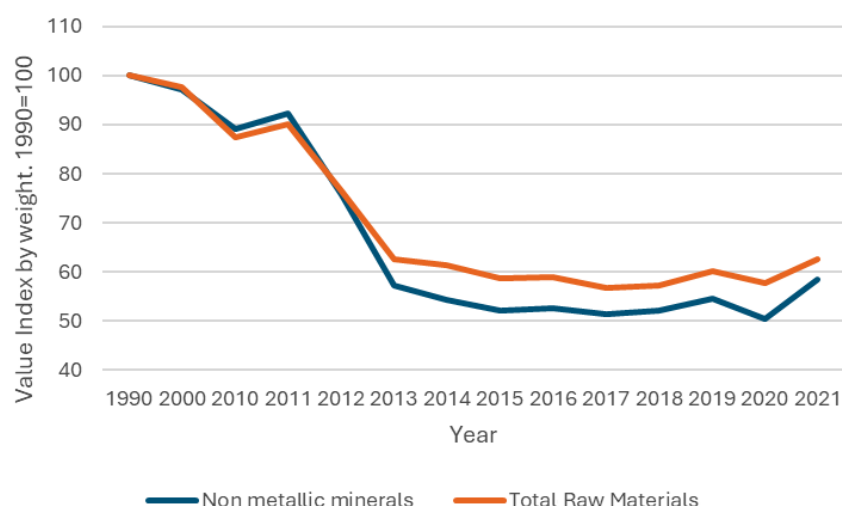


FIGURE 2: Internal raw material extraction in Italy (by weight, 1990=100) - Source: our calculations on data from ISTAT (2024a).

even if the State can still establish minimum standards according to its exclusive competence in the field of environmental protection and worker health and safety. Regions oversee the adoption of provincial plans, establishing the identification of exploitable sites, quantity, and quality of materials to be extracted, exploitation techniques, and the final management of the sites where the quarrying activities end up.

In this regulatory framework, four main actors are involved: the Region, the Province, the Municipality, and the site owner (EEA, 2008). Exploitation by site owners must be based upon an agreement between the owner and the Municipality in which the site is located and authorised by the Province. These agreements include details on quantity and quality of materials to be extracted, allowable exploitation techniques, and the commitment to carry out restoration of the site after the cessation of the activity.

Regional governments are in charge of defining concession (mainly authorization) fees for the extraction and to decide the allocation of revenues. According to national legislation (Leg. Decree 112/1998), maximum limits of concession fees should be set by the State, but these coordination measures have never been established, allowing Regions to benefit from a large degree of freedom. This is partially due to the hybrid nature of the levy, representing a borderline case between taxes and user charges, in that it shows clear connections with benefit and compensation principles, without fully relying on a cost-recovery approach (Zatti, 2023). This ambiguity is confirmed by the fact that the Italian Constitutional Court has recognised the non-tax nature of the aggregate fee (e.g. Judg. 89/2018), while a different approach is found in the OECD tax guide, according to which: 'any severance payments that are imposed on the extraction of minerals and fossil fuels from reserves owned privately or by another government should be classified as taxes' (OECD, 2021, p.9).

The overall implementation process shows a great heterogeneity and fragmentation of rates and procedural requirements (Auci et al., 2013; Zatti, 2023). Generally, the fees cover a wide range of aggregates (sand and gravel,

limestone, clay, peat, ornamental stones) and are set on metre cubic of materials extracted⁶; in some cases (in Apulia, Sicily and in the Autonomous Province of Trento), the area of the extraction site is also considered for the quantification of the due amount. Planning and authorisation procedures generally integrate considerations of the heterogeneity in land use and values, but even fee rates are sometimes differentiated according to the supposed ecological value of the land. For example, Lombardy and Emilia-Romagna apply higher fees for extractions within the boundaries of regional parks, while Calabria applies a higher fee (x 8) for extractions from riverbeds. Even without the definition of a ceiling by the State, a race to the top has not occurred: three Regions out of 21 adopt a 'zero rate' on all aggregates and the average rate applied to aggregates in Italy are below average values present in the OECD Database (Figure 3).

Revenues of the fees are generally assigned/left to the Municipalities and communities involved to grant environmental restoration and more general territorial compensation; in some cases, Regions and Provinces retain a share to cover administrative, monitoring and enforcement costs. In actual fact, the process of earmarking funds for rehabilitation projects at the municipal level is far from being transparent and well documented and there is no available data gathered on the regional or national scale (EEA, 2008; Legambiente, 2021; Mazzanti, Zoboli, 2013).

The Italian experience clearly reveals a limited consideration of the role of this price instrument in influencing the behaviours of the economic agents involved.

The fee is not included in the ISTAT database on environmental taxes and even values comprised in the annual report on quarries of the environmental NGO Legambiente are only indirectly estimated, relying on rates and quantities extracted but not on true budget data (EEA, 2008; Zatti, 2023). Time series on prices of aggregates are not available and values published by the Chambers of Commerce are far from being entirely representative, due to the presence of many direct private transactions (EEA, 2008). As a matter of fact, the full regionalisation of competences

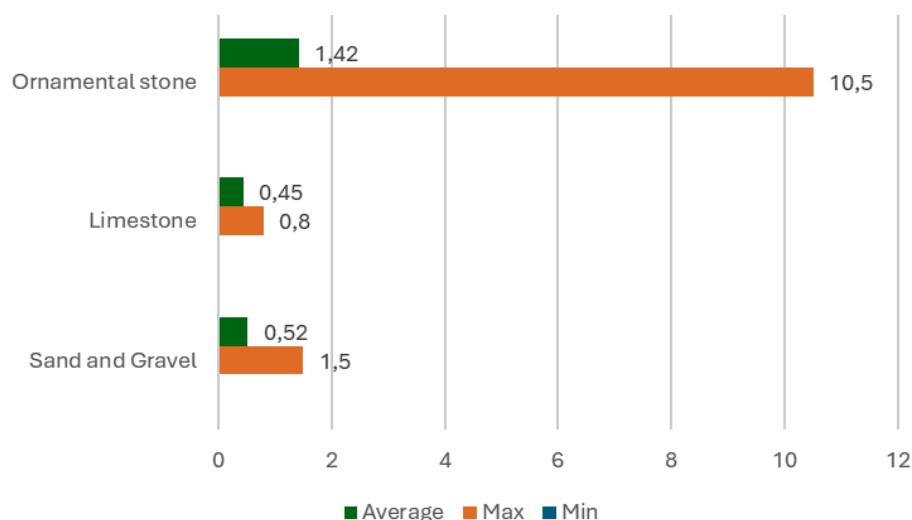


FIGURE 3: Fees applied on the principal materials extracted in the Italian Regions (€/m³, year 2021). The three materials described are estimated to be nearly the 90% of total non-metallic minerals extracted (ISTAT, 2024a). Minimum values are = 0 for all the three categories. - Source: own elaborations on Legambiente (2021).

in 1998 has aggravated the situation, bringing about fragmentation and missing data (Auci et al., 2012).

Moreover, the Italian approach to extraction is mainly based on a command & control logic, in which the planning of sites and quantities exploitable are based on the priority of keeping pace with demand in the construction and infrastructure sectors, while the substitution of virgin materials is not the main concern (EEA, 2008; Mazzanti, Zoboli, 2013). With this consideration, price signals are not aimed at influencing volumes extracted or to create real incentives for recycling and substitution, but mainly to cover administrative costs and to finance compensation for the affected local communities for land restoration and environmental improvement.

The level of extraction fees is esteemed to be inadequate in influencing market evolution (Auci et al., 2012; EEA, 2008; Legambiente, 2021⁷) and the only quantitative exercise conducted on the Italian data concludes that 'the effects of extraction charges on aggregate market development in Italy are very limited' (Mazzanti, Zoboli, 2013 p. 910). All in all, the decentralisation of responsibilities and the absence of a minimum standard has fostered a race to the bottom, testifying how, in terms of potential distortive effects on competition, the downward trend is the most likely result to occur.

4. LANDFILL (AND INCINERATION) TAX

Landfill and incineration taxes are implemented in several countries to drive waste away from the less desirable management options, to promote recycling, and to encourage the construction of plants for alternative treatments. Several case studies demonstrate how the combination of waste disposal taxes with taxes on raw materials provides strong incentives to economic agents to employ recycled materials rather than extracting virgin materials and disposing of old ones (Ettliger, 2016; Eunomia, 2022; Soderholm, 2011).

A well-known case study in the EU is that of the UK, where the tax has been in place since 1996 to internalize the external costs of landfilling (Eunomia, 2022). The tax is charged by weight, with a standard rate (122€/t in 2024) and a lower rate (3.3€/t) applying to inert or inactive waste, while a Tax Escalator was introduced in 2015 to regularly update tax rates and maintain the effectiveness of the tax. Revenues raised from the tax were used initially to reduce National Insurance Contributions and, since 2003, to finance environmental regeneration projects and promote biodiversity (ibidem). The tax had environmentally positive outcomes since the increase on the cost of final disposal incentivised local authorities and private actors to reuse products as well as develop remanufactured products from recycled goods. At the same time, some flaws in terms of misclassification of waste streams and illegal waste sites are reported.

In Italy, landfilling and incineration have, till today, a non-negligible role in waste management. In 2022, 18% of municipal waste generated (5.2 million tonnes) was landfilled and roughly the same percentage was incinerated (ISPRA, 2023a). As far as non-municipal waste is concerned, 10.2 million tonnes were sent to be landfilled in 2021 (5.7% of the total production), whilst nearly 3 million tonnes were incinerated (1.6% of the total) (ISPRA, 2023b). The construction sector proves to have a non-negligible role since C&D wastes correspond to about ¼ of the total weight of non-municipal waste landfilled (2.5 million tonnes), a percentage that rises to 30% if also waste coming from quarries and pits is included.

The landfill tax has been introduced in Italy in 1996 by the national law with the explicit environmental aim (Pigouvian tax) of increasing prevention in waste production, the recovery of matter and energy from waste, as well as containing the use of landfill, considered as the less desirable method of handling waste. Any operator conferring solid waste or sludge to a landfill or to an incineration plant without energy recovery is taxed.

Revenues are assigned to the Regions that may also fix rates (differentiated by quantity and the kind of waste stream) within upper and lower limits defined by the national law (Table 2). Since 2015 it has been established that all the revenues must be imputed by regional governments to an 'environmental fund' to be allocated to the following aims/topics: reduction of waste production, energy and material recovery, land remediation, creation and preservation of natural areas, and the financing of regional environment agencies. A percentage can also be assigned to Municipalities where the landfills are located or even to nearby territories that may be negatively affected by their presence. Total revenues generated were 115 million euros in 2022 (ISTAT, 2024b) but, even in this case, a transparent national (and even regional) reporting and assessment of the destination of funds is almost completely inexistent.

Lower and upper limits fixed by the national law have remained the same since 1996 and the unitary real value of the tax declined by nearly 1/3 in the period 1996-2018⁸, a clearly counterintuitive development for a price signal introduced to penalise the less preferable option in the waste management hierarchy. Figure 4 shows wide differences in the tax rates applied to inert wastes, with several Regions still adopting the lower limit (1 €/ton) fixed almost 25 years before.

At the same time, the State law (Leg. Dec. 152/2006, as modified by Law 221/2015) fixed new national targets for the separate collection of waste, with a bonus/malus mechanism in terms of landfill tax rates (Table 3). This represents an intrusive and questionable measure, since it introduces a sort of incentive redundancy and, at the same time, brings about strong discontinuities in the economic rationale of the economic operators⁹.

Even in this case, values are estimated to be too low to concretely influence the waste management chain, while the inertia of the national legislation has favoured the application of rates significantly lower than the European best practices (Ref Ricerche, 2020). In this perspective, a continued and progressive raise of minimum and maximum rates, even considering results obtained along the whole

TABLE 2: Landfill tax rates set by the Italian National law (L. 549/1995) - source: own elaborations.

	Lower and upper limits (€/ton)
Inert wastes	[1-10]
Other wastes	[5,17-25,82]
Incineration without energy recovery	20% of rates applied to landfilling
Incineration with energy recovery	0

TABLE 3: Landfill tax rates according to the fulfilment/not fulfilment of the national targets for the separate collection of waste - source: Leg. Dec. 152/2006, as modified by law 221/2015 art. 32.

	Tax rates
Not fulfilment	+ 20%
Over-fulfilment	
Up to 10%	- 30%
From 10% to 15%	- 40%
From 15% to 20%	- 50%
From 20% to 25%	- 60%
More than 25%	- 70%
% difference with respect to standard rates	

waste chain, may represent a policy option to be taken into consideration by the State legal framework. A useful example is that of the UK landfill tax which has been steadily increased since its introduction through an automatic escalator mechanism (Ettlinger, 2016).

Two other contents of Italy's approach to waste management are worthy of some reflections. Firstly, the conditions applied to incineration appear to be too generous, even in the case of energy recovery¹⁰. The Italian Ministry of Environment has classified this measure as an environmentally harmful subsidy (MASE, 2022) since the lack of an incineration tax does not provide incentives to prefer more virtuous management options in the EU waste strategy. Secondly, it may be wondered if the definition of an upper limit to tax rates adopted at the regional level is necessary for a levy that, due to the self-sufficiency principle, mainly

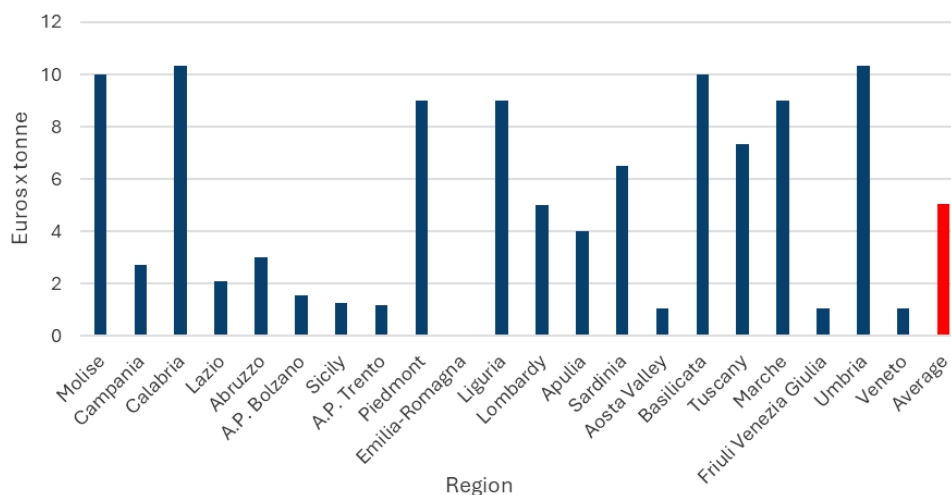


FIGURE 4: Landfill tax rates for inert wastes in the Italian Regions (€/ton, 2020) - Source: own elaborations on Ref Ricerche, 2020.

affects resident people and enterprises and, therefore, generate few risks in terms of tax exporting (Millimet, 2013).

5. DISCUSSION

The analysis conducted on the two price instruments implemented in Italy at the beginning and at the end of the material stream - i.e the aggregates levy and the landfill tax - shows how there are ample margins to improve their effectiveness and coherence, by rationalising multilevel governance and reforming some of the main implementation mechanisms of the levies.

In the Italian fiscal system, the space for Regions to introduce new environmental taxes autonomously is limited by several binding principles and by the extensive role granted to the State government in terms of fiscal coordination. In this perspective, the definition of a coherent policy setting seems to be particularly important both to valorise the decentralised ownership of environmental policies and, at the same time, to prevent possible distortions and opportunistic behaviors (Zatti, 2023).

The State action seems to be important, firstly, to guarantee deeper and updated knowledge of the main data and variables characterizing the value chain of raw materials (quantities extracted, revenues obtained, rate of penetration of recycled materials, etc.); without this pre-condition, it is very difficult to derive any conclusions or policy recommendations. This requires having a well-structured and organised monitoring of extraction and disposal sites and, at the same time, coordinating and checking activities carried out by lower levels of government. Many studies observed discontinuity and non-homogeneity in data collection, above all on mining and quarries, leaving space to significant improvements from this point of view.

Secondly, the national framework, while allowing flexibility across the Regions, turns out to be decisive to set minimum levels both in terms of technical standards and of tax rates, thereby preventing race to the bottom trends and giving Regions the opportunity to 'go beyond' with more ambitious targets ('meet or exceed' approach). In this direction, it will be important that increases in minimum tax rates are planned well in advance, for instance with an annual automatic escalator, to minimise delays in the revision of tax rates and to ensure predictability and transparency to industry and waste management operators.

In terms of the strategies of implementation, the effectiveness of aggregate/landfill taxes cannot be achieved in isolation from the other features and instruments implemented. Pricing instruments become effective in combination with additional regulatory and voluntary instruments, adopted upstream and downstream (planning of sites, selective demolition, minimum requirements of recycled contents, better quality of waste sorting, labels on secondary materials, controls against illegal disposal, raising-awareness campaigns, etc.), to design a well-structured and coherent policy framework. Otherwise, the economic incentive/disincentive alone may prove to be ineffective to stimulate operators to improve their environmental performance and to substantially increase the diffusion of recycled materials.

Finally, the earmarking of revenues to improve the acceptability and effectiveness of the tax can be seen a reasonable policy option. At present revenues are mainly left to regional and municipal governments for environmental and community purposes (compensation logic), but greater attention can be directed to promote the production and use of recycled materials. As displayed above, secondary aggregates are often not the first choice of construction operators - due to economic, cultural, and technical barriers - thus leaving space to a more effective policy-driven stimulus to close the loop of the sectoral material flow.

6. CONCLUSIONS

There is a wide and long-lasting consensus regarding the importance of fiscal instruments for boosting transformations aiming at meeting environmental challenges and promoting the transition to a circular economy. The rationale of this approach is based on the concept of *extrafiscality*, that emphasizes that taxation is not limited to raising revenues for the public budget but can also contribute to other objectives such as health protection, territorial development, environmental protection or social cohesion. In this direction, the circular economy action plan of the European Union (COM(2020)98 final) explicitly includes a section about 'getting the economics right' in which it encourages the implementation of several economic instruments at the different life-cycle stages of the value chain.

As a matter of fact, several barriers (informative, economic, legal and political) hampered the full development of economic instruments so that the focus was largely concentrated on administrative interventions and soft measures. The Italian experience confirms this statement: anti-tax approaches, inertia and overlapping of competences favoured the definition of low rates and a generally inadequate policy setting. Paradoxically, the existing taxation system often penalizes labour-intensive activities, including many activities (remediation, refurbishment, repair, recycle, maintenance, reuse) favourable to the circular economy paradigm.

Future steps towards the circular transition require a gradual but systemic change in the overall architecture of the tax system: a green tax shift able to substantially increase the tax base by broadening the degree of coverage or resources and activities (at present mainly concentrated on energy production and use) and by raising the respective tax rates. Economic disincentives can be combined, for political acceptability reasons, with tax benefits and other subsidies able to incentivize circular consumption and to mitigate more negative redistributive and social impacts.

Two further policy options can be profitably discussed in this scenario.

On the one hand, the progressive assessment and elimination of tax benefits and subsidies harmful to the environment or that promote polluting, unsustainable, and linear activities may become a key focus of the policy agenda. The fifth edition of the Italian Catalogue of subsidies that have an impact on the environment (MASE, 2022) has developed, for the first time, a specific section on 'subsidies harmful for the circular economy', identifying 52 measures

(tax expenditures), for a total amount of 18.8 billion euros: a promising starting point for the definition of future strategic environmental taxation's reform.

On the other hand, also quasi-economic instruments must be carefully taken into consideration. For example, the development and update of mandatory minimum environmental criteria in public procurement may represent an important step of policy packages. In 2015, Italy was a forerunner in implementing mandatory Green Public Procurement criteria, including a minimum certified recycled content in major construction materials and products, to be reached not only in new buildings and infrastructure but also in the renovation of existing ones. These obligations and technical standards could be expanded to all construction works, including the construction and renovation of private buildings, ensuring at the same time their continuous update together with a more effective control.

To conclude, even if the potential of green fiscal reforms has been widely investigated in the past, its extension to the circular economy seems to be still in infancy and requires further research both at the individual instrument and at the framework level.

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¹ Domestic material consumption (DMC) refers to the amount (in terms of weight) of materials used in an economy, i.e., materials extracted or harvested in the country, plus imported materials and products, minus exported materials and products.

² Which cover up to 110% of costs of interventions adopted to improve energy efficiency and reduce seismic risks in private buildings

³ See for example: Judgements 108/2005, 246/2006, 378/2007, 187/2011.

⁴ In the Italian legal system only Regions, among territorial governments, have a legislative power, while other levels (Municipalities,

Provinces, Metropolitan cities) have only statutory and regulatory power. The division of legislative competences between State and Regions is detailed in art. 117 of the Constitution: a first list of specific fields of exclusive jurisdiction of the national Parliament and a second set of specific subjects referring to concurrent legislation for which legislative powers shall be vested in the Regions, except for determination of fundamental principles, to be established by State legislation. Moreover, Regions shall have legislative power in all subjects not expressly attributed to the State.

⁵ Not energy-related materials.

⁶ In the Liguria Region the tax base is expressed in tonnes.

⁷ According to the values here reported, average extraction fees represent in Italy nearly 3,5% of the market values for sand and gravel and 3,2% for ornamental stones.

⁸ Calculated at the aggregate national level as the ratio of total revenues on total landfilled wastes.

⁹ For a waste operator increasing the rate of separate collection of 1% above the national target, for example, the mechanism provides for a 30% reduction of the due amount on the whole wastes landfilled. The same discontinuity can be found at each of the bands' limit reported in Table 3.

¹⁰ As a matter of fact, the country does not have incineration plants without energy recovery for municipal waste.

ASSESSING THE BUSINESS POTENTIAL, PRICE VOLATILITY AND SUPPLY CHAIN CHALLENGES OF SILICON METAL IN THE EUROPEAN UNION

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ABSTRACT

The European Union (EU) is heavily reliant on raw materials, some of which pose critical supply challenges. The EU's Critical Raw Materials (CRM) policy, initiated in 2011 and recently reinforced by the 2023 Critical Raw Materials Act, seeks to ensure the sustainable supply of crucial raw materials due to scarcity or economic importance. One such sector significantly affected is photovoltaic (PV) energy production, which faces challenges due to the scarcity of materials like silicon metal, crucial for PV cell manufacturing. Despite efforts to mitigate supply risks, challenges persist, including limited recycling and dependency on imports. This study evaluates annual silicon prices and volatility to assess the business potential of silicon metal production in the EU under current import reliance and 2030 targets. Analysing production trends and applications reveals a concentrated supply chain and diverse industrial uses of silicon metal. Price analysis highlights specific events influencing silicon metal prices, emphasizing its criticality to various economic sectors.

1. INTRODUCTION

The sustainability of the European Union (EU) in social, economic and environmental terms depends to a large extent on raw materials. These are necessary for our daily life as well as for modern technologies, but the supply is a matter of concern not only in the EU but globally. Among them, some are particularly important for the EU economy due to the high supply risk. Critical Raw Materials (CRM) are particularly linked to European industry, modern technologies and clean energy production (European Commission, 2023). Since 1977, the EU has been increasingly concerned about the supply of raw materials and the price volatility of some of them (Unión Europea, 1977). 2011 was the start of the EU's CRM policy (Comisión Europea, 2011) which aims to guarantee the sustainable supply of certain raw materials that are particularly important due to their scarcity or economic importance. The Critical Raw Materials Act announced in 2023 is the latest point of this policy (European Commission, 2024).

One of the more affected sectors is photovoltaic (PV) energy, which represents a way of clean energy production (Li et al. 2023), but it requires raw materials, which are in

short supply in the EU. The most representative example is silicon metal, which is needed for the manufacture of PV cells and for which up to 60% of the demand is imported in Europe (Grohol & Veeh, 2023; Liu, Phang, & Macdonald, 2022). Although, other CRM such as copper, borates, gallium, germanium and indium are used in this sector.

Despite the dependence of third countries on silicon metal, it was not until the third list in 2017 that it was considered CRM (Comisión Europea, 2017). At that time, the dependence on silicon metal was 64% (imports concerning non-EU countries). In addition, the substitution rate was 0.99 out of 1, indicating that there was no substitute for the technological applications, and the recycling rate was zero (Comisión Europea, 2017). Despite EU efforts to improve this situation in recent years, the current state remains difficult for silicon metal (Table 1). The recycling rate has not improved, dependence has fallen to 60%, and there are no realistic substitutes (0.97 out of 1) (Grohol & Veeh, 2023). It should be highlighted that only the processing stage is assessed for the silicon metal and not the extraction.

To improve this situation, there are three possibilities: improving silicon metal production in Europe, increasing



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TABLE 1: Mains indicator of the silicon metal.

Indicator	2017	2023
Substitution index (SI)	0.99	0.97
Recycling rate	0.00 %	0.00 %
EU import reliance	64.00 %	60.00 %

recycling or finding substitutes for this raw material (Rizzo et al., 2020). These practices could contribute positively to increasing the value of European industry and limiting the effect of silicon price volatility.

The aim of this article is to analyse the situation of Si metal in relation to the CRM policy being developed by the EU and to study the evolution of Si metal supply and recycling in order to understand the effectiveness of this policy. Furthermore, the research focuses on the price volatility of Si metal. CRMs suffer from high price volatility due to various factors. However, this aspect has not been extensively studied and analyses and policies often focus on the economic importance of CRMs or their supply, ignoring the uncertainty due to constantly changing prices. The aim of this article is to study the changes in the price of this metal in order to have more information on this important aspect. This short study assess the business potential of silicon metal production in the EU under the current import reliance and the 2030 targets (European Commission, 2023).

2. SILICON METAL PRODUCTION AND APPLICATIONS IN EUROPE

The silicon metal in the EU comes mainly from third countries, although the dependency has been partially reduced since 2017. At this time, Brazil and China were two of the main suppliers, with 12% each. But, currently, this supply has been reduced to 9% and 2%, respectively (Comisión Europea, 2017; Grohol & Veeh, 2023). The main suppliers to the EU are shown in Table 2.

It is possible to observe that only the consumption from France and Germany has increased, which indicates a diversification to other countries, such as Bosnia and Herzegovina, Russia, Australia, Iceland, South Africa, Kazakhstan or Malaysia (Grohol & Veeh, 2023). It should be considered as a positive practice that allows to minimize the risk derived from singular factors from specific countries.

The silicon metal production globally is around 3,1 Mt, and just in the EU is around 400 kt (BRGM, 2019; CRM Alliance, 2022). It is concentrated in a few companies: Elkem, Wacker and Finnfjord AS in Norway, Ferroglobe in Spain and France and RW Silicium in Germany (AMG Silicon, 2023; Elkem, 2024; Ferroglobe, 2022; Finnfjord, 2022; Kristiansen & Van der Eijk, 2020; Wacker, 2017). The main applications of the silicon metal are:

- Chemicals production: such as silicone rubber, foam, sealants, adhesives, lubricants, additives, coatings... (Ferroglobe, 2024).
- Aluminium alloys: Silicon is used in aluminium alloys to improve their strength for structural purposes (Belmont, 2022; Ferroglobe, 2024).

TABLE 2: Si metal supply (only countries with a share > 5 % are shown).

Country	2017	2023	Change
Norway	23 %	33 %	+10 %
France	19 %	28 %	+9 %
Brazil	12 %	9 %	-3 %
Germany	5 %	6 %	+1 %
Spain	9 %	4 %	-5 %
China	12 %	2 %	-10 %
Non-EU members	64 %	60 %	-4 %

- Photovoltaics: It is used in PV cell wafer production (Ferroglobe, 2024).
- Electronic components. Mainly in microchips due to their semi-conductivity properties (Belmont, 2022; Ferroglobe, 2024).

Globally and in terms of material consumption, chemical production and aluminium alloys are the most important (25% and 32% respectively). This means that the silicon metal market is related to the aluminium alloy one. Periods with a high urbanistic or mobility development are characterised by an increase in aluminium alloy prices and, therefore, by an increase in silicon metal prices. Energy prices can also deeply affect the silicon metal since aluminium and silicon are electro-intensive industries.

The PV industry accounts for less than 7% of silicon consumption, and electronics for around 1% (BRGM, 2019). However, their impact on the economy is significant. The PV and electronics sectors have a combined value in the EU of €77,000 million similar to the manufacture of basic metals at almost €65,000 million. Chemical production is the most important one with a value of €132,000 million (Grohol & Veeh, 2023). The PV sector could affect the silicon price in the future due to its dependence on energy prices. Vast PV installations in the EU may transform the energy mix, which could change energy prices, affecting the aluminium alloys sector and silicon metal production.

3. SILICON METAL PRICES AND VOLATILITY

In addition to the supply risk of silicon metal, there is also the issue of its price and, in particular, its fluctuation. Although silicon metal is not the most volatile CRM (Rare Earth Elements are likely), its price is deeply affected by specific issues in specific countries and sometimes as a result of national policies of third countries or internal decisions of some companies, which can be not aligned with the EU interests. Figure 1 shows the price evolution in comparison with gold from 1991 to 2022 based on the annual reports of the United States Geological Survey (USGS, 2024) (Metallurgical-grade silicon metal is considered in this study).

The most important conclusion to be drawn from the figure is the variability of the silicon metal price since 2008. If the gold price is taken as the economic benchmark, the greatest stability would be achieved with a symmetric

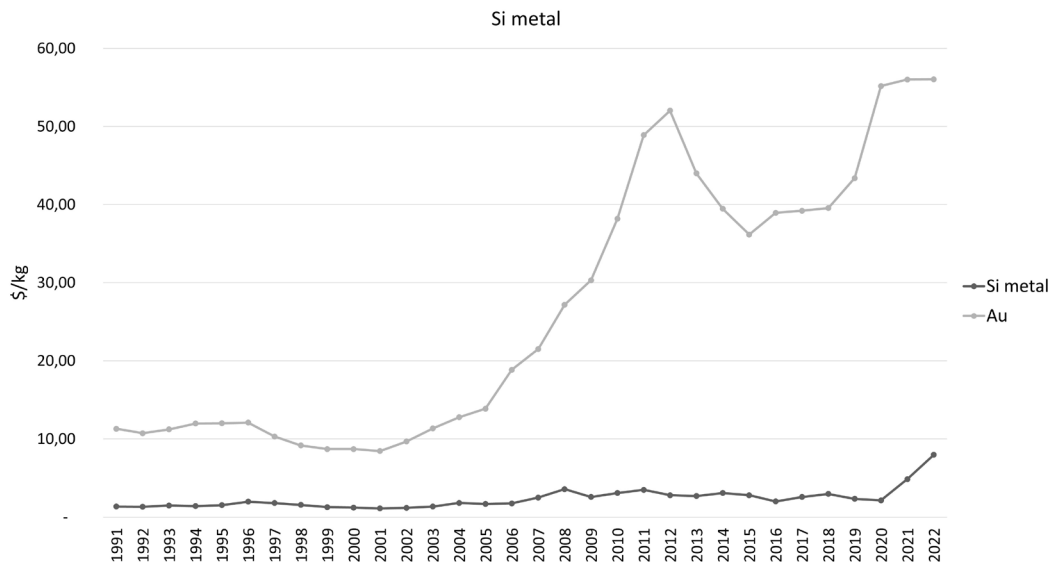


FIGURE 1: Silicon metal prices.

graph. It can be seen that around 2005, the gold price started a steady rise until 2012 and then another stage of price increase in 2020. There are two main reasons for the increase in the gold price:

- Economic recessions: When the global economy is in recession, investors prefer safe options like gold to financial markets. This was the situation in 2007-2008 with the global economic crisis and in 2019-2020 with the COVID-19 crisis.
- Devaluation of the US dollar and high oil prices: The devaluation of the US dollar and the high price of oil also contribute to the rise in the price of gold, as large companies use gold to offset the fluctuations in the US dollar. Additionally, since 2001, gold mine production was decreasing until 2007, which increased the gold price.

In the 1990s, prices for silicon metal were considered to be heavily dependent on energy costs, accounting for up to 20% of the final price in addition to the ore. In 1996, the peak coincided with the maximum of steel production, which indicates a period of strong demand related to the industrialisation of China, Brazil and India (Comisión Europea, 2011; USGS, 2013).

The silicon metal prices were stable until 2004, when PV energy started its development. The trend seemed to be upwards, but in 2005 and 2006, the price declined. This is because the demand forecasts of the PV sector were met by the processing of the domestic reserves of the wafer producers and the Chinese government's permissiveness for the export of silicon metal in order to promote PV energy (PHOTON International, 2004). However, in 2007, China increased tariffs on silicon metal exports, which affected global prices until 2009, when the global economic crisis deeply reduced the consumption of aluminium (USGS, 2013). However the aluminium market rebounded since 2010, when the economy started to recover. This recovery was supported by the integration of silicon producers by primary consumers (USGS, 2013).

In 2011, there was a change in EU policy that stopped supporting the development of PV, which led to a decrease in polysilicon supply in 2012 (Wang, Z. & Wei, 2017). This was countered by an increase in exports from China, which allowed for an increase in production in countries such as China, Russia, Malaysia, Sweden and Uzbekistan, which led to a slight decrease in the price until 2013 (USGS, 2012; USGS, 2013). The situation remained stable until a decrease in 2016 related to a weak global aluminium alloy demand coupled with an oversupply of silicon metal (USGS, 2017).

In 2018, there was a small spike that could be due to market dominance by Ferroglobe, which would have acted as a virtual monopoly by positioning the price of silicon metal as a way to put up trade barriers to certain countries (US International Trade Commission, 2020).

The huge price increase since 2021 has been driven by two determining factors. Firstly, due to the Chinese government's carbon reduction policy. Many coal-fired silicon metal plants were forced to stop production. This was compounded by rising energy prices in Europe, particularly affecting production in Norway (Sadhna Gupta, 2023).

In addition to annual price variability, silicon price volatility within the same year must be considered. The reports of the German Geological Survey (BGR, 2023) show how in the last 10 years it has been relatively stable with variations of 10% except for the last three years: 17.9% in 2020, 107% in 2021 and 24.5% in 2022. These are the years when the price rises sharply, so it is possible to correlate volatility with rising silicon metal prices.

4. BUSINESS POTENTIAL

The EU's 2030 targets for CRM imply the processing of at least 40% of CRM in Europe and a recycling rate of 15%. With regard to silicon, processing in Europe is currently already at 40%, with a 4% increase in the period 2017-2023 (Grohol & Veeh, 2023; Comisión Europea, 2017), so it is not ambitious to think that it is possible to improve by another

er 4% by 2030. Although, these goals are for all CRM as a whole, considering a current demand of 6 million tonnes (Mt) and a projected demand of 10.7 Mt in 2030 (European Commission, 2023), silicon processing and recycling in Europe both current and desirable in 2030 are illustrated in Figure 2.

With these targets, in terms of production, the EU would produce more than twice as much as it is currently producing. However, the recycling rate is optimistic as there are currently no industrial processes and only research (Moen, et. al., 2017).

Based on the price analysis carried out, a realistic price for silicon metal can be set as an average of the values between 2007 and 2020, which remained between 9000 and 14000 \$/kg. This is 12500 \$/kg (11576 €/kg) with a positive variability of 30% and a negative fluctuation of 27%. In addition to the 2022 value of 36186 \$/kg (33506 €/kg). The business potential of silicon processing and recycling is shown in Table 3.

However, it is necessary to consider that these are gross amounts without considering production costs and therefore these values are not profit. It has already been seen that the energy cost alone account forat least 20% and is probably higher today. The other costs would also be relevant (but they have not been assessed by the Commission).

Another important issue is the recycling rate. Today, there are several recycling processes under research (Deng, et. al., 2022; Moen et al., 2017; Riech et al., 2021; Wang, et.al., 2022). However, it does not seem realistic to think that by 2030 the processes under research will have reached industrial scale and will be able to achieve the 15% recycling rate.

In any case, an increase of 19 billion euros (minimum realistic scenario) or even 26 billion euros (AVG realistic scenario) is an important economic volume to be considered. Even if it is not the final profit, this economic volume opens up a business opportunity that would remain in the EU instead of in other third countries where the industry may not be environmentally or socially sustainable.

TABLE 3: Business potential of silicon metal processing and recycling in 2030.

Stage	Realistic scenario in 2030 (M€)			2022 value
	AVG	MAX	MIN	
Processing	54,501	70,620	39,669	157,744
Recycling	18,580	24,075	13,524	53,776
Processing (increase)	26,292	34,068	19,137	76,098
Recycling (increase)	18,580	24,075	13,523	53,776

5. CONCLUSIONS

Several interesting conclusions at different levels can be drawn from this research.

The EU relies heavily on CRM to sustain its economy and technology. The EU's CRM policy, launched in 2011 and updated in 2023, aims to ensure the sustainable supply of essential raw materials. In the case of silicon metal, dependency has been improved, but recycling is an ongoing challenge that needs to be addressed. Diversification has been observed in the countries supplying silicon metal to the EU, suggesting a risk mitigation strategy. Silicon metal is essential for PV energy production, but is in short supply in the EU, posing a risk to energy and economic security.

One of the issues with silicon metal is its price volatility, which is not as high as other CRM (REE). Despite this, some important price changes are observed occasionally motivated by specific factors such as Chinese domestic policies, business decisions or energy crises.

The improvement of the silicon metal processing situation in the EU under the targets set for 2030 could bring an enormous business potential with an estimated increase of 19 billion euros (B€) in the most favorable scenario for transformation and 13.5 B€ in the case of recycling.

However, further analysis of silicon metal prices and volatility is needed to try to anticipate price behaviour and minimise investment risks in order to develop a stronger EU industry to reduce imports, and mitigate price fluctuations.

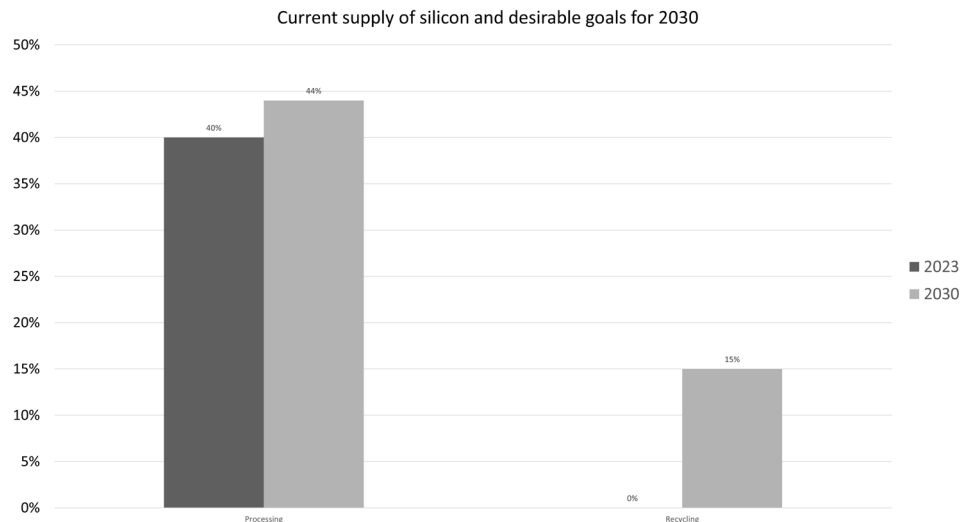


FIGURE 2: Current supply of silicon and desirable goals for 2030.

It is necessary to study the different silicon metal products and to understand their intra-year volatility (rather than considering solely annual average values). Furthermore, this price analysis has been carried out on the American price basis, so there is a need to analyse the differences with the European market. Finally, a study of silicon metal processing costs is required to understand the potential direct economic benefits of increasing silicon metal production in the EU.

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ECOTOXICOLOGICAL ASSESSMENT OF WASTE SCRUBBER WATER IN UNICELLULAR ALGAE (*TETRASELMIS SUECICA*) AND BLUE MUSSEL (*MYTILUS EDULIS*) LARVAE

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ABSTRACT

Wastes from shipping pose a substantial environmental challenge at local, regional and global scales, particularly in terms of adverse impacts on climate change and biodiversity. There is an increase in installations of exhaust gas scrubbers to reduce the emissions from ships following international regulations on sulphur content in marine fuel from 2020. Marine scrubbers can be classified into wet and dry scrubbers. Scrubber water (also known as washwater) from both wet scrubber systems has been found to release acidic water containing nutrients and contaminants back to the marine environment, including metals, aromatic hydrocarbons, and soot particles. This is especially true for the open-loop scrubbers that utilize the natural alkalinity of seawater and keep a high flow of process water in order to reduce SO₂ in the exhaust and the washwater is discharged to sea, most often without substantial treatment. Little is known about potential impact of the discharged washwater on the marine environment. In ecotoxicological tests, a number of marine organisms have shown negative effects after acute and chronic exposures to varying concentrations of scrubber water, but the main pollutants involved in these responses are not clear yet. The aim of this study was to evaluate the effect of the main pollutants found in open-loop scrubber discharge water for survival, feeding and development of different species at the base of the food web after acute exposures to gas scrubber effluent. Toxicity, mortality, and physiology have been evaluated in marine microalgae, *Tetraselmis suecica*, and blue mussel (*Mytilus edulis*) larvae. Direct exposure to scrubber water appears to adversely affect biological and reproductive parameters in invertebrates, raising substantial concerns about ongoing open-loop exhaust gas scrubber system deployment.

1. INTRODUCTION

Waste materials from international shipping pose a significant threat to our oceans and their delicate ecosystems, particularly in terms of impacts on climate change and biodiversity. Ships produce oily waste from engines, machinery, and fuel systems. Oil spills harm marine life, contaminate water, and damage coastlines. Discharged washwater contains sulfur and other pollutants, impacting air and water quality. Ballast Water, used to stabilize ships during loading and unloading, introduces invasive species, disrupting local ecosystems. Sewage (so-called "black water"), from crew and passengers, contaminates seawater,

spreads diseases, and harms aquatic organisms. Grey-water (non-sewage wastewater from sinks, showers, and laundry) contains detergents and chemicals, affects localised water quality. Leftover cargo materials and cleaning water pollutes water, especially if cargo contains hazardous substances. Discarded rubbish from onboard activities, including plastics, paper, and food waste is ingested by marine animals, causing entanglement and habitat destruction. Efforts to manage ship-generated waste involve regulations, proper disposal, and sustainable practices to safeguard our oceans.

In addition, international shipping is responsible for the emissions of a variety of air pollutants, including volatile



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carbon-based gases, particulate matter (PM), sulphur oxides (SO_x), and nitrogen oxides (NO_x), with marine shipping currently predicted to account for 5-15% of global anthropogenic SO_x emissions (Viana et al., 2014; Lindstad & Eskeland, 2016; Sofiev et al., 2018). Ship emissions are regulated by the International Maritime Organization (IMO) which has recently set a stricter limit on the sulphur content of shipping fuels. Exhaust gas cleaning systems, also known as scrubbers, remove SO_x from exhaust gases before they are emitted to the atmosphere, with studies suggesting scrubbers are able to remove up to 95% of SO_x from the exhaust gas (Andreasen & Mayer, 2007). Due to the increased costs of these higher quality fuels, many shipping companies opt for the use of scrubbers that offer a viable solution at a lower economic cost (Endres et al., 2018; Teuchies et al., 2020). The majority of scrubbers installed on marine vessels are wet scrubbers using either a 'closed-' or 'open- loop' system. Remnant washwater is discharged into the surrounding surface water at a flow rate between 200-500L/s for a 15MW vessel from an open-loop scrubber, and only ~0.5-3L/s for a 15MW vessel from a closed-loop scrubber due to the recirculation process (Teuchies et al., 2020). The washwater generated can contain a variety of pollutants, including polycyclic aromatic hydrocarbons (PAHs), trace metals and nitrates, thus displacing potential atmospheric pollutants to the marine environment (USEPA, 2011; Lunde Hermansson et al., 2021). Washwater is highly acidic, with the median pH of open-loop scrubber water reported as 5.63, due to the dissolved sulfuric and nitric acids (Comer, Georgeff & Osipova, 2020; Teuchies et al., 2020).

The chemical composition and concentrations of contaminants in discharged scrubber water are dependent on a number of factors including the type of scrubber, scrubber design, removal efficiency, composition of fuel and lubricant oil, and the operating conditions of the ship (Hassellöv, et al., 2020). Several PAHs - such as anthracene, phenanthrene, naphthalene, fluorene and fluoranthene - have been identified in multiple studies (Magnusson, et al., 2018; Schmolke, et al., 2020), and may originate from the fuel itself or the fuel combustion process. Metals are an important component of scrubber water with the highest recorded concentrations reported as vanadium, zinc, copper and nickel (Teuchies, et al., 2020). Vanadium and nickel originate from, and according to previous studies, correlate strongly to the sulfur content of the used fuel (Hansen & Aalborg, 2012; Teuchies, et al., 2020). However, the high concentrations of copper and zinc may be related to the composition of other equipment such as a ship's anti-fouling system, the sampling tube, and corrosion protection anodes (Turner, et al., 2017; Ushakov, et al., 2020). A report by Carnival corporation & PLC (2019) found similar metals in their washwater effluents, with additional metals such as lead (Pb) and chromium (Cr) present at lower concentrations.

Despite the rapid increase in the use of scrubbers, knowledge about the impacts of scrubber effluent on the marine environment remains insufficient (Lange & Markus, 2015). Very few studies have investigated the effects of scrubber water on marine organisms. Some argue that the environmental impacts of the discharge of scrubber

water will be negligible, with the resulting concentrations of hazardous substances in the surface water predicted to be orders of magnitude below levels of concern (Kjølholt et al., 2012). However, this modelling was completed on the assumption that background concentrations of contaminants remain constant. This scenario is unlikely to be representative, especially close to ports, marinas, and areas where sources of diffuse pollutants may accumulate, and as substances such as PAHs and metals can persist for long periods in the environment, thus increasing in concentration over time. Initial studies have also demonstrated lethal and sub-lethal effects on various marine organisms when exposed to different dilutions of scrubber washwater (Koski, Stedmon & Trapp, 2017; Magnusson, Thor & Granberg, 2018; Ytreberg et al., 2019). Furthermore, there is particular concern over the effects of scrubber washwater within ports, estuaries, and coastal waters, that experience low water exchange rates and are some of the most heavily trafficked areas (Ytreberg et al., 2019; Teuchies et al., 2020; Thor et al., 2021). Studies analysing the contents of scrubber discharge suggest that the use of scrubbers directly conflicts with the Water Framework Directive's goals of achieving good environmental status within European waters (Lunde Hermansson et al., 2021). Therefore, understanding the potential effects of scrubber water discharge, as well as some of the main pollutants such as PAHs and heavy metals, on phytoplankton and zooplankton is important for assessing the environmental risks associated with increased scrubber usage in the future.

Many studies that have investigated the effects of PAHs and metals have used concentrations of individual compounds that are orders of magnitude higher than those found in scrubber water. Existing studies tend to focus on assessing the impact of single, isolated chemicals on aquatic organisms. This approach, while informative, does not reflect the real conditions in the environment faced by marine life when exposed to scrubber washwater. Scrubber discharge contains complex mixtures of various contaminants, including metals, sulfur compounds, polycyclic aromatic hydrocarbons (PAHs), and other potentially harmful substances. As a result, marine organisms are often subjected to combined chemical interactions and synergistic effects that can amplify toxicity beyond what would be expected from exposure to individual compounds alone. This complexity highlights the need for our study; original research that evaluates the cumulative and interactive effects of multiple pollutants in scrubber washwater in order to provide a more comprehensive understanding of its true impact on aquatic ecosystems. Our study also examines the effects of individual pollutants present in scrubber wash water, such as phenanthrene, anthracene, and fluoranthene, compared to their combined impact.

A multispecies approach toward ecotoxicological testing is fundamental for the purposes of risk assessment and getting accurate environmental management and ecological risk assessment procedures. For this reason, the ecotoxicological experiments in this report include two different species reflecting different trophic levels in the marine food web. Part I of this paper provides evidence of the effects of scrubber water and some of the main pollutants

on the growth and mortality of marine microalgae, *Tetraselmis suecica*. Part II shows results obtained with the blue mussel larvae exposed to different dilutions of scrubber water. These results provide information about the potential effect of scrubber water discharges in the marine environment and which trophic levels could be more sensitive to the components of this water.

2. PART I: EVALUATION OF THE TOXICITY OF SCRUBBER WATER AND SOME OF THE MAIN POLYCYCLIC AROMATIC HYDROCARBONS (PAHS) AND HEAVY METALS FREQUENTLY FOUND IN SCRUBBER EFFLUENTS ON THE GROWTH AND MORTALITY OF THE MARINE MICROALGAE, *TETRASELMIS SUECICA*

2.1 Introduction

Marine microalgae are important components of the aquatic environment due to their role as primary producers in the aquatic food web and as key players in biogeochemical cycling (Ebenezer & Ki, 2013). They can be one of the first groups affected by contamination and can thus be sensitive indicators of environmental change (Debelius et al., 2009; Kumar & Shin, 2017). For this reason, they have frequently been used in ecotoxicological assays, results of which have been widely used as a basis of risk assessment and the development of environmental regulations (Kumar & Shin, 2017).

Tetraselmis suecica is a species of unicellular green algae with a high growth rate and a global distribution (Ebenezer & Ki, 2013; Kumar & Shin, 2017). *T. suecica* is a commercially important algae species due to its widespread use as a basic food organism in aquaculture (Kumar & Shin, 2017). Due to its unicellular nature, it is likely to be sensitive to potentially toxic compounds (Debelius et al., 2009). Microalgae are often the main primary producers within aquatic environments and thus they represent the main energy input into the ecosystem (Campanella, et al., 2001). Therefore, the impacts of the components of scrubber washwater on a species such as *T. suecica* are of great interest, due to the potential wider environmental impacts on the marine environment, as detrimental effects to algal species are likely to affect organisms at higher trophic levels, and thus potentially impact ecosystem health (Chen et al., 2018). This species shows short generation times (Cid, et al., 2012), is easily cultivated within a laboratory (McCormick & Cairns, 1994), and is closely connected with other marine organisms (Champan, 2013), making it an ideal test species.

The objectives of the ecotoxicological assays using *T. suecica* were:

- To determine the effect of scrubber water at different concentrations on the survival and growth of *T. suecica*.
- To evaluate the effect of phenanthrene, anthracene and fluoranthene as single compounds and combined on the survival and growth of *T. suecica*.
- To assess the effect of heavy metals on the survival and growth of *T. suecica*.

2.2 Methods

2.2.1 Algae culture conditions

The green unicellular marine algae *T. suecica* was cultured in the laboratory under standardised conditions (24 hours of continuous light at 20°C), following the internal standard protocol at the National Oceanographic Centre, Southampton. The subculturing process was carried out using 20 L bottles containing filtered seawater, which was enriched with nutrient solution. Cultures of *T. suecica* were supplied with 0.2 µm-filtered air and were routinely monitored by microscope observation to confirm their growth and check for contamination.

2.2.2 Experimental solutions

All glassware was acid-washed (HNO₃ 10% vol.) and rinsed with distilled water and acetone prior to the experiments.

Scrubber water discharge

The scrubber water was collected from the open scrubber system of "Catherine C" by DANAOS. Sample was taken on 19th of August 2021; location: at sea; position: 51 04.6N; 001 37.9W. The scrubber discharge water was collected from the overboard unit of the ship (scrubber outlet) directly into acid-washed glass containers; at the same time, inlet water was collected from the seawater pump system onboard (scrubber inlet). All containers were shipped to Southampton, UK, upon returning to the harbour and were subsequently stored at 4°C and dark until use in experiments. The scrubber water concentrations for the experiments were 0.001, 0.01, 0.1, 1.0, 2.0, 5.0, 10.0, 20.0, 40.0 and 100% of the original outlet scrubber discharge water. The dilutions were made using autoclaved and filtered sea water. Each treatment was replicated three times and the whole experiment was repeated twice.

PAHs

PAHs were tested individually at concentrations equivalent to those identified in scrubber water. Phenanthrene and fluoranthene are some of the most prevalent PAHs identified in washwater (USEPA, 2011; Teuchies et al., 2020). Anthracene has been identified as a priority substance by the European Water Framework Directive (Teuchies et al., 2020), and studies have suggested it is one of the most toxic hydrocarbons (Aksmann & Tukaj, 2004). Multiple studies have reported median effect concentrations at the magnitude of micrograms per litre (Gala & Giesy, 1992; Aksmann & Tukaj, 2004, 2008), with one study reporting a median lethal concentration (LC50) of 1.93 µg/L (Weinstein & Polk, 2001). This low lethal concentration is a similar concentration to those recorded in scrubber water, suggesting that anthracene could be one of the more potent pollutants present within scrubber discharge.

Analytical grade phenanthrene, fluoranthene and anthracene (Sigma-Aldrich) were dissolved in 100% acetone to produce stock solutions at concentrations of 1 mg/mL. These were subsequently diluted using autoclaved seawater, to obtain a final concentration of 100 µg/L of phenanthrene, and 20 µg/L for both fluoranthene and anthra-

cene. Appropriate volumes of final stock solutions were then added to experimental vials, and diluted with artificial seawater, to obtain desired concentrations of PAHs. The concentrations used in this study were selected to reflect those found in scrubber water at various levels of dilution, equating to a 1%, 10%, 40% and 100% concentration of washwater. The nominal test concentrations were therefore 0.05, 0.5, 2 and 5 µg/L for phenanthrene, and 0.01, 0.1, 0.4 and 1 µg/L for both anthracene and fluoranthene. Each treatment was replicated three times and the whole experiment was repeated twice.

When assessing the toxicity of PAH combinations, the ratio of the individual mixture components was kept constant while the total concentration of the mixture was varied.

Metals

Four metals commonly found in scrubber effluents were our focus for the tests. These are V, Ni, Zn and Cu. All metal sources were of analytical grade ($\geq 99\%$ purity) and purchased either from Merk (Germany) or Sigma-Aldrich (Germany). Substitutes were used as sources for each metal to simplify the homogeneity of the solution. Vanadyl sulphate hydrate (H_2O_6 SV) was used as a vanadium source; copper sulphate pentahydrate ($\text{CuSO}_4 \cdot 5\text{H}_2\text{O}$) was used as a copper source; zinc sulphate heptahydrate ($\text{ZnSO}_4 \cdot 7\text{H}_2\text{O}$) was used as a zinc source; nickel sulphate hexahydrate ($\text{NiO}_4 \cdot 6\text{H}_2\text{O}$) was used as a nickel source.

A stock solution of each metal was produced with an initial concentration of 1000 ppm. This initial stock solution was serially diluted in autoclaved seawater to obtain stock solutions of: 500 µg/l for V; 500 µg/l for Zn; 100 µg/l for Ni; 50 µg/l for Cu. The appropriate volume of each solution was added to a final solution of 20 ml. The final concentrations of metals were: 350 µg/l for V; 250 µg/l for Zn; 50 µg/l for Ni; 40 µg/l for Cu. These concentrations were determined from the average of metals found in open loop systems noted by Teuchies et al. (2020) as well as preliminary results obtained by the EMERGE consortium. The [100%] concentration represents the concentration of metals taken directly from a tap near the scrubber outlet. Three additional diminishing concentrations were tested to consider the dilution of the effluent over time as well as analyse the trend in cell concentration and viability. These concentrations are: [40%]; [10%] and [1%]. Each treatment was replicated three times and the whole experiment was repeated twice.

2.2.3 Experimental design

Toxicity assays were performed following the proposed standard protocol for conducting ecotoxicological studies on algae from ISO standard 8692 (ISO, 1989). Toxicity testing was carried out over a 96-hour period for scrubber water discharge and 72-hour period for PAHs and metals exposure. Static, non-renewal tests were carried out at $20 \pm 0.7^\circ\text{C}$, under continuous illumination. Tests were conducted in 50 mL beakers; test solution volume was 20 mL. Stock cultures in logarithmic growth phase (cell density of 1×10^5 cells mL^{-1}) were used as inoculum. During exposure, beakers were shaken daily. Two independent tests were

carried out with triplicate repeats for each treatment. A negative control (i.e. a group that does not receive any type of treatment and, therefore, should not show any change during an experiment) was included with autoclaved seawater only. A solvent control was included with a diluted acetone for PAHs exposure and a solution of seawater with 2% of HNO_3 for metals exposure. 40 mg/L copper(II) sulfate pentahydrate ($\text{CuSO}_4 \cdot 5\text{H}_2\text{O}$) and 200 µg/L tributyltin chloride (TBT-Cl) were used as the reference toxicants for PAHs and metals experiments, respectively (Tam, Wong and Chong, 2001).

2.2.4 Cell concentration

At the end of the 72-hour exposure period, growth was determined by cell counts using a Neubauer Improved haemocytometer at 10 x magnification under an optical microscope.

2.2.5 Trypan blue cell viability test

Growth was determined by cell counts using a Neubauer Improved haemocytometer and a light microscope. After the incubation period of 72 hours, 10 µl of Trypan blue solution was added to 10 µl of each treatment to stain the sample for microscopic evaluation. The number of viable and non-viable individual algal cells were identified and enumerated using a haemocytometer under a light microscope (Olympus CH30) with 20X magnification. Non-viable algae cells were distinguishable from viable cells as they were stained blue unlike the green viable cells without the penetration of the stain. In an attempt to avoid unconscious bias when counting and identifying viable cells, this was performed blind, and treatments and concentrations were only made known post-counting.

2.2.6 Statistical analysis

The data were tested for normality and homogeneity of variance. One-way analysis of variance (ANOVA) was subsequently utilised to test for significant differences in cell concentration and mortality between treatment groups. Where a significant difference was observed, Tukey's HSD post hoc test was used to assess pairwise differences. When data did not meet the assumptions of normality and homogeneity of variance, the non-parametric Kruskal-Wallis one-way ANOVA was employed, followed by Dunn's test, to compare individual treatment groups. Percentage mortality data was arcsine transformed before any statistical analysis was carried out (Bellas et al., 2008).

2.3 Results

2.3.1 Effect of exposure to scrubber water on cell concentration and mortality of *T. suecica*

After a 96-h exposure, a significant difference in algal cell concentration between treatment groups ($p < 0.001$; Figure 1a) was observed. Cell concentration and mortality showed a concentration-dependent effect with the maximum reduction observed for the highest concentrations of scrubber water. The 96-h LC_{50} was calculated as 60.17% at the end of the experiment.

2.3.2 Effect of exposure to PAHs on cell concentration and mortality of *T. suecica*

After the 72-hour exposure, there were no significant differences in the algal cell concentration or mortality between the negative seawater controls and the acetone controls, in the second experiment ($p > 0.05$). A significant difference in algal cell concentration between treatment groups ($F = 15.61$, $p < 0.001$; Figure 2a) was observed. Exposure of *T. suecica* to all concentrations of phenanthrene, fluoranthene and anthracene resulted in a significant reduction in the cell concentration, compared to the control treatment ($p < 0.01$). However, cell concentration did not decrease with increasing concentration of toxicant, with no significant differences in cell growth between concentrations of all the PAHs tested ($p > 0.05$).

The mortality significantly differed between treatment groups ($F = 11.8$, $p < 0.001$; Figure 2b). None of the PAH treatment groups resulted in a significantly different mortality compared to the negative controls ($p > 0.05$). However, the resulting mortality upon exposure to 100%, 10% and 1% phenanthrene, as well as 100% fluoranthene, significantly differed to all other PAH treatments ($p < 0.05$; Figure b). One of the individual PAHs tested (phenanthrene) showed significant toxicity (significantly greater algal mortality

than the solvent control, $p < 0.05$) at the 100% concentration, however the other two PAHs (anthracene and fluoranthene) did not show significant toxicity at the any of the tested concentrations.

All 4 possible combinations of the 3 PAHs at all concentrations were significantly different ($p < 0.05$) to the control. Kruskal-Wallis tests were performed at a 95% confidence interval to determine if there was a significant difference in mean percentage mortality between the individual PAHs and when in combination at 4 relative concentrations. Each individual PAH was compared to its respective combinations (e.g., Anthracene (A) 1% was compared to A+F 1%, A+P 1%, and A+F+P 1%) (Table 2).

Post-hoc Dunn tests were performed at a 95% confidence level. In the case of all 3 PAHs, the effects of PAHs in combination were significantly different from the individual effect (Figure 3). For each combination tested there appeared to be a positive correlation between increasing concentration and the level of toxicity on *Tetraselmis*. The resulting p -values were all < 0.05 demonstrating that the concentration of each combination tested had a significant impact on the toxicity of that combination.

The post-hoc tests demonstrated significant differences between the 1% and 100% concentrations for all four

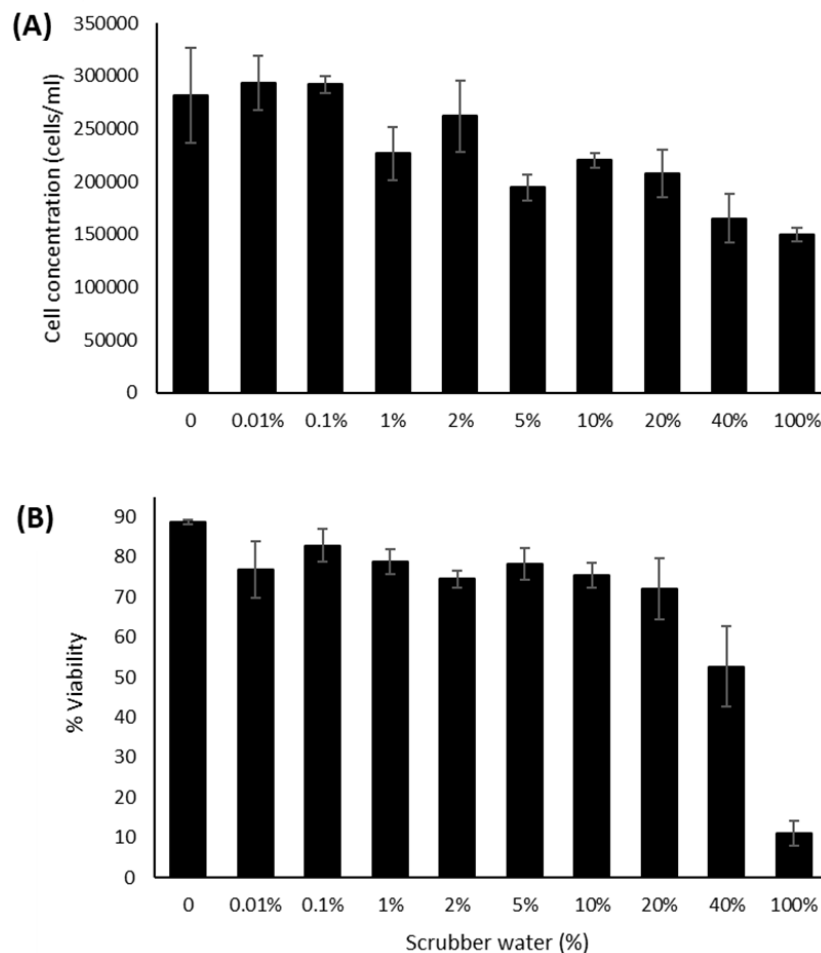


FIGURE 1: The effect of scrubber water on (A) the cell concentration and (B) mortality of the marine algae, *Tetraselmis suecica*, over a 96-hour period. Values plotted indicate the mean $\pm$ SD ($n=3$).

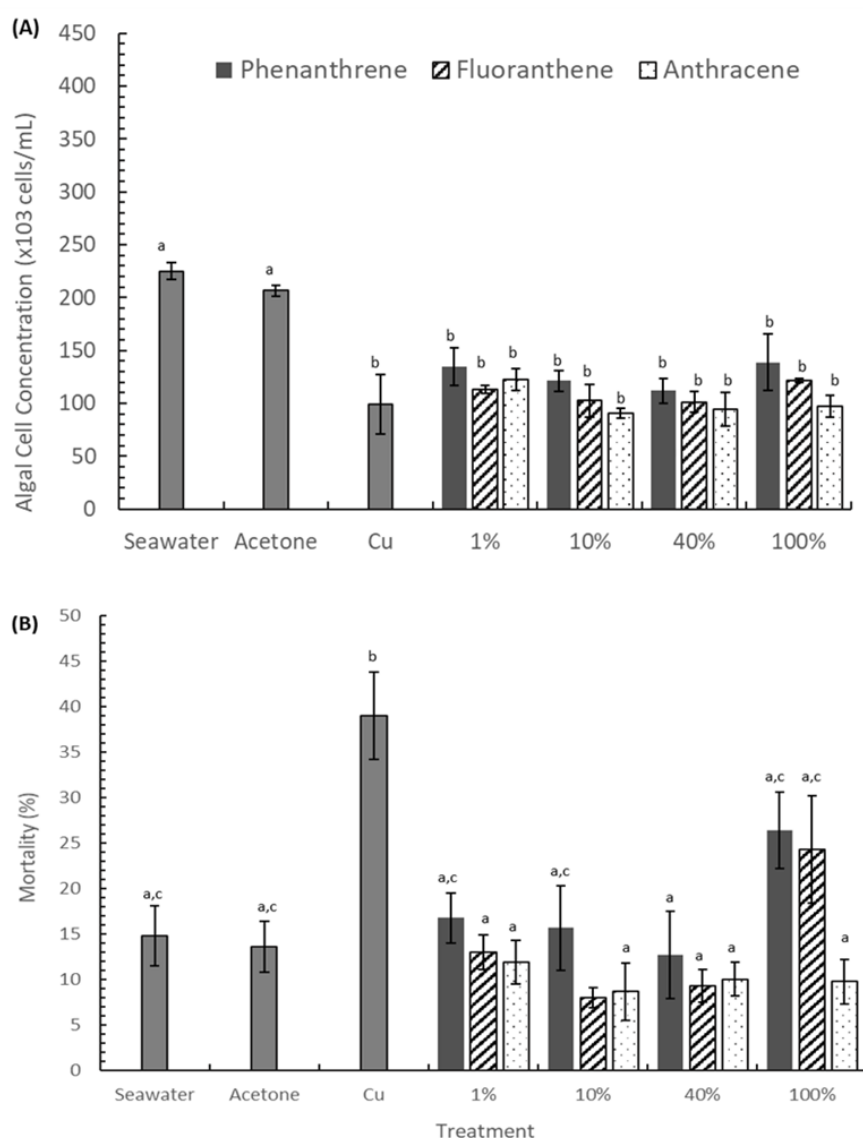


FIGURE 2: The effect of the polycyclic aromatic hydrocarbons (PAHs), phenanthrene, anthracene and fluoranthene on (A) the cell concentration and (B) mortality of the marine algae, *Tetraselmis suecica*, over a 72-hour period. Values plotted indicate the mean $\pm$ SD (n=3). Letters represent significant differences ($p < 0.05$) between treatments, whereby if bars share the same letter, they are not significantly different.

combinations illustrating this large variation in concentration has a significant impact on their toxicity. However, the toxicity of the various intermediate concentrations differed between the PAH combinations. The toxicity of both the A+F and F+P mixtures was significantly different between the 1-10%, 1-40%, and 10-100% treatments. This suggests there is a significant relationship between increased concentration and toxicity, however the impact of increasing concentration appears to decrease at the highest concentrations. The toxicity of A+P is only significantly different between the 1-10% treatments. This suggests that for this combination of PAHs, the greatest increase in toxicity occurs between 1-10% concentration, and the toxicity of the three highest concentrations does not differ significantly, illustrating a weaker relationship between concentration increase and toxicity, with a plateau in toxicity apparent across these higher concentrations. Finally, the A+F+P mixture shows no significant difference between the 1-10%

treatments unlike the other 3 mixtures, however, does show significant difference between all other combinations of concentrations. Hence concentration significantly impacts the toxic effects of this PAH combination above 10% with each increase in concentration resulting in an even greater toxic effect.

To assess the interactions between the PAHs examined, the extent to which the effect of the PAHs in combination exceeds the effect of each PAH considered individually can be explored, a method outlined by Rothman (1986). This extent can be measured utilising the equation below:

$$p_{03+n} - p_{01} - p_{02} - p_n + p_{00} \quad (1)$$

where:

p = probability of the outcome from 0-1 (in this case, the probability of algal mortality)

03+n = effect of the toxicants in combination

01 = effect of first individual toxicant

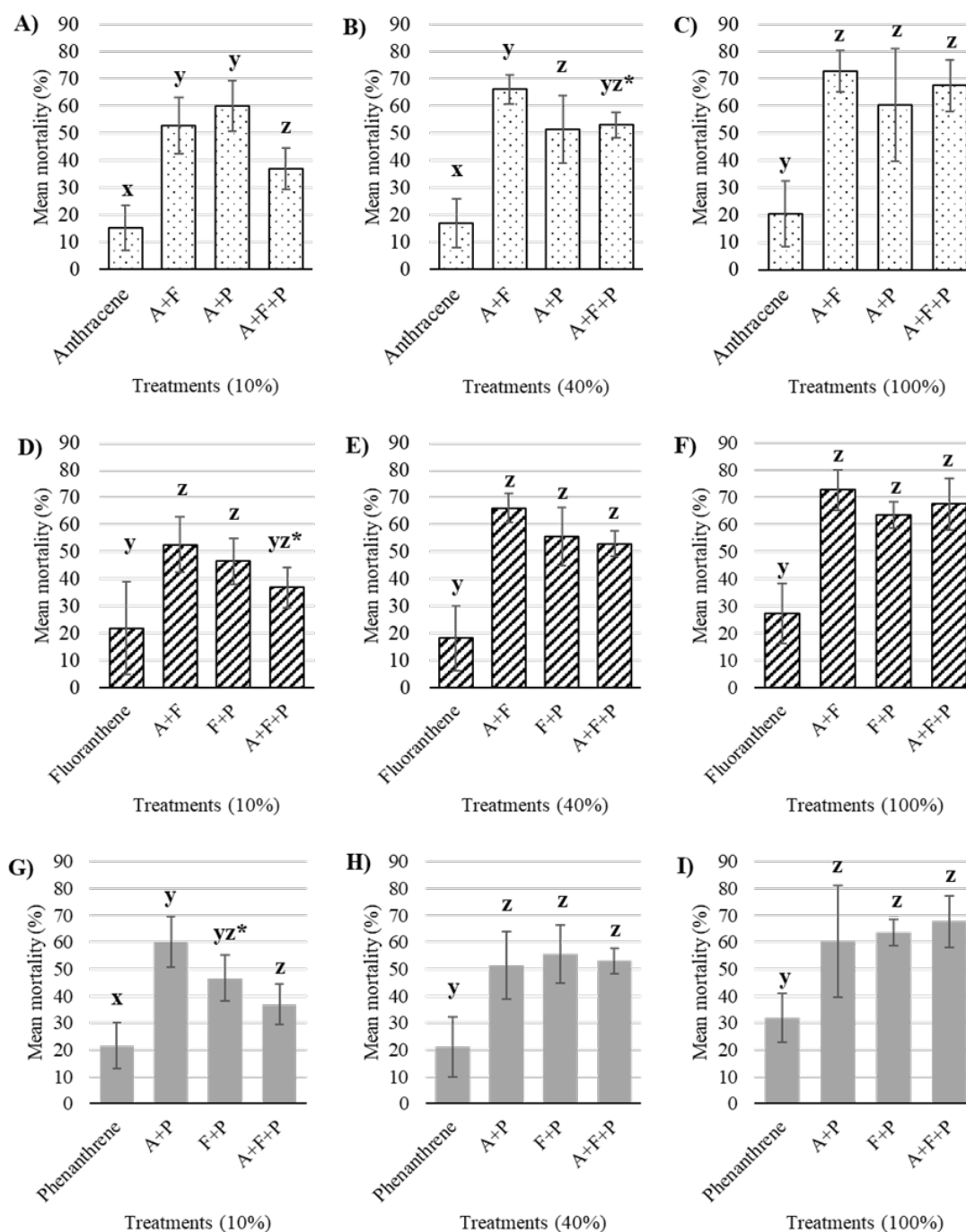


FIGURE 3: Mean percentage algal mortality of different treatments under a range of concentrations and combinations of PAHs. Anthracene and respective mixtures at A) 10%, B) 40%, C) 100%. Fluoranthene and respective mixtures at D) 10%, E) 40%, F) 100%. Phenanthrene and respective mixtures at G) 10%, H) 40%, I) 100%. Error bars = $\pm$ 1 standard deviation. Different letters signify significant difference. *These treatments are unusually not significantly different from 2 treatments which ARE significantly different from one another, thus are labelled accordingly.

02 = effect of second individual toxicant

n = effect of nth individual toxicant

00 = effect resulting from no toxicant (e.g., solvent control)

If $p_{03+n} - p_{01} - p_{02} - p_n + p_{00} > 0$, the interaction is said to be positive or "super-additive".

If $p_{03+n} - p_{01} - p_{02} - p_n + p_{00} < 0$, the interaction is said to be negative or "sub-additive".

A value of exactly 0 would represent a standard additive effect.

The interaction effect was calculated at all 4 concentrations in order to get a broader understanding of the interactions between PAHs (Table 2).

Table 2 gives insight into the nature of the interaction of PAHs within mixtures at different concentrations. A+F shows super-additive (aka synergistic) effects at all four concentrations alongside A+P and F+P at 10, 40 and 100% concentration, and A+F+P at 40% concentration. This suggests that at many concentrations, the relative toxic effects

TABLE 1: Effects of individual and mixtures of PAHs on algal mortality at the 4 relative concentrations. Kruskal-Wallis p-values at 95% are displayed with levels of significance illustrated (* = $p \leq 0.05$, ** = $p \leq 0.01$, *** = $p \leq 0.001$).

PAH (& respective combinations)	Relative concentration	Kruskal-Wallis p-value
Anthracene	1%	0.07
	10%	0.0005 ***
	40%	0.0005 ***
	100%	0.004 **
Fluoranthene	1%	0.06
	10%	0.008 **
	40%	0.0007 ***
	100%	0.002 **
Phenanthrene	1%	0.5
	10%	0.0003 ***
	40%	0.007 **
	100%	0.009 **

TABLE 2: Interaction index for the 4 PAH mixtures using equation above. >0 represents a super-additive effect, <0 represents a sub-additive effect (highlighted in grey). Percentage mean mortality was converted to probability from 0-1 for calculation purposes.

PAH mixture	Relative concentration	Interaction calculation result
Anthracene + Fluoranthene	1%	0.004
	10%	0.249
	40%	0.402
	100%	0.342
Anthracene + Phenanthrene	1%	-0.014
	10%	0.327
	40%	0.226
	100%	0.172
Fluoranthene + Phenanthrene	1%	-0.002
	10%	0.124
	40%	0.257
	100%	0.137
Anthracene + Fluoranthene + Phenanthrene	1%	-0.186
	10%	-0.432
	40%	0.061
	100%	-0.028

of the PAHs are greater in combination than when tested individually. However, this finding was not universal with both A+P and F+P at 1%, and A+F+P at 1, 10, and 100% concentrations showing sub-additive (aka antagonistic) effects.

2.3.3 Effect of exposure to heavy metals on cell concentration and mortality of *T. suecica*

For cell viability, the negative and solvent controls were non significantly different (Dunn Test: $p = 0.76$) whilst they were both significantly different to the positive control (Dunn Test negative-positive: $p = 0.0054$; Dunn Test solvent-positive: $p = 0.0086$).

Significant differences in cell density among the treatments for 100, 40, 10 and 1% of the scrubber effluent metal concentrations were observed. The cell density for metals which had concentrations equal to 100% of scrubber effluent were significantly different from the negative control (Kruskal-Wallis: $n = 6$, $p = 0.001624$). All the treatments had lower cell concentrations than the negative control (from ~100,000 to ~40,000 cells/ml) but weren't significantly different from one another (Figure 4).

The cell density of treatments when concentrations were at 40% of the reported scrubber effluent concentration were significantly different from one another (Kruskal-Wallis: $n = 6$, $p = 0.000403$). All the 40% treatments had significantly different and lower averages compared to the negative control, except Cu (Dunn test Cu-negative: $p = 0.287$). Whilst observing the treatments, V had the lowest cell density (33,000 cells/ml) but was not significantly different to Zn (48,500 cells/ml; Dunn test V-Zn: $p = 0.029$) or Ni (52,100 cells/ml; Dunn test V-Ni: $p = 0.029$). V was, however, significantly different to Cu (83,125 cells/ml; Dunn test V-Cu: $p = 0.0082$). The other treatments (Ni, Cu, Zn) were not significantly different from one another.

The cell density of treatments where concentrations were at 10% reported significant differences between themselves (Kruskal-Wallis: $n = 6$, $p = 0.002471$). Ni, Cu, Zn and the negative control were not significantly different from one another. V was significantly different to both Cu and the negative control (Dunn test V-Cu: $p = 0.0015$; Dunn test V-negative control: $p = 0.0004$).

Finally, when metal concentrations were at 1% of the reported scrubber effluent concentrations, cell density of the treatments were significantly different (Kruskal-Wallis: $n = 6$, $p = 0.0420$). The only significant difference was between Cu and V (Dunn test V-Cu: $p = 0.003$). Every other treatment was not significantly different to the negative control.

Significant differences in cell viability among the treatments for 100, 40, 10 and 1% of the scrubber effluent metal concentrations were observed (Figure 5). Viability of cells when the metal concentrations were equal to 100% of scrubber effluent had significant differences in their average (Kruskal-Wallis: $n = 6$, $p = 0.00034$). At 100% concentrations, all of the metals had cell viability ranging from 0.35 to 0.5 compared to the negative control, which had a viability of 0.89. Whilst all of the metals were significantly different to the negative control, no significant differences were found between themselves (Dunn test: $p > 0.05$).

The cell viability of treatments when concentrations were at 40% of the reported scrubber effluent concentration were significantly different (Kruskal-Wallis: $n = 6$, $p = 0.0075$). Just like for the 100% concentration, all the treatments had significantly different and lower averages compared to the negative control. The metals had similar viabilities (~ 0.65) and were not significantly different from one another.

The cell viability of treatments when concentrations were at 10% reported significant differences between themselves (Kruskal-Wallis: $n = 6$, $p = 0.01155$). Just like the 40% and 100% concentrations, Metals had similar viabilities (~ 0.75) and were not significantly different from one another. However, they all had lower and significantly dif-

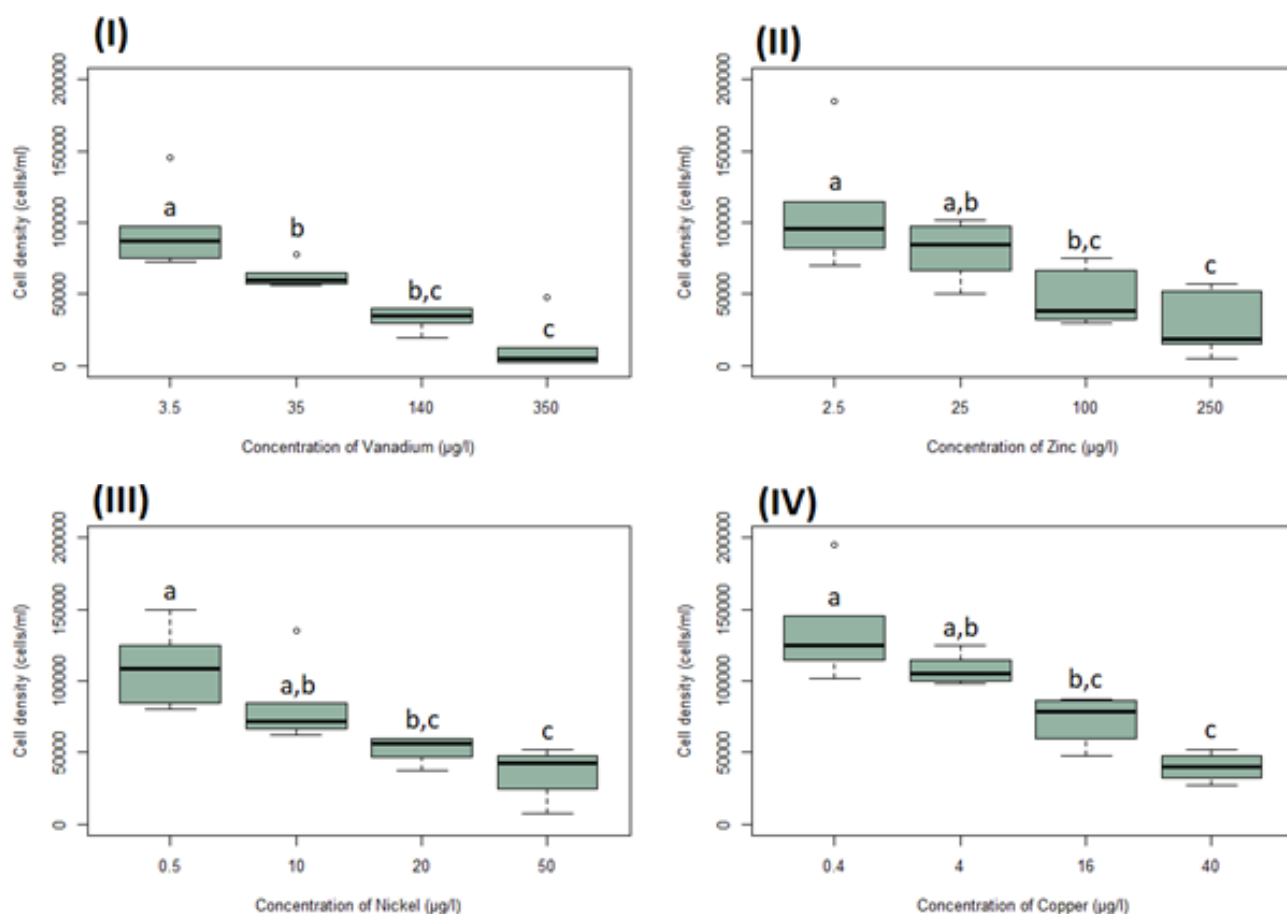


FIGURE 4: *T. suecica* average cell density after 72h of exposure to different concentrations of vanadium, zinc, copper and nickel. Average obtained from the three culture replicates per bioassay done twice. Different letters designate values that are statistically significantly different from one another (ANOVA and Tukey's test with a $p \leq 0.05$). (I). Vanadium; (II). Zinc; (III). Nickel; (IV). Copper.

ferent averages in viability when compared to the negative control.

Finally, when metal concentrations were at 1% of the reported scrubber effluent concentrations, cell viability of the treatments were still overall significantly different (Kruskal-Wallis: $n = 6$, $p = 0.01429$). Zn, Cu and the negative were not significantly different from one another. However, the negative control and Zn were significantly different to V (Dunn test V-negative control: $p = 0.0123$; Dunn test V-Zn: $p = 0.021$) and Ni (Dunn test Ni-negative control: $p = 0.007$; Dunn test Ni-Zn: $p = 0.014$). Cu, V and Ni had averages of cell viability that weren't negatively different from one another (Dunn test $p > 0.05$).

3. PART II: EVALUATION OF THE TOXICITY OF SCRUBBER WATER ON THE EARLY DEVELOPMENT OF THE BLUE MUSSEL (*MYTILUS EDULIS*)

3.1 Introduction

It has been reported that contaminants present in scrubber discharge water could induce biological effects at low concentrations (Koski et al., 2017; Ytreberg et al., 2019). Negative effects of scrubber water discharges on plankton have been reported with more toxic effects com-

pared to single compounds (Koski et al., 2017). This indicates synergistic effects of the scrubber discharge water mixture, particularly with direct - not dietary - exposure to discharge water (Koski et al., 2017). If this is a common situation for organisms at the basal levels of the food web, it could have substantial consequences for the food web structure and ecosystem functioning.

For many aquatic species, early life stages are the most vulnerable to environmental stressors, and therefore embryos and larvae are commonly used for toxicity tests to determine the sensitivity of organisms to pollutants (Connor, 1972; Martin et al., 1981). Mussels are an ideal test species due to their high sensitivity to pollutants, ecological importance, vital ecological niches, susceptibility to pollutants uptake and close connection with marine predators and human health. However, for the purposes of risk assessment, ecotoxicological tests and organisms used should be standardized.

Embryos and larvae are less tolerant to toxic compounds than adults of the same species and therefore represent the critical life stages for toxicity tests (Connor, 1972; Martin et al., 1981). For this reason, embryo-larval bioassays have proved to be very sensitive indicators of seawater contamination (Bay et al., 1983). *Mytilus* larvae has been frequently used in different studies due to its

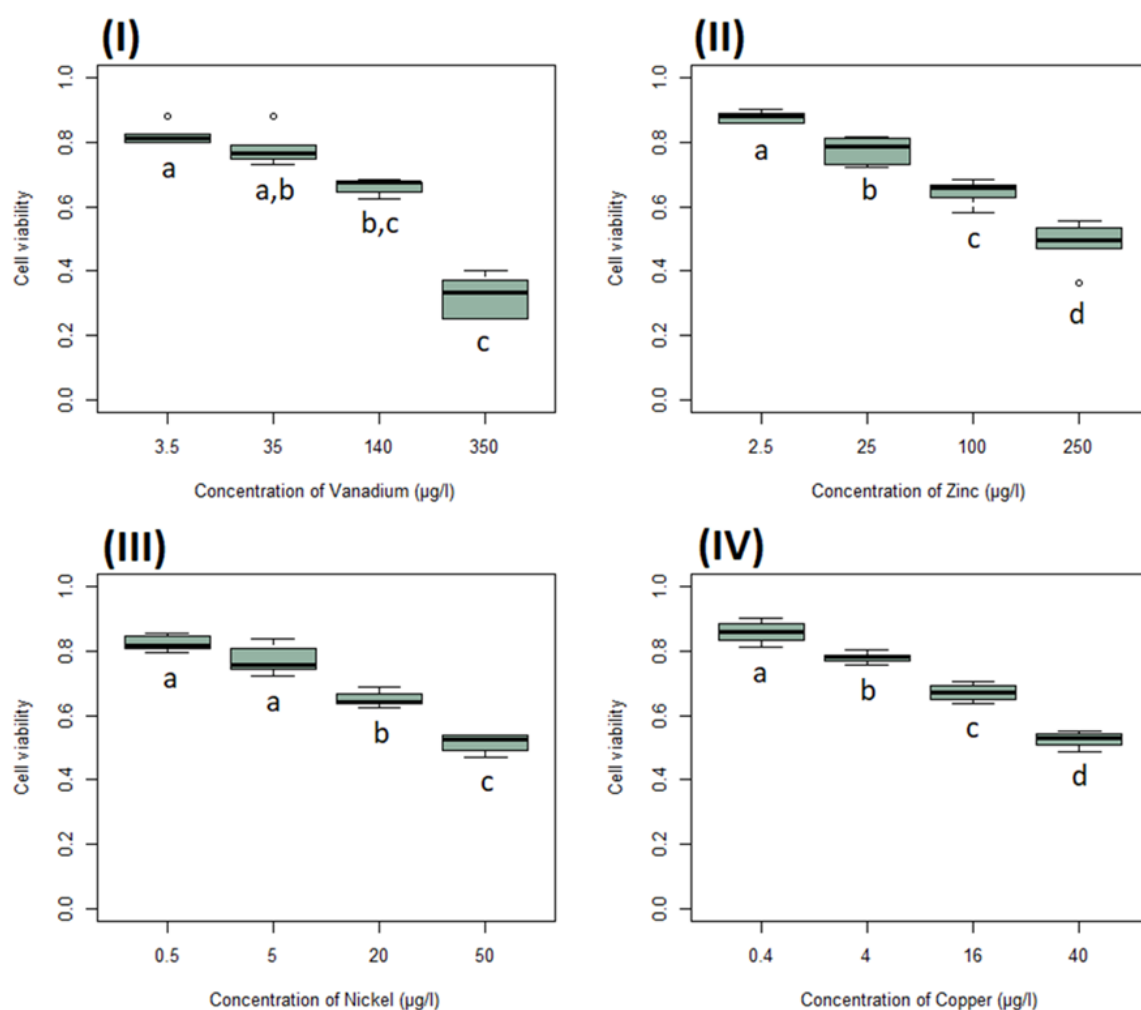


FIGURE 5: *T. suecica* average cell viability after 72h of exposure to different concentrations of vanadium, zinc, copper, and nickel. Average obtained from the three culture replicates per bioassay done twice. Different letters designate values that are statistically significantly different from one another (ANOVA and Tukey's test with a $p \leq 0.05$). (I). Vanadium; (II). Zinc; (III). Nickel; (IV). Copper.

good response to environmental contaminants (Annamaria et al., 2005; Beaumont et al., 1987).

The aim of our study using *M. edulis* larvae was to investigate the effects of open-loop scrubber discharge water on physiological parameters of *M. edulis* developing larvae in order to find the potential effect on their survival and development after exposure to this discharge water.

3.2 Methods

3.2.1 Animals

Adult blue mussels (shell length 4.15 ± 0.25 cm) were collected from the pontoon at the National Oceanography Centre Southampton (NOCS) and were transported to the hatchery. The mussels were cleaned of any epifauna, epifloral, and other attached materials with a plastic brush before acclimating them indoors in a tank with filtered Sea Water under a static condition and with continuous aeration. Eighty percent of the water was exchanged twice per week and before the addition of microalgae. Animals were kept at 14°C and salinity 32 ppt corresponding to the natural conditions of seawater at the collection site, and were fed with a combined diet of *Tetraselmis suecica* and *Isoch-*

rysis galbana. Microalgae were cultured in 20-L flasks in the algae room under standardised conditions (24 hours of continuous light at 20°C), following the standard protocol at the National Oceanographic Centre, Southampton. The subculturing process was carried out using filtered seawater enriched with nutrient solution. Cultures were supplied with 0.2 µm-filtered air, and were routinely monitored by microscope observation to confirm their growth and check for contamination.

3.2.2 Spawning induction

The mussels were induced to spawn by the temperature shock method. Six blue mussels were placed in a spawning tank containing filtered seawater at 30 ppt, with continuous aeration. Every 30 min, the mussels alternate between warm seawater tanks (25°C) and cold seawater tanks (14°C). Once the spawning of the mussels had completed, the adults were returned to the acclimation tank. Fertilised eggs were collected using a 30 µm sieve, placed and maintained in a glass beaker (5 L) filled with SW, filtered through 1-µm filter, with continuous aeration. Before starting the exposure to the different dilutions, fertilised

eggs were counted using a counting chamber under a microscope, diluted to a density of 200 eggs·cm⁻² in seawater into a flat-bottomed glass dish (See Annex 1).

3.2.3 Scrubber water discharge

The scrubber water was collected from the open scrubber system of "Catherine C" by DANAOS. Sample was taken on 19th of August 2021; location: at sea; position: 51 04.6N; 001 37.9W. The scrubber discharge water was collected from the overboard unit of the ship (scrubber outlet) directly into acid-washed glass containers; at the same time, inlet water was collected from the seawater pump system onboard (scrubber inlet). All containers were shipped to Southampton, UK, upon returning to the harbour and were subsequently stored at 4°C and in the dark until use in experiments. The scrubber water concentrations for the experiments were 0.001, 0.01, 0.1, 1.0, 2.0, 5.0, 10.0, 20.0, 40.0 and 100% of the original outlet scrubber discharge water. The dilutions were made using autoclaved and filtered sea water. Each treatment was replicated five times and the whole experiment was repeated twice.

3.2.4 Experimental design

The fertilized eggs of *M. edulis* under the different treatments were kept undisturbed in the dark without aeration and food addition for a period of 48 h at 16°C. Experiments were performed using a final volume of test solution of 20 mL. Five replicates for each treatment were tested. Due to the end of the spawning season, we had a lack of spawning animals and just one experiment could be performed. Cop-

per sulfate pentahydrate (CuSO₄·5H₂O) was used as reference substance with test concentrations included in the range 0 µg/L to 100 µg/L of CuSO₄·5H₂O. The acceptability of test results was based on negative control for a percentage of normal D-shaped larvae ≥80% (His et al., 1999).

3.2.5 Biological measurements

Larvae were inspected under a stereoscope and an optical microscope after approximately 48 h and assessed for the percentage which exhibited normal morphology (normality, D-veliger shape) from random samples of approximately 100 larvae (direct sampling, using a light microscope (Olympus CH30) with 20X magnification). Morphological deformity was determined. Deformities were considered when shell abnormalities, hypertrophy of the mantle, and/or hinge abnormalities were found.

3.2.6 Statistical analysis

Data on the percentage of developing veliger larvae were arcsine-square root transformed before analysis. ANOVA or the Student t-test were used to determine if there were significant differences in the mean percentage of normal D-larvae and larval growth rate among treatments. Data analysis was carried out using the software SPSS version 27.

3.3 Results

Morphologically normal veliger larvae developed at all treatments (Figure 6), although the relative proportion

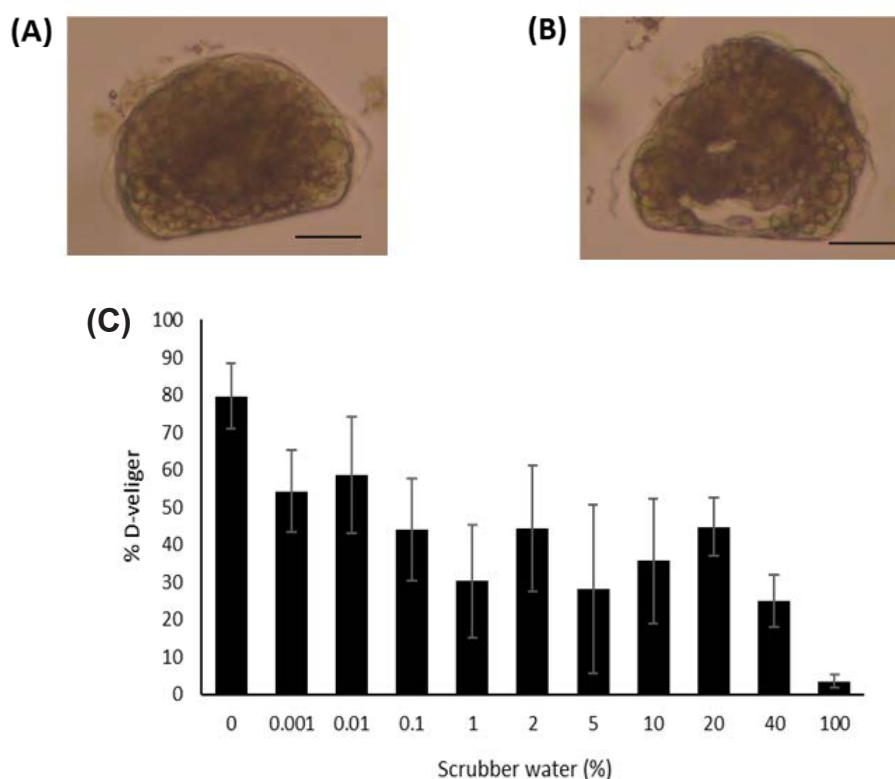


FIGURE 6: *M. edulis* (A) Normal D-shaped larvae and (B) Abnormal D-shaped larvae with hypertrophy of the mantle. (C) Percentage of D-veliger larvae after 48 h exposure of *M. edulis* fertilized eggs to different dilutions of scrubber water. Error bars represent standard deviations. Scale bar: 20µm.

statistically ($p < 0.05$) varied among treatments. Figure 6 shows the percentage of normal morphological development of veliger *M. edulis* 48 h after fertilisation at different scrubber water dilutions. All the dilutions inhibited the embryonic development of *M. edulis* in a concentration-dependent manner. The EC_{50} after 48 h exposure to scrubber water was 0.94%.

4. DISCUSSION

4.1 Effects of scrubber water on *Tetraselmis suecica* and *Mytilus edulis* larvae

The results demonstrate that the normal *Mytilus edulis* larvae development can be affected by exposure to scrubber water at different dilutions. A high sensitivity of embryo-larval stage of the blue mussel larvae was found. This high sensitivity has been reported for *Mytilus* by other authors confirming the use of this species as a bioindicator useful for ecotoxicological testing focused on ecological risk assessment (Boukadida et al., 2016). On the other hand, the high LC_{50} value obtained for *T. suecica* exposed to scrubber water indicates that this species is more tolerant to this kind of pollution. Different sensitivities can be expected between species (Croxtton, Wikfors & Schultenbrandt-Gragg, 2015) making a multispecies approach fundamental for the purposes of risk assessment and environmental management. Marine invertebrates are particularly sensitive to environmental pollutants. Exposure to scrubber water can lead to physiological stress, impaired reproductive success, and even increased mortality rates (Koski et al., 2017; Thor et al., 2021; Picone et al., 2023). The acidic nature of the wash water can disrupt shell formation in mollusks by altering calcium availability (Gazeau et al., 2013), while the presence of toxic metals and PAHs can interfere with enzyme function and cause oxidative stress in a variety of invertebrate species (Letendre et al., 2011; Vlahogianni & Valavanidis, 2007; Vitorino, 2023). Given their role as key components of marine ecosystems and their position in the food web, negative impacts on invertebrates from scrubber discharge can have cascading effects on biodiversity and ecosystem health, making it crucial to carefully manage and regulate scrubber water emissions in marine environments.

4.2 Effects of PAHs on algae growth and mortality

The results demonstrate that algal growth was impacted by exposure to PAHs, where exposure to all concentrations of each PAH resulted in a reduction in cell concentration. This is consistent with the responses shown in many other studies, whereby PAH exposure results in reduced cell densities and growth inhibition (Gala & Giesy, 1994; Jiang et al., 2002; Šepič, Bricelj & Leskovšek, 2003; Wang, Zheng & Meng, 2008). However, this did not occur in a dose-dependent manner, as cell growth did not significantly differ with increasing toxicant concentration. This contrasts with the majority of the literature, where growth is inhibited in a dose-dependent manner to PAH exposure (Aksmann & Tukaj, 2004, 2008; Wang, Zheng & Meng, 2008).

The apparent lack of toxicity of PAHs presented in this study could be due to the fact that toxicity assessments

using marine algae can be more challenging compared to assessments using freshwater species (Ebenezer & Ki, 2013). The tolerance of *T. suecica* to chemical exposure likely affected the toxicity of these chemicals. *T. suecica* specifically has been shown to be more tolerant to chemical stress, due to its ability to bioaccumulate higher concentrations, compared to other species of marine algae (Ebenezer & Ki, 2016). For example, *T. suecica* has the ability to withstand total PAH concentrations approaching 400 $\mu\text{g/L}$ (Garr, Laramore & Krebs, 2014), which is multiple orders of magnitude greater than the concentrations used in this study. It has been reported that marine algae could have a higher tolerance to toxicant contamination compared to freshwater species, and other species such as invertebrates, especially at lower concentrations (Gala & Giesy, 1992; Ebenezer & Ki, 2013).

Cell size has also been shown to be an important factor in determining PAH toxicity in phytoplankton, with smaller cells being the most sensitive to chemical exposure (Echeveste, Agustí & Dachs, 2011). This is a result of smaller cells having an increased capability to incorporate contaminants into their cells, due to their higher surface to volume ratio (Echeveste, Agustí & Dachs, 2010). This has been demonstrated by the LC_{50} of phenanthrene ranging from 30.2 to 189 $\mu\text{g/L}$, depending on the cell size of the phytoplankton species (Echeveste, Agustí & Dachs, 2010).

Another factor that is likely to have significantly affected the toxicity of the PAHs in this study is the light conditions. Treatments were stored in an incubator where they were exposed to standard light conditions. However, many studies have shown that the toxicity of PAHs increases upon the exposure to UV light, with the sensitivity of organisms increasing two to three-fold under UV light exposure (Grote, Schüürmann & Altenburger, 2005; Okay & Karacik, 2007). The growth rate upon exposure to anthracene of the green algae *Selenastrum capricornutum* was inversely correlated to the intensity of UV-A radiation, with EC_{50} ranging from 3.9 to 37.4 $\mu\text{g/L}$ (Gala & Giesy, 1992). This is due to photomodification, which causes the formation of multiple oxygenated products which are typically more toxic than the parent PAH, and photosensitisation; this causes the generation of oxygen species, resulting in oxidative damage (Aksmann & Tukaj, 2004). Anthracene is predicted to be one of the strongest and most efficient photosensitisers, and its photomodification results in the formation of more than 20 products, the majority of which are quinones (Aksmann & Tukaj, 2004, 2008). Due to their similarity with naturally occurring components of living cells, they can often disrupt cellular processes, such as the electron transport chain, resulting in impaired photosynthesis (Tukaj & Pokora, 2006). This in turn leads to the production of reactive oxygen species production (Aksmann & Tukaj, 2004; Tukaj & Pokora, 2006). Therefore, this study likely underestimates the toxicity of the PAHs tested at the chosen concentrations, especially anthracene, as in the marine environment, these chemicals will be exposed to UV light, thus increasing their toxicity. Analysis has estimated a threshold of photo-induced toxicity as low as 1.5 $\mu\text{g/L}$ (Gala & Giesy, 1992), therefore the concentrations which occur in this study are likely to result in more significant

effects on morality and growth, when combined with UV-light exposure.

The PAHs tested in this study are all lower weight PAHs, and are therefore more volatile in nature (Croxtton, Wikfors & Schulerbrandt-Gragg, 2015). This implies that evaporation needs to be considered, which could have resulted in lower toxicity of PAHs, as they may have evaporated at a higher rate within the crucial period for their toxicity.

Scrubber water is known to comprise of a multitude of pollutants, including various PAHs, metals, nitrates, and is also highly acidic (USEPA, 2011). Exposure to a multiple stressors has the potential to cause toxic effects, despite individual contaminants being below levels of concern (Beyer et al., 2014). For example, when testing the toxicity of scrubber effluent to copepods, none of the contaminants occurred at concentrations that could individually explain the toxic effects observed (Thor et al., 2021). This demonstrates that scrubber water toxicity cannot be predicted by comparing the concentrations of individual pollutants to toxicity thresholds (Magnusson, Thor & Granberg, 2018).

Exposure to multiple PAHs has been shown to have varying effects, with a combination of phenanthrene and anthracene, as well as paired exposure to anthracene and fluoranthene, resulting in synergistic effects on algal cell growth (Wang, Zheng & Meng, 2008).

The results of this study show a significant difference between the toxic effects of the 4 PAH mixtures at different concentrations, illustrating the dose-response we might have expected to see with the individual PAHs, with the greater PAH concentrations resulting in greater algal mortality. Specific studies into the effects of PAHs on plant cells have shown anthracene is especially disruptive to the photosynthetic elements of plant cells, causing distinct changes in the pigment composition (Jajoo, et al., 2014). However, the phenanthrene-containing mixtures did not appear to be significantly more toxic which may be explained by the fact phenanthrene is not as toxic as anthracene and fluoranthene, thus a greater concentration does not necessarily correlate with greater toxic effects as illustrated by greater effect and lethal concentration endpoints in several other studies (Abernethy, et al., 1986; Spehar, et al., 1999; Verbruggen, 2012). This suggests when in combination, the individual toxicity of PAHs is less important, and the combined effects/interactions dominate the toxicity of the mixture.

Additionally, we may have expected the treatments containing 3 PAHs to produce greater toxic effects at all concentrations due to the predicted interactive effects of the PAHs. However, at the highest and lowest concentrations, adding a third PAH to the mixture did not result in significantly different levels of toxicity, while unexpectedly at the 10% and 40% concentrations, the mixture containing 3 PAHs demonstrated weaker toxic effects than mixtures containing 2 PAHs.

The lack of difference between mixture toxicity at 1% may be due to insufficient concentration and chemical exposure to result in mortality, and thus equal mortality was observed across all mixtures from natural causes or potentially from exposure to the acetone solvent which was shown to result in a small increase in algal mortality (Ver-

bruggen, 2012). Furthermore, the lack of differences between mixture toxicity at the 100% concentration may be a result of excessive exposure time resulting in all 4 mixtures reporting equal algal mortality. Toxicity begins to decrease and plateau after a certain length of time, allowing less-toxic mixtures to 'catch up' with the other more rapid-acting toxic treatments (Connell, et al., 2016). In future, shorter exposure experiments may be beneficial to get a better understanding of the magnitude of toxicity of these different mixtures at the 100% relative concentration.

The effects of scrubber water on marine organisms are difficult to predict based on individual chemical action. While it is important to compare the effects of different mixtures with each other, the results above have illustrated that it is equally important to understand how the effects of these mixtures differ from the individual effects, and furthermore to understand the interactive effects/mechanisms of PAHs in combination with one another.

PAHs are considered to be general narcotics and are presumed to be similarly acting substances, meaning their toxicity in combination is expected to be additive. This additive interactive effect has been demonstrated in numerous previous studies (Swartz, et al., 1995; Landrum, et al., 2003). Deviation from additive toxicity has also been reported, with both synergistic and less than additive effects have been found in both marine and freshwater organisms (Boese, et al., 1997; Olmstead & LeBlanc, 2005), and further illustrated in the present study, indicating that the concentration addition model may not accurately represent the toxicity of PAHs in combination. Numerous studies suggest the variation in interactive effects may be due to the differing toxic mechanisms of chemicals within mixtures (Chen & Lu, 2002), however in the present study this should not be the case as the toxic mechanisms of the three PAHs are presumed to be identical. Environmental pollutants can disrupt the antioxidant enzymes system through closely related mechanisms, indirect interactions, and cascade effects that influence processes from pre-transcriptional stages to catalytic activity (Regoli et al., 2014). Chemical pathways can significantly increase the intracellular production of reactive oxygen species (ROS) and, at the same time, excessive oxyradical levels can suppress xenobiotic metabolism, leading to significant environmental consequences for organisms exposed to complex chemical mixtures. Although some studies have shown up-regulations in genes such as CYP450s, GST, SOD, GPx, CAT, and HSPs, establishing their characterization as PAH-sensitive genes (Onyena et al., 2024), the mechanisms are still unclear because PAHs can affect organisms depending on exposure time, kind, matrix, and pollutant dose. In microalgae cells, it seems like exposure to PAHs (as single compounds or in mixture) can result in a reduction in growth rates and biomass, alter photosynthetic activity, cause damage in the membrane integrity, chloroplast and thylakoids, increase lipid cellular content and antioxidant-response (Othman et al., 2023). The joint action modes of PAH mixtures are complicated and thus the reasons behind the complex pattern of the joint action mode of PAH mixtures requires further research.

4.3 Effects of metals on algae growth and mortality

Metals such as As, Cr, Co, Cu, Ni, Se, Va and Zn are essential as nutritional requirements at trace amount for many organisms but are toxic when present in greater amounts (Inthorn, 2001). We observed an elevated reduction in cell density and cell viability of *T. suicica* when exposed to [100%] of the metal concentrations found in open loop scrubber effluents. Further dilutions of the metal concentrations resulted in no significant reduction in cell density. However, V and Ni still had significant differences in average cell viability when compared to the control even at concentrations of [1%].

Dilution occurs at a higher rate in open sea systems than in inland systems. In open sea, washwater will be diluted up to 2000 times when 50 m behind the vessel, and 1750 times whilst in a port at low speed (Buhaug et al., 20006). At this dilution rate, the acute toxicity of the metals is not expected. However, ports contain high volumes of vessels at a single time which could result in an increase in overall discharge and thus increase the concentration found in the port. Whilst open water systems might not be directly affected by the scrubber effluents, closed inland systems such as harbours or rivers might be more affected by the washwater. Teuchies et al. (2020) reported that metal concentrations that originated from scrubber washwater is higher in inland waterbodies (i.e.: estuaries, harbours) than in open marine systems. For example, it is expected that the concentration of V could increase by 46% when in docks. The overall concentration of these metals depends on the speed and manoeuvre/activity of the vessel. Lower use of fuel results in a lower volume of scrubber water discharge, and vice versa.

Whilst studies can be found regarding the EC50s of *T. suicica* with some of these metals such as copper (Davarpanah and Guilhermino, 2015), not all of the metals have their EC50s reported. Examining the combined effect of these metals on *T. suicica* could help paint a better picture of the overall effects of scrubber water.

The co-exposure to metals and PAHs has frequently reported synergistic action, with nearly 45% of research on their combined toxic effects demonstrating synergistic mortality (Gauthier et al., 2014). Multiple studies have shown additive or synergistic effects of anthracene and cadmium, both of which are found in scrubber water, on the growth of algal species (Tukaj & Pokora, 2006; Pokora & Tukaj, 2010; Baścik-Remisiewicz et al., 2011). A synergistic effect between copper and a mixture of PAHs has been reported, which adversely affected the development of kelp species (Espinoza-González et al., 2021). It is believed that the effect of PAHs on membrane integrity influences permeability to metals, and that metals inhibit the detoxification of PAHs (and vice-versa), therefore resulting in increased toxicity upon combined exposure in aquatic systems (Gauthier et al., 2014). However, the combined action of PAHs and metals can depend on the concentrations, as the effects of cadmium and phenanthrene switches from synergism to antagonism with increasing concentrations (Zhu, Lu & Chen, 2008). Therefore, further research is required to understand how the contaminants present

in scrubber interact to affect their toxicity on the marine environment.

Overall, our results have given reason for concern regarding the discharge of scrubber water due to the potential detrimental impacts to the *Tetraselmis* species tested. As previously mentioned, this algae species was used as it is widely distributed and it not only plays its own important ecological role, but also represents the role of a multitude of other algal species. In the same manner, invertebrate larvae together with algae species form the basis of a food chain/web, occupying the position of producer. This role makes them immensely important as they act as a food source to numerous aquatic organisms, which in turn are eaten by larger organisms further up the food chain. We have identified the toxic potential of a handful of components of scrubber water discharge resulting in increased mortality of the test algae species and abnormal development of blue mussel larvae. On a larger scale, such as the increasing use and associated washwater discharge of scrubbers worldwide, these observations have the potential to massively impact marine food webs and subsequently marine ecosystems. Even at low concentrations, the discharge of scrubber wash water has the potential to effect marine organisms via the processes of bioaccumulation and biomagnification. The results report a lower mortality rate at lower PAH concentrations, however the hydrophobic character and low biodegradability of PAHs cause accumulation in organisms beyond their concentration levels within the environment (Van der Oost, et al., 1988). Those algae which don't die at low PAH concentrations still have the potential to absorb or take up the toxic PAHs, meaning, if consumed in large volumes, PAHs can accumulate within higher consumers (Alexander, 1999) and transfer these pollutants through the food chain.

The concentrations *T. suicica* were exposed to in this study represent the PAH levels found in scrubber effluent from a single ship. However, marine organisms will be exposed to scrubber discharge from multiple ships, as their installations increase. One study found that 30 scrubber-equipped ships emitted approximately 35 million tonnes of scrubber water in one year, which was set to increase to 47 million tonnes as a result of the introduction of the global sulphur cap (Comer, Georgeff & Osipova, 2020). Furthermore, coastal regions often already have higher background concentrations of pollutants, as they are susceptible to larger inputs of PAHs from other sources, including atmospheric deposition, direct spillage of petroleum from vessels and industrial emissions (Wolska et al., 2012; Cerezo & Agustí, 2015). The drastic rise in the application of scrubber technology on marine vessels may result in the increased accumulation of contaminants to already polluted areas, which in the long term may see PAHs, and other pollutants, reaching toxic concentrations (Kjøholt et al., 2012; Ytreberg et al., 2019). Additionally, coastal marine environments are home to some of the greatest biodiversity and ecologically protected areas, which can be highly sensitive to chemical stress (Thor et al., 2021). Therefore the increase in substances such as metals and PAHs, from scrubber effluent, will likely have an adverse effect on these sensitive species and habitats (Lange &

Markus, 2015). Therefore, as a result of a combination of these factors, scrubber water may present a greater threat than this study initially suggests, and its effects on the marine environment should not be dismissed.

Although this study provides valuable insight into the potential effects of scrubber effluent, further experimentation is required to gain a better understanding of scrubber water toxicity. Additional studies should investigate the interaction between multiple pollutants identified in scrubber water, as mentioned above, due to their combined action having the potential to be more toxic than the exposure to single pollutants. Furthermore, the effects of the components of scrubber water, and their mixtures, need to be tested on more sensitive species.

More sensitive techniques are proposed using marine microalgae species such as flow cytometry to determine cell density, changes in cell size, formation of ROS, Mitochondrial membrane potential, and changes in DNA content. Additional experiments exposing blue mussel and sea urchin larvae to scrubber water will be carry out. This includes more experiments to confirm EC₅₀, fertilization success, and reduce the range of scrubber water concentrations to clarify the effects obtained below 0.94%.

5. CONCLUSIONS AND RECOMMENDATIONS

Wastes from shipping may be very damaging at local, regional, and global scales, particularly in terms of impacts on climate change and biodiversity. The use of both closed- and open-loop scrubbers on marine vessels is increasing rapidly, resulting in the discharge of large volumes of potentially toxic scrubber water into the marine environment. This study has successfully evaluated the cumulative and interactive effects of multiple pollutants in scrubber wash-water to provide a more comprehensive understanding of its true impact on aquatic ecosystems. We assessed the toxicity of a small proportion of PAHs found within scrubber water, which as a group, constitute an even smaller proportion of the total number of pollutants found within scrubber water. Selected metals were tested in conjunction with PAHs to understand the potential environmental impacts on key organisms. The characteristics of discharged water, such as its pH and temperature, further complicate the combined effects of chemicals, and thus the full scrubber water toxicity. Despite only testing a very small proportion of the chemicals found within scrubber water, this study has demonstrated increased algal mortality rates when exposed to concentrated scrubber water, PAHs in combination (and more so than when exposed to individual PAHs) and selected shipping-related metals. Blue mussel larvae were highly sensitive to scrubber water showing effects on development. This raises concern over the impacts scrubber water may have on the marine environment both now and in the future as the installation of scrubbers to vessels continues to increase. If scrubbers continue to be considered an effective and preferred method to meet emission restrictions, then the need for significant investment in technological advances, improved monitoring protocols, and evidence-based regulations on scrubber water discharge limits and restrictions is extremely urgent to pre-

vent further potentially irreversible damage to the marine environment.

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SOLID WASTE POLICIES AND CLIMATE CHANGE – THE CASES OF FEDERAL BRAZIL AND CANADA

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ABSTRACT

The objective of our paper is to scrutinize the ongoing dynamics surrounding waste and climate policies within the federal democracies of Brazil and Canada. Our research design is guided by four primary inquiries. Firstly, we look at the integration of the circular economy concept within waste and climate policies. Secondly, we explore the extent to which concerns regarding solid waste are assimilated into climate mitigation and adaptation strategies. Thirdly, we assess the degree to which social inclusion is upheld as foundational principle within climate and waste policies. Lastly, we investigate the governance practices that have been put in place to promote effective intergovernmental and state-society cooperations. Initially, we illuminate the commonalities and disparities between Brazilian and Canadian federalism, emphasizing the challenges of policy coordination—a critical prerequisite for a smooth integration and successful execution of solid waste and climate policy initiatives. In the core section of our paper, we adopt a comparative lens to analyze both policy domains, focusing on (a) institutional frameworks, competencies, and the characteristics of the policy-making processes in each country; (b) legislative measures, programs, plans, and policy instruments to elucidate the policy fields and the linkages between them. We conclude our paper by juxtaposing the experiences of both countries and suggesting potential ways for mutual learning to improve federal democratic responses in tackling these complex and interlinked challenges.

1. INTRODUCTION

Establishing links between solid waste management (SWM) and climate policy does not seem obvious at first glance, yet they are intrinsically related. In the most recent IPCC reports, SWM is mentioned as important within urban services and connected to infrastructure. While recognized as a “short-term adaptation option” (IPCC, 2022, p. 46 (ch. 15)), waste management does not receive sufficient attention as a long-term structural requirement within climate policy. On local and regional scales, lacking waste collection, improper disposal practices, and unsuitable facility locations have negative effects both on the environment and the climate. Globally solid waste plays a significant role in exacerbating climate change, to a large extent linked to the greenhouse gas emissions from the organic fraction in waste dumps and landfills (Kaza et al., 2018) as well as from the plastic content in incineration.

According to Yang et al. (2023), the circular economy focuses mainly on conserving resources and optimizing

the energy and material cycles across various levels: at the micro-level, involving enterprises and consumers; at the meso level, where economic agents collaborate symbiotically; and at the macro-level, encompassing cities, regions, and governments. They consider that the increase in solid waste and the worsening of the climate crisis is due to global industrialization and excessive dependence on non-renewable energy sources. The authors assume that by promoting the circular economy, carbon emissions can be reduced by 45% by 2030 and carbon neutrality can be achieved by 2050. They analyzed circular economy strategies focusing on waste management, climate change, energy, air and water quality, land use, industry, food production, life cycle assessment and economic routes and noted that increasing the use of bio-based materials is a challenge in terms of land use and land cover. They pointed out that carbon removal technologies are still not economically viable and conclude that although circular economy strategies can be implemented in various sectors, such as industry, waste, energy, buildings and transport, life cycle

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assessments are necessary to optimize new systems. Other studies suggest that integrating waste management into future smart cities with complete product life cycles could be a potential step towards achieving “zero waste” (Lee et al., 2021) and to achieve this goal, three steps must be taken: waste prevention, accurate waste collection and recovery of the functional value of the collected waste (Yang et al., 2023). Fang et al., (2023) conclude that using Artificial Intelligence (AI) in waste logistics can reduce transportation distance by up to 36.8%, cost savings by up to 13.35%, and time savings by up to 28.22%. Furthermore, AI would also allow for identifying and sorting waste with high accuracy, ranging from 72.8 to 99.95% (Fang et al. 2023). The conditions for effective waste management are particularly precarious “in cities of the global South where the capacity to tax or charge for services is limited, waste is often an unmanaged and unregulated environmental blight, and its management is often a physical manifestation of political and social challenges of urban governance and urban service delivery” (Myers, 2014, p. 448).

In the global debate, policy initiatives such as the promotion of zero waste and circular economy through reverse logistics incentives, and the strengthening of the bioeconomy are some of the strategies which presumably could make an important contribution to the necessary transition to a sustainable and resource-efficient economy (Cosenza et al., 2020; UNEP/UN-Habitat, 2021). Yet, the Circularity Gap Report 2019 finds that just 9% of the 92.8 billion tonnes of minerals, fossil fuels, metals, and biomass that enter the economy are in fact re-used annually (Circle Economy, 2019).

Material use and climate change are closely linked. Waste accounts for 3-5% of global greenhouse gas emissions (GGE) and is the third source of methane emissions, after natural wetlands and agriculture. CO₂ is responsible for 74.4% of GGE while methane is the second most potent greenhouse gas, accounting for 17.6% of global emissions (IPCC, 2014). Yet, most of the global GGE (62%) are released during the extraction, processing and production phases (Circle Economy, 2019). Wilson, Filho and Ramola (2023) point out that calculations by the International Solid Waste Association (ISWA) suggest that better waste and resource management could mitigate 15% to 25% of global GGE and should therefore be included in the Nationally Determined Contributions (NDCs) of all countries.

Nevertheless, frequently, SWM is still confused with waste disposal and recycling at the local level, whereas Solid Waste Policies (SWP) should be considered part of an overall strategy of “transformational, systems-level change” (Anthony, 2022, p. 7), “building net zero, circular, resilient, inclusive and just cities” (UNEP/UN-Habitat, 2021, p. 39). We anticipate at least four approaches that go beyond technical strategies for improving systems and involve enforcement through climate and solid waste policies and their interfaces:

1. Divert materials from the waste stream into a circular system, engaging citizens and various production chains.
2. Integrate solid waste concerns into mitigation and ad-

aptation strategies within climate policy.

3. Foster fair and inclusive income opportunities in waste management, particularly for socially disadvantaged groups.
4. Ensure cooperation among different governmental levels, specifically related to environmental sanitation and climate change.

In our paper we examine ongoing experiences with waste and climate policies in the federal democracies of Brazil and Canada with focus on these four approaches. To achieve this goal, we conducted a literature search on relevant policy texts, analysing them for key categories and themes and comparing them across the two countries. Our paper is structured as follows. After introducing our methodology, we address the nexus between the management of solid waste and its contribution to global warming. Then we point out some commonalities and differences between Brazilian and Canadian federalism with a focus on the challenges of policy coordination, a crucial requirement for the integration of SWP and climate policy, and its successful implementation. In the main part of the paper, we look at both policy fields from a comparative perspective, highlighting (a) institutions, competencies, and the characteristics of the policy-making process in both countries; (b) legislation, programs, plans, and policy instruments, to elucidate the policy fields and the linkages between them. We conclude the paper comparing the experiences of both countries and highlighting possible mutual learning opportunities for federal democratic action to deal with such wicked problems.

2. RESEARCH METHOD

Data gathering for this research involved an online search for relevant legislation, programs, plans, and policy instruments as well as documents that describe the institutions, their competencies and the characteristics of the policy-making process in both countries. We consulted government websites, specialized policy databases and academic databases (Web of Science, Scopus). After sifting through the literature encountered, we applied thematic analysis: identifying, analyzing, and reporting relevant patterns or themes within our data that speak to our research question. We manually identified key categories revealing descriptive, interpretive and conceptual information of the data relevant to our research question. These categories were then organized into emerging themes across our set of texts to allow for comparison. Reading through the texts supported the review and refinement of these themes to ensure we accurately represent the identified codes and the overall data set. Our data analysis was an iterative process with constant comparison, whereby new data was compared to the existing categories. In this research phase we synthesized the findings in a spreadsheet, which helped visualize and organize the information (see Annex). We then explored the relationships, themes, and patterns that had emerged from the comparative analysis (Bauer & Gaskell, 2000).

3. THE NEXUS BETWEEN WASTE AND CLIMATE CHANGE

In many ways solid waste is one of the drivers for climate change; some aspects are already well known while others are less obvious or hidden and happen even before something is discarded. As with every material, before becoming a consumer object, resources have to be extracted and transported to industrial plants where they are transformed into the ingredients for manufacturing the final product, which then is transported to close and far retailers and consumers, generating emissions and waste with all its climate impacts along the chain (Pollans, 2021). These extraction and production processes involve the use of energy, affecting natural environments by contaminating soil, water, and air and generating GGE, leaving behind contaminated trails of what Lepawsky (2022) calls “discard-scapes”.

For example, oil is the primary ingredient necessary to produce the polymers required to make plastics and during its extraction waste is generated, environmental impacts occur and greenhouse gases are emitted. The extraction and processing/refining of oil and gas consumes large volumes of water, produces wastewater and contaminates surface and groundwater with benzene, methane, radiation and other chemicals, clearly affecting the environment and climate. Long-term pollution and habitat destruction affect wildlife, biodiversity and consequently also reduce ecosystem services, thereby avoiding their mitigating effects on climate change (Finer et al., 2008).

The case highlights possible hidden climate impacts not considered in the debate that links waste to climate change, generally limited to analysing different waste management options, including recycling, composting, landfilling, incineration (Kaza et al., 2018), or strategies less well reported such as open burning and waste export.

While recycling is presented as the basic solution to our waste problem, it is not exempt from GGE. The main source of greenhouse emissions linked to the recycling of materials like plastic and metal is the usage of energy (Astrup, Fruergaard, & Christensen, 2009a; Damgaard et al., 2009). Direct emissions refer to activities and processes that take place on-site at the Material Recovery Facility (MRF), whereas indirect emissions refer to the processes and activities that are affected by the operations at the MRF upstream and downstream (Damgaard et al., 2009).

Incineration as waste disposal strategy is portrayed as consistent with the circular economy idea based on energy recovery (Di Foggia & Beccarello, 2021; Kaza et al., 2018; Lausset et al., 2017; Malinauskaite et al., 2017). However, due to high contents of fossil energy in municipal solid waste (plastics, synthetic textiles, synthetic rubber, etc.) waste-to-energy is regarded as a mostly fossil energy source, thus a source of GGE (Zero Waste Europe, 2020).

Astrup, Møller & Fruergaard (2009) posit that incineration is characterized by the almost total oxidation of organic materials to CO₂ and that it is possible that some waste components are not going to be completely oxidized and may end up being released as CO₂ and other hydrocarbons. Furthermore, nitrous oxides (N₂O) may be

released, depending on the flue gas cleaning system of the incinerator (Chen & Lo, 2016). In comparison, Chen and Lo (2016) show that landfill sites generate fewer GGE than waste-to-energy facilities. According to Yifei Ma et al. (2024), incineration has become the largest source of GGE from the waste sector in China, with plastic representing the largest impact.

A particularly powerful greenhouse gas is methane (CH₄). Despite its brief atmospheric lifetime, methane has a global warming potential (GWP) of 72 over 20 years and 25 over 100 years, where a GWP of 72 means that 1 ton of methane released today is equivalent to 72 tons of CO₂ (O'Neill, 2019). GGE are measured throughout a product's manufacturing and use, but they are also connected to the techniques used to handle waste at the product's end of life (Braschel & Posch, 2013). Methane emissions from landfills have historically been the main source of GGE in the waste sector with a contribution rate ranging from 69% to 95% (Bogner et al., 2008).

Open burning of waste, which is common in low-income countries, is a low-temperature combustion method that emits large levels of contaminants, especially black carbon, a by-product of incomplete combustion, which also has an adverse effect on the environment (Reyna-Bensusan et al., 2018). Overall, we know that methane, carbon dioxide, nitrous oxide, and black carbon are some of the greenhouse gases connected to solid waste (UNEP/ISWA, 2015).

The waste-energy-climate nexus that becomes evident in waste policy, reveals, on the one hand, the associated technical and systemic complexity and, thus, the need for extending technical-scientific knowledge, but, at the same time, the strong economic vested interests that exert pressure on political decision-makers. Therefore, there is a need for strong governmental regulatory capacity to provide incentives and ensure a CO₂-reduced and climate-mitigating waste policy, and to establish clear responsibilities between the national, state, and municipal governments.

4. BRAZILIAN AND CANADIAN FEDERALISM: COMMONALITIES AND DIFFERENCES

It seems reasonable to assume that the state's capacity to act and to respond to challenges like environmental and climate crises is dependent on the effectiveness of the general political order. Particularly in federal states, it is the division of power between the constituent units, the rules and institutional arrangements that shape the federal dynamics (Benz & Broschek, 2003). In view of climate change, the degree of cooperative behaviour is considered crucial for the effectiveness of climate policies (Jordaan et al., 2019).

Brazil and Canada are both federal democracies with different historical traditions, sharing the common challenge of how to simultaneously “maintain and generate both unity and diversity” (Elazar, 1987, p. 64), which has been a common necessity of both federations due to their large territorial extension, their significant socioeconomic heterogeneity and cultural diversity (Abrucio, 2005; Lederman, 1975). Considering the institutional framework is

therefore crucial to understanding the different federal dynamics and governance patterns that manifest themselves in the waste and climate policies of the two countries and which are analysed comparatively in the next section.

In Brazil the country's democratic Constitution from 1988 aimed to reconcile the restoration of local autonomy – in order to guarantee diversity and favour citizen participation – with the promotion of greater territorial equity through federal authority – in order to maintain unity and ensure national governability (Abrucio, 2005; Souza, 2019; Watts, 2006). Municipalities even gained the status of federative entities. Since the 1990s, the federal government has increasingly taken the lead in shaping public policy within the framework of shared public responsibilities. National public policy systems were created, encouraging local authorities to adapt to the federal policies, particularly through financial incentives, which has contributed significantly to improving public policy indicators (Arretche, 2012). In their evaluation of Brazilian federalism, Souza and Fontanelli (2021, p. 139) conclude optimistically that “the design of Brazilian federalism contemplates institutional instruments capable of managing the tensions and dilemmas of the federation”.

However, it is important to recognize that under Jair Bolsonaro's right-wing government (2019-2022), intergovernmental relations have deteriorated sharply, leading to a general decline of democratic practices (Avritzer et al., 2021), and, specifically, the dismantling of environmental policy (Hochstetler, 2021a, 2021b). The first months of the government of Luiz Inácio Lula da Silva, though indicate important attempts to re-establish the former cooperative practice of governance, in the form of a kind of “executive federalism” with the president seeking to reassume direct dialogue with state governors and mayors in order to pacify the previous tense intergovernmental relations.

In Canada, the functioning of the institutional political order laid down in the 1982 Constitution has also been designated as “executive federalism”, defined as “the processes of intergovernmental negotiation that are dominated by the executives of the different governments within the federal system” (Watts 1989, quoted in Howlett & Migone (2019, pp. 64-65)).

It is the strong federal executive, sometimes even referred to as “an elected dictatorship” (Sayers & Banfield, 2013, p. 186), which should ensure the executive control of the political agenda. However, despite the institutionally strong position of the federal government, it has proved unable to unify the institutional-political practices and public policy-making. Different views on how a federation should work make it extremely difficult to come to a more consensual, consolidated decision-making and implementation process of public policies (Schertzer, 2020).

The “lack of jurisdictional clarity” (Schertzer, 2020, p. 165) in a country marked by extreme societal diversity contributes to what Howlett and Migone named “institutionalised ambivalence” – meaning a system in which political institutions in the country often make it difficult to clarify and achieve objectives, leading in turn to ‘ambivalent’ policies with often unclear impacts and effects” (Howlett & Migone, 2019, p. 142).

Hence, as highlighted by Cameron and Simeon (2002), Canadian federalism has undergone important variations concerning intergovernmental changes. In the last two decades, “oscillations between collaborative and conflictual relations in very short spans of time” (Schertzer, 2020, p. 175) has been observed, influenced by the specific political style of each Prime Minister, as well as by the general political configurations.

The following section analyses and discusses how these specific institutional conditions and governance practices have shaped waste and climate policies in the two federations.

5. SOLID WASTE AND CLIMATE-RELATED POLICIES AND LEGISLATION SPECIFIC TO BRAZIL AND CANADA

This section centres on the data found on the current policy and legal approaches to climate change and waste management in both countries. The focus will be primarily on federal and overarching jurisdictions. The following table (Table 1) provides a brief overview of the constitutional authorities and regulations related to solid waste in Brazil and Canada. A more detailed presentation with the description of these policies can be found in the attached Appendix.

5.1 Waste and Climate Policies and Plans in Brazil

In Brazil, the National Policy of Climate Change (PNMC), Federal Law 12.187 from 2009, and of Solid Waste (PNRS), Federal Law 12.305 from 2010 (Brasil, 2009; 2010a) and their respective regulatory decrees (Brasil, 2018; 2022a) and National Plans (Brasil, 2016; MMA, 2022) are in force.

In 2021 Brazil would rank fifth among the eight largest greenhouse gas emitters in the world (UNEP, 2022). It was estimated that from all land use sectors the Forestry represented 49%, Agriculture and Livestock 25%, Energy 18%, Industrial Processes and use of products 4% and Waste also 4% of the country's emissions (Observatório do Clima, 2023) Methane gas accounts for about 97% of emissions from the waste sector. Of the total, the main contribution is associated with the disposal of solid waste in controlled landfills, dumps and sanitary landfills (64.1%), followed by the treatment of domestic liquid effluents (28%), treatment of industrial liquid effluents (6.1%), incineration or open burning (1.7%) and, with a smaller contribution, biological treatment through composting (<1%). The impact of organic waste in dumpsites and landfills stands out for the emission of greenhouse gases, especially methane (O'Neill, 2019). In the country, about 45.3% of the waste collected is composed of food scraps and green waste (MMA, 2022).

Signatory of the United Nations Framework Convention on Climate Change (UNFCCC), Brazil has committed to regularly submit its National Communications every four years, including the National Inventories of GGE. Updates of the Nationally Determined Contribution (NDC) in 2022, confirm its commitment to reduce GGE in 2025 by 37%, and in 2030 by 50%, compared with 2005, and to achieve climate neutrality by 2050 (Brazil, 2022). Yet, the NDC does not give specific actions for the waste sector.

TABLE 1: Institutional arrangements and policies for solid waste management in Brazil and Canada (source: compiled by the authors from multiple official online sources).

Federal Level	Brazil	Canada
Constitution	Brazilian Constitution (1988)	Canadian Constitution Acts (1867 to 1982)
Policies	Federal Law on Solid Waste (Law N° 12.305/2010) Federal Law on Basic Sanitation (Law N° 11.445/2007) New Federal Law on Basic Sanitation (Law N° 14.026/2020)	Canadian Environmental Protection Act – CEPA National Parks Act (Garbage Regulations) Indian Act (Reserve Waste Disposal Regulations) Sanitary Landfilling Guidelines for Federal Agencies
Major agencies & institutions	Ministry of the Environment and Climate Change Ministry of Cities Brazilian Institute of Environment and Renewable Natural Resources (IBAMA) National Environment Council (CONAMA)	Canadian Council of Ministers of the Environment (CCME) Environment and Climate Change Canada (ECCC)
Key instruments	National Solid Waste Plan (Decree N° 11,043/2022) National Program of Reverse Logistics (Decree N° 10,936/2021) National Solid Waste Management Information System National Sanitation Plan (PLANSAB) Sector Agreements	Canadian Environmental Protection Act (CEPA) National Packaging Protocol (1989) Single-Use Plastics Prohibition Regulations (2022)
State/Provincial	Brazil	Canada
Constitution	National Constitution (Art. N° 23), State constitutions Specific laws approved by the state legislative assemblies	Provincial amendments to the National Constitution
Key policies	State solid waste policy Solid Waste plans	Environmental statutes Provincial recycling regulations Provincial deposit refund systems
Major agencies & institutions	State agencies for environmental protection and control State environmental councils	Ministry or Department of Environment
Key instruments	State solid waste plans	Provincial solid waste management programs
Municipal	Brazil	Canada
Organic Law	As federative unities municipalities have their own organic laws (local constitutions)	Federation of Canadian Municipalities (FCM)
Key policies	Municipal basic sanitation law Municipal solid waste policies	Shared constitutional authority
Major agencies & Institutions	Municipal agencies for environmental policy / control Municipal Environment Council	Local departments or agencies overseen by ECCC
Key instruments	Municipal/Intermunicipal Solid Waste Plan Municipal Sanitation Plan Municipal Environmental Fund	Specific regulation, legislations and bylaws Extended Producer Responsibility Programs (EPR) Waste Diversion Programs

The PNMC created the Brazilian Forum on Climate Change and respective governance structure (Brasil, 2009). The Forum was initially very active, but lost space and relevance during Bolsonaro's term in office, which was accompanied by a general dismantling of the environmental and climate policy (Hochstetler, 2021a).

With regard to the national inventories, the IPCC (2019) found that these do not effectively reflect the contribution of the waste sector to GGE mitigation. If the recycling and energy recovery potential were included as a contribution of the waste sector, its potential contribution would reach up to 20% of total GGE (Brasil, 2021). At the interface between PNMC and PNRS, in 2011, the Action Plan for Sustainable Production and Consumption was developed by the Ministry of Environment, whose implementation could significantly contribute to reduce emissions from waste and leverage the circular economy, but was discontinued by the Bolsonaro government.

It is up to all spheres in Brazil, to establish public policies, norms and guidelines related to waste management. The state sphere is responsible, besides the formulation of policies and norms, for the inspection and monitoring of municipal and private treatment and final disposal systems. The attribution of the waste management system,

collection, treatment and final disposal of waste is municipal. Federal conflicts have arisen as in the case of the ban of the use of plastic bags in supermarkets, adopted in some cities of the country and contested in court, with the ruling that, in matters of concurrent legislation, the municipalities have broad power to regulate local systems on issues not regulated by federal or state law.

Unlike the PNMC, the PNRS does not have a governance system at the federal level (TCU, 2016) and relegates the development of state and local plans to the States and Municipalities. Yet, the PNRS establishes several principles to guide waste management in Brazil.

First, it defines a hierarchy from non-generation, reduction, reuse, recycling, waste treatment to environmentally appropriate final disposal for all types of waste.

Secondly, it stipulated that open dumping of waste must end by 2014 - a goal which is not yet achieved. Consequently, the Federal Law on Basic Sanitation No. 14.026/2020 had to establish a new deadline depending on the characteristics and size of the municipalities, between 2020 to 2024 (Brasil, 2020).

A third principle relates to the shared responsibility for the product life cycle of producers, importers, distributors and retailers, consumers, as well as providers of public

services for urban cleaning and solid waste management (Brasil, 2010). The tasks include the collection of products and residual waste after use and their subsequent environmentally sound final destination and the obligation to structure and implement reverse logistics systems through the return of products after use by the consumer. The aim is to reach sectoral agreements for different types of waste. However, the Federal Court of Accounts (TCU) warned in its report that shared responsibility fails to specify the parties accountable for the stages of the product life cycle related to environmentally appropriate disposal, without specifying the allocation of costs for each stage (TCU, 2016).

The fourth principle prioritizes contracting waste picker organizations for municipal recycling programs and facilitating reverse logistics for the private sector (Brasil, 2010). This involves a large number of waste pickers who work independently or organized as associations, cooperatives and networks (Gutberlet et al., 2020; Pisano et al., 2022). Brazil is a pioneer in the development of organized recycling and has inspired many other countries to provide opportunities for the generation of better working conditions for informal waste pickers (Besen & Fracalanza, 2016; Nogueira Zon et al., 2020).

A decree issued by President Bolsonaro's government (MMA, 2022), in 2022, integrates the PNRS into the National Environmental Policy, with the National Basic Sanitation Policy, however, without mentioning the National Climate Policy. The Decree launches the National Reverse Logistics Program, which is of great interest to the business sector and opens doors to regulating the incineration of municipal waste in the country. This action, however, contradicts the waste management model advocated by the civil society, which is based on participatory waste management and in which waste pickers play a pivotal role.

5.2 Waste and Climate Policies and Plans in Canada

In Canada, the provinces have constitutional authority to deal with most activities relating to solid waste. Federal authority extends primarily to federal lands, federal facilities, and 'Indian lands' (lands reserved for First Nations peoples under treaties, agreements, or other legal means). The Canadian Council of Ministers of the Environment (CCME) has produced guidelines on a variety of subjects related to municipal and hazardous wastes, including composting; waste incineration, alternative fuels for cement kilns; and lubricant recycling.

Climate change is already severely impacting Canada in many different and complex ways. The country has warmed up by around twice the rate of the rest of the world, and some communities have already suffered substantial harm from this phenomenon (Greaves, 2021).

While the oil and gas (26%) as well as the transportation sector (25%) continue to be Canada's largest sectoral GGE sources, the waste sector (4%) should not be overlooked as direct contributor (Government Canada, 2020). Several steps to combat climate change by addressing waste generation have been initiated, besides reducing GGE and assisting communities to adapt to their effects. (Guo et al., 2022).

In Canada, primarily municipalities handle solid waste generated by households, businesses and institutions.

Cities also manage landfill operations, waste collection, sorting and composting facilities. Local governments can lead environmental efforts by involving households and businesses in reducing waste and greenhouse gas emissions. They can also create low-emission energy sources and utilize resources from organic waste and biosolids to enhance sustainability. Landfills are the most widespread disposal form in Canada and regulations frequently call for landfill covers and landfill gas collection.

As a result of the relative municipal autonomy in policy implementation, the outcomes can be uneven and as a negative result, emissions have not decreased in most parts of the country for over a decade (Government Canada, 2020). Municipalities are bound to existing hierarchies in jurisdiction which can become a hurdle for progressive governments. For example, in 2018 the city of Victoria produced a bylaw to ban plastic bags. This bylaw was challenged by the Canadian Plastic Bag Association and the final ruling of the Supreme Court of Canada established that Victoria had exceeded its legal authority by implementing the ban without provincial approval.

Created in 1971, the Ministry of Environment and Climate Change Canada (ECCC) is responsible for coordinating environmental policies and programs, as well as preserving and enhancing Canada's natural environment and renewable resources. With the Canadian Environmental Protection Act (CEPA), from 1999, a consistent set of environmental protection standards have been set in place throughout the country.

The country's recent Climate Plan elaborated by ECCC (2022) points to: 1) Landfill methane regulations, 2) Food waste reduction and 3) Investments in waste and recycling infrastructure, as measures to reduce GGE. Landfill methane is generated as a result of decades of landfilling of biodegradable waste which still makes up more than 60% of the waste generated by households in Canada. It is most concerning that Canada wastes more than half of its food supply, most of which could be avoided, but instead releases large quantities of methane (ECCC, 2019). To address this challenge, the Government of Canada has launched in 2020, the first national plan to reduce food waste.

Canada's experiences with GGE reduction in the waste sector concentrate on capturing methane from landfills and directing food waste away from landfills. Nevertheless, a study by Williams et al. (2022) asserts that after Canada implemented its waste management rules through CEPA, GGE are being monitored in engineered and managed active landfill sites, but not in historic landfills, like those in Montreal and many other cities. These environmental passives are not part of current policies.

Plastic waste is also considered one of the key challenges for the country. Only 9% of the approximately three million tons of plastics discarded in Canada in 2016 was recycled (ECCC, 2019). In 2018, the federal, provincial, and territorial governments approved a Canada-wide Strategy on Zero Plastic Waste (Government Canada, 2020) and endorsed two aspirational waste reduction goals: to decrease waste by 30% by 2030, and up to 50% by 2040. Furthermore, the Single-use Plastics Prohibition Regulation (ECCC, 2023) in force prohibits the manufacture, import

and sale of specific single-use plastic items (e.g., check-out bags, cutlery, foodservice ware, ring carriers, stir sticks, straws). To achieve zero plastic waste, the Government has committed to innovation and investment, including in the design and construction of recycling facilities, the development of new recovery technologies, remanufacturing processes, and product delivery, reuse and repair systems, through Canada's Plastics Science Agenda.

Since 2021, the Government of Canada is preventing exports of plastic waste to countries that have established import restrictions (such as China) and taking steps to implement new controls under the Basel Convention. However, Canada continues to export plastic waste, shipping it illegally to the U.S. with unknown fate, but also to non-OECD countries in Asia, Central and South America (CBC, 2022; Basel Action Network, 2023; Stahl et al., 2023).

Canada is now developing a Federal Greenhouse Gas Offset System under the Greenhouse Gas Pollution Pricing Act (GGPPA), by adding a charge on fuels and by encouraging voluntary project activities to reduce GGE (e.g., capturing landfill methane) or to remove greenhouse gases from the atmosphere, translating them to "offset credits" (ECCC, 2022). In contrast, the provinces and territories in consultation with Indigenous peoples, developed a Pan-Canadian plan to act on climate change, primarily through technology development.

Canada's Clean Air and Climate Change Act gives no specific attention to waste-related air pollution, generated by solid waste incineration. The Regulatory Framework for Air Emissions even provides an extension of the 2020 existing tax incentive for waste-to-energy.

The 2030 Emissions Reduction Plan (ECCC, 2022) is an ambitious and achievable roadmap that outlines a sector-by-sector path for Canada to reach its emissions reduction target of 40% below 2005 levels by 2030 and net-zero emissions by 2050. Yet, a growing body of evidence is pointing to waste prevention, reuse, and recycling as key sources of untapped potential for GGE reductions. Countless local initiatives, found throughout the country, value discardable materials and products through community recycling centres, free stores, repair cafes, zero waste initiatives and projects that appreciate lifestyle and value change (Lee & Khan, 2020). In 2021, ECCC launched a public engagement process for the 2030 Emissions Reduction Plan (ERP) to seek submissions from interested persons, communities and organizations. The Canadian public's main concerns are about waste reduction, and proper waste management/disposal, specifically endorsing circular economy approaches to achieve net-zero, including generating less waste and changing consumption habits.

6. COMPETENCES, LEGISLATION, POLICIES, INSTRUMENTS AND EXPERIENCES IN WASTE AND CLIMATE POLICIES IN COMPARATIVE PERSPECTIVE

In the table in the Appendix the Canadian and Brazilian constitutional and legal competences and regulations, the main waste-related policies and competent governmental

institutions and agencies, as well as the main policy instruments applied at the different government levels to handle waste management are presented.

First, it is noticeable that in the Brazilian case the federal constitution provides a special chapter for the protection of the environment and obliges both government and community "to defend and preserve it for present and future generations", although there are no specific clues regarding solid waste or climate change issues. In contrast, in the case of Canada, there are no such provisions on environmental matters at the constitutional level. The Canadian Constitution is still based on the British North America Act from 1867, having the country achieved its full sovereignty only with the Constitution Act from 1982. While thus the constitution is not exactly up to date, having suffered since then only minor amendments, in Brazil the rupture with the authoritarian regime has provided the opportunity for a broad constitutional process with significant participation of civil society and social movements. As a result, far-reaching societal demands, e.g., environmental protection, were introduced in the Constitution. Although there have been no specific decisions by the Brazilian Supreme Court regarding waste management, it is worth noting that the Constitution has proven a significant tool during the government of Jair Bolsonaro, to slow down the degradation of environmental policies, obvious in the deforestation of the Amazon, the main source of CO₂ emissions in Brazil.

In Canada, CEPA is also the most important law for waste and climate-relevant policies. Several regulations refer to waste management and the definition of basic waste production related principles like the Extended Producer Responsibility and Product Stewardship and provide guidelines for municipal and hazardous waste management. We also find clues related to the importance and diffusion of best practices in reducing pollution and GGE from waste management.

In Brazil, in accordance with the attempts of providing nationally valid framework legislations for each policy field, "the principles, objectives, instruments, and guidelines related to the integrated management of solid waste" have been determined by the 2010 Federal Law on Solid Waste (N° 12.305/10), giving guidance to state and local governments of their responsibilities in the field.

A significant achievement lies in the recognition and prioritization of reverse logistics practices. This includes establishing shared responsibility among manufacturers, importers, distributors, and traders, and placing a central emphasis on nationally organized waste pickers associations. These efforts have been crucial in integrating social considerations into national policy legislation.

Solid waste concerns also entered the federal law on basic sanitation (N° 14.026/2020), amending the National Solid Waste Policy, setting time-limits for the environmentally adequate final disposal of waste and determining the collection of fees and rates for solid waste services.

This overlap of legislation finds its correspondence in the governmental and administrative division of responsibilities, which are shared between the Ministry of the Environment and Climate Change with the National Secretariat of the Urban Environment and Environmental Quality being

responsible for proposing policies, plans and strategies, as well as coordinating, monitoring and evaluating the implementation of the National Solid Waste Policy; and the Ministry of Cities with its National Secretariat of Environmental Sanitation providing for the elaboration of studies and projects related to solid waste.

As part of the framework legislation regarding the environment – the National Environmental System (SISNAMA) – the National Council of the Environment (CONAMA) is a consultative and deliberative body, gathering representatives from the government, the private sector and civil society, responsible for establishing criteria for environmental licensing and standards for the control and maintenance of environmental quality.

During the Bolsonaro Government this forum of participatory governance was 'dried out', reducing its composition from 96 to 23 members, concentrating the decision-making power in the government representatives and taking away votes from civil society, a decision contested by the Federal Supreme Court. In the current Lula government this Council was reconstituted with now 114 members, reestablishing the voice of civil society in environmental issues.

In Canada, the major institutional body is the Canadian Council of Ministers of the Environment (CCME) as an intergovernmental forum consisting of the 14 federal, provincial, and territorial ministers responsible in their states for environmental concerns. Waste management and climate change are part of the agenda, however due to lack of enforcement authority their recommendations are not binding for the provinces. This shows the difficulties the federal government has to unify public policymaking and it highlights the dependence on voluntary agreements to arrive at collective political strategies and practices. Some agreements have been negotiated within the Waste Management Committee of the CCME as for instance the preparation of a common packaging policy for Canada.

Despite ECCC – Environment and Climate Change Canada's broad mandate, there is a lack of enforcement capacity on the part of the federal government. ECCC works mainly with producing guidelines and recommendations. Ultimately, the treatment of environmental issues hinges on the decisions made by the Minister, who can subsequently be held accountable for those decisions.

In Brazil, there are national plans for waste and sanitation to complement and operationalize the corresponding federal laws, specific programs (reverse logistics) and information systems (National Solid Waste Management Information System – SINIR) to support, coordinate and conduct the waste policy in the federation. It is important to note the sectoral agreements, which are contractual acts, although their outcomes vary, as seen in the sectoral agreement on packaging. For instance, the packaging development paradigm prioritizes competitiveness over recyclability, which may impact the promised outcomes of the agreement (Demajorovic & Massote, 2017).

At the subnational level, the institutional structure follows the regulations provided in the constitution and the federal laws and policies, having their own environmental regulating laws and plans. Even though the federal government due to the local autonomy does not have the author-

ity to oblige municipalities to create their own Integrated Solid Waste Management Plan and their own Basic Sanitation Plan, the municipalities can only apply for federal resources in the case such plans already exist. Such kind of conditionality is a very popular tool used by the Brazilian government to make municipalities adopt the federal policy (Arretche, 2012). They also used to have their own Public Policy Council related to the environment allowing for the participation of civil society in environmental policy, dispose of a secretariat and – mostly – an environmental agency to implement the policies, and an environmental fund to support environmental initiatives and projects.

In Canada, by contrast, the subnational governments dispose of the constitutional authority to conceive or shape their own solid waste policies, favouring a greater plurality in initiatives and programs. At the provincial level there are Environmental Ministries and at provincial and municipal level agencies responsible for environmental control and climate change policies. It is also worth noting the role of the city council with specific committees created to tackle specific waste related issues and climate actions and of community councils which can be activated to collaborate with specific waste related issues and climate actions.

There are still other issues insufficiently addressed by waste policy like the consumer and citizen side of waste production and treatment. In view of the strong influence of business on waste policy, there is a need for environmentally aware citizens who increasingly base their consumption decisions on ecologically and socially relevant criteria. This strategy is insufficient to drive the necessary transformative change. Therefore, the political role of citizens and civil society needs to be strengthened by considering social mobilization and political participation as central. Politically conceived environmental education must therefore become an essential component of a comprehensive approach to waste and climate policy, which is still severely underexposed in the policy approaches examined.

It is the proximity between citizen and city hall that favours the role of local governments in environmental policy, especially waste policy. Whereas climate policy is mainly driven by global governance, assimilated and adapted to the national context by national governments, to be translated into regional and local strategies by subnational governments, in waste policy, the pressure to act rests primarily on local authorities, which must contend with the effects of the externalities produced by households and companies.

7. CONCLUSIONS

In this paper we worked out the commonalities and differences of Brazil and Canada, two federal democracies, in handling the challenge of integrated solid waste management in view of climate change. A true comparison of public policy in contexts as different as Brazil and Canada is a fundamental challenge given the different federal legal systems, the different legacies of bureaucratic and political action patterns in public policy making, the different stages of social, economic and democratic development, and the

different financial and technological resources available to these two governments. Initially, we identified four fundamental challenges that need to be addressed in order to achieve greater urban sustainability. Our assessment of the Brazilian and Canadian experiences with waste and climate policy speaks to these challenges.

1. We can say that in both countries the circular economy concept has been incorporated – not always explicitly – into the normative regulations of solid waste policy. However, several limitations have been identified, *inter alia*: (i) Implementation deficits in the municipal systems of selective collection. In both countries selective waste collection for recycling is the responsibility of local governments. In Brazil, the lack of policy capacities (Wu et al., 2018) – technological, financial and political –, poses a significant constraint, particularly for smaller municipalities, hindering effective delivery of environmental services (Frey et al., 2022); whereas in Canada insufficient enforcement capacity at the federal level results in varied performance across subnational waste management, by local governments; (ii) the inclusion of energy recovery from waste (waste-to-energy) in the circular economy concept, despite knowing of its climate impacts. The debate around waste incineration shows that highly influential business interests advocate for technical remedies like waste-to-energy, sidelining more socially and environmentally sustainable practices such as waste reduction and recycling, which include the work of waste pickers. Despite the existence of environmental movements in the global North, the economic-technological approach of ecological modernization or the “gospel of eco-efficiency” (Martinez-Alier, J., 2002) has always prevailed in environmental policy in highly developed countries like Canada, in Brazil the hegemony of this “Global North” approach has always been challenged by the “environmentalism of the poor” or environmental justice movement (*ibid.*; Guha, 2014). For its part, however, the latter has come under increasing pressure from the eco-efficiency approach fuelled by neoliberalism to the detriment of socially and ecologically compatible approaches. As a result, local authorities are increasingly resorting to supposedly more efficient technological solutions such as waste incineration, giving in to the pressure of powerful economic interests. (iii) the avoidance of getting to the root of the problem, namely curbing waste generation. This is a more fundamental challenge as it counters the basics of growth-oriented capitalism and our consolidated consumer practices. Thus, apart from some conceptual advances that have been incorporated into policies, we remain entrenched in a political landscape where the priorities and agendas of economic growth and competitiveness overshadow environmental and social considerations, hindering genuine structural change.
2. Both countries pay little attention to the problem of waste as a climate-related issue. The hidden effects of climate change resulting from the production pro-

cess and the corresponding supply chains tend to be ignored. Relying solely on seeing waste through the economic lens is inadequate and may even be counterproductive, as it tends to encourage more waste production rather than aiming to reduce it.

3. The Brazilian experience of explicitly including and promoting waste pickers as an essential component of waste policy is an important achievement in a policy field where technological and business concerns tend to take centre stage. The important role of social movements in the process of democratic transition, the new Citizen's Constitution of 1988, and the emergence of left-wing governments at local and later at federal level were fundamental contextual conditions for the creation of policies for the productive inclusion of waste pickers. Nevertheless, given the simultaneous expansion of waste incineration, the spreading of this experience seems difficult to achieve due to the powerful interests boosting this highly profitable business. Its continuity depends fundamentally on the political development in the country and municipalities, where, in view of the current political and ideological polarisation, such socially oriented policies have increasingly become the plaything of hostile opponents.
4. The different federal institutional arrangements make a difference in the design of waste policy in both countries. Due to the different historical legacies and the related differences in political and administrative structures, diverse strategies are being pursued to meet the challenge of “unity in diversity”. In Brazil, the constitutional process has favoured the strengthening of municipal autonomy and the principles of cooperative federalism, with the federal government increasingly taking the lead in public policy at the expense of the states, which have lost relative financial strength and governance capacity. This general condition has led to a national SWP where the federal government develops a general legal policy framework that establishes basic principles, institutional requirements and policy tools, and provides financial support for policy implementation by subnational governments, at least to the extent that they comply with the basic requirements stipulated by the federal government. In principle, participatory arrangements provide for the necessary democratic legitimacy of such policy measures. However, a lack of effective horizontal policy integration has been identified in the areas of waste, basic sanitation and climate policy and as has been seen with President Bolsonaro, federal governments that are not committed to environmental issues, can easily undermine such comprehensive approaches by omitting or obstructing them. In the case of the Canadian “executive federalism” the federal government's capacity to unify public policy-making is historically low and depends on the government's ability to negotiate joint agreements with the provincial ministers. In the event of differences of opinions or perceptions, the ECCC can only try to work with guidelines and recommendations lacking integration and enforcement capacities.

This is reflected in the waste and climate policy, where besides the general recommendations and guidelines there are only specific regulations that could be agreed upon and, in addition, quite different province-specific regulations, progressive in some provinces and more conservative and restraining in others. However, this results in a greater diversity of waste policy proposals and practices in the various sub-national governments. In summary, it has proven worthwhile to take the similarities and differences of federalism in both countries as a starting point for our analysis, in order to better understand the respective experiences themselves, as well as to draw lessons that may point beyond the individual cases. The complexities involved in our cases, especially if we look at solid waste and climate change as interdependent policy fields, reveal their character as wicked problems (Head, 2022). However, whereas such problems demand coordination, cooperation, innovation and policy integration, political-administrative systems in general tend to be siloed. We identified progress and obstacles in taking an integrative approach to the nexus waste-climate to address the related wicked problems in both countries.

Our research highlights the difficulties faced by both national governments in designing and enforcing national waste policies at the local level. In the case of waste policy, local administrations are in principle in better conditions to design locally adapted public policies through innovative contextual approaches that can only emerge from local experiences. At the same time, subnational governments must be supported in their waste policies and these must be integrated into federal climate policy. Effective multi-level and cross-policy governance is therefore critical to successfully coordinate these policies. Further empirical research is needed to better understand the potential and challenges for democratic federal systems in tackling such wicked problems. The climate contributions from specific waste management practices supported by public policies need to be further studied in order to learn about their impact and either expand their application or redesign these policies more effectively.

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ROBOTS IN WASTE MANAGEMENT

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ABSTRACT

There are different strategies for urban waste management. At present, solutions are being sought for example to the noise that the collections cause to the inhabitants in the cities, as well as the environmental impact. The introduction of technology and particularly robots in the last time, open an opportunity to develop different applications for the robots last mile. This paper highlights and discusses properties of robotic waste collection, based on existing literature of the environmental and economic impacts of different waste collection and management systems. It explores how a robotic system could be introduced into public spaces. For this purpose, a new deployment protocol for public spaces has been developed, addressing these interactions. Surveys and personalized interviews with pedestrians in the presence of the robot in an urban center were conducted. Based on the collected data, a protocol has been designed, along with a proposed design applicable to urban centers, small towns, or isolated residential areas. Many challenges remain, including a comprehensive environmental analysis of the robotic waste collection system, and calculating the actual costs of implementation, operation, and maintenance over time.

1. INTRODUCTION

Urban waste management (Muntasir Shovon et al., 2024) is a practice that has been carried out for a long time in cities due to the importance of sanitation in preventing the spread of diseases (Moanaro Ao and Nzanthung Ngulie, 2024). Throughout history, solid waste has been handled in a simple manner, for example, using horse-drawn carts to collect the waste that accumulated in the streets and industries, and recycling those residues that could have value. Examples of this approach can be found in ancient Rome, where circular economy was valued (Brunner and Morf, 2024), and recycling was done as much as possible.

The big revolution in waste collection happened in the late 19th century with the emergence of the first steam-powered trucks, initially simple open-box carts. Later, the internal combustion engine, containers, and the establishment of routes that we are familiar with today arrived, now enhanced by the latest technologies (Di Maria et al., 2020).

Today, the amount of waste generated in cities remains a significant problem (Rossit and Nesmachnow, 2022), both from a public health and environmental perspective. One way to address this problem is through the implementation of more efficient and sustainable waste collection systems.

When comparing different methods of waste collection, including manual collection, automated waste collection using robots, and specifically last-mile robots, several advantages and disadvantages can be identified (Table 1). Manual waste collection is labor-intensive and time-consuming (Sushma et al., 2021), while automated waste collection using robots can significantly increase efficiency and reduce the need for manual labor (Raihan et al., 2020).

Underground garbage collection containers offer numerous advantages such as preventing the spread of odors and water, reducing the risk of illnesses from bacteria, saving space, enabling hygienic waste collection, and facilitating on-site waste separation (Delice et al., 2019). These systems utilize innovative features like ultraviolet disinfection to ensure cleanliness and deodorization while maintaining safety (Morakabatchian et al., 2021). Additionally, underground containers help in efficient waste collection and contribute to recycling efforts due to their compact design (Delice et al., 2019). However, some disadvantages include the initial installation costs and potential maintenance requirements associated with the complex mechanisms involved in these systems.

Last-mile robots, in particular, offer the advantage of autonomously navigating to collect waste, reducing human intervention and potentially lowering operational costs



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TABLE 1: Advantages and disadvantages of integrating robots into urban spaces.

Waste collection methods	Advantages	Disadvantages
Manual waste collection (Sushma S C et al. 2021)	Flexibility and adaptability	Labour-intensive
	Capacity to handle bulky waste	Risk of injury and exposure to hazardous substances
	Low initial cost of equipment	Inefficiency in large or densely populated areas.
	Direct human interaction and quality control	Variability in the quality of work
		Time-consuming
Mechanised Collection (Raihan N. et al.2020, Badve M. et al 2024)	High efficiency in the collection of large volumes	Require initial investment and maintenance efforts
	Reduction of manual time and effort.	Requires adequate infrastructure (wide streets, accessibility).
	Reduced exposure of staff to hazardous waste.	Emission of gaseous pollutants and noise.
	Greater consistency and reliability of service	Difficulty in areas of difficult access or complex geometries
Underground Containers (Delice E. et al. 2019,Morakabatchian M. at al. 2021)	Improved urban aesthetics.	the initial installation costs and potential maintenance requirements associated with the complex mechanisms involved in these systems
	Reduction of odours and pests.	Need for underground infrastructure.
	Increased storage capacity.	Difficulty of collection in case of system failures.
	Less frequent collection required.	Requires specific technology for harvesting.
Harvesting with Last Mile Robots(Tomitagawa, K. et al. 2024)	Autonomous operation, reduces the need for manual labour.	High cost of development and initial implementation
	Possibility of operating at non-conventional times	Limitations on carrying capacity and types of waste
	Reduced emissions and noise compared to larger vehicles.	Dependence on advanced technologies and connectivity.
	Accuracy in collection and route optimisation.	Need to adapt urban infrastructure
	Improved operational efficiency in dense urban areas	Possible technical failures and complex maintenance.
	Potentially lowering operational costs	Challenges such as energy consumption efficiency .
		Path optimization exist for waste collection robots, which can impact their overall effectiveness in waste management tasks.

(Tomitagawa et al., 2024). However, challenges such as energy consumption efficiency and path optimization exist for waste collection robots, which can impact their overall effectiveness in waste management tasks (Tomitagawa et al., 2024). Therefore, while last-mile robots offer automation benefits, considerations regarding energy efficiency and path optimization are crucial for maximizing their effectiveness in waste collection operations. This is still in the process of being studied and optimized.

Due to the current door-to-door selective waste collection (PaP) (Laso et al., 2019; Baltrocchi APD., 2024; Dotsenko et al., 2022) implemented by the Catalan Waste Agency, which involves delivering waste to the municipal service at your doorstep, it has complicated and duplicated the work of waste collection services, as well as their costs. The implementation of a home waste-collecting robot in Catalonia offers numerous advantages, such as cost reduction through a decrease in the number of collection vehicles on the streets, as well as a reduction in maintenance and operational costs. There would also be an improvement in the efficiency of the collection process and a reduction in the environmental impact of the activity, enhancing the residential landscape and reducing unpleasant odors. Currently, robots can be found in numerous controlled environments:

- Airports (Hwang et al., 2023): Robots have been used in airports to perform tasks such as floor cleaning, waste removal, baggage delivery, and passenger transportation. For example, Incheon Airport in South Korea utilizes a passenger and baggage transport robot.
- Restaurants (Molinillo et al., 2023): In some restaurants, robots are used to take orders and deliver food to tables. For instance, the Spyce restaurant in Boston features a fully automated kitchen that uses robots to prepare and serve food.
- Hospitals (Huang et al., 2023): Robots have been employed in hospitals to transport supplies and food, as well as perform cleaning and disinfection tasks. For instance, Texas Children's Hospital in Houston uses robots called Tugs to transport medications and supplies to different areas of the hospital.
- Parking lots (Soori et al., 2023): Robots have been utilized in certain parking lots for autonomous parking and retrieving of vehicles. For example, at the Volkswagen Autostadt in Wolfsburg, Germany, visitors can leave their cars in an automated parking structure called Car Towers, which uses robots to retrieve and deliver vehicles.

The evolution of urban waste management has seen significant advancements, transitioning from histori-

cal methods to modern technologies like autonomous waste-collecting robots (Liang-Bi Chen et al., 2023; Bharanikumar et al., 2022, Sushma et al., 2021). These robots offer a sustainable solution by reducing operational costs, minimizing environmental impact, and enhancing efficiency in waste collection processes. By integrating autonomous robots into urban environments, particularly in areas like Catalonia, where door-to-door waste collection systems are in place, numerous benefits can be realized, including cost reduction, improved operational efficiency, and a reduction in environmental footprint. The implementation of waste-collecting robots not only addresses the challenges of waste management but also contributes to job creation in the technology sector and promotes the adoption of sustainable technologies, marking a significant step in the robotics industry's evolution towards more complex urban applications.

These robots, equipped with artificial intelligence and robotic systems, aim to autonomously collect and dispose of garbage while monitoring waste levels and contents in smart trash cans. By utilizing technologies such as AIoT (Artificial Intelligence of Things), ROS (Robot Operating System) (Sunil and Shanavas, 2023), image sensors, and ultrasonic sensors, these robots can navigate designated areas, detect trash, and efficiently manage waste disposal processes, contributing to cleaner environments and improved waste management practices (Gupta et al., 2022). The ability of a waste-collecting robot to operate in these challenging environments requires a combination of technical skills, such as autonomous navigation, obstacle detection, and coordination of multiple simultaneous tasks, as well as excellent adaptability. The robot must be capable of planning efficient collection routes, safely maneuvering through streets and sidewalks, accurately picking up the waste, and depositing it in the appropriate location (Azza et al., 2023). Additionally, it must interact safely and respectfully with citizens, given its operation in public settings. This article will explore these interactions, focusing on how the robot engages with the city and its residents.

Research questions:

What environmental and social analyses have been conducted on garbage collection robots? What challenges and intersections arise when introducing last-mile garbage collection robots in cities, particularly in terms of pedestrian reactions, public space sharing preferences, and potential physical conflicts with existing infrastructures? What protocols should be followed to minimize any inconvenience during their integration?

The study aims to address these research questions by conducting a comprehensive systematic review of last-mile waste collection robots, including economic, environmental (Life Cycle Assessment), and social analyses, and proposing a new framework for implementing a robotic waste collection system in urban areas. The literature mainly addresses technology and human-robot interaction (Tuli et al., 2022; Fischer et al., 2022; Hwang et al., 2007), but research on how these robots interact with urban infrastructure is lacking. This study aims to design an innovative robot tailored for urban environments, particularly residential complexes with shared spaces and waste collection

points. It will also examine the robot's interaction with city infrastructure and residents, focusing on potential physical conflicts with existing structures. Additionally, the study will compare door-to-door and robotic waste collection systems to propose a new protocol for integrating these robots into urban centers, promoting widespread adoption.

2. METHODOLOGIES

This paper offers a comprehensive systematic bibliographic review focusing on three key aspects. Firstly, it examines the diverse range of robots developed for last-mile waste collection, incorporating economic, environmental (LCA), and social analyses. Secondly, it proposes a framework for the implementation process of a last-mile robotic system within urban areas. Lastly, it presents a novel robot proposal tailored for urban environments with residential complexes or blocks featuring shared spaces and waste collection points.

In the systematic literature review, we explore the various existing robots that have been developed specifically for last-mile waste collection. This includes examining their functionalities, capabilities, and performance in real-world scenarios. We analyze the navigation systems used by these robots to maneuver through complex urban environments, their ability to detect and avoid obstacles, and their collection mechanisms. We also assess their efficiency in terms of time and energy consumption, as well as their adaptability to different types of waste.

For the purpose of conducting this comprehensive review, we will utilize the extensive resources available within the Web of Science (WOS) database. This database offers a vast collection of peer-reviewed literature spanning various disciplines, ensuring a thorough and in-depth analysis of the relevant research findings.

To assess the current state-of-the-art in last-mile waste-collecting robots, a comprehensive literature review was conducted using the Web of Science database (2023). The search utilized the terms "trash collection robots," "autonomous waste collection," "robotic waste collection," and "waste management robots," combined with the Boolean operator "OR" to retrieve relevant articles. As depicted in Figure 1, although research in this field began in 1992, there has been a significant increase in published studies over the past four years.

When we delve deeper into the analysis of the results, it becomes evident that among the six most productive countries, China, the United States, and India significantly outshine the rest. Spain, as the leading European nation in scientific production within this context, holds the fourth position.

If we look at the document types, we can see that there are 211 journal articles, 144 proceeding papers, and 17 reviews, resulting in a total of 369 documents. When filtering by these three document types, we reduce the search to 369 results.

If we further refine the search by adding "AND LCA" to consider the environmental impact through a life cycle assessment, we narrow down our search to 205 documents. Then, if we select only articles and literature reviews, we

Year distribution

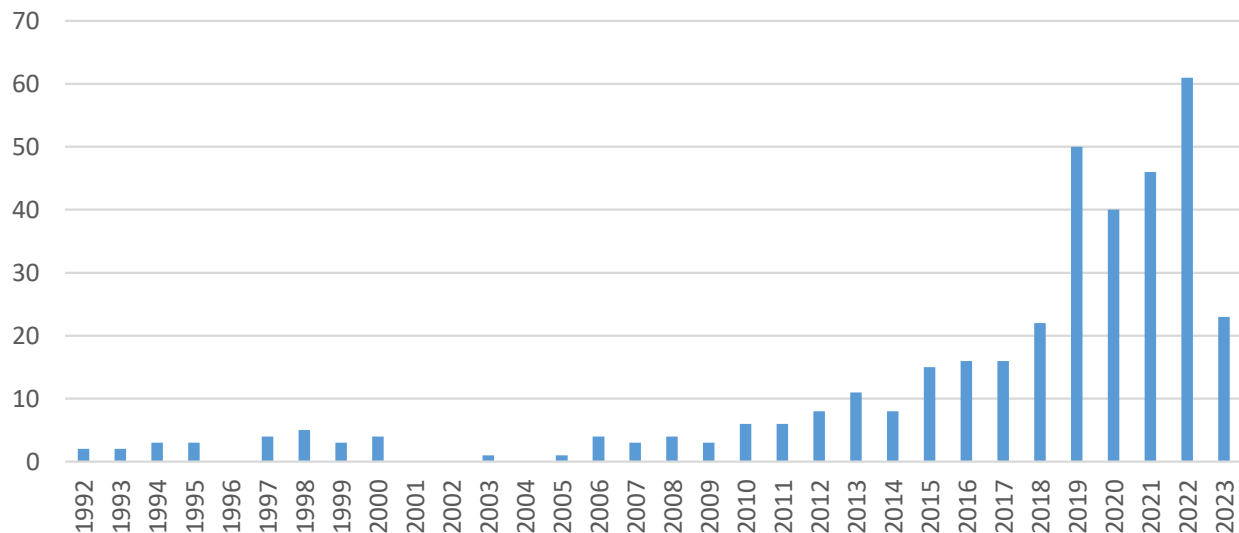


FIGURE 1: Year distribution of the number of publications on waste-collecting robots.

are left with 141 documents. From these, we can take the top 5 most cited and the 5 most recent, which are presented in Table 2.

In these articles, we can see that digitization and artificial intelligence are considered important aspects in waste management and the involvement of waste-collecting robots. The circular economy is mentioned as a key approach to waste management and the application of advanced technology. Environmental sustainability is a recurring theme, focusing on waste management as well as other related areas such as recycling and reverse logistics. Challenges and opportunities associated with the implementation of waste-collecting robots are mentioned, such as the need to adapt to different contexts and the automation of tasks. Furthermore, the exploitation of waste management in specific contexts is discussed, including the generation of alternative fuels and biogas from waste materials, among other elements.

If we analyze the 17 articles that are reviews and cover the entire period up to the present day, we can highlight that the first article discusses how to digitize the waste management value chain, always considering the circular economy. The second article highlights artificial intelligence as a key factor in waste management in cities and presents a use case in a smart city. The third article, found in Table 2, focuses on waste classification and discusses the challenges and opportunities that arise. The fourth article presents a case study in China and explains the characteristics of waste management in a rural area. The sixth article discusses the informal management of urban waste in eight Latin American countries. The thirteenth article discusses motion control and forces of robots, although it could be applicable to any similar application, it is very specific about the functioning and autonomous movement of the robot. Lastly, the seventeenth article presents an analysis of digitalization driven by urban circular metabolism and examines circular city initiatives.

We made a proposed implementation protocol for said last mile robot dedicated to urban waste collection. To define the protocol, our approach draws upon existing literature on robot integration and leverages insights from previous experiences documented by the Institute of Robotics and Industrial Informatics (IRI) at the Universitat Politècnica de Catalunya (UPC).

To gauge user acceptance, in the last stage of the protocol, we employed two strategies. Firstly, we conducted a survey to enable quantitative analysis of the level of user acceptance regarding the integration of robots into urban areas for waste collection. This survey, hosted on Google Forms, was initially distributed through the researchers' social networks, targeting individuals within their respective circles. Subsequently, the Esplugues de Llobregat city council, a participating entity in the project, was approached to disseminate the survey on the city hall website, thereby reaching the local populace for whom the system is intended. The questionnaire comprised six sections encompassing a total of 26 questions. These sections addressed various aspects such as users' perceptions of the robot's interaction with citizens, the tasks performed by the robot, the necessary infrastructures to support last-mile distribution robots (including connectivity, data management, monitoring, and energy recharging), the design and organization of urban public space, and the sustainability implications of introducing distribution robots into urban environments. To enhance respondents' engagement and comprehension, the questionnaire incorporated videos and images. The survey questionnaire is accessible via the following link: <https://forms.gle/9ojMG6P1rCAYNbodA>

Another strategy involved conducting structured interviews in two distinct environments:

- The first set of interviews took place in controlled areas over several days. During this phase, the robots were deployed across the university campus, where

TABLE 2: Most cited and most recent paper about Smart Trash Robot System. Literature reviews on last-mile garbage robots.

Title	Authors	Journal
Most cited and most recent paper about Smart Trash Robot System		
Smart technologies for collection and classification of electronic waste	Ada, E; Ilter, HK; Kazancoglu, Y	International Journal Of Quality & Reliability Management (2023)
The impacts of autonomous vehicles on local government budgeting and finance: case of solid waste collection	clark, by	National Tax Journal(2020)
Smart Trash Can Robot System with Integration of Internet of Things and Mobile Applications	Chang, BR; Tsai, HF; Yin, TK	Sensors And Materials (2019)
Stb: Child-dependent Sociable Trash Box	Yamaji, Y; T; Okada, M	International Journal Of Social Robotics (2011)
Digitalisation and intelligent robotics in value chain of circular economy oriented waste management - A review	Sarc, R; Curtis, A; Pomberger, R	Waste Management(2019). Citations 151
Public willingness to pay and participate in domestic waste management in rural areas of China	Han, ZY; Zeng, D; Mou, ZS	Resources Conservation And Recycling. Citations 54
Does performance evaluation help public managers? A Balanced Scorecard approach in urban waste services	Guimaraes, B;Simoes, P and Marques, RC	Journal Of Environmental Management(2010). Citations 54
A mobile robot based system for fully automated thermal 3D mapping	Borrmann, D; Nuchter, A;; Velagic, J	Advanced Engineering Informatics (2014). Citations 53
Optimal Route Recommendation for Waste Carrier Vehicles for Efficient Waste Collection: A Step Forward Towards Sustainable Cities	Ahmad, S; Imran; Kim, D	Ieee Access (2020) Citations 30
Literature reviews on last-mile garbage robots		
1.-Digitalisation and intelligent robotics in value chain of circular economy oriented waste management- A review	Sarc, R; Curtis, A; Pomberger, R	Waste Management (2019)
2-Artificial intelligence for waste management in smart cities: a review	Fang, BB; Yu, JC; Yap, PS	Environmental Chemistry Letters(2023)
3.-A state-of-the-art review on robotics in waste sorting: scope and challenges	Satav, AG; Kubade, S; Pawar, A	International Journal Of Interactive Design And Manufacturing - Ijidem (2023)
4.-Characteristics and management modes of domestic waste in rural areas of developing countries: a case study of China	Han, ZY;Ye, CW; Shi, GZ	Environmental Science And Pollution Research (2019)
6.-The Role of Informal Waste Management in Urban Metabolism: A Review of Eight Latin American Countries	Espinosa-Aquino, B; Durany, XG and Vargas, RQ	Sustainability (2023)
13.-Hybrid motion/force control: a review	Ortenzi, V; Stolkin, R; Mistry, M	Advanced Robotics (2017)
17.-Digitalisation driven urban metabolism circularity: A review and analysis of circular city initiatives	D'Amico, G; Arbolino, R; Ioppolo, G	Land Use Policy (2022)

students, professors, administrative staff, maintenance workers, security personnel, and cleaning staff were approached for interviews. Although this initial approach resulted in a somewhat biased sample, with the majority comprising students from a technological university, it provided valuable insights into the perspectives of the younger generation.

- The second approach to assess robot acceptance involved traveling to Esplugues de Llobregat, a city chosen for its diverse urban environment. Here, the research team transported the robot to various locations within the city and conducted interviews with pedestrians encountered along the way. The interview questions mirrored those used in the survey, facilitating data extraction and comparison with quantitative analysis. Despite maintaining a structured format, participants were encouraged to openly express any concerns, ensuring comprehensive feedback.
- Finally, based on the findings from the literature review and the feedback of the users, we propose a specific robot design tailored for urban developments with communal areas and waste collection points (Saragih et al., 2023). This robot is designed to operate within the con-

finer of the urbanization, collecting waste from designated collection points. We outline the key features of our proposed robot, including its size, maneuverability, collection mechanism, and navigation system. We also discuss the potential benefits of implementing such a robot in terms of cost reduction, improved efficiency, and enhanced cleanliness in communal areas.

3. RESULTS AND DISCUSSION

When analyzed together all the papers (Table 2), we see the importance given to digitalization, artificial intelligence, and other advanced technologies in waste management and the development of more efficient waste-collecting robots. Most of the articles mention the circular economy as a key approach to waste management and how waste-collecting robots can play a role in transitioning to a circular economy model. The importance of environmental sustainability in waste management and the development of environmentally friendly technologies and robots is addressed. Waste-collecting robots (Chen et al., 2023). are considered efficient and automated tools for waste collection and sorting, which can increase efficiency and reduce costs associated with waste management. Chal-

allenges and opportunities associated with the implementation of waste-collecting robots are highlighted, such as autonomous navigation in complex urban environments, safe interaction with citizens, and coordination of multiple simultaneous tasks. We have also seen examples of waste management in specific contexts, such as rural areas in developing countries or in smart cities, emphasizing the need to adapt waste-collecting robots to different environments and requirements.

Andreasi Bassi et al. (2017) compare the environmental impacts of household waste management in seven European countries (Germany, Denmark, France, the UK, Italy, Poland, and Greece), highlighting the benefits and challenges of different waste management strategies (as we previously present in table 1). Notably, Germany shows the best environmental performance, with high recycling rates and minimal landfilling. Recycling generally yields the largest environmental benefits, particularly for paper, metals, and glass, while waste-to-energy (WtE) plants can either contribute to environmental savings or loads, depending on their efficiency and the energy they replace. Denmark stands out for achieving significant environmental savings through energy recovery due to its extensive district heating network. However, France's high recycling rates are offset by the environmental burden from WtE plants, and Greece and Poland show high environmental impacts due to heavy reliance on landfilling. The study emphasizes that as Europe moves away from landfilling, a shift in focus is needed from managing waste inputs to maximizing the value of recovered materials and energy outputs. It also notes that increasing recycling rates does not always correlate with better environmental performance, depending on national contexts and technological factors.

In parallel with the information search for the design of the waste collection robot, a protocol for the deployment of these robots in public spaces has been developed. There are numerous studies examining the logistical integration of last-mile robots, primarily for parcel delivery (Demir et al., 2022; Hyeong et al., 2021; Deloitte, 2023). Conversely, some studies focus exclusively on robot design (Boysen et al., 2021; Park et al., 2023; Burde et al., 2023). Other researchers focus on making deliveries more sustainable and optimizing resources to the maximum (Aljohani and Thompson, 2019; Ranieri et al., 2018), including authors who analyze the interactions between robots, people, and public space (Puig-Pey et al., 2023). However, there is still a need for subsequent adaptation of the robot or public space to ensure the seamless integration of the robot. Therefore, we propose in this paper a new protocol with different phases for its proper implementation, taking into account not only the interaction with public space but also with people and current mobility systems.

Before proceeding to analyze the integration of a robot within the urban space and recognizing its technical viability with necessary adaptations yet to be made, we have assessed its environmental sustainability as a waste collection system. This assessment is part of a broader exploration into waste management strategies, which have been subject to various studies evaluating their environmental

impact and efficiency. For instance, Mora et al. (2014) conducted a Life Cycle Analysis (LCA) to assess the environmental implications of different waste management methods, including street-side containers, subterranean containers, doorstep collection, and pneumatic collection systems. Conversely, Gilardino et al. (2017), integrated operational research methods and LCA to devise an efficient collection-route system for garbage compactor trucks, aiming to reduce environmental impacts in Lima. Pérez et al. (2017) outlined a methodology for assessing the environmental impact of urban containerization systems using LCA, while Pires et al. (2017) examined the economic aspects alongside the environmental impact of curbside and exclusive bring systems. Social perspectives on common collection systems were also considered (Yildiz-Geyhan et al., 2017). Introduction of new technologies facilitated comparisons between conventional and novel systems such as vacuum collection.

Teerioja et al. (2012) compared a stationary pneumatic waste collection system with a traditional vehicle-operated door-to-door collection system in a densely populated urban area, focusing on economic aspects. Punkkinen et al. (2012) and Aranda-Usón et al. (2013) utilized LCA to evaluate the environmental sustainability of both collection methods. Iriarte et al. (2009) employed LCA to quantify and compare the potential environmental impacts of three selective collection systems: mobile pneumatic, multi-container, and door-to-door. Previous findings suggest that pneumatic systems tend to generate more air emissions due to electricity consumption and installation materials. However, when loads approach full capacity, the vacuum system exhibits superior environmental performance compared to conventional methods. Additionally, from an economic standpoint, pneumatic collection is estimated to be six times costlier than traditional systems (Teerioja et al., 2012).

As observed, studies conducted thus far have primarily focused on comparing pneumatic or door-to-door urban garbage collection systems (Iriarte et al., 2009; Laso et al., 2019; Baltrocchi APD., 2024; Dotsenko et al., 2022), predominantly using traditional traction vehicles rather than electric or green energy vehicles (Canals Casals et al., 2016; Canals Casals et al., 2017; Aryan et al., 2023) but according to Laso et al. (2019), pneumatic collection could be an environmentally friendly option for MSW management under a circular economy approach in urban centers, taking into account waste as an opportunity for a material or energy source. If the proposed waste collection robot functions as an electric vehicle, particularly one designated for door-to-door collection, as demonstrated by Yang et al. (2022), it is reasonable to anticipate a substantial reduction in emissions. This shift would make it a more viable option for waste management.

When assessing the collection systems from an economic perspective, Laso et al. (2019) and Yang et al. (2022) underscore the cost-effectiveness of the door-to-door system. However, a comprehensive comparison must be made between the initial installation costs and potential maintenance requirements of the complex mechanisms involved in pneumatic systems and the robots' operational

TABLE 3: Protocol for the Introduction of Robots.

Project Definition - Public Space - Stakeholders (Human Robot Interaction Roles: robot supervisor, robot operator, peer teammate, peer enduser, bystander, robotic urban set designer, researchers) - Need or Activity to be Developed by the Robot	Initial Studies - Studies on robot typology - Studies on public space - Study of interactions between "City" and "Robot" - Study and analysis of regulations and applicable legislation in the project, as well as the necessary licenses for it.	Technological Development of the Robot - Design and implementation of the robot - Testing of it in the laboratory Design of the public space that will host the robot - Implementation of the zones - Design of the communication equipment necessary for orientation/communication with the robot - Design of specific recharging/repair spaces for the robot in the city - Design of the pick-up and delivery areas.	Prototype Implementation - Analysis of proper integration - User Acceptance Analysis (surveys/interviews/real experiences with prototypes) - Analysis of suitable schedules and spaces - Implementation in a controlled environment - Implementation in the city, controlled part. - Implementation in a real environment.
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expenses, maintenance requirements, and long-term sustainability.

Shifting focus to the framework for implementing a last-mile robotic system in urban areas, Table 3 outlines the procedure used in this research for integrating a robot into public spaces.

The Table 3 show the protocol to develop and introduce a new technology in the city. This protocol is divided into four sections. The first section outlines the project's definition, including technical requirements, identification of interacting actors, and specific locations for deployment. The second section delves into an in-depth study of various robot typologies to determine the most suitable option. Additionally, this section examines the city and its characteristics, along with relevant project regulations.

The third section focuses on the technical development of the robot, design considerations for the public spaces it will interact with, communication systems, parcel delivery mechanisms, and other related aspects. Finally, the last section encompasses the realization of the prototype and its interaction with the environment. This stage involves analyzing citizen acceptance of the robot, making necessary adjustments for adaptation, and addressing any identified issues.

As previously mentioned, surveys were conducted with users prior to introducing the prototype into the urban environment, followed by personalized interviews after conducting robot implementation tests in the city. The vehicle utilized for these tests was the ONA (Human-Robot Interaction (HRI) Operational Navigation Autonomous), as described by Puig-Pey et al. (2023). ONA is an electric vehicle with approximately 8 hours of autonomy and dimensions similar to those of the proposed waste collection robot.

As seen in Table 3, it is essential to be in a connected city to implement robotic systems, as the device must communicate at all times to follow the predefined route without conflicts.

In analyzing the city, it is necessary to distinguish between different urban areas. In this study, we chose to classify the city into three parts: urban center, industrial zone, and neighborhoods/developments due to their different urban characteristics.

Simultaneously, we analyzed various types of robots currently available that could fit the activity we were devel-

oping. At this point, we opted to design a waste collection robot for developments and used ONA for the test, as seen earlier.

In the interviews as we could see, conducted during the implementation tests, we found that bystanders indicated, on a scale of 1 to 7, that 90% (taking into account ratings of 5, 6, and 7) would accept the presence of a robot in the public space (Figure 2). The scale from 1 to 7 indicates the degree to which the robot is accepted in the urban space. Where 1 is unfavorable and 7 is feasible).

They emphasize that the most feasible implementation they see at present would be in residential areas or industrial zones (university campuses, industrial estates, pedestrian urban centers, or small urban developments). They point out that it would be advisable to segregate urban spaces and that nighttime mobility would be the most suitable to minimize interference with bystanders. Therefore, we believe that the proposed activity is appropriate based on the suggestions of passersby.

The dimensions of the ONA were deemed suitable, suggesting that our waste collection robot would encounter no issues in this regard.

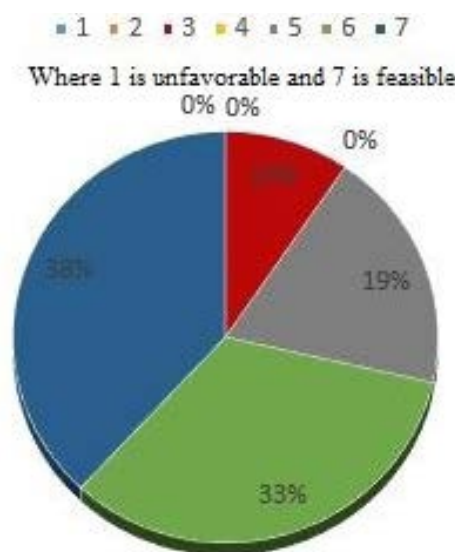


FIGURE 2: To what extent would you accept the presence of a robot in the current urban public space?



FIGURE 3: Year distribution of the number of publications on waste-collecting robots.

Regarding the city's preparedness for robot integration, participants expressed the need for significant investment. While acknowledging the potential reduction in unpleasant nighttime noise, concerns were raised about potential job displacement as a significant drawback to implementation.

Based on all these conclusions, also based on previous studies of waste disposal robots (Sushma et al., 2021; Gupta et al., 2022; Chen et al., 2023) and the type of robots at IRI (Institute of Robotics and Industrial Informatics) (ONA robot, (Puig-Pey et al., 2023)) and with the help of students from the Master's program in Industrial Electronics and Automation Engineering at the ESEIAAT (The School of Industrial, Aerospace and Audiovisual Engineering of Terrassa from Universitat Politècnica de Catalunya-BARCELONATECH), a waste-collecting robot design proposal has been developed, which can be seen in Figure 3.

This robot is designed to operate in the residential neighborhoods of Esplugues de Llobregat, where door-to-door waste collection is currently implemented with a different type of waste collected each day. The robot is equipped with caterpillar-like tracks, allowing it to navigate various surfaces with ease. The specifications of both the robot and the container are detailed in Table 4.

As the design of the robot and the different software and hardware components is not yet available, an information search has been carried out to determine the key factors for the design. Specification sheets are prepared based on the factors to be considered for a more sustainable design. A survey of 33 leading robotics companies was

conducted, as used in Zerkane et al. (2015). Some of the key factors identified include energy saving, durability optimization, recyclability, biodegradability, renewability, and reusability. However, there is still a lot of work to be done in this area.

As shown in Tables 3 and 4 and Figure 3, the preliminary design indicates that each container will have a capacity of 500 liters, and it is estimated that there will be one robot for every 300 households or more (considering that the average garbage production per family per day is 1.5 kg), forming a network of robots in more populated environments. The system is electric, equipped with lithium batteries, and rechargeable through solar panels located on the top of the container, if conditions permit. The current robot can receive instructions from an operator, follow a programmed route, includes both manual and automatic emergency stops, and is equipped with a trash compactor (Dalmasso et al., 2023).

It is clear that the production and disposal of batteries significantly contribute to the overall environmental footprint of BEVs (Koroma et al. 2022). However, as demonstrated in the work by Yang et al. (2022), compared to combustion engines, the environmental impact is lower, although it is clearly influenced by the energy mix of the country where the BEV or robot is implemented.

Therefore, as a final reflection we could say that the integration of robots into urban spaces presents a myriad of advantages and disadvantages (Table 5 presents a first analysis of them). On one hand, robots can streamline various urban processes, such as waste management, transportation, and infrastructure maintenance, leading to increased efficiency and reduced operational costs.

Additionally, robots can mitigate risks to human workers by performing tasks in hazardous or challenging environments. However, their introduction may also raise concerns regarding job displacement and social acceptance. Moreover, the complexity and cost associated with implementing and maintaining robotic systems in urban environments pose significant challenges. Therefore, while robots hold promise for enhancing urban life, careful considera-

TABLE 4: The technical specifications of the robot and the container.

Robot specifications		Container specifications	
Speed(m/s)	4	Speed(m/s)	4
Mass (Kg)	50	Mass(Kg)	150
Speed-up(m/s ²)	1.6	Speed-up(m/s ²)	1.6
Power(W)	320	Power (W)	960
Torque (Nm)	40	Torque (Nm)	120

TABLE 5: Advantages and disadvantages of integrating robots into urban spaces.

Aspect	Advantages	Disadvantages
Technological	- Automation of repetitive tasks. - Increased precision and efficiency in waste collection. - Integration of advanced technologies.	- Vulnerability to technical failures and cyberattacks. - High initial and ongoing costs associated with development and maintenance. - The delayed implementation of technology in connected cities poses challenges for current robot navigation.
Social	- Improvement of working conditions. - Potential for creating new jobs. - Reduction of pollution and noise.	- Concerns over job loss. - Challenges of acceptance and adaptation by the population.
Environmental	- Reduction of greenhouse gas emissions. - Improvement in waste management and collection efficiency.	- Environmental impact of production and disposal of electronic components. - Energy consumption during manufacturing and operation.
Economic	- Long-term reduction in operational costs. - Savings generation through efficient resource management.	- High initial costs. - Uncertainty about return on investment and long-term economic viability.

tion of both their benefits and drawbacks is essential for successful integration into urban spaces.

4. CONCLUSIONS

As we have analyzed, in reviewing the collection of studies (Table 2), the critical role of digitalization, artificial intelligence, and other advanced technologies in waste management emerges prominently. These technologies are pivotal in developing more efficient waste-collecting robots. Many papers as we saw, underscore the circular economy as a central approach to waste management, highlighting how waste-collecting robots can facilitate the transition to this model. Additionally, the importance of environmental sustainability in waste management and the creation of environmentally friendly technologies and robots is emphasized.

Waste-collecting robots, as noted by Chen et al. (2023), are viewed as efficient, automated tools for waste collection and sorting. They can significantly enhance efficiency and reduce the costs associated with waste management. However, challenges such as autonomous navigation in complex urban environments, safe interaction with citizens, and the coordination of multiple tasks simultaneously present significant obstacles yet.

Parallel to the information search for the waste collection robots, a deployment new protocol for these robots in public spaces has been developed. As we have seen, this includes phases to address interaction with public spaces, people, and existing mobility systems.

Environmental sustainability assessments, such as those conducted by Mora et al. (2014), Gilardino et al. (2017), Pérez et al. (2017), and Pires et al. (2017), provide insights into the environmental implications of waste management methods. These studies, employing Life Cycle Analysis (LCA), reveal that while pneumatic systems may generate more air emissions due to electricity consumption, they exhibit superior environmental performance when operating at full capacity. Economic evaluations, such as those by Teerioja et al. (2012), suggest that pneumatic collection systems are significantly more costly than traditional methods. Robotic systems could balance cost and emissions, but comprehensive studies are still needed. The key factors necessary for the design of robots have been analysed,

based on the survey developed by Zerkane et al. (2015), of 33 leading companies in the development of robots.

Studies primarily comparing pneumatic and door-to-door collection systems (Baltrochi APD., 2024; Dotsenko et al., 2022) typically use traditional traction vehicles rather than electric or green energy vehicles (Canals Casals et al., 2016, 2017). Transitioning to electric vehicles for waste collection, particularly for door-to-door systems, is likely to result in substantial emission reductions, making it a more viable waste management option.

The economic perspective, as discussed by Laso et al. (2019) and Yang et al. (2022), highlights the cost-effectiveness of the door-to-door system. However, a comprehensive evaluation of the economic advantages of integrating robotic systems compared to pneumatic facilities requires a more in-depth analysis. Factors such as initial setup costs, operational expenses, maintenance requirements, and long-term sustainability must be thoroughly examined.

In implementing last-mile robotic systems within urban areas, it is essential to follow a structured protocol as we have presented. This includes defining the project, studying various robot typologies, focusing on the technical development of the robot, and analyzing citizen acceptance and necessary adjustments for adaptation.

In conclusion, integrating robots into urban spaces offers numerous advantages, including increased efficiency and reduced operational costs. However, challenges such as potential job displacement, social acceptance, and the complexity and cost of implementation must be addressed. Careful consideration of these factors is crucial for the successful integration of robots into urban environments, especially based on data obtained from pedestrians who would share public space with robots.

The consideration of environmental, social, economic and efficiency factors is crucial for the successful integration of robots in urban environments, as we present in the protocol and it is necessary to obtain some of these from pedestrians who would share public space with robots.

In future work, we will conduct a technical comparison with systems proposed by other institutions. Furthermore, we will perform environmental and social analyses to evaluate the device's integration and incorporate diverse proposals from stakeholders.

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COMMUNICATING OUTPUTS, OUTCOMES AND IMPACTS OF WASTE-RELATED RESEARCH: THE “EMERGE” CASE STUDY

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ABSTRACT

Communication, dissemination and exploitation of research findings is vital to maximise research impact. This is increasingly important as society faces sustainability challenges that require people to be educated about a broad range of issues spread across academic disciplines, and this is especially true of the waste management industry. Despite this need, effective approaches to communication, dissemination and exploitation can be challenging. This paper presents the EMERGE project as a case study, presenting the approaches used, identifying areas of success and highlighting areas for further improvement. This reveals a successful strategy to communicating with scientific audiences but highlights the considerable potential for improvements in the approach to communicating with the general public. New approaches to social media usage for research projects need to be considered, such as working with social media professionals, using a greater variety of apps and proactive collaboration with influential users/accounts.

1. INTRODUCTION

In an era of sustainability concerns and climate emergency, environmental education is of critical importance if the transition to a sustainable society is to be achieved. Environmental education is defined by the Tbilisi declaration (1977) as “a learning process that increases people’s knowledge and awareness about the environment and its associated challenges, develops the necessary skills and expertise to address the challenges, and fosters attitudes, motivations, and commitments to make informed decisions and take responsible action.” (Gillett, 1977). The United Nations’ Educational, Scientific and Cultural Organization (UNESCO, 2021) has highlighted the key role that environmental education can play in encouraging sustainable development, by increasing public awareness and respect for nature. The research community can play its part in this by adopting effective strategies to science communication, dissemination, and exploitation of the research findings (Clark et al, 2016). It is therefore vital that the latest research relating to environmental and sustainability issues is communicated in the most effective way possible, and to the widest possible audience. This is particularly applicable for the waste management sector, as it links very strongly to the three pillars of sustainable development: so-

ciety, environment, and economy. Effective strategies for communication, dissemination and exploitation of results are therefore essential components of future research projects.

This paper presents the approach taken to communication, dissemination, and exploitation of research findings by the European Union’s Horizon 2020-funded project “Evaluation, control and mitigation of the environmental impacts of shipping emissions” (EMERGE). This EU project focuses on the impact of wash water discharges from exhaust gas cleaning systems (EGCS) as a waste-related case study. It explains the approaches taken to communication, dissemination, and exploitation of research findings, whilst evaluating successes and areas for improvement.

2. BACKGROUND

Science communication is the act of conveying scientific knowledge to diverse audiences, often with varying level of expertise (Garvey, 2014). This may involve communicating with other scientists, policy makers, relevant stakeholders, as well as the general public. This broad communication is critical if the research is to improve awareness among wider society and make a meaningful contribution to environmental education (Cox, 2013). Sci-



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ence communication creates a channel of communication between scientists and wider society, enabling research to provide greater impact (Lamb et al, 2018). In the past, this was mainly the responsibility of teachers, science writers, journalists, and outreach coordinators. However, scientists are now increasingly taking an active role in science communication themselves (Cooke et al. 2017).

Science communication uses a variety of methods, including traditional approaches such as publication in scientific journals, presentations and publications at conferences, seminars, or public lectures, as well as dissemination through printed and broadcast media. In addition, digital media is increasingly employed as a tool for science communication, encompassing email lists, digital newsletters, and social networking sites such as X (formerly Twitter), Facebook, Instagram, LinkedIn, and YouTube, as well as podcasting (Fox et al. 2021). Research can make an impact on policy through the distribution of policy briefs to governmental bodies and relevant stakeholders (Massey et al, 2013). These methods are summarised in Table 1. The most effective science communication strategies use a combination of all these methods to disseminate the results in a targeted way to different audiences (Dudley-Brown, 2019).

Dissemination of research allows the work to make a meaningful contribution to the discipline, influence policy and lead to evidence-based decision making. Ensuring these findings are available and visible to a broad audience creates opportunities for collaboration, new ideas, and innovation, as well as enhancing transparency and accountability (Dienlin et al. 2021). This is particularly important for research that receives public funding (Suber, 2012). Communicating research findings to the public and key stakeholders not only helps create trust in science, but

also raises awareness of the issues involved (Weingart and Guenther, 2016).

Sustainable development requires complex decision-spanning social, environmental, and economic aspects. Research can no longer be siloed within academia and separated from wider society. This entails a complete change in how knowledge is consumed, eliminating barriers to information and fragmenting disciplines. Education for sustainable development (ESD) has been encouraged as a way to address this, with UNESCO (2021) stating that ESD “gives learners of all ages the knowledge, skills, values, and agency to address interconnected global challenges including climate change, loss of biodiversity, unsustainable use of resources, and inequality. It empowers learners of all ages to make informed decisions and take individual and collective action to change society and care for the planet.” ESD has been highlighted as an essential tool for achieving the sustainable development goals by 2030 (UNESCO 2021).

As a further step beyond communication and dissemination, the exploitation of research enables a sustained impact after the project concludes. This may include the use of research findings in published literature, further development of tools, models and approaches created during the research, the implementation of the research via policy, and the incorporation of the research findings into education, such as via university and schools’ lectures.

Despite the clear benefits and importance, challenges still remain in relation to the communication, dissemination, and exploitation of research. Overcoming these challenges is vital if the cutting-edge of research is to be translated into meaningful outcomes. International research projects such as those undertaken by the European Union require a clear approach; however, this is often difficult to achieve. To overcome these challenges, it is essential to conduct a detailed study of the successes and failures of existing approaches. By identifying paths to success and learning from failures, we can pave the way for effective solutions. There is a lack of publications that presents comprehensive examples of these strategies for other researchers to learn from. This will be explored in this paper in more detail by presenting the EMERGE project as a case study.

3. EMERGE PROJECT

EMERGE is a 4-year project funded by the European Union’s Horizon 2020 research and innovation programme. It aims to systematically analyse the complex interactions between technological options for pollution reduction, pollutant emissions and dispersion, and the environment. This especially focuses on the effectiveness of exhaust gas cleaning systems (EGCS), or scrubbers, used on ships to reduce pollutants, and the impact the discharge of the resulting wash water may have on the marine environment. Open-loop scrubbers discharge wash water as a waste product directly into the environment. Closed-loop scrubbers store this wash water and need specialist facilities in ports to manage this waste. The EMERGE project therefore has a clear potential contribution to sustainable development and the wider waste management sector.

TABLE 1: Methods used in science communication and dissemination.

Key approaches	Examples
Academic articles and presentations	- Journal papers - Conference papers and presentations - Seminars - Events
Direct communication with stakeholders, governmental bodies, and potential users	- Press releases - Policy briefs - Events - Collaboration with and feedback from stakeholders
Digital communication	- Social networking sites - Online Newsletters and email lists - Podcasting - Online distribution of media (digital versions of printed media) - Blogs - Websites - Members/stakeholder network enrolment - Online survey
Printed media	- Magazines - Leaflets - Posters - Booklets
Mass media	Use of television, radio, magazines, and newspapers

The EMERGE project's broad aims were, firstly, to comprehensively quantify and evaluate the effects of a range of potential emission reduction solutions for shipping in Europe, and secondly, develop more effective strategies and measures to reduce the environmental impacts of shipping. This research has been undertaken by a consortium containing eighteen partners from ten European countries, and a further five international partner institutes. This network was expanded by 136 stakeholders and thirteen advisory board members recruited in the preliminary stages of the project. Details are available on the project's website (<https://emerge-h2020.eu/>).

While the EMERGE project benefits from a large network, granting access to resources and stakeholders that may not be possible in smaller-scale research projects, its approach to communication, dissemination and exploitation can provide useful insight to projects of all sizes.

3.1 Approach to communication and dissemination

The EMERGE project has deployed a wide range of approaches to communication and dissemination, which targets a diverse range of audiences. The target audiences identified in the communication strategy are:

- The EMERGE consortium partners
- Academic, research and general scientific/engineering community
- Shipping sector and community
- End users
- Policy makers
- Education community
- General public

These audiences have been reached in a variety of ways. In particular, the academic, research and scientific/engineering community has been targeted primarily through the publication of academic journal articles and conference papers. This approach has reached many key audiences, notably the shipping sector and community, end users and policy makers. The academic papers produced by the EMERGE project have been specifically aimed at open-access journals, resulting in 43 papers published and 3 accepted for publication by end May 2024 (Appendix 1). At least another 30 papers are in preparation for publication in peer-reviewed journals. Utilising open-access journals ensures that the research findings are accessible to all the target audiences highlighted above. A total of 68 papers were presented at international conferences, with presentations also given at a wide range of international conferences, workshops, webinars, working groups and committees. A full list is available (Williams and Neophytou, 2024).

The EMERGE consortium agreed to produce eight 6-monthly newsletters and four annual press releases from the project. These were successfully delivered – on time and on budget – and are available to view on the website. The aim of the newsletters was to disseminate information about the project and its progress to a wide array of stakeholders as. It provided updates on past and upcoming project events, international events where the EMERGE

Consortium participated or was planning to participate in, and project news. It was an important communication tool as it contributed to building support for the project from all target groups and the public. The project's first newsletter included the following sections:

- Short project description
- Project's Mission
- Project's Objectives
- Project's Scope
- Project Consortium
- Project News (i.e. the starting of the project and its kick-off meeting)
- Future events (in which the EMERGE Consortium will be participating)
- Join the EMERGE Information Network (invitation to stakeholders)
- The project's social media accounts
- EU disclaimer

The structure for the next newsletters remained the same by removing the introductory details of the project.

Press releases were intended to communicate the project's developments or announce important achievements. They were released annually and aimed to update the targeted group within EMERGE's Information Network stakeholders, along with the public, on the project's progress. The project's press releases included project research findings and selected outcomes (i.e. case studies outcomes) accompanied by a bespoke Infographic that depicted the scope and approach of the EMERGE project. Additionally, the press releases included case study maps showing the areas of sampling and measurements for each case study.

Numerous tools have been used to communicate and disseminate these findings beyond academic journals. The project website was used as an information platform with the aim of organising key project information for partners and visitors, including stakeholders, members of the public and scientists. In addition, the project has benefitted by utilising the EMERGE member's websites, blogs, and media channels to further promote the research. Project newsletters, press releases and a summary booklet have been created and distributed to subscribers (Figure 1). Posters, briefing sheets, leaflets, roll-ups, and presentations have been created for distribution/display at events and online. Eight short documentary videos have been produced and have been made available via social media and websites. A stakeholder's workshop from 2021 was recorded and uploaded to YouTube as a long video.

Project events were meticulously coordinated and organised, encompassing stakeholder workshops (e.g., EMERGE Stakeholder Workshops in March 2021 and February 2024), conferences (e.g., the 13th International Conference on Air Quality), as well as seminars (e.g., a collaborative seminar with the SCIPPER project, EMERGE's sister project, held within the framework of the Posidonia 2022 Exhibition). These gatherings provided EMERGE partners with the opportunity to showcase their findings and engage stakeholders. Additionally, project representatives actively



FIGURE 1: Illustrative examples of an EMERGE project press release, newsletter and summary booklet.

participated in various conferences, where they presented the EMERGE project and their work within its framework, thereby broadening its reach to academic and professional communities. Other outreach efforts encompassed diverse communication campaigns including radio and TV interviews, increasing the project's visibility. The results of the project have been incorporated into the creation of educational lectures and seminars aimed at students, along with relevant training materials.

The use of social networking sites is a key aspect of the communication strategy (Table 2).

Numerous forms of social media were used to reach difference audiences. LinkedIn was used to target a professional audience. Other forms of social media were used more generally with the public in mind as a target audience. YouTube was used only as a platform to host videos which were then be shared to the other social media platforms.

The hashtags and profiles tagged on X (Twitter) are also shown in Table 2. This highlights that the chosen hashtags are all specific to the issues investigated.

3.2 Approach to exploitation

The EMERGE project developed a comprehensive strategy to enable continued exploitation of the research findings after the project has finished, with the following aims:

- Use of the project's results and policy recommendations in national, regional, and international conferences, fora, and organizations
- Use of the project's findings in scientific journals
- Use and further development of the project's data and tools (decision support tool) available on the website
- Development of new lectures and seminars for presentation to students

The target audiences for the exploitation strategy are organisations, entities or individuals that can benefit from EMERGE's results were identified:

- The EMERGE consortium partners
- Academic, research and general scientific/ engineering community
- Shipping sector and community
- Other end users (local authorities, port authorities, regulatory bodies)
- EU and other international policy makers
- Education community
- General public
- Research funding agencies
- Experts
- Media groups

This strategy aimed to maintain the project website's availability, further disseminate project's results at conferences and forums, and sustain the availability of the social media/printed media posts created to communicate findings. Considerable efforts were made to provide input to key stakeholders and policy makers, including:

- National governments
- the International Maritime Organisation (IMO)
- HELCOM (Baltic Marine Environment Protection Commission)
- the OSPAR Commission (so named because of the original Oslo and Paris Conventions ("OS" for Oslo and "PAR" for Paris))
- the International Council for the Exploration of the Sea (ICES)
- the Baltic Sea Action Group (BSAG)
- and EU expert groups (i.e. the European Maritime Safety Agency (EMSA)).

TABLE 2: Social media channels, target audiences and content type.

Social media	Target audience	Type of content	Hashtags/Tagged Profiles
X	General public, key stakeholders, local authorities and governmental bodies, marine/maritime/shipping organisations & companies.	Infographics, videos, links, pictures, news, documents.	#EuEMERGE #H2020 #Shipping #Marine #Maritime #MarinePollutionPrevention #INEA #scrubbers #MitigationScenarios #ShippingEmissions #EnvironmentalImpacts #EmissionReductionSolutions @EU_H2020 @EUscienceInnov @EU_Commission @EUSciComm @EU_MARE @EU_ENV @Transport_EU @inea_eu
LinkedIn	Scientific community, professionals from related areas, local authorities and governmental bodies, marine/maritime/shipping organisations & companies.	Infographics, pictures, videos, links, news, documents.	
YouTube	Not specified	Audio-visual contents that will be shared on other media and platforms	For example, "Tidelines" video: https://youtu.be/raDZvo8rF70
Facebook	Not specified	Infographics, videos, links, pictures, news, documents.	

This was also achieved via attendance at meetings and provision of policy briefs, as well as working directly with the shipping industry and producers of abatement equipment. This strand of the exploitation strategy was made possible by the large EMERGE consortium and may not be possible for smaller research projects; however, it illustrates very clearly the benefits derived from collaborative efforts.

The strategy aimed to achieve the short-term targets of maintaining a network of key experts via collaboration between consortium members and the use of professional networks such as LinkedIn and promoting and supporting existing online tools such as the website, social media, newsletter, and documentary videos. The mid- and long-term targets were designed to enable working with international groups and surveying project partners to identify further opportunities that have arisen from the EMERGE project.

3.3 Outcomes

Whilst the project was only completed at end May 2024, there is unmistakable evidence of success. The production of promotional and educational material is shown in Table 3 (and available via the website <https://emerge-h2020.eu/>), demonstrating that the most used approach has been the production of short videos to highlight the research findings. The second most used approach is the distribution of newsletters twice a year, aiming at providing updates on the project, news, publications, and future events. These newsletters were distributed to a global network of subscribers and were presented in the form of short and succinct documents with a focus on conveying key information effectively, with a range of 2 to 10 pages, with the majority falling between 3 and 5 pages long. This prevents recipients being overwhelmed with too frequent and too long newsletters and losing interest. These newsletters have encouraged further exploitation of the research findings by highlighting publications produced by project partners and published to peer-reviewed journals. This communication strategy makes a direct contribution to the exploitation plan.

This strong focus on videos has been highlighted in the literature as an effective tool for science dissemination,

particularly for use in the education sector as an effective tool for helping non-specialist users understand scientific knowledge (Giron-Garcia and Fortanet-Gomez, 2023). The policy briefs due to be created will also help implement the exploitation plan. Table 5 highlights a preference for digital dissemination over printed and physical media. This is essential for an international research project such as this and allows the research to reach a wider audience.

Eight short documentary videos have also been produced, with one video per work package. These videos have been distributed on the project's social media channels and may make a valuable contribution to communication and dissemination. In addition to this, the stakeholder workshops have been recorded and are available via YouTube (e.g. <https://www.youtube.com/watch?v=dQrdnYvt360>).

The total numbers of people reached using all dissemination and communication activities according to internal mid-term review reports from the EMERGE project are shown in Table 4, also highlighting areas of success.

The high number of over 129 million members of the general public reached is according to project partner Chalmers, with their press release reaching approximately 129 million people. The press release received significant attention from global media outlets, earning it a spot-on Chalmers' top-ten list for widespread coverage. This highlights the importance of multiple avenues to reaching the

TABLE 3: Outcomes of dissemination strategies.

Approach	Quantity
Briefing note distribution	1
Roll up banner	1
Flyers and posters	1 of each
Newsletters	8 (2 per year)
Press releases	3 (1 per year)
Videos	1 project introduction, 8 short documentary videos and 2 long recorded stakeholder workshop videos from 2021 and 2024.
Policy briefs	Summary booklet published, briefs due at project end.

TABLE 4: Type of audience reached by dissemination & communication activities according to mid-term reports as of March 31st, 2023.

Type of audience reached in the context of all dissemination & communication activities (as of 31st of March 2023)	Estimated number of persons reached
Scientific community (Higher education, research)	7202
Industry	5052
Civil Society	983
General Public	129,007,442
Policy Makers	1795
Media	145
Investors	732
Customers	732
Other	1177
Total	129,025,260

general public, with press releases able to make a significant contribution.

The approach of the EMERGE project to publishing findings has clearly been successful, leading to 43 academic papers published to date, with a further 3 accepted for publication, against an overall target of 50. In terms of achieving the objectives of the exploitation plan relating to use of the projects results and findings, Overall (by end May 2024), the papers have been viewed close to 50,000 times, downloaded more than 11,000 times, and cited over 500 times. The exact counts for views and downloads will be higher as not all journals provide detailed information about downloads and view counts.

These results produce a mean citation count per paper of 14.6 citations (at the end of May 2024), which suggests the work has been highly impactful thus far, especially considering many of the papers have only been available for two years or less. The use of open-access journals has made the research papers more accessible to a larger audience and has contributed to achieving the exploitation plan. Young and Brandes (2020) found that open-access papers receive more citations and attract a more diverse, interdisciplinary audience, which is beneficial for research aiming at producing more sustainable outcomes, as it necessitates consideration across various disciplines.

The number of attendees at stakeholder workshops was 140 at the online stakeholder workshop in 2021 and 66 at the online stakeholder workshop in 2024. These events enabled the EMERGE project's progress and results to be disseminated to a range of stakeholders in multiple countries. A bespoke EMERGE EU stakeholder workshop was initially planned to be held in a hybrid format, with both in-person participation in Brussels and online attendance, scheduled for May 2024. Participants were representatives from the EMERGE project, the EMERGE EC Project Officer, and stakeholders from parent DGs, CINEA, and EMSA. The agenda, detailing arrangements for both in-person and online participation, along with the WebEx invitation for the online attendees, were sent to the EC Project Officer for

dissemination to EU stakeholders prior to the event. However, due to the Project Officer's inability to attend the event in person, and at their suggestion, the format was shifted to an entirely online event.

Prior to the workshop, the EMERGE e-booklet was distributed to EMERGE partners and the EMERGE EC Project Officer for further dissemination to EU stakeholders. This e-booklet provided a comprehensive overview of the project's key accomplishments, including scientific achievements, policy recommendations, and guidance. The EMERGE EU stakeholder workshop took place online on May 21, 2024, and lasted approximately three hours. During the event, project results and policy recommendations were presented to 30 attendees, including the EC Project Officer, one stakeholder from EMSA, one from DG MOVE, and the EMERGE team. The event was recorded, as per the EC Project Officer's request.

An innovative dissemination activity was a creative collaboration between the EMERGE team, three university faculties, the University of Southampton's (UoS) Widening Participation and Social Mobility (WPSM) team, three schools based in Southampton, England, and a local professional artist. The goal of "Tidelines" was to promote knowledge exchange on nature- and engineering-based solutions to climate change and decarbonisation in the marine environment based on the findings from the EMERGE project.

A knowledge exchange (KE) programme was developed to inform international policy-making and future, sustainable, collaborative projects that support levelling up and Equality, Diversity & Inclusivity skills & capability development in disadvantaged communities. This was achieved in February-June 2024 by combining science/engineering and the arts to inform, inspire and encourage engagement in solutions to contemporary issues that affect coastal communities.

"Tidelines" is an innovative multimedia project that placed international researchers, professional poets, and student poets ('Shadow Poets') in targeted youth settings in Southampton, primarily disadvantaged young people in the IntoUniversity Southampton West partnership. EMERGE researchers interacted with primary and secondary school children to discuss the complex interactions between shipping and coastal environments. There was a focus on discussing sustainable solutions to societal impacts, technological options, pollutant emissions and dispersion, and environmental costs/benefits. Additional benefits included the development of improved literacy, self-confidence, and a range of soft skills in the children. University- and Student Poets led research-based and creative workshops to inspire young people, teachers, and teaching assistants (TAs) to write and perform poetry about i) their current everyday lives in a coastal city and ii) how they feel nature- and engineering-based solutions could transform coastal areas for the better. The workshops culminated in a performance and interactive eBooks that featured poetry, audio, and videos created by the participants. The outputs will be hosted on the EMERGE project's website to ensure international communication/dissemination and input into policy-related discussions (Figure 2). We led pedagogical and practical training with the teachers and TAs, focussing on

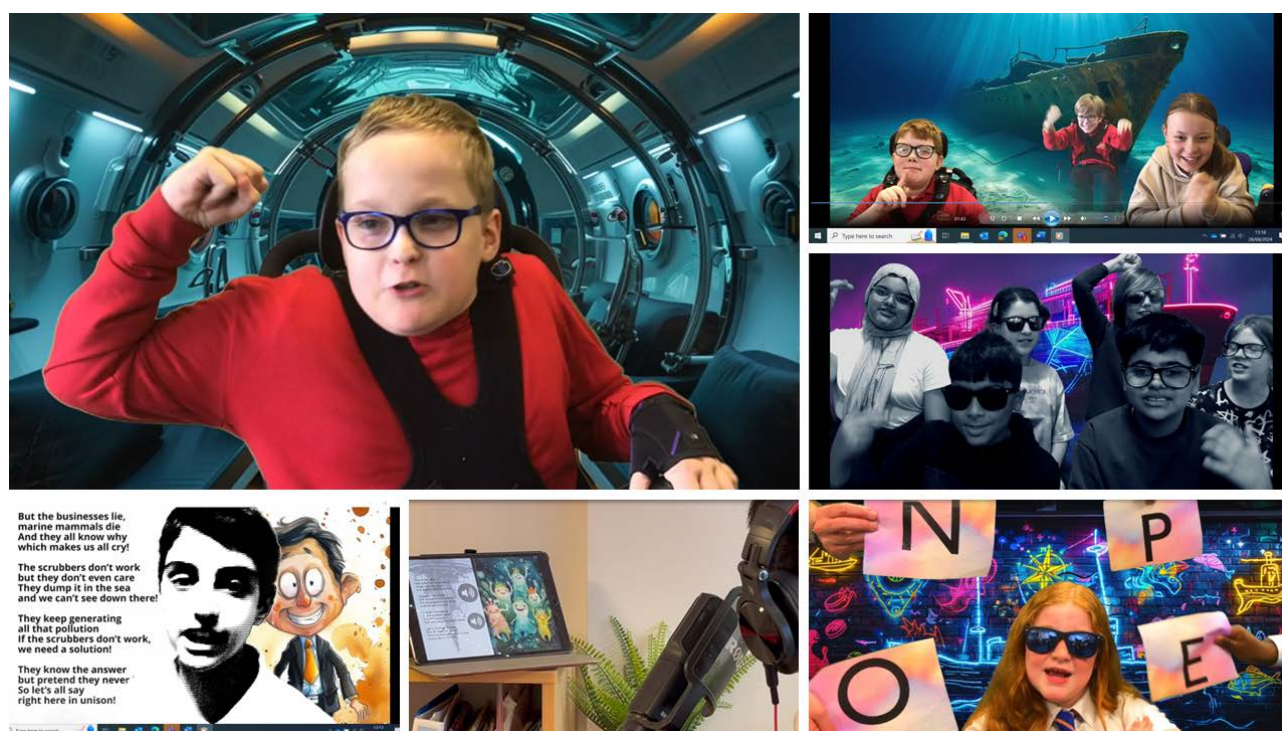


FIGURE 2: Screenshots from the Tidelines video featuring the rap “Listen Up Businesses” created by the school children as their response to what they learned from the EMERGE project’s findings.

Action Competences, speedy filmmaking techniques and iBook creation. The iBooks of poetry were be written and performed by the young people, the teachers, and the Shadow Poets. One iBook from each school and an anthology of selected poetry from the entire project will be freely available to download. Each interactive iBook contains poetry in accessible written, audio and video form. A final celebratory showcase event was held involving each school. The showcase celebration featured original poetry videos from the project, performances from the Young People and Shadow Poets, and launched the iBooks. The EMERGE project team will include the outputs and outcomes in their recommendations to international marine/maritime organisations.

Appendix 1 and the data above shows that the EMERGE project has successfully communicated its findings via peer-reviewed academic journals and clearly reached an academic and professional audience. This effective communication with professionals can be seen in the social media accounts, with LinkedIn performing the best, exhibiting a greater number of likes and comments than X, Facebook and YouTube combined, as shown in Table 5.

These statistics suggest that the EMERGE project has succeeded in communicating with other scientists and professional stakeholders. However, the lack of interactions on social media accounts targeting a broader audience indicates that, concerning outreach to the general public, the project has been eclipsed by e.g. immediate news stories, entertainment, and content from celebrities and so-called influencers. This remains a challenge for the scientific community but may be partially addressed by further work as the project comes to its end and all the outputs, outcomes and impacts become available. The exploitation plan will further increase the interaction with the general public after the project has finished.

Table 5 highlights that despite considerable efforts to build a considerable social media presence, with over 1000 posts overall, this has led to little interaction, with an average interaction of only 6.8 likes and comments per post. If likes and comments of those directly involved with the project were removed, this average would drop even further. This poses the possibility that if the interactions of those directly involved were not included, the actual interaction

TABLE 5: Interactions with social media accounts. Accurate 29.05.2024.

Social media	Followers	Number of posts	Total Likes and comments	Average likes and comments per post (mean)
X (formerly Twitter)	141	118	614	5.2
Facebook	123	94	510	5.4
LinkedIn	761	88	1508	17.1
YouTube	18	11	9	0.9
Overall	1,043	311	2,641	7.15

with the general public via social media may be negligible. This, however, does not necessarily reflect a failure of the EMERGE project, as other industries aiming at public engagement have also been shown in the literature to receive similar levels of interaction on social media (Roberts et al. 2021), and the EMERGE project is a project most useful to policymakers rather than the general public. A social media engagement rate of 1-5% is considered to be a good rate, although this becomes harder to achieve as the number of followers increases (Sehl and Tien, 2023). The average likes and comments per post shown in Table 5 reveals an average engagement rate of 3.7%, 4.4%, 2.2% and 5%, for X, Facebook, LinkedIn, and YouTube respectively. Thus, despite these numbers appearing to be low, they are still good engagement rates for social media.

The short documentary videos uploaded to YouTube further highlight the challenges. These provide an introduction, 7 videos for different work packages, and then a summary video. These videos have 47 views on average and 0.5 likes on average. Two key issues that are present in the YouTube approach are the names of the videos, which are named after the work packages, such as EMERGE WP1, WP2, etc, and the fact comments are turned off. These names make the videos difficult to find for the general public, who may use generic search terms such as “the impact of scrubbers” rather than searching for specific work packages. Having comments disabled also prevents interaction and may make the videos less visible, due to social media algorithms prioritising content with high levels of interaction. If interaction is disabled, then the videos are always going to struggle to be seen. The EMERGE project has used YouTube simply as a site to host videos and then share them on other platforms, which is where the main interaction with these videos is to be found.

Another aspect to consider is the information that viewers receive even when they do not directly interact with social media content. Chung et al. (2020) found that when social media was used for environmental education, viewers often did not interact with the posts or share them, but they did still acquire environmental information. This suggests that evaluating the effectiveness of social media simply by interaction levels may be ineffective, as it will fail to capture the exchanges of information that still take place. This highlights the challenges of science communication and despite its benefits, social media’s effectiveness is difficult to evaluate. Whilst scientists have taken an increasingly active role in science communication, and digital communication has become an increasingly widely used tool for science communication, the complexity of social media and the algorithms involved may mean that a lot of effort from the scientists may lead to little actual measurable public engagement. Posts on social media that are successful are often referred to as having gone viral, where the post amasses significant interactions in a short space of time. The reality of social media is that the vast majority of posts fail to take off in this way and achieve little interaction (Sehl and Tien, 2023). In order to go viral a post needs to satisfy the requirements created by an algorithm that then decides to what extent the post is prioritised, filtered, ranked, and recommended to other

users (Poyry et al, 2018). This makes using social media an art in itself. The rise of social media has led to the era of influencers and those who make a career using social media, attracting vast numbers of people to their content, and generating enough revenue to make a living. This shows that social media can be highly effective if used correctly and the potential for public engagement is vast, but identifying these strategies is challenging.

The social media sites of X, Facebook and YouTube may limit the reach of the research, with apps such as TikTok, Instagram and Snapchat proving more popular with young people (Hogan, 2022). This may be because apps such as TikTok have rapidly become popular during the years that this research project has taken place. This suggests that future research projects should consider using new apps as they emerge. For example, Ofcom (2023) found that TikTok was the most favoured source of news for teenagers in the UK, followed by YouTube and Instagram. Research investigating the potential role of TikTok in science communication has found that it was a useful platform for providing summaries of research papers in video form, and that 84% of users felt higher levels of trust in science and scientists after following (Rein, 2023). TikTok may represent a useful tool for this and other projects to use in the future to distribute short documentary videos.

The EMERGE project has only posted 11 times on YouTube, and has mostly focused on X, Facebook, and LinkedIn. X and Facebook both have declining user numbers and are less popular with young people (Canales, 2021), and LinkedIn is largely a professional network. This means the reach of this project is limited in regard to the general public and future projects should consider a more diverse range of apps for disseminating the findings if young people are to be more effectively included. Another option to increase the reach of social media posts would be to pay to advertise, which is something most businesses actively do on social media already (Fulgoni, 2015). If academic work is to achieve the same visibility it may have to be promoted in the same way and invest some of the budget assigned to communication and dissemination, if any exists, into actively promoting the work on social media via paid advertising and/or collaboration with influential users. Artificial intelligence (AI) is likely to become a useful tool in helping make engaging social media content in the near future.

The hashtags identified in the communication and dissemination strategy are shown in Table 2. However, in reality, the majority of posts on the X account contain no hashtags, reducing their potential reach. Whilst these chosen hashtags may have been useful for engaging with key stakeholders, government authorities, maritime industries etc, one issue of using these highly specific hashtags is that it is not going to engage the general public. These hashtags will be followed by those with an existing interest in shipping and maritime issues, rather than those with a general willingness to learn who may follow more general hashtags relating to the environment, science, and sustainability. The largest potential audience on social media is those who are not already actively engaged with the accounts, however with these hashtags there is no obvious

way to engage them, as most of them are very content specific. One way a project could improve its exposure to the general public would be to use more general hashtags broadly relating to the issue presented, such as #environment #sustainable #sustainability #green #ecofriendly etc. (Matthews, 2011).

Another issue which will impact the EMERGE project's online visibility is the fact that another European Union Horizon 2020 funded research project shares the same name; the evaluating mHealth technology in HIV to improve empowerment and healthcare utilisation: research and innovation to generate evidence for personalised care (EmERGE). Whilst it is inevitable that similar names may be used over time, it makes it more difficult for the project website and associated social media accounts to stand out, which may have a detrimental impact on long-term legacy exploitation.

4. CONCLUSIONS AND RECOMMENDATIONS

The approach taken by the EMERGE project demonstrates a successful example of science communication, dissemination, and exploitation. A large number of academic articles have been published and there is a large social media presence. Rapid publication of results has allowed the research to be well-cited before the project has even finished, leading to a highly impactful research output. This highlights the importance of prompt publication of research findings and the benefits of publication in open-access journals. The large consortium of members and wider stakeholder network has ideally positioned the EMERGE project to implement its exploitation plan successfully. For smaller projects that lack this network, collaboration should be encouraged wherever possible to expand the potential strategies available.

The project has reached the public successfully via press releases which were communicated by the media, with over 129 million views. One aspect of the strategy that could be elevated further is a potential lack of engagement with the public via social media, which is a common issue with public engagement approaches. The EMERGE project outcomes are most useful for stakeholders, and there is a limit to the impact the general public could have on encouraging the implementation of research findings, however, this will not be the case for many other waste-related and sustainability-related research topics. Improving engagement with the general public will be highly important if these projects are to be translated into behavioural change.

Social media represents the most widely used tool used in the approach to engaging with the public, and it is possible that if interactions from consortium members were to be removed from social media, there may be a low level of engagement. The documentary videos may also fail to reach the public due to the names assigned to these videos. Whilst this issue is not unique to the EMERGE project, it illustrates that even large projects can still struggle to engage with the public. To address this, science communication strategies should consider an innovative approach to social media use. Further research could at-

tempt to identify the most suitable strategies to increasing social media engagement, which may potentially include approaches such as paid advertisement, collaboration with influencers/successful accounts, better use of titles and hashtags, employing social media professionals, and content designed with social media algorithms in mind.

A greater variety of social media apps could be used to better reach the general public, especially young people who prefer apps such as TikTok, Instagram and Snapchat which are less frequently used for science communication. Podcasting could be used to increase the impact further. These tools may also be easier to use for smaller projects that lack a large stakeholder network.

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Extra contents

COLUMNS AND SPECIAL CONTENTS

This section comprises columns and special contents not subjected to peer-review

Environmental Forensic

UNRAVELLING THE LINK BETWEEN SOLID WASTE MISMANAGEMENT AND AIR POLLUTION - THE ROLE OF ENVIRONMENTAL FORENSICS

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1. INTRODUCTION

Open dumping and open burning are the primary waste treatment and disposal methods used in many developing countries (Ferronato & Torretta, 2019). Approximately 40% of the world's waste ends up in open dumpsites, especially in cities within middle and low-income countries that lack adequate waste collection and treatment systems. For instance, in many African cities, up to 90% of waste is openly dumped (United Nations Environment Programme, 2024). Even in cities where waste collection systems are well-established and offer extensive coverage, the lack of adequate treatment facilities and efficient processing systems often results in the collected waste being sent to landfills. Landfilling is linked to several environmental pollution issues, including (a) groundwater contamination from the leaching of organic, inorganic, and other harmful substances contained in the waste; (b) air pollution due to the suspension of particles and gaseous pollutants; (c) odour pollution affecting nearby communities; and (d) surface water pollution caused by potential runoff (Siddiqua et al., 2022). In addition, the accumulation of waste in landfills can pose long-term risks to public health and contribute to climate change, exacerbating health and environmental challenges globally.

From the air pollution perspective, in uncontrolled landfills and dump sites, waste can spontaneously combust due to factors like methane emissions from decomposing waste. Burning waste releases harmful carcinogens like dioxins, furans, and a short-lived climate pollutant, the black carbon, posing serious health risks. The combustion processes can also release polycyclic aromatic hydrocarbons (PAHs) and other volatile organic compounds (VOCs), further deteriorating the air quality. In recent years, the number of landfill fires has increased due to excessive heat, driven by climate change. From an environmental justice perspec-

tive, communities living near landfills are disproportionately exposed to severe air pollution and other attendant health and safety hazards. Growing awareness of these issues and the associated health impacts has led to a rise in citizens taking legal action, demanding cleaner air in cities, and raising the concerns of those living near waste dumpsites. However, there are no standard approaches, besides expensive real-time source apportionment tools, to understand the change in air quality levels attributable to waste dumping. The high cost associated with these tools makes legal redress a daunting task in developing countries.

Although, in a developing country context, in addition to poor waste management, sources like residential cooking and heating, transportation, and industries are also significant contributors to air pollution, actions are already in place to control emissions from these sectors. For example, in the case of residential cooking, several measures have been taken over the past few decades that have reduced exposure to household pollution (Health Effects Institute, 2024). In the case of transportation, we have vehicular emission standards in place, and for industries, we have industrial emission standards. However, waste management is an area yet to be addressed from an air pollution perspective and hence merits a calibrated strategy to address the issue. Despite the substantial impact of waste mismanagement on air pollution, strategies specifically addressing this issue have not received adequate attention in the past. Environmental forensics, as a techno-legal approach, has an important role to play in this situation.

2. THE ROLE OF ENVIRONMENTAL FORENSIC TOOLS

Environmental forensics in air pollution is used to identify pollution sources, track pollutant dispersion, assess environmental and health impacts, and quantify the



damages. The goal is to determine who is responsible for the pollution, assess the extent of environmental damage, and support legal or regulatory actions (Abdul Samad et al., 2020). Environmental monitoring plays a critical role in environmental forensic investigations. There are two primary approaches to environmental monitoring: one, the conventional air quality monitoring methods, or reference techniques, established by federal or state agencies. These monitoring methods are typically available for a limited range of pollutants. In developing countries, courts often rely on these standardised methods to assess pollutant releases. However, this practice has sometimes proven to be a limitation, as investigations are restricted to the pollutants listed by federal agencies, leaving other potentially harmful substances unexamined. The second approach is to look for chemicals that can act as fingerprints of the sources of pollution. This approach often requires monitoring at the potential sources (for emission inventORIZATION) and the receptor locations. The monitoring data along with appropriate modelling tools (dispersion/receptor) help to identify the sources, their relative contribution, and the time of release.

The first approach can be implemented by installing ambient air quality monitoring systems in landfill sites. Such systems can detect a range of harmful substances, including volatile organic compounds (VOCs), particulate matter, and greenhouse gases like methane. Several developed countries have made it mandatory to monitor air quality in landfill sites. However, this is not a common practice in developing countries. Moreover, when there are other possible sources for the monitored pollutants, establishing the contribution of landfills can be controversial.

The second approach involves using techniques like fingerprinting and modelling, where specific compounds act as indicators for various decomposition processes in landfills. For instance, methane serves as a fingerprint for anaerobic decomposition of organic waste, while hydrogen sulphide indicates the breakdown of sulphur-containing materials. Ammonia is a marker for the decomposition of nitrogenous organic matter. Specific pollutants like siloxanes point to the degradation of personal care products, medical waste, and other silicone-based materials. Compounds like polychlorinated biphenyls (PCBs), dioxins, and furans are linked to emissions from the breakdown of certain plastics, electrical components, and industrial waste. Further, the studies by Bakkaloglu et al. (2021, 2022) point to the possibility of using the difference in carbon isotopic signature of methane (CH_4) emitted from landfills of different ages to identify the specific landfill contributing CH_4 to the atmosphere. Landfill fires are not uncommon, and the use of compounds emitted during the fire incidents can serve as signatures for identifying the areas affected by these events. For instance, Polybrominated diphenyl ethers (PBDEs) were used as a signature compound to identify the areas within the city impacted by emissions from a major landfill fire in Santiago, Chile (Pozo et al., 2023).

Air pollution models are useful for quantifying emissions, identifying pollutant sources, and assessing health impacts from landfills. By inputting data on waste composition, decomposition rates, and other site-specific pa-

rameters, the emission models can estimate the rate of pollutants emitted. For example, the LandGEM (Landfill Gas Emissions Model), developed by the United States Environmental Protection Agency (USEPA), uses first-order decay equations to estimate emissions of methane and other gases from landfills based on waste acceptance data (United Nations Environment Programme, 2005). This model is used to calculate emission rates for total landfill gas, methane, carbon dioxide, non-methane organic compounds, and specific air pollutants from municipal solid waste (MSW) landfills.

Source apportionment models are valuable for assessing the impact of landfill emissions on local air quality by determining the proportion of pollutants originating from landfills using techniques such as fingerprinting. This knowledge is crucial in understanding the potential health risks associated with landfill emissions, including respiratory and cardiovascular diseases, and consequently in liability allocation.

Dispersion modelling tools predict how landfill emissions move through the atmosphere. Gaussian dispersion models, such as AERMOD, consider meteorological data (e.g., wind speed, direction, temperature), landfill topography, and surrounding land use to simulate pollutant dispersion patterns (Perry et al., 2005).

These approaches discussed above are not just relevant for the forensic applications but also for the regular air quality management efforts as discussed below.

i) Compliance with national standards: National Ambient Air Quality Standards (NAAQS) is a legal tool available in the majority of the countries, and the states or provinces are mandated to take necessary steps to maintain these standards. However, in developing countries, the exceedance of these standards is very common. There are provisions under the Air Acts to monitor the NAAQS, but there is no way to identify the sectors or activities contributing to it on a day-to-day basis. Enhanced monitoring can identify pollution-causing activities at landfills and support mitigation strategies. Air quality models, such as AERMOD recommended by the USEPA, are vital for demonstrating regulatory compliance. Developing countries can use these models to assess landfill impacts and ensure emissions stay within limits, aiding the permitting process. These tools help operators and regulators make informed decisions to protect public health and the environment.

ii) Community exposure and health assessment: Although not a replacement for exposure monitoring, ambient air monitoring can provide a reasonable estimate of pollutants in the breathing zone. Thus, it reflects the community's exposure to air pollution. However, in developing countries, ambient air quality monitoring data near landfill sites is very limited. Even in developed countries, air quality monitoring (AQM) can be spatially limited, potentially failing to adequately represent the impact if they are not positioned downwind of the landfill. Mobile AQM stations can be used to fill in the gaps; however, due to their high cost, they cannot significantly increase spatial sampling density. Any lack of data can hinder the ability to accurately assess community exposure and implement effective health interventions for the communities living in the close vicinity of

a landfill site. A cost-effective approach for AQM would be the establishment of mixed networks including both reference-grade monitors as well as emerging sensor technologies (Gani et al., 2022). Further, dispersion model predictions help in identifying areas at higher risk of exposure to harmful pollutants, establishing buffer zones around landfills, and strategically placing monitoring stations.

iii) Intervention assessment: Assessing the effectiveness of interventions at landfill sites to reduce pollution is often challenging. Installing air quality monitors would help determine which measures have been successful and which have not. Dispersion models are used to evaluate the effectiveness of various mitigation strategies, such as gas collection systems, flaring, and biocovers. By simulating scenarios with and without these measures, models can provide quantitative assessments of emission reductions.

3. THE IMPORTANCE OF REGULATORY SUPPORT

In developed countries, environmental forensics is being utilized to ensure that environmental regulations are strictly adhered to and hold polluters accountable for air pollution caused by waste mismanagement. In the European Union, directives such as the Waste Framework Directive and the Ambient Air Quality Directive mandate the member states to ensure comprehensive monitoring and reporting of emissions from various waste management facilities, including landfills. These directives require member states to implement national laws and strategies that ensure continuous air quality monitoring and enforce strict penalties for non-compliance. In response, local authorities are driven by national legislation to adopt environmentally sound practices of waste management.

On the other hand, environmental regulators in developing countries have to contend with deficiencies in legislative framework and state capacity. For instance, in India, air pollution is regulated under the Air Act, and waste is managed under the Environmental Protection (EP) Act and different rules, including municipal, plastic waste, biomedical, hazardous, e-waste, and construction waste management rules. Under the Biomedical Waste Management Rules, it is mandated to install air quality monitors in biomedical waste processing centers across the country. Yet, these rules are unevenly implemented. Even rights-conscious judicial interventions for upgrading enforcement capacities could produce only mixed outcomes (Gill, 2024). Furthermore, India's air pollution control legislation does not penalize local government officials for contravening their duties. Since municipal authorities are primarily responsible for solid waste management, there is a worrying gap in existing accountability mechanisms. Similar deficiencies in the legislative regime are compounded by structural limitations of local government, resulting in the release of toxic air pollutants on account of waste mismanagement. Thus, it is critical to harmonize the enforcement and penalty regimes under air quality and waste management laws. Under the provision of waste management rules and the Air Act, it can be mandated to install air quality monitors near landfills and dump sites, regularly monitor, and take

necessary measures if it exceeds the NAAQS. For all other sources of pollution, it is possible to establish emission standards, but in the case of open landfills, this can be a suitable way, and the residents around the landfill will also be aware of the prevailing air pollution levels.

Effective accountability should involve not only penalizing immediate violations but also ensuring the implementation of robust waste management practices based on source segregation, efficient collection and scientific treatment and disposal, and regular air quality monitoring. This comprehensive approach is essential for addressing the environmental and public health impacts associated with landfill operations and for ensuring long-term improvements in waste management practices.

4. ENFORCEMENT

The effectiveness of environmental regulations is significantly compromised by the lack of a stringent penalty regime. In developing economies, including India, the enforcement of environmental laws related to waste management and air quality are restricted by inadequate penalty mechanisms. While penalties for specific episodic and visually evident violations, such as landfill fire outbreaks, are occasionally imposed by regulatory bodies or judiciary systems, regular issues, such as PM emissions from vehicular movement while transporting waste, ongoing smouldering of waste, and constant methane emissions from decomposing waste, often go unnoticed and unaddressed.

In addition to air monitoring, critical preventive measures designed to minimize air pollution and reduce landfill fire hazards need to be enforced. These measures include daily compaction of waste using dozers and covering waste with a layer of soil or debris of sufficient thickness to prevent exposure to the environment. These practices are intended to control emissions, prevent fires, and manage waste more effectively. However, compliance with these protocols is often lacking in developing countries, leading to poorly managed sites that do not adhere to scientifically sound operational standards. This gap in compliance highlights the urgent need for robust accountability mechanisms for municipal government and waste generators.

5. CONCLUSIONS

Developing countries face significant challenges with solid waste management and air pollution. Ineffective waste management practices, such as open dumping and burning, significantly contribute to air pollution. Despite the clear link between waste mismanagement and air pollution, targeted strategies to address this issue have often been overlooked. Environmental forensics, which includes tools such as monitoring, modeling, and fingerprinting, plays a crucial role in this context. These techniques can help identify emission levels, sources, and timing of emissions. To effectively tackle the issue of air pollution from waste management practices, it is essential to align enforcement and penalty measures across air quality and waste management regulations. Environmental forensics has a significant supporting role to play in this.

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Grassroots Innovations from Youth

REDUCING FOOD WASTE: INSIGHTS FROM STUDENTS AT THE UNIVERSITY OF VICTORIA AND JARAMOGI OGINGA ODINGA UNIVERSITY OF SCIENCE AND TECHNOLOGY

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1. INTRODUCTION

This column explores the strategies students at the University of Victoria (UVic), Canada, and at Jaramogi Oginga Odinga University of Science and Technology (JOOUST), Kenya, are adopting to reduce food waste on their campuses. Through interviews, the research highlights the creative and practical ways students address food waste while identifying systemic changes universities could implement to support these efforts. From grassroots initiatives to institutional suggestions, this analysis highlights the role of students as both agents of change and advocates for sustainability.

1.1 Background on Student-Led Food Waste Solutions

Food waste is a pressing global issue (FAO, 2015), and universities often reflect the complexities of these challenges within their campuses. Functioning as small-scale models of society, campuses are shaped by the interactions of various actors; students, faculty, administrators, and food service providers operating within a network of institutional policies, infrastructures, and external market pressures (Boschini et al., 2018; Marais et al., 2017). These environments face constraints such as limited food options, inefficient dining practices, including poorly prepared meals, and inadequate facilities for food storage and management. Despite these challenges, students actively develop innovative strategies to reduce waste, establishing themselves as vital contributors to sustainability efforts.

Student-led solutions are often rooted in the concept of grassroots innovation—practical, community-driven approaches that are tailored to the immediate needs of their environment (Gutberlet, 2023). These efforts typically involve small-scale, low-resource initiatives that address specific campus challenges while aligning with broader goals of sustainability and waste reduction. For example, students often reimagine traditional practices or adopt creative approaches to food preservation and redistribution, reflecting both their adaptability and their awareness of systemic inefficiencies.

Unlike top-down solutions, these grassroots innovations focus on addressing everyday issues students encounter, such as food spoilage because of the lack of storage facilities. They showcase students' ability to respond proactively to challenges while advocating for longer-term changes, such as improved infrastructure or institutional policies. These efforts demonstrate not only practical action but also critical reflection, as students seek to both mitigate waste in their immediate surroundings and inspire systemic change across their universities.

2. METHODS

The study used a qualitative approach with semi-structured online in-depth interviews involving twenty undergraduate students: ten each from UVic and JOOUST. These sites were selected due to researchers' location and established international connections in Kenya. Semi-structured interviews offered flexibility for in-depth discussions (Dunn, 2021). Data collection occurred between November 2023 and May 2024, with ethics approval from the University of Victoria's Research Ethics Board (Protocol #23-0328). Participants at JOOUST included six females and four males (eight first year and two third year students). At UVic, participants comprised seven females and three males (nine first-year and one third-year student) and one international student.

Interviews were conducted via Zoom, leveraging its ability to overcome challenges related to time, cost, and location (Dunn, 2021; Thunberg & Arnell, 2021). Sessions lasted 30 to 45 minutes, providing students with the opportunity to share detailed insights. While several open-ended questions were posed, the ones most relevant to this grassroots innovation column focused on two key areas: practices students use to reduce food waste and what they believe their universities could do to support these efforts.

Convenience sampling (Acharya et al., 2013) was used to recruit undergraduates aged 18–24 who lived on-campus and regularly ate at residence dining halls. At UVic, recruitment involved posters (Figure 1), direct approaches in common areas, and assistance from professors. At

RESEARCH PARTICIPANT RECRUITMENT

Exploring Food Waste in Universities: A Case Study of the University of Victoria and the Jaramogi Oginga Odinga University of Science and Technology.

UVic students are invited to participate.

What is involved?:

A 30 - 40 minute-long individual in-depth interview with UVic students to share their experiences with food and food waste on the university campus.

Eligible participants:

- Students between the ages of 18 - 24.
- Students who frequently visit eateries on UVic campus at least twice a week.
- Students residing in on-campus dorms.

More information:



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Disclaimer: This study has received approval from the UVic Review and Ethics Board (23-0328).



FIGURE 1: Recruitment poster.

JOOUST, a research assistant employed snowball sampling to identify eligible participants (Kirchherr & Charles, 2018).

Participants received interview questions in advance to facilitate thoughtful preparation and reflection. Informed consent was obtained before interviews, with participants briefed on their rights, including the ability to withdraw at any time.

During data collection, we used Zoom's "View Full Transcripts" function to generate real-time transcripts of each session. These transcripts were reviewed for accuracy by cross-referencing them with audio and video recordings to correct any errors or omissions. Familiarizing ourselves with the material allowed us to efficiently evaluate and extract key information.

Data analysis followed a systematic process: open coding was used to identify and categorize significant concepts relevant to the research objectives (Glaser, 2016). This was followed by focused thematic coding of group related data, and finally, selective coding to develop themes and establish connections between categories.

3. STUDENT INNOVATIONS: INSIGHTS FROM THE UNIVERSITIES

This section explores how students at JOOUST and UVic are tackling food waste through distinct yet impactful strategies. From traditional knowledge to collaborative food recovery initiatives, their approaches highlight the power of adaptability and collective action in addressing one of the most pressing global sustainability challenges.

3.1 Jaramogi Oginga Odinga University of Science and Technology

At JOOUST, students use traditional knowledge to address infrastructural challenges that limit access to refrigeration. Restrictions on personal refrigerators, due to concerns about fire risks, limited space, and high electricity costs, have led students to develop innovative food preservation methods. For example, vegetables are left outdoors overnight to stay fresh through the cooling effect of natural dew. This creative adaptation to environmental conditions leverages local resources to minimize food waste.

This approach demonstrates the competence and adaptability of students in resource-scarce settings, turning constraints into opportunities for sustainability. By relying on traditional practices, students address modern challenges while ensuring food security, providing a compelling example of grassroots innovation in action.

3.2 University of Victoria

At UVic, students employ different strategies rooted in community action and sustainability. The Community Cabbage initiative, a student organization at UVic, exemplifies this, as students engage in food recovery by rescuing surplus produce from local stores and preparing free weekly hot meals at campus from the reclaimed food items, serving students affected by food insecurity. This project raises awareness about food waste, fosters a sense of community, and promotes food security through collective action. By turning food that would otherwise be discarded into nutritious meals, students highlight the potential of collaborative efforts in reducing waste and addressing systemic issues.

UVic students also participate in other grassroots movements like freeganism and dumpster diving, also recovering edible food discarded by grocery stores and restaurants throughout the city. These practices reflect a broader cultural commitment to resourcefulness and sustainability, often shaped by personal values and upbringing in households that emphasize minimizing waste. Through these actions, students not only reduce food waste but also challenge societal norms around consumption, wastefulness and disposal, advocating for a more mindful approach to resource use and avoidance of waste.

3.3 Connecting Local Solutions to Global Challenges

The initiatives at JOOUST and UVic illustrate the power of student-led, grassroots innovations in addressing food waste. Whether through the application of traditional knowledge to preserve food or creative community-based recovery strategies, these efforts demonstrate how students can lead in developing sustainable, locally relevant solutions. By blending resourcefulness, adaptability, and critical reflection, these practices offer valuable lessons for tackling food waste at both institutional and societal levels in institutions both in the global North and in the global South.

4. INSTITUTIONAL SUGGESTIONS FROM STUDENTS

While students actively use innovative ways to reduce food waste within their campus and local communities, certain systemic issues remain beyond their direct control. Factors such as large portion sizes, poorly prepared meals, limited food variety, and inadequate infrastructure often lead to unnecessary waste. Recognizing these challenges, students at JOOUST and UVic have proposed several actionable measures their universities could adopt to foster a more sustainable and waste-conscious dining environment. Promoting this dialogue at the institutions further pushes for necessary change and adaptation. The following sections outline their key recommendations.

4.1 Community Fridges

Students at both universities identified communal fridges as a practical solution to reduce food waste and address storage challenges. At JOOUST, where students are not permitted to have personal refrigerators due to high electricity costs, limited space, and fire safety concerns, communal fridges could offer a centralized alternative. By reducing the need for individual appliances, this approach could lower energy consumption and alleviate space constraints in student housing.

At UVic, while students are allowed to bring personal fridges, not all of them do, and those who have them often underutilize the storage capacity. Communal fridges could help optimize food storage, prevent waste from forgotten or spoiled items, and save electricity costs across residences.

To succeed, communal fridges would require shared responsibility. Universities could implement clear usage guidelines with weekly cleaning schedules, and oversight to ensure these resources remain hygienic and effective. By promoting collective ownership, these fridges could serve as a cost-effective, sustainable solution to address food waste on campuses.

4.2 Portion Control

Fixed portion sizes in campus dining facilities, like those at UVic, often result in over-serving and unnecessary waste. With a “one-size-fits-all” approach to meal portions, students frequently end up with more food than they can consume. To address this, students at UVic proposed introducing sliding scales and customizable portion options, allowing individuals to select the quantity that best suits their appetite.

Implementing plate size choices or a “build-your-own-portion” system could significantly reduce waste while enhancing student satisfaction by catering to diverse needs. Universities could also pair this initiative with awareness campaigns promoting mindful consumption, encouraging students to take only what they need and fostering a culture of sustainability.

4.3 Fair Treatment in Food Service

At JOOUST, students highlighted concerns about favoritism in food distribution, noting that this practice not only causes dissatisfaction but also leads to unnecessary food waste. Unlike UVic, where fixed portion sizes are standard, JOOUST operates a pay-as-you-eat system, allowing students to buy food based on their individual needs. However, some food vendors develop close relationships with certain students, giving them extra food they may not finish, contributing to waste.

To address this, students suggested introducing measures to promote fairness in food service. Universities could establish clear guidelines for portion sizes within the pay-as-you-eat model to ensure all students receive equitable treatment. An easy and fair approach would involve weighing plates with food servings and charging a per-kilo price, allowing students to pay based on the exact quantity they choose. Additionally, training food vendors on impartiality

and accountability could help eliminate favoritism, while regular monitoring and anonymous feedback systems would encourage transparency. Specific support programs targeted at food insecure students could make the difference for low and middle-income students. By addressing these issues, universities can foster a more inclusive and sustainable dining environment.

4.4 Food Quality and Variety

Students at both UVic and JOOUST emphasized the need for better food quality and greater menu variety as key measures to reduce dissatisfaction and associated food waste. Poorly prepared meals, such as undercooked meat or bland dishes, and repetitive menu options often result in students discarding uneaten food. At JOOUST, the limited options in the campus mess—mostly rice, beans, cabbage, ugali (cooked maize meal) and occasionally chapati—leave students feeling bored and unsatisfied. Similarly, at UVic, repetitive vegan options and a lack of variety during dinner hours were frequently cited as issues.

To address these concerns, universities could invest in sourcing higher-quality ingredients and providing diverse menu options tailored to different dietary needs, such as vegan, gluten-free, and allergen-free meals. Rotating menus could reduce monotony, while partnerships with local suppliers would ensure fresh, high-quality produce to be used in meal preparation. Expanding dining hours, particularly for dinner, could also cater to students with busy schedules, offering them more opportunities to access fresh and appealing meals.

4.5 Food Labeling

Students at UVic identified the lack of clear food labeling as a significant factor contributing to food waste on campus. Without clear expiration dates or indicators of freshness, students often discard food prematurely, unsure whether it is still safe to consume. This is especially true for items like fresh produce such as dairy products, fruits and vegetables that might look fine but have already passed an undefined storage period.

To address this issue, universities could implement standardized and consistent labeling practices across all dining and retail outlets. Clear “use by” and “best before” dates would provide students with confidence in food quality. Adding freshness indicators, such as color-changing labels or packaging seals, could further help students assess the usability of food items. However, the effectiveness of these labels depends on students understanding their meaning. Educational initiatives could accompany these labeling efforts, clarifying the differences between “use by” (indicating a health safety threshold) and “best before” (relating to quality). Additionally, universities could provide general guidelines, such as how many days certain foods remain safe to consume past their labeled date, to help students make informed decisions. Pairing this with signage in dining halls could reduce confusion and empower students to better manage their food, minimizing unnecessary waste.

Collaboration with suppliers is equally important to ensure food items arriving on campus are properly labeled

and meet these standards. Such measures would not only reduce waste caused by uncertainty but also foster a culture of sustainability, empowering students to take more responsibility for their food choices while minimizing unnecessary waste.

4.6 Feedback Mechanism

Students at both UVic and JOOUST expressed concerns about the effectiveness of feedback mechanisms in addressing issues with campus food services. At UVic, the smiley-face system, where students select their level of satisfaction from options ranging from happy to angry, was criticized for being too simplistic to capture detailed concerns. While a dedicated email channel (eat@uvic.ca) exists for submitting more specific feedback, many students were unaware of its availability, highlighting the need for better communication about existing tools.

At JOOUST, feedback mechanisms, while present in the form of quarterly questionnaires, are not frequent or comprehensive enough to consistently capture student input on food quality, service, or variety. This limited approach can hinder efforts to address recurring concerns effectively or identify broader trends in student satisfaction.

To address these challenges, both universities could improve their feedback systems by making them more accessible and user-friendly. At UVic, increasing awareness of existing options, such as the feedback email channel, would encourage greater participation and ensure more students to feel heard. At JOOUST, complementing the quarterly questionnaires with additional tools like suggestion boxes or digital platforms could provide students with more frequent and flexible opportunities to share their input. Strengthening these systems would enable both universities to gather more detailed and actionable feedback, fostering a responsive and inclusive approach to improving dining services.

5. FINAL CONSIDERATIONS AND NEXT STEPS

The strategies and suggestions presented in this study illustrate the vital role students play in addressing food waste on university campuses. Their grassroots innovations and practical solutions not only tackle immediate challenges but also highlight opportunities for institutions to adopt more sustainable and equitable food systems. By integrating student-led efforts with institutional support, universities can make meaningful progress toward reducing food waste while fostering a culture of sustainability and collective responsibility. To build on these findings, the following steps could be taken:

- *Enhance Institutional Commitment*
Universities should formalize their commitment to sustainability by implementing the student-suggested measures, such as introducing communal fridges, improving food quality and variety, charging for food by weight, and adopting clear food labeling practices. These steps can reduce waste while improving the overall campus dining experience.

- *Strengthen Collaboration*
Partnerships between students, faculty, food service providers, and local suppliers are essential for the success of these initiatives. Universities could create advisory committees or task forces that include student representatives to ensure their voices remain central in shaping policies that reduce food waste.
- *Expand Awareness Campaigns*
Universities should invest in educational initiatives to promote mindful consumption and sustainability. Workshops, campaigns, and informational sessions can help students better understand food labeling, portion control, and storage practices, empowering them to make more informed decisions and be informed to avoid food waste.
- *Monitor and Evaluate Progress*
Regular assessments of implemented changes, supported by enhanced feedback mechanisms, will help universities measure the effectiveness of these efforts. Transparent reporting of outcomes can build trust and encourage further participation from the campus community.
- *Expand the Research*
Future research could investigate the long-term impacts of these student-led and institutional initiatives, assessing how they influence food waste reduction and overall campus sustainability. Comparative studies across other institutions could provide broader insights into best practices and scalability.

By prioritizing these next steps, universities can align themselves with global sustainability goals while directly addressing the concerns and contributions of their students. These efforts not only enhance the campus experience but also position universities as leaders in combating food waste at both local and global levels.

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DETRITUS & ART / A personal point of view on Environment and Art

by Rainer Stegmann

In this version of the Detritus & Art page, I ask our readers how far the high consumption of resources is acceptable when it comes to creating art. To say it in a simplified way, does art justify contributing to a high waste and energy production? I share some of my thoughts with you and would appreciate your feedback (stegmann@tuhh.de).

I will explain what I mean by using two examples: We all know about the wrapping of buildings from the artist couple Christo and Jeanne-Claude (<https://christojeanne-claude.net>). In the case of wrapping the German Reichstag 109 400 m² (1 094 ha) thick aluminium-coated Polypropylene (PP) textile was used. For fastening the plastic material, 15.6 km of blue 3.2 cm wide polypropylene-rope (550 m²) was produced.

People could visit the wrapped Reichstag from 24.6 to 7.7. 1995, i.e., for about two weeks. The used material is certified using the Oeko-tex® test procedure for textiles to make sure that the material is acceptable for human use (Wikipedia).

For the wrapping of the Arc de Triomphe in Paris, about 4 ha of aluminium-coated Polypropylene (PP) tissue was produced, from which 2.5 ha material was used. As mentioned by the organizer, Polypropylene, after use, can be recycled by melting and pelleting; the pellets can be used to make new PP. Unfortunately, I could not determine how the aluminium coating was removed from the PP surface and recycled. The plastic material was recycled by the German company Luftwerker in Lübeck. Also, due to the aluminium coating, recycling of the wrapping material must have been demanding. Except for some environmental groups, there was no significant discussion in the public about the potential waste of resources because- and that is often the argument that everything is fine- it is recycled. However, looking at more details, we learn that PP recycling is downcycling. Due to UNPD 2021, PP can only be recycled one to two times. That means about 30 to 50% of PP waste is produced. Recycling uses, in addition to its production, about 2/3 of the energy has been used for the new production of the PP (US EPA, <https://archive.epa.gov/factoid>). When the Al is separated from the Polypropylene, aluminium recycling works well, combined with a significant energy saving of about 95%. On the other hand, energy for transport, installation, de-installation, and more has to be considered.

The Italian-born Rudolf Stingel is another example of an artist using - or shall I say wasting? - enormous amounts of material. I saw his exhibition in 2013 in the Palazzo Grassi in Venice, Italy (<https://www.pinaultcollection.com/palazzo-grassi/en/rudolf-stingel>).



Christo and Jeanne-Claude: The wrapped German Reichstag, Berlin, 1995 (Qwesy/Wikimedia Commons).

He covered floors, ceilings and walls on the ground floor and the two upper floors of the Palazzo with oriental carpets (not hand-made). In total, around 5000 m² of carpets were used. I asked the museum guard - he may not have all the information - where the carpets would end up after the exhibition; he said they all would become waste. Even if the organizers once intended to recycle the carpets, receiving useable recycling rates, if any, will be difficult or practically impossible. Machine-produced carpets are primarily made of different kinds of plastics and are generally disposed of in landfills, or they are - as in Germany - almost entirely incinerated.

I cannot make material and energy balances because of lacking data but from the here presented information



Rudolf Stingel, Exhibition in the Palazzo Grassi, Venice, 2013.



the reader gets an impression of the dimension of the produced waste and used energy. Coming back to my initial question, can such an effort in material and energy consumption be justified for a temporary art exhibition? Of course, we have to protect the freedom of art, but are there no limits....? Please give me your thoughts; I will then also present mine.



In the next edition, I will report on my summer weekend in Varel, Germany, where four artists created an outdoor exhibition titled "The Guards of Sustainability".

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