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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

WASTE MANAGEMENT WITHIN EUROPEAN UNION: MUST THE DISCOURSE CHANGE?

From a legal perspective, waste management is about dealing with the potential risks that waste poses for human health and the environment. Within the European Union (EU), waste is primarily governed by the Waste Framework Directive (2008/98/EC) (WFD) and associated legislation. In the advent of the circular economy, an additional purpose has come to the center of attention, namely that of utilizing waste as a valuable resource. This purpose is explicitly mentioned in the recital of the 2018 revision to the WFD where it is stated that "[i]mproving the efficiency of resource use and ensuring that waste is valued as a resource can contribute to reducing the Union's dependence on the import of raw materials and facilitate the transition to more sustainable material management and to a circular economy model." (Directive 2018/851, recital 2). While similar ideas were present in the 2008 WFD (2008/98/EC), the scope was limited to strengthening the economic value of waste (Directive 2008/98/EC preamble 8). It is thus clear that the policymakers of the European Union have incorporated the notion of a circular economy in their law-making process. However, the dichotomy of environmental protection and resource utilization is challenging to realize in practice.

The legal demarcation between waste and resource lies within the interpretation of the concept of waste. While the interpretation of what waste is within the EU is neither in its infancy nor definitive, a recent verdict from the Court of Justice of the European Union (CJEU), Case C-239/21 Porr Bau, showcases how a potential waste may, in an appropriate manner, substitute primary resources. The background for the dispute is an inquiry from a group of farmers to obtain excavated materials for soil adaption purposes from the construction company Porr Bau. At the time of the farmers inquiry the company was not in possession of such materials, which led Porr Bau to select an appropriate on-going construction project and extract the wanted materials. In short, this implies that materials which otherwise might not have been put to use is utilized for a beneficial purpose. Following this, Porr Bau requested primarily that the local authorities declare that the excavated material did not constitute waste, and secondly that it had ceased to be waste. The authorities, however, found that the excavated materials did constitute a waste, and also that it did not meet the requirements for end of waste, whereupon Porr Bau appealed the decision. The question is then how the authorities could reach the conclusion that the materials constituted a waste when Porr Bau could clearly show that the materials were of the highest grade, and commercially retailed to the farmers for a beneficial purpose.

Waste is, according to EU law, something that its holder discards, intends to discard or must discard (Article 3(1) WFD). Based on the letter of the law, it should thus be fairly simple to determine whether an object constitutes a waste: has it been, or will it be, discarded? Or, as in this case, is Porr Bau discarding the excavated materials by selling it to the farmers? My first reaction was: surely not. But the interpretation of waste has generated a complex body of law, according to which, to this day, waste is not clearly defined. While the rulings of the Court partially and incrementally have clarified the meaning of waste, the idea of the circular economy implies new challenges. In Porr Bau, the Court clearly expressed that that re-use of excavated materials must not be hindered by formal criteria which have no bearing on environmental protection, as this would counteract the effectiveness of the WFD, and in turn breach EU law. And yet it was not due to formal criteria that the materials was seen as a waste, but the fact that it was considered discarded. No consideration was given to the quality of the material and that the use was not assumed to entail any adverse environmental impacts.

Thus, while the court clearly takes a stance in favor of both environmental protection and the circular economy as the classification of waste would imply that the excavated materials could not be used for a beneficial purpose the outcome of this case also highlights two problems:

First, the wide, and at the same time, ambiguous definition of waste results in considerable uncertainty because a correct assessment requires a combination of both objective and subjective criteria. What this in practices implies is that government authorities often adopt a better safe than sorry approach. While this approach is in line with the precautionary principle (i.e., in dubio pro natura), the flipside is that such a cautionary stance may in many cases constitute a detriment to the realization of waste as a secondary resource. In our case, for instance, Porr Bau were contacted by the farmers in early 2020 and the dispute was solved in late 2022. With such protracted processes, customers (in this case the farmers) are likely to obtain the desired materials elsewhere.

Second, while the Court directly refers to undermining circular economy as undermining the WFD, the case does not shed much light upon the dichotomy of environmen-

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tal protection and resource use as the court only states that formal criteria, which are irrelevant for environmental protection, would undermine the circular economy. This of course begs the question of which parts of the waste legislation that could undermine the circular economy with the same argument, that is, where there are legal requirements that has no bearing on environmental protection, for example that a "market" is required for end of waste. Furthermore, it begs the question of if it is at all possible to make an assessment that favors resource utilization, or if waste as a resource will never outweigh environmental protection?

Environmental law, and by extension waste law, will always entail tradeoffs, and both the legislator and practitioners must find a balance between environmental protection and other interests. These tradeoffs have been illustrated in various ways in the last ten years, where the CJEU has produced rulings in waste law, covering the concept of waste, shipment of waste, final storage of waste through landfilling, and end of waste. Throughout the Court's history, a common denominator of the rulings have been the maintenance of a high level of environmental protection. While this is both understandable and rational – as EU waste legislation is based upon article 192 TFEU – it has resulted in an extensive interpretation of the concept of waste through a precautionary approach (Van Calster, 2015). The necessity of a wide interpretation of waste in order to uphold the environmental protection purpose of the waste regime is also repeatedly referred to by the Court.

To conclude, while the core of the concept of waste (discard) has remained unchanged for over thirty years, in light of the idea of the circular economy, it can be argued that the waste discourse has changed, or needs to change, in order for the new goals to be reached. It is time to reflect upon the consequences of the current regime. If waste, as the legislator points out, should be valued as a resource, it may be high time to narrow the definition of waste.

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FOOD WASTE COLLECTION VERSUS FOOD WASTE DISPOSERS: A CASE STUDY OF A PORTUGUESE CITY

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ABSTRACT

The European Union (EU) prioritises sustainable development and aims to achieve climate neutrality by 2050 through investments in green technologies and a legislative climate framework. Waste collection and treatment systems, including biowaste like food waste, protect natural capital and citizens' well-being. Directive 2008/98/EC defines biowaste and sets the stage for exploring alternative solutions. This study focuses on the city of Guarda (Portugal) and analyses the use of disposers for food waste treatment. By comparing storage/collection systems with disposers regarding environmental and economic costs and benefits, the study aims to determine the most effective solution for collecting food waste. The analysis utilises a simulator provided by the Portuguese Fundo Ambiental®. The results show that while operating costs are higher for food waste collection systems, disposers require a more significant initial investment. Nevertheless, the cost-benefit ratio favours disposers, while the net present value analysis suggests that the food waste collection system is more favourable. Additionally, food waste collection systems contribute to higher greenhouse gas (GHG) emissions per tonne of food waste compared to using disposers.

1. INTRODUCTION

Population growth and the development of society, coupled with rapid urbanisation, are projected to result in 68% of the world's population living in urban areas by 2050 (Ritchie & Roser, 2018). As a result, will be a significant escalation in the production of municipal solid waste (MSW). In many cases, the amount of solid waste generated should exceed the management capacity of municipal authorities, potentially causing adverse impacts on the environment, human health, and quality of life. Since land resources around urban centres are limited, finding environmentally safe and socially and politically acceptable sites for landfill construction has become a problem for technical and political decision-makers.

The European Union's (EU) growth strategy advocates investments in green technologies and presents a legislative climate framework that allows society to achieve climate neutrality by 2050. The protection, conservation, and enhancement of the natural capital of the EU and the defence of the health and well-being of the citizens against risks and impacts related to the environment are one of the EU's priorities. The existence of waste collection and treatment systems, including biowaste, particularly food waste, is another example of the search for sustainable development to improve living conditions and preserve the environment (Mehta et al., 2018; Laso et al., 2019). Also, Directive 2008/98/EC of the European Parliament and the Council of November 19th, 2008, defines biowaste as: "biodegradable garden waste, food and kitchen waste from homes, restaurants, catering and retail units and similar waste from food processing plants". Unlike Europe, Central Asia, and North America, which tend to generate more significant amounts of dry waste, most regions produce around 50% or more organic waste on average (Kaza et al., 2018).

Until now, in Portugal, the organic fraction has been collected in an undifferentiated model. However, starting January 1st, 2024, the organic fraction will be collected separately through a door-to-door system, public road collection, or treatment and recovery at the source, such as home or community composting, or a combination of these solutions. In the door-to-door collection system, households are encouraged to separate their organic waste from other types of waste, such as recyclables or non-biodegradable items. Residents use specific bins (typically of brown colour). These receptacles are designed to prevent odours and facilitate easy handling. Waste collection personnel visit residential areas on a scheduled basis and collect organic waste directly from each household. After the collection procedure, the organic waste is processed through composting or anaerobic digestion to ultimately obtain a

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duly stabilised fertiliser that can be incorporated into the soil. When collecting on the public roads system, the objective is the same. However, instead of each house having a container, one or more containers with greater capacity are distributed on public roads and serve a wide range of homes. The concept of home or community composting is a decentralised approach to composting organic waste that takes place in the yards, balconies, or designated spaces. The basic idea is to convert kitchen scraps and garden waste into nutrient-rich compost, which can then be used to enrich soil and promote plant growth. Home or community composting is a sustainable waste management practice that empowers individuals and communities to actively reduce organic waste, promote environmentally friendly habits, and avoid waste collection vehicles. The alternative to minimising waste at the source is using food waste disposers. These devices allow the separation of a significant fraction of food waste from the entire MSW stream by grinding the waste with mechanical means and adding tap water, allowing mixing in the drainage. As a result, recycling at the source can be diverted from the waste collection system by introducing a food waste disposer that directs the flow of food waste to the wastewater collection and treatment system (Iacovidou et al., 2012; Zan et al., 2022).

In this scope, based on the case study of the Portuguese city of Guarda, this paper analyses the possibility of using disposers send solid food waste to wastewater treatment plants (WWTP) but not about the subsequent treatment there. What municipalities do with food waste frequently harms the environment and public health. For example, food waste is sometimes stored inside buildings, piled in bags on sidewalks, collected in trucks and shipped to distant landfills, generating leachate and greenhouse gases (GHG) emissions. This process is not cheap, hygienic, environmentally friendly, or sustainable. Contrariwise, the food waste disposers reduce the incidence of vector attraction by decreasing the amount of MSW garbage collection, promoting a healthier municipal solid waste environment, and reducing truck collection costs and greenhouse emissions. Thus, the comparative study between food solid waste storage/collection and food waste disposers, in a context of environmental and economic costs-benefits, applied to the Portuguese city of Guarda, aims to analyse whether food waste disposers are the best environmental and economic solution to implement, ensuring that the food waste is sent adequately for treatment and recovery for this type of waste. Using the simulator provided by the Fundo Ambiental (Department of the Portuguese Ministry of Environment and Climate Action), two scenarios were conducted to evaluate the most appropriate solution for the city of Guarda (Portugal), considering the analysis period from 2023 to 2030. One scenario considers food waste collection on public roads, door-to-door and recycling at source and another contemplates using food waste disposers.

Therefore, the structure of the research is organised as follows. Section 2 focuses on the food waste disposers' history and features. Section 3 presents the solution for food waste, presenting the technical parameters and the simulation developed. Section 4, centred on results and discussion, sets out the capture rate of the food waste, collected and disposers, and their future potential. Section 5 summarises the main findings and presents the conclusions.

2. FOOD WASTE DISPOSERS

2.1 History

Food waste disposers were first introduced in the United States of America (USA) in the 1930s and have since become prevalent, with over 94% of all cities and nearly half of all American households using them as standard items (Kelemen & Furlong, 2020). After the 2nd World War, with the real estate boom, the effectiveness of disposers was evaluated by municipalities as an alternative to solid waste collection and treatment systems. They conducted thorough evaluations to determine their effectiveness. However, their use faced scepticism in some large cities, such as New York, which banned their use due to concerns about the city's old sewer infrastructure's ability to handle the additional load. It was not until 1995 when the city was forced to address waste disposal concerns after the closure of its main landfill, that it commissioned an extensive pilot project to study the impact of food disposers on sewage. The study yielded positive results, and the city lifted the ban and legalised installing food waste disposers in residential buildings in 1997 (Shure, 2021). Towards the end of the 20th century, disposers became ubiquitous in American kitchens. Today, approximately 50 countries, including England, Ireland, Italy, Spain, Japan, Canada, Mexico, and Australia, sell food waste disposers to homes with limited or no restrictions. France also banned food waste disposers, but it was lifted in 1986 following an in-depth investigation by French authorities (Nilsson et al., 1990). With growing concerns about the environmental impact of depositing food waste in landfills and increasing interest in finding sustainable alternatives, food waste disposers are gaining global acceptance.

Although food waste disposers help divert organic waste from the solid waste stream and reduce associated management costs, their use raises questions about the additional energy required to operate the units, the amount of additional tap water needed for the transfer of particulates in the sewer, the alteration of sewer quality due to the addition of suspended solids (SS) and organic substances, and the additional loads in the sewer network and sewage treatment plants. Several studies have been conducted to investigate these issues. The increasing international acceptance of food waste disposers is a response to the significant concern of diverting organic food waste from landfills and finding beneficial uses, such as land application. Traditional municipal methods of managing food waste, such as storing it inside buildings (even refrigerated), piling it up in bags on sidewalks, collecting it in trucks, and sending it to distant landfills, be harmful to the environment. This process is not cost-effective, hygienic, ecological, or sustainable.

2.2 Features of food waste disposers

Food waste disposers, also known as garbage disposals or waste disposers, serve as kitchen appliances designed to efficiently grind food scraps and other kitchen waste for easy disposal with water. Typically installed beneath the kitchen sink, these devices connect seamlessly to the plumbing system. At the core of the food waste disposer lies an electric motor, carefully sealed to prevent water damage. The grinding chamber constitutes the primary space where food waste accumulates and undergoes the grinding process. This chamber is constructed from robust materials such as stainless steel and ensures durability throughout the grinding operation. The grinding mechanism employs a shredder ring or grinding wheel featuring sharp blades that rotate at high speeds, effectively transforming food waste into fine particles. It is crucial to emphasize that not all types of waste are suitable for disposal in the unit. Items like bones, fibrous vegetables, and non-food materials have the potential to cause damage and should be avoided.

Numerous studies have examined the operational, economic, and environmental aspects of using disposers to treat the organic fraction of domestic MSW by diverting food waste to wastewater treatment systems and sludge. Based on the available literature, the benefits of implementing this approach outweigh the drawbacks. The literature highlights several advantages of food waste disposers, including reduced noise, improved traffic flow due to the absence of collection vehicles, lower GHG emissions, and transport costs. Additionally, waste disposers are a less complicated, more hygienic, and sustainable system compared to other waste-collected disposal methods (Saraiva et al., 2016). It also reduces the frequency and amount of MSW collection (Sahu, 2020) and eliminates the need for storing food waste at home or on public roads, which helps reduce disease vectors, contributing to public health.

Compared to other waste disposal methods, disposers and sending food waste through the drainage system is a natural way of disposing of organic waste. Since food waste is mostly made up of water, the WWTP is the most natural system for processing food waste (Battistoni et al., 2007; Trabold & Nair, 2018). Additionally, the high carbon content of food waste improves the nitrogen and phosphorus nutrient removal process in WWTP (Tang et al., 2019; Feng et al., 2021). Grinding food waste also enables energy recovery using biogas resulting from anaerobic digestion (Caffaz et al., 2008; Morelli et al., 2020). The use of food waste disposers (FWD) can significantly reduce the potential for uncontrolled biochemical processes in landfills, lower the generation of leachates, and decrease the contamination of groundwater (Battistoni et al., 2007; Iacovidou et al., 2012). This is particularly beneficial in small towns and decentralized areas, where FWD can be integrated with the management of municipal wastewater and household organic waste (Battistoni et al., 2007). However, the impact of FWD on the wastewater system, particularly in terms of chemical oxygen demand (COD), biological oxygen demand (BOD), and suspended solids, needs further investigation (Thomas, 2011). Additionally, the recycling of nutrients from organic residues, such as the sludge resulting from treatment at WWTP, through composting and application to the soil, can further enhance the environmental benefits of FWD (Nowak, 2006).

Although food waste disposers can divert organic waste from the solid waste stream and reduce associated management costs, their usage raises several concerns. These include the additional energy required to operate the units, the amount of extra tap water needed to transfer particles in the sewer, changes to sewer quality due to the introduction of suspended solids and organic substances, and the added strain on sewer networks and treatment plants. Extra water use due to disposers is negligible, averaging 4.3 L/person.day and representing only 2.2% of total domestic water use. Additionally, the cost of electricity to run food disposers is relatively insignificant, with estimated consumption lower than that of a 75 W light bulb used for 10 minutes (Marashlian & El-Fadel, 2005; Iacovidou et al., 2012).

Disposers significantly change the amount of organic matter and suspended solids at the WWTP. In a study, Thomas (2011) reported that the largest impacts will be on COD, BOD and solids, with food waste adding an additional of 24%, 28% and 18%, respectively, compared with the design standard when a figure of 142g/person/day food waste is applied, or 48%, 59% and 39%, respectively, when a figure of 303g/person/day food waste is applied. Another study in Shurammar (Sweden) did not show increased BOD, nitrogen, or phosphorus (Evans et al., 2010).

In long-term tests assessing obstructions, Evans et al. (2010) demonstrated that the optimised use of disposers over 15 years did not present any operational problems within the drainage system. Regular inspections and annual network filming showed that outside sewer pipes worked well when using food disposers. In the WWTP, theoretical calculations have shown that using food disposers increases the amount of sludge by 57%, mainly affecting the biological stage and treatment of sludge (Thomas, 2011). Similar theoretical estimates by Galil and Yaacov (2001) reported that the contribution of food waste disposers increases the weight of raw sludge by about 60% in the case of biological treatment and 62% in the case of primary sedimentation followed by biological treatment. However, another study indicated an increase of only 18.1% in sludge production due to the introduction of food disposers (Marashlian & El-Fadel, 2005). Therefore, it is crucial to consider the specific characteristics of the region and its food waste generated, as there may be differences in the generation of sludge with possible environmental and economic implications depending on the location.

3. SOLUTION FOR THE FOOD WASTE

3.1 Technical parameters

The Portuguese Biowaste Collection Systems Simulator - Version 1.3, licenced by the Portuguese Fundo Ambiental®, allows the determination of the investment flows (tangible and intangible) and exploration flows (tariff income, avoidable costs, and other types of expenses) so that, in the end, it is possible to determine the economic-financial sustainability and environmental viability of the proposed solutions. In addition to the economic-financial and technical inputs, other global characterization data were necessary and obtained from various sources, such as AMCB (2021), APA (2020), EGF (2020), ERSAR (2020), CMG (2021), INE (2020), the following values were gathered for the year 2019 to construct scenarios for the collection of food waste on public road, door-to-door and recycling at the source, along with simulations involving the use of disposers in kitchens:

- The resident population in the city of Guarda is 26,446 inhabitants;
- The rate of change in the resident population according to projections for the central region of Portugal exhibits negative variations ranging from 0.077% to 0.504% until 2030;
- There are 14,619 accommodations, with an average of 1.8 inhabitants per accommodation;
- Non-domestic producers total 139, including 128 from the hotels and catering industry and 11 from other sectors;
- Urban waste collected varies between 15,908 tons and 15,312 tons in the period under review; undifferentiated waste varies between 13,818 tons and 8,309 tons; food waste varies between 4,249 tons and 464 tons; and green waste varies between 1,695 tons and 490 tons;
- The potential of biowaste ranges between 5,945 tons and 6,162 tons;
- Service reach 100% of the population and accommodation;
- Non-domestic producers are served at a 100%;
- The tariff charged to the Municipality for undifferentiated waste is €42.84 per ton;
- The Waste Management Fee related to the deposition of undifferentiated waste in landfills, is avoided, with an average amount of €32.2 charged to the Municipality;
- The percentage of undifferentiated waste sent to landfills is 87%;
- Operating income results from the application of fixed tariffs (€55.84), variable tariffs and auxiliary services (€4.73), for the provision of the selective biowaste management of service, by producer (domestic and non-domestic);
- The tariff charged to the Municipality by the high system related to the forwarding of biowaste, set at €0.00 per ton, has not yet been defined.

For the collection of food waste on public roads, door-to-door and recycling at source, the following parameters were considered:

- Annual value for awareness campaigns, proposed by the Simulator, amounts to €20,328.03 per year, from 2022 to 2030. An except is made for the year 2021, with a foreseen value of €135,520.22;
- 254 Containers of 770 litres each, with a unit price of €261.25;
- 2,961 Containers of 340 litres each, with a unit price of €88.28;
- 5,157 buckets of 7 litres, with a unit price of €3.70;
- 176 containers of 110 litres each, with a unit price of €56.09;
- A biocomposter (volume of 333 litre) with a unit price of €60,000.

Food waste will be collected three times a week, with an estimated annual diesel consumption of 8,300 litres. The estimated outsourcing value is €79,658. The planned acquisition of a container washing vehicle with a total value of €262,728, includes a frequency of 6 washings per year. The annual global cost of human resources is estimated at €40,824. Furthermore, for the simulation involving the use of a food waste disposer, the following base parameters were considered:

- The capture rate of household food waste is 90%;
- The capture rate of non-domestic food waste is 80%;
- The capture rate of domestic green waste is 70%;
- For non-domestic producers,139 food waste disposers were considered, with a unit price of €1,140.21;
- For domestic producers, 14,619 food waste disposers were considered, with a unit price of €320.

3.2 Simulations

Table 1 presents the food waste capture rate, categorised by producer type (domestic and non-domestic) and collection system (public road, door-to-door, and recycling at the source), as well as considering an alternative system of capturing food waste through disposers (considered equivalent to recycling at the source for model convenience). In an optimistic scenario representing selective collection or recycling at the source from 2021 to 2030, collection rates range between 38% and 57% for collection on public roads, 56% and 75% for door-to-door collection, and 70% for recycling at the source. An estimated increase in the global capture rate of biowaste from 2021 to 2030 is projected to reach 65.1%, with a domestic food waste at 65.2% and a non-domestic food waste at 63.4% by 2030.

Table 1 also presents results considering an alternative system of capturing food waste through disposers (considered equivalent to recycling at the source for model convenience). In this scenario, capture rates for recycling at the source are 90% for domestic food waste and 80% for non-domestic food waste. The higher rates in this context reflect the assumption that individuals are more likely to participate in a system where they do not need to separate and place waste in containers, whether on the public road, through door-to-door collection, or by using domestic or community composters. It should be noted that capture rates for public road collection and door-to-door collection are 0% in this alternative system. Globally, the estimated capture rate for food waste from 2021 to 2030 is approximately 89%.

Table 2 exhibits the potential for food waste collection from 2023 to 2030, considering two different systems: food waste collection and food waste disposers. In both scenarios, the potential amount of food waste decreases over the analysed period, from 4,468 tons in 2023 to 4,347 tons in 2030 (-2.7%). This decrease is attributed to the expected population decline until 2030 in the city of Guarda, following the trend observed in the central region of Portugal over the past decade.

Regarding the projection of the amount of food waste collected, an inverse trend is observed in both systems. In the food waste collection system, the amount collected

TABLE 1: Food waste capture rate (2021-2030).

		2021	2022	2023	2024	2025	2026	2027	2028	2029	2030
Waste collection	Food waste (%)	50.2	51.8	53.5	55.2	56.8	58.5	60.1	61.8	63.5	65.1
	Domestic food waste (%)	50.0	51.7	53.4	55.1	56.8	58.5	60.2	61.9	63.6	65.2
	Public roads (%)	38.0	40.1	42.2	44.3	46.4	48.6	50.7	52.8	54.9	57.0
	Door-to-door (%)	56.0	58.1	60.2	62.3	64.4	66.6	68.7	70.8	72.9	75.0
	Recycling at source (%)	70.0	70.0	70.0	70.0	70.0	70.0	70.0	70.0	70.0	70.0
	Non-domestic food waste (%)	52.5	53.7	55.0	56.2	57.4	58.6	59.8	61.0	62.2	63.4
	Public roads (%)	38.0	40.1	42.2	44.3	46.4	48.6	50.7	52.8	54.9	57.0
	Door-to-door (%)	56.0	58.1	60.2	62.3	64.4	66.6	68.7	70.8	72.9	75.0
	Recycling at source (%)	70.0	70.0	70.0	70.0	70.0	70.0	70.0	70.0	70.0	70.0
Waste disposers	Food waste (%)	89.4	89.4	89.4	89.4	89.3	89.3	89.3	89.3	89.3	89.3
	Domestic food waste (%)	90.0	90.0	90.0	90.0	90.0	90.0	90.0	90.0	90.0	90.0
	Public roads (%)	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Door-to-door (%)	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Recycling at source (%)	90.0	90.0	90.0	90.0	90.0	90.0	90.0	90.0	90.0	90.0
	Non-domestic food waste (%)	80.0	80.0	80.0	80.0	80.0	80.0	80.0	80.0	80.0	80.0
	Public roads (%)	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Door-to-door (%)	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Recycling at source (%)	80.0	80.0	80.0	80.0	80.0	80.0	80.0	80.0	80.0	80.0

increased from 2,391 tons in 2023 to 2,831 tons in 2030 (+18.4%). Furthermore, the amount collected for the food waste disposer system varies from 3,992 tons in 2023 to 3,884 tons in 2030 (-2.7%).

4. RESULTS AND DISCUSSION

About the distribution of households, hotels and catering industry, and other entities served by two food waste management systems, i.e., the food waste collection system and the kitchen disposer system (here equivalent to recycling systems at the source, for simulation purposes only), both systems were assumed to have 100% coverage from the outset for the simulations conducted. In the food waste collection system, the distribution corresponds to 49% through the collection on public roads, 31% through door-to-door collection, and the remaining 20% through recycling at the source. Additionally, for the kitchen disposer system, the simulation assumes that all households in the city, the hotels and catering industry, and other entities are equipped with kitchen disposers to ensure complete population coverage.

Comparing the economic and financial sustainability between the collection of food waste and the use of disposers (Table 3) allows us to conclude that operating costs are substantially higher for collection systems (public road, door-to-door and recycling at source), with values in 2023 of €250,074 and €191,774 in 2030, while in the implementation of domestic disposers, the expected value for 2023 is €162,061 while for 2030 the value is reduced to €96,731. It is also possible to observe that the investment in implementing food waste disposers throughout the city is much higher. In 2023 the estimated value is €2,430,917 and €4,836,569 in 2030, while for the food waste collection system, the figures point to just €679,435 in 2023 and €679,435 in 2030. The food waste disposer system requires a much higher investment than implementing a collection system. However, the cost-benefit ratio is much more favourable for food waste disposal systems. In other words, the cost-benefit ratio for implementing a food waste disposal system is 372% in 2023 and 636% in 2030, while for the food waste collection system, it is 133% and 204%, respectively.

Additionally, the feasibility analysis of the two studied alternatives presented in Table 3 shows that the food waste collection system is more favourable by comparing the net present value (NPV), which considers the time value of money to assess investment viability. In 2030, it will reach a value of €1,456,277 and recover the initial capital investment in 5 years. Moreover, within the same time frame, the domestic waste crusher system amounts to €1,312,153 in 2030 with a capital recovery period of 8 years. Considering each pro-

TABLE 2: Potential for food waste collection and food waste disposers (2023-2030).

	Food waste collection			Food waste disposers		
	2023	2027	2030	2023	2027	2030
The potential amount of food waste (ton)	4,468	4,409	4,347	4,468	4,409	4,347
Amount of food waste selectively collected (ton)	2,391	2,652	2,831	3,992	3,939	3,884

TABLE 3: Decision support for food waste collection and disposers (2023-2030).

		Food waste collection			Food waste disposers		
		2023	2027	2030	2023	2027	2030
Economic and financial sustainability	Operating expenses (€)	250,074	202,482	191,774	162,061	138,188	96,731
	Benefit/cost (%)	133	182	204	372	444	636
	Investment (€)	679,435	679,435	679,435	2,430,917	4,836,569	4,836,569
Economic and financial sustainability	Net present value (€)	-281,517	625,804	1,456,277	-620,157	-543,680	1,312,153
	Payback invested capital (years)	I.N.C.	5	5	I.N.C.	I.N.C.	8
	Profitability Index (%)	-41	92	214	-26	-11	27
Economic and financial sustainability	Emission of greenhouse gases (kgCO_2/ton)	7.4	6,4	5,9	2.9	2.9	2.9

I.N.C. - Investment Not Covered

posed system's net present value and investment requirements, the profitability index favours the food waste collection system. In 2030, its profitability index stands at 214%, while the food waste shredding system only reaches 27%.

Table 3 also presents the forecasted GHG emissions per tonne of food waste for the two waste management systems: waste collection (including public road collection, door-to-door collection, and recycling at the source); and the shipping system through the basic sanitation system, using food waste disposers. In the latter case, the calculations are based on an average daily operation time of 10 minutes, with a power consumption of 75W, and considering the current energy mix in Portugal (APA, 2022). For the household waste collection system, the projected emissions in 2023 are expected to be 7.4 kg/tonne, decreasing to 5.9 kg/tonne by 2030. On the other hand, using food waste disposers is estimated to result in 2.9 kg/tonne emissions in 2023 and subsequent years. This value will likely decrease further due to the anticipated increase in renewable energy sources in the Portuguese energy mix in the coming years.

The results support the conclusion that collecting food waste through food waste disposers and subsequent treatment at WWTP through the sewage system is the most environmentally favourable solution. It should also be noted that considering the average increase in water consumption of 4.3 litres per person per day resulting from using food waste disposers and considering the population covered in the conducted simulation, the estimated daily increase in water consumption would be approximately 11.3 m^3.

5. CONCLUSIONS

Based on the provided results, a conclusion can be drawn regarding comparing food waste collection systems and using food waste disposers. The analysis, focused on economic and financial sustainability, feasibility, and GHG emissions, shows that the operating costs for food waste collection systems (public road, door-to-door, and recycling at source) are significantly higher compared to the use of disposers. Moreover, the investment required for implementing food waste disposers is considerably higher than for collection systems. However, the cost-benefit ratio favours the food waste disposal system, indicating higher economic efficiency.

The NPV analysis indicates that the food waste collection system is more favourable, reaching a higher value in 2030 and recovering the initial capital investment in 5 years. The food waste disposers have a lower NPV and a more extended capital recovery period, making them less financially feasible. Also, food waste collection systems result in higher GHG emissions per tonne of food waste than food waste disposers. The emissions associated with food waste disposers are lower and will potentially decrease further with the anticipated increase in renewable energy sources in the energy mix.

Based on the provided data, collecting food waste through food waste disposers and subsequent treatment at the WWTP via the sewage system offers the most environmentally favourable solution. While higher upfront costs are associated with implementing the disposer system, it demonstrates better cost-benefit ratios, lower operating costs, and reduced GHG emissions compared to food waste collection systems. Finally, to ensure the successful implementation of food waste collection through disposers, it is essential to approve new regulations in the licensing process for constructing and renovating buildings, which specifically mandate the installation of these devices to maximise their effectiveness, considering factors such as building size and type (residential, commercial, among others), contributing to sustainable waste management practices, and reducing the environmental impact of food waste.

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WASTE EGGSHELLS AS CATALYSTS: AN ENVIRONMENTALLY-FRIENDLY APPROACH

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ABSTRACT

The survival of mankind, the ecosystem and the development of society is inextricably linked to sustainability. The land has become heavily polluted following the uncontrolled dumping of huge quantities of waste. To address this issue, an environmentally-friendly protocol capable of yielding heterogeneous catalysts of immense value from waste eggshells whilst minimizing pollution has been designed. These synthesized heterogeneous catalysts developed from waste eggshells will replace the existing homogeneous process. This article highlights the key aspects of preparing heterogeneous catalysts from waste eggshells for use in the synthesis of CaO, to be applied in Biodiesel synthesis, environmental remediation, and organic transformation. It moreover addresses key issues relating to future recommendations for the development of more efficient CaO catalysts from waste eggshells. These CaO eggshell catalysts are characterized by X-ray diffraction (XRD), Fourier-transform infrared (FT-IR), Thermogravimetric-Differential Thermal Analyzer (TG-DTA), Scanning electron microscope (SEM), and Brunauer-Emmett-Teller (BET) surface area, showing the structure, morphology, porosity, thermal stability, reusability, of catalysts. Indeed, eco-friendly heterogeneous catalysts developed from eggshell wastes via biochemical methods represent a cleaner and reusable method and, over the next two decades, are destined to become increasingly popular and to promote the global economic growth of Namibia.

1. INTRODUCTION

The modern era has been facing severe challenges of climate change and energy depletion (Malins et al., 2014). Due to industrialization and rapid population growth, energy consumption has increased rapidly over the last two decades and will continue to grow at an alarming rate in the future (Uslu et al., 2008). Every year, millions of tons of potentially valuable eggshell waste is trapped in landfills. However, in the European Union, by the year 2030 an estimated 44 million tons of MSW will have been deposited, whilst oil reserves will be exhausted over the next 63 years (Tabatabaei et al., 2015). Although 8.7 million tons of energy have been recovered from household waste, the presence of incineration plants in the EU increased by a negligible 0.7% in 2013 (Tabatabaei et al., 2015). Jayathilakan et al. (2012) reported the higher viability of readily available animal bones and seafood wastes as feedstocks for the production of calcium-based catalysts. Global production of mollusks and meat is estimated to exceed 160 and 300 million tons per annum, respectively, with up to 28 wt% of the livestock ending up as non-edible solid waste, including bone. Waste generated from the shrimp, crab and lobster market currently averages between 6 and 8 million tons per annum. However, a recent increase in the cost of waste export and storage has been observed, leading to the notion whereby waste incineration plants might start to monopolize the waste management market. Regulations established for the energy sector advocate the use of more efficient sources of primary energy and a substantial reduction in the use of energy from alternative fuels (Jayathilakan et al., 2012). The storage of waste is strongly discouraged due to possible pollution of the air (greenhouse gas (GHG) emissions), soil, and vegetation, ultimately placing wildlife at harm as they consume waste or rotten food.

Furthermore, at times components from unprocessed waste are discharged into water reservoirs resulting in the pollution of water bodies and posing a significant threat to the aquatic environment. For example, garbage patches (clusters of garbage floating freely on the surface) that form in streams, ponds, lakes, rivers, seas, and oceans are one of the greatest threats to the aquatic environment (Fava et al., 2015). Data present in literature illustrate how a new area of catalysis is starting to attract more attention.

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The use of homogeneous catalysts in the field of biodiesel production is well known, although difficulties are encountered in separating these catalysts; this type of problem however may be mitigated by replacing homogeneous catalysis with solid heterogeneous catalysts.

However, in transesterification reactions, calcium compounds, solid base catalysts, have been exploited in diverse forms (oxide, hydroxide, carbonate, and diglyceroxides). CaO is an environmentally friendly material classified as a heterogeneous base catalyst generally derived from $CaCO_3$, $Ca(NO_3)_2$ or $Ca(OH)_2$ as raw materials. Limestone is frequently used to obtain CaO, although the less expensive but lengthy process required for the synthesis route is not easy to achieve. Accordingly, this option is not a preferred option due in large part to its non-renewable nature (Correia et al., 2014). Based on this limitation, eggshell waste catalysts have been reported to be excellent heterogeneous catalysts for use in the transformation of a series of organic compounds, including the production of biodiesel from waste cooking oil, the synthesis of bioactive compounds and environmental applications (Fava et al., 2015).

2. ECONOMIC FACTORS

In 2023 Lanzon et al. reported a continued growth in the production of hen eggs in recent decades, reaching a total of 82.4 million tons in 2019, i.e. approximately 1480 billion eggs per year (Lanzon et al., 2023). The European Commission predicts a projected increase in the per capita consumption of eggs to 15.0 kg (14.2 kg in 2021), with production expected to increase up to 6.77 million tons (with an average annual growth rate of 0.7%) by 2030 (Arizzi et al., 2015). Moreover, from a green chemistry and global point of view, the reaction illustrated in below in equations 1 and 2 results in a higher atom economy (62.7%) compared to both conventional routes using NaOH and $CaCl_2$ (38.74%) or those involving polymeric exchange resins (51.03%) (Anastas & Warner, 1998). Finally, the CO_2 emission factor is comparable to or even lower than that associated with manufactured chemical reagents. Although the first step (eq. 1) entails CO_2 release, $Ca(OH)_2$ is later transformed into $CaCO_3$, making the CO_2 balance stoichiometrically neutral for the chemical reaction. In addition, the CO_2 emission factor associated with CaO fabrication from $CaCO_3$ is estimated to be about 0.79 kg of CO_2/kg (Anastas & Warner, 1998).

$$CaCO_3 \rightarrow CaO + CO_2 \uparrow \quad (1)$$
$$CaO + H_2O \rightarrow Ca(OH)_2 \quad (2)$$

Due the huge quantities involved, eggshell waste constitutes a considerable burden, leading to economic loss and environmental pollution, although, as reported by Fava et al. (2015) this waste is readily convertible into a series of valuable commodities for domestic and industrial applications. Transition metal oxides (TMOs) such as zirconia and titanium oxide were initially studied to evaluate the feasibility of their use as solid acid catalysts in the field of biodiesel production (Xie & Li, 2006). The main limitations of using acid catalysts are associated with long reaction times and high reaction temperatures (Xie & Li, 2006). Solid base catalysts such as calcium oxide (Kim et al., 2010), magnesium oxide (Noiroj et al., 2009), alkali alumina (Freedman et al., 1986), hydrotalcite (Ilgen et al., 2009), zeolite (Zabeti et al., 2015) are promising for use as heterogeneous catalysts to substitute acidic and basic catalysts. However, the majority of processes use very expensive synthetic chemicals, although literature data report how eggshell wastes are well known for their use in a range of applications as heterogeneous catalysts. One notable example is biodiesel production by means of a conventional transesterification process using a homogeneous alkaline catalyst (Ghosh et al., 2016). This process is fast and efficient, achieving a conversion rate of more than 95% at 60°C over a 1 h reaction time. However, the process also has a number of limitations, including formation of side products and soaps (Ravindran et al., 2012) and being a complicated process to achieve separation of the product using homogeneous catalysts (Loehr, 2012). Indeed, a series of different methods have been developed or are undergoing development for the production of catalysts from eggshell wastes.

3. METHODS

The synthesis of calcium oxide and calcium oxide NPs differ, invariably favoring the process which yields the best performance over a high surface area (Habte et al., 2019). Synthetic methods are generally classified into physical, chemical, and biological. Physical methods include thermal decomposition Sadeghi et al., 2022), the sono-chemical method (Alavi & Morsali, 2010), microwave processing (Roy et al., 2013), co-precipitation (Ghiasi & Malekzadeh, 2012), and the sol-gel method (Darcanova et al., 2015). Ball milled grinding methods are used for the preparation of CaO nanoparticles (Darcanova et al., 2015). Synthesis of Nano CaO from waste eggshell is achieved using plant extracts such as Mentha piperita (Ijaz et al., 2017), benign papaya leaf extract and green tea extract (Anantharaman et al., 2016), and rhododendron arboretum extracts (Ramola et al., 2019). These plant extracts contain bioactive compounds including antioxidants, phenols, enzymes, amino acids, and sterols acting as reducing agents (Aljabali et al., 2018). CaO from waste eggshell was prepared by boiling in water, followed by calcination of the waste eggshell at 900°C for 3 hours (Awogbemi et al., 2020; Lanzon et al., 2023). Initially, the eggshells were washed with deionized water and dried at 130°C for three hours. The dry material was heated at 550°C for five hours to eliminate the remaining organic matter. In the final step, the dry shell was calcined at 1100°C for two hours to form CaO (Awogbemi et al., 2020). Osman et al. (2022) reported grinding dried eggshells using an electrical milling machine for 5 min, after which the fine powder was calcined at 900°C for one hour, resulting in CaO.

This novelty of this article focuses on the benefits represented by synthesizing CaO catalysts from eggshell waste with the aim of minimizing environmental pollution and reducing the amount of wastes forwarded to land fill, thus complying better with environmental regulations; this eco-friendly process therefore plays a major role in converting eggshell waste into a valuable heterogeneous catalyst for use in a range of applications. CaO from waste eggshells is produced in large quantities by simply calcining eggshells in air at high temperatures - the method used is simple, the reaction time is relatively short, and the product yield is high, with a large surface area. Eggshell waste catalysts can be used in a wide range of applications in biodiesel production, civil engineering, medicine, water treatment, and synthesizing bioactive compounds.

3.1 Relevance

A wide range of physical, chemical and biological techniques are adopted in the synthesis of CaO from waste eggshells, yielding a compound featuring a relatively high catalytic activity, good thermal-electrical properties and chemical stability. The synthesis of CaO focuses on environmentally friendly and economical processes. The synthesis of waste eggshells is moreover a highly economical and socially-acceptable process. Going forwards, additional efforts will focus on waste prevention and the recycling of waste eggshell into high-quality CaO catalysts as secondary raw materials.

3.2 The Rationale

A series of different methods are available for use in the synthesis of CaO, all of which may be further modified changing reaction parameters. CaO is used in synthesizing composites, cosmetics, cement production, and $CaCO_3$ particles as fillers in polymer composites, construction materials, metal and antimicrobial activity. Eggshell waste yields a range of potential benefits including use as an additive in calcium phosphate bio-ceramics as a calcium supplement (Sakai et al., 2017); Siemiradzka et al., 2018), adsorbent material (Witoon, 2011), as a solid catalyst for biodiesel production (Gupta et al., 2018; Huang et al., 2020), bone graft substitute (Salama et al., 2019), and for environmental remediation (Mignardi et al., 2020). Ok et al. (2011) investigated the use of CaO in heavy metal removal, whilst Choi & Lee (2015) looked at the use of CaO to produce green hydrogen in renewable energy streams. In the medical field, CaO NPs have been successfully applied in drug delivery. Ali et al. (2015) conducted a study in rats using manual hematoxylin and eosin staining protocols, reporting a finding of toxicity of CaO nanoparticles featuring acute tubular and hepatic degenerations in the rat kidneys and liver, respectively. Tangboriboon et al., 2012, reported how CaO prepared from waste eggshells is characterized by the use of physical methods such as XRD, SEM, TGA and FTIR. From this study, the organic (C/H/N) composition of raw duck eggshells were investigated using an elemental analyzer. The average percentages detected of carbon, hydrogen, and nitrogen amounted to 13.64%, 0.49%, and 0.63%, respectively.

4. FOOD WASTE MATERIALS FOR THE PRODUCTION OF NANOPARTICLES

The development of efficient and cost-effective approaches aimed at decreasing organic waste and developing better food waste operational practices, capable of improving overall management of the food supply chain is paramount. Several techniques for the processing of waste fruits, vegetables and grains have been explored (Ghosh et al., 2016); Ravindran & Jaiswal, 2016). These approaches aim to exploit the value and practical advantages of food waste, eventually reducing the amount of waste deposited at landfills (Loehr, 2012). Food waste and nanoparticle synthesis may, at first sight, appear an unlikely partnership. However, recent studies reported in the literature have revealed the active role that biomolecules play in reducing precursor metal ions in aqueous solutions to produce nanoparticles (particles less than 100 nm) featuring exceptional medicinal and pharmaceutical properties. The biomolecules also act as reducing agents/capping agents that promote particle growth in different angles and prevent nanoparticle agglomeration (Shah et al., 2015). The synthesis of nanoparticles from food waste has the potential to deliver consistently sustainable and green chemistry-based technologies that are eco-friendly, hence reducing the human health and environmental risks generally associated with the use of toxic solvents and chemicals with conventional processes (Anastas & Warner, 2000). In recent decades, considerable interest has been generated amongst the scientific community in nanotechnology research due to their exceptional physiochemical properties and application in several fields such as pharmaceuticals, biomedicine, electrical, environmental, food, and agriculture industry. The small size and large surface area to volume ratio are two substantial features of nanoparticles, which display improved properties compared to conventional processes (Willems, & van den Wildenberg, 2005). For example, gold (Au) nanoparticles are now widely used in medicine for diagnostics (Torres-Chavolla et al., 2010), targeted delivery of pharmaceuticals (Bhumkar et al., 2007), Paciotti et al., 2004) and tumor destruction via hyperthermia (Huang et al., 2006). Silver (Ag) nanoparticles display a broad range of antimicrobial activity against numerous human and animal pathogens (AshaRani et al., 2009); as a result, they are used as antimicrobial agents in a wide variety of medical and consumer products (Tran et al., 2013). Moreover, platinum (Pt) and palladium (Pd) nanoparticles have been used as catalysts in the production of value-added products (Cheong et al., 2010; Lin et al., 2011).

Furthermore, metal oxide nanoparticles such as copper oxide (Cu_2O, CuO) and zinc oxide (ZnO) have displayed antimicrobial activity (Abboud et al., 2014); accordingly, ZnO nanoparticles have been used in a variety of food packaging applications (Espitia et al., 2012). Dubey et al. (2010) reported the formation of 16-nm Ag spheres and 11-nm Au nano-triangles when Tanacetum vulgare (tansy fruit) extract was reacted with aqueous solutions of $AuCl_4^-$ ions and Ag^+ ions, respectively. In addition, numerous other food waste extracts, including Pyrus sp. (pear fruit) and Mangifera indica (mango peel), have displayed the potential to reduce Au (III) ions to form Au nanoparticles (Yang et al., 2014; Edison & Sethuraman, 2012). Citrus sinensis (orange peel) and Ananas comosus (pineapple) extracts have revealed potential in reducing Ag^+ ions in aqueous solutions to form Ag nanoparticles (Kaviya et al, 2011). Similarly, Pt and Pd nanoparticles have been synthesized via Musa paradisiac (banana peel), tea, coffee extracts, and lignin (Namvar et al., 2015; Nadagouda & Varma, 2008). Kaviya et al. (2011) studied the formation of Ag nanoparticles using Citrus sinensis (orange) peel extract as the reducing agent; the reduction of Ag nanoparticles was formed swiftly in 20 min, with the observation that NPs formation changed swiftly from colorless to yellowish-brown. Consequently, the characterization of Ag nanoparticles from SEM analysis revealed a spherical nanoparticle morphology, with a profoundly changed size depending on reaction temperature. At 25°C, mean particle size was 35 ± 2 nm; at 60°C, size was 10 ±1 nm (Kaviya et al., 2011). Furthermore, Ateeq et al. (2021) reported the synthesis of Ag NPs from *Pennisetum Glacuum* rice husks for the removal of algae from wastewater. In this study, AgNPs were doped on charcoal for the oxidation of benzyl alcohol to benzaldehyde. A significant role is played by food waste due to the presence of a wide range of phytochemicals, such as polyphenols, which act as capping agents, influencing the orientation and assembly of smaller particles during subsequent growth. The modelling action designed by surfactants during biological synthesis indicates that the change in NPs occurs following the pathway (Chiu & Ruan, 2013; Gao et al., 2020).

Nanoparticle growth ultimately leads to the formation of morphological structures such as spheres, cubes, triangles, hexagons, pentagons, and wires (Akhtar, Panwar, & Yun, 2013). The following experimental parameters play a significant role in the formation of nanoparticles: (1) the nature of food waste extract, (2) the concentration of food waste extract in the reaction mixture, (3) metal ion concentration in the source solution; (4) reaction mixture pH; (5) reaction mixture temperature; (6) contact time and (7) choice of solvent. All these parameters are vital for nanoparticle formation and exhibit their physiochemical properties (Dwivedi & Gopal, 2011; Obadiah et al., 2013).

Nasrollahzadeh et al. (2016) reported the synthesis of nanocomposites with eggshell catalysts by reduction method, i.e. mixing dried eggshell powered with $CuCl_2$ and $FeCl_3$ and aqueous extract of Orchis mascula L.in alkaline medium. The removal of dyes from wastewater was observed by heating at (70°C), stirring and filtering (Nasrollahzadeh et al., 2016). Vandeginste (2021) reported the synthesis of nanocomposites from corn starch, bioplastics, bacterial cellulose, polypropylene carbonate, polylactic acid, poly (diethylene glycol succinate) (PDEGS), polypropylene (PP), and polyvinyl alcohol (PVA) using eggshell, $NaClO_2$ solution, NaCl solution, NaOH solution, methanol, acetone or dimethylformamide. After washing, the eggshells were dried at room temperature or in an oven (either in air or under vacuum) at an elevated temperature, 50-110°C. Different parameters are used for the synthesis of nanocomposites derived from eggshells (Huang et al., 2020). In recent years, there has been an extensive search for natural antimicrobial agents capable of overcoming the limitations of currently available synthetic antimicrobial pharmaceuticals. For example, copper (Cu) and Cu oxide nanoparticles are natural antimicrobial agents that have been biosynthesized using a variety of plant-derived extracts. Lee et al. (2011), reported the synthesis of Cu nanoparticles from leaf extract of Magnolia Kobus. Spherical particles of Cu nanoparticles formed in the 40 to 100 nm range and displayed excellent antimicrobial properties towards Escherichia coli (Lee et al., 2011). From a marine phytoplankton perspective, Abboud et al. (2014) reported the biosynthesis of copper oxide nanoparticles using brown algal extract from Bifurcaria bifurcata. The resulting Cu nanoparticles were spherical, ranged in size from 5 to 45 nm and displayed good antibacterial properties towards both Enterobacter aerogenes and Staphylococcus aureus (Abboud et al., 2014). Nagarajan & Kuppusamy (2013) reported the synthesis of zinc oxide (ZnO) nanoparticles from a brown seaweed extract (Sargassum myriocystum) that displayed excellent antimicrobial properties against several bacteria and fungi. In an attempt to provide a solution to the above shortcomings, the following should be achieved: (1) development of a mechanism for nanoparticle formation, (2) investigation of experimental parameters on nanoparticle size, shape and solvent dispersion, (3) refining of the biosynthesis process to improve reproducibility and activity of NPs for antimicrobial activity, any biochemical reactions and environmental remediation (Jha et al., 2009). Moreover, to promote commercialization the following are required: (1) development of new technologies to overcome the limitations of scaling up the biosynthesis process; (2) development of a continuous supply route for suitable aquacultural and horticultural food waste; and (3) optimization of the waste management chain to utilize biomolecules and bioactive chemicals present in aquacultural and horticultural food waste to produce nanoparticles (Parfitt et al., 2010).

4.1 Eggshell waste as catalysts

Eggshell represents approx. 10% of a hen egg, an elementary foodstuff widely consumed worldwide for domestic and industrial uses. Eggshells are generally produced in large amounts by egg processing industries. Vast quantities of this solid residue are still disposed of as waste in landfills without any pretreatment, which leads to organic pollution of the environment (Gao & Xu, 2012). Eggshell is made up of 96% calcium carbonate, 1% magnesium carbonate, 1% calcium phosphate, organic materials (mainly proteins) and water (Oulego et al., 2020). The structure of an eggshell can be divided into three layers: a thin mammillary inner layer without pores, a thick palisade middle layer comprising many large pores, and a thin cuticle outer layer with no pores (Tan et al., 2015). However, the eggshell contains approximately 7,000-17,000 funnel-shaped canals distributed unevenly on the eggshell surface, which makes it a highly feasible option for energy and mass transfer (Guo et al., 2019). However, eggshell waste is an excellent bio-template that can be recycled as an economical and eco-friendly carrier of metal nanomaterials. Eggshell waste has been used in numerous studies performed to date as a low-cost catalyst for biodiesel production and an adsorbent for the removal of metal ions from wastewater (Goli & Sahu, 2018), as displayed in Table 1. In addition, the use of eggshell membranes as a catalyst has been studied in wastewater treatment processes (Oulego et al., 2020). Al-Ghouti & Salih (2018) reported how the use of waste eggshells as a catalyst eliminates waste whilst

TABLE 1: Comparation of eggshell catalysts and applications.

Product	Process	Egg catalysts preparation	Reference
Hydrogen/Syngas	Methane oxidation/Wood gasification	Methane oxidation Wash with deionized water and calcined (100°C, 4 h). Remove the membranes, wash and dry. Crush, sieved (250 mm), dry and calcine (900°C, 2 h).	Karoshi et al., 2015
Wood	Coal gasification	Wash and dry (70°C, overnight). Ground and sieved to 210 mm. Calcine (800°C, 2 h).	Taufiq-Yap et al., 2013
Biodiesel	Oil transesterification		Shinha & Murugavelh, 2016
Hydrogen	Syngas	Methane oxidation Wash with deionized water and calcined (1000°C, 4 h).	Shinha & Murugavelh, 2016
	Coal gasification	Wash and dry (70°C, overnight). Ground and sieved to 210 mm. Calcined (800°C, 2 h).	Shinha & Murugavelh, 2016
Dimethyl carbonate	Propylene carbonate and methanol transesterification	Wash and remove the membranes. Dry (120°C, 1 h). Mill and mix the eggshell powder with CH_2Cl_2 and place in an ultrasonic bath (60 Hz, 50°C, 1 h). Filter and dry at room temperature. Wash and dry at room temperature. Crush and mill by mortar.	Gao & Xu, 2012
Bioactive Compounds	Chromenones Pyrans	Heterogeneous base catalyzed organic synthesis. Wash, clean, dry and treat by ball-mill. Wash and remove the membranes. Wash and dry (120°C, 1 h). Mill and calcined (900°C, 1 h).	Mosaddegh & Mosaddegh, 2013
	Pyrans Heterogeneous based catalyzed organic synthesis	Dry (100°C), crush and calcined (900°C, 3 h)	Hassankhani et al., 2016
Benzylidenepropanedinitrile	Knoevenagel condensation Derivatives	Wash and dry (80-90°C, 24 h). Ground by mortar and calcined (900°C).	Patil et al., 2013
Polyhydroquinoline	Hantzsch condensation	Wash and dry (80-90°C, 24 h). Ground by mortar and calcined (900°C) 3-4 hours.	Patil et al., 2013
Benzothiazoles	Heterogeneous base catalyzed organic synthesis	The raw eggshell powder was calcined up to 900°C, and this temperature was maintained for 3 h. Wash, dry (105°C, 4-24 h), ground (~5 mm)	Borhade et al., 2016
Lactulose Lactose	Isomerization	The egg shells were washed with water; subsequently the eggshells were dried at 105°C for 12 h, grounded and sieved by 80 mesh screen (mesh size, 177 lm).	Nooshkam & Madadlou, 2016
Nanocomposites Nano composites (PP, Corn starch, Bioplast, bacterial cellulose, Polypropylene carbonate, Polylactic acid, poly(diethylene glycol succinate) (PDEGS), polyvinyl alcohol (PVA).	Reduction	Eggshells with a $NaClO_2$ solution, NaCl solution, NaOH solution, methanol, acetone or dimethylformamide. After washing, the eggshells are dried at room temperature or in an oven (either in air or under vacuum) at elevated temperature, generally between 50 and 110°C, and there are a few studies that report drying at 200 or 250°C. The drying time varies from half an hour to several days.	Vandeginste et al., 2021
Egg shell supported CuO catalysts	Blue 19 degradation	Egg shell washed with deionized water and dried in oven at 80°C, powdered sieved through 100 mesh. Then known amount egg shell and $CuSO_4$ solution is is added and stirred for 16hr at room temperature. Then the blue solids were washed with ethanol and deionized water several times and dried in oven at 80°C. And then CuO was obtained by calcining the prepared CuO egg shell at 600°C for 2 hours.	Xu et al., 2022
Valorization of eggshell waste	Construction materials: Concrete, clay bricks, ceramic tiles, Compressed stabilized blocks (CSBs)	The Egg shell washed with deionized water and dried in oven at 100°C for 24 hours. And calcined between 600-1000°C in oven for 2-4 hours.	Ngayakamo & Onwualu, 2022
Waste mud crab shell	Transesterification reaction.	Waste shell is washed with water and dried in oven at 105°C for overnight. Then calcined in furnace at 900°C for 2 hours.	Boey et al., 2009
Egg shell waste	For removal of phenol from aqueous solution	Eggshells were prepared and sieved with a sieve of 425 µm. And the egg shells were at different temperatures at 200˚C, 400˚C, 600˚C, 800˚C and 1000˚C for 2 h.	Chiraibi et al., 2016

Valorization of eggshell waste	Removal of Co^{2+} ions from aqueus solutions	Eggshells were washed several times with tap and they were dried at 105˚C for 24 h in an oven, then the catalysts were calined in furnace at 600˚C for 2 hours.	Mignardi et al., 2020
Waste egg shell catalysts	Biodiesel production	Egg shell poweder was calcined at 900˚C for 2 hours to produce CaO catalysts.	Ari et al., 2017
Egg shell catalysts	Synthesis of 2-arylbenzothi-azole derivatives	Egg shell powder is shwahse with water and ried in oven at 100˚C for 2 hours. Then calciend in furnace at 900˚C for 3 horus.	Borhade et al., 2016
Calcined waste egg shell	Material science in catalysis, pharmaceutical and mechano-chemistry	Waste egg shell is converted to CaO cata-lysts by many methods ie calcined at 900˚C in furnace for 2 hours.	Balaz et al., 2021
Monetite micro particles from eggshell waste	Environmental applications, fuel additive	Calcined egg shell and disodium phosphate were added into it and pH 2 was adjusted with concentrated hydrochloric acid and stirred for 30 min at room temperature and refluxed at 80˚C for 2 h. Then it was dried in an oven at 80˚C for 24 h and ground to fine powder.	Khan et al., 2020

concomitantly producing low-cost catalysts. This contributes towards maintaining a clean environment and, at the same time, reducing prices of the end-products. Shan et al. (2016) reported how the development of CaO-based catalysts for biodiesel production from waste eggshells represents a promising technology. However, the present review focuses on recently developed studies, showing the potential of eggshell waste to be used as a low-cost catalyst in the manufacture of various compounds. CaO from waste eggshell catalysts is characterized by XRD, FTIR, TG-DTA, SEM spectroscopy and BET surface area (Shan et al., 2016). Table 1 presents the details of egg catalysts developed for use in a wide range of applications.

4.2 Synthesis of biodiesel with eggshell catalysts

A rise in global warming and fossil fuel energy crises have prompted scientists/researchers to seek alternative fuels, with proving to be the most prominent alternative fuel. Biodiesel is renewable, nontoxic, and environmentally-friendly, being synthesized from oils and animal fats (Goli & Sahu, 2018). However, the use of expensive catalysts has made this process unsustainable in the long-term. The first use of eggshell as a catalyst in the production biodiesel with a 98% yield was reported by Shan et al in 2016 and Al-Ghouti & Salih in 2018. Several modifications have since been developed for the synthesis of CaO catalysts from waste eggshells to improve the yield of biodiesel (Oulego et al., 2020). Calcination temperature has emerged as a pivotal factor in the production of efficient solid base catalysts from mollusk/eggshells comprised of $CaCO_3$. The Thermal Gravimetric Analysis (TGA) and X-Ray diffraction (XRD) characterization analysis shows the decomposition of $CaCO_3$ to CaO at temperatures of 700°C, resulting in active transesterification catalysts, typically offering >90% conversion of palm, soybean and Karanja oils in 2-6 hours at 60-70°C reaction temperatures with alcohol:oil molar ratios around 10, and catalyst loadings of 2-5 wt%. Regeneration of deactivated eggshell-based catalysts is usually achieved through re-calcination at 900°C. Another major drawback with these catalysts is the leaching of Ca^{2+} into the alcohol phase of the reaction media, leading to a gradual loss of activity (Boey et al., 2009). However, deactivation of CaO catalysts is caused by exposure to air (Birl et al., 2012). As reported by Corrie et al. (2014) when compared with the calcined crab waste, the high Ca content present on calcined surface of eggshells is attributed to a better catalytic performance in the transesterification of waste cotton cooking oil with calcined eggshells. However, using pistachios, shell yield was 84% (catalyst 3 wt%, methanol/oil molar ratio 9-12:1 at 60°C). Roschat et al., 2017, reported the potential of use of rubber seed oil in the production of biodiesel using coral 10 wt% CaO-based catalysts in methanol in 180 minutes, resulting in a 97% yield. With 100% coral catalysts, the transesterification reaction took longer (210 min) under optimum conditions (catalyst 9 wt%, 12/1 methanol/oil molar ratio and 65°C). Asri et al. (2017) reported how using eggshell catalysts, biodiesel yields gradually increased from 72.05 to 75.92% over a period of 6-7 hours at 65°C. Crushed eggshell chips were powdered and passed through 100 and 200 micron sieves and egg catalysts were then calcined at 900°C for one hour (Asri et al., 2017). Farooq et al. (2018) reported the preparation of an effective catalyst from waste chicken eggshells at three different temperatures (800°C, 900°C and 1000°C). Date seed oil, 5 wt.% catalyst, 12:1 methanol: oil ratio and conversion efficiency of biodiesel was 93.5% at the end of 1.5 hours (Farooq et al., 2018). However, a similar study conducted by Obadiah et al. (2012) reported order of magnitude increases in surface area from 680 m^2g^{-1} to 885 m^2g^{-1} of calcined sheep bone with calcination temperature, 600-800°C (Obadiah et al., 2012). Chakraborty et al. (2010) studied the preparation of calcium-based catalysts from ash scales using a similar route, with calcination at 900°C for 2 hours yielding a porous material comprised of mainly b-tricalcium phosphate with a surface area of 39 m^2g^{-1}, resulting in 98% fatty acid methyl esters (FAME) and transesterification of soybean oil for 5 hours at 70°C using merely 1 wt% of eggshell catalyst and low methanol: oil ratio of 6. In another study, Zn/CaO was synthesized from waste hen eggshells and investigated for use in the synthesis of biodiesel using waste frying oil (WFO) as a feedstock (Rahman et al., 2021; Rahman et al., 2022). The waste eggshells were calcined at 900°C to obtain CaO, with Zn impregnated on CaO via the wet incipient impregnation method, resulting in a yield ranging from 64 to 91% of biodiesel oil. Khan et al. (2022) reported on the preparation and application of mussel shell CaO catalysts impregnated with praseodymium to obtain CaO-praseodymium catalysts 3, 5, 7, 9 & 11 wt% for use in the production of biodiesel by transesterification reaction from castor oil. However, Khan et al. (2022) reported that the optimization of reaction time, temperature, and transesterification was carried out at different temperatures: 50°C, 55°C, 60°C, 65°C with different intervals of time from two to 5 hours, 2.5% catalyst loading rate and 8:1 methanol to oil ratio. The maximum yield obtained was 87.42% over a 4-hour period (Khan et al., 2022). Anantharaman et al. (2016) studied biodiesel production using eggshell catalysts prepared by washing with deionized water and oven drying at 100°C for 24 hours. The catalysts were calcined in the range of 700-1000°C in a furnace for 2-4 hours, resulting in an 80% yield of biodiesel.

5. APPLICATIONS OF EGGSHELL CATALYSTS

5.1 Synthesis of bioactive compounds

Eggshell-derived catalysts have been used for the synthesis of bioactive compounds such as benzothiazoles, bioactive molecules scaffolds with antitumor, antiallergic, antidiabetic, and antimicrobial agents (Balaz et al., 2021). Eggshell CaO catalysts have been applied to produce critical industrial chemicals, such as dimethyl-carbonate, oximes, and glycerol oligomers. CaO eggshell catalysts were simply prepared by milling using low-cost tools such as mortar and pestle. Other methods used in the preparation of eggshell catalysts include mechanochemical methods, ultrasound treatment, and impregnation approaches (Balaz et al., 2021). Oulego et al. (2020) reported the conversion of eggshell wastes to CaO catalysts and subsequent use in the synthesis of different bioactive compounds such as chromenones, pyran derivatives, polyhydroquinolines, aromatic aldehydes, and carbohydrates. Nitrogen-containing phthalazine compounds and other N-heterocycles are potentially indicated for a range of antimicrobial, antioxidant, and anti-convulsant applications. These compounds are synthesized through a multicomponent reaction using heterogeneous catalysts (Balaz et al., 2021). In a study by Balaz et al. (2021), eggshell-derived catalysts were employed to synthesize pyrozolo-phthalazine derivatives through four-component condensation, such as Knoevenagel-Michael reaction. This reaction was conducted in water as a green solvent under moderate conditions (temperature, shorter reaction time). The study revealed a good catalytic performance for eggshell catalysts, achieving up to 98% yield of the end product (Balaz et al., 2021). Chromenes and Pyrans were produced using heterogeneous base catalysts prepared from eggshells. The eggshell catalysts were synthesized by washing and removing the membranes and were then dried at 120°C for one hour. The milled eggshell powder was mixed with CH_2Cl_2 and placed in an ultrasonic bath at 60 Hz, 50°C for one hour. The catalysts were filtered and dried at room temperature before being calcined at 900°C for one hour to yield CaO catalysts (Mosaddegh, 2013; Mosaddegh & Hassankhani, 2014). In another study, the benzylidene propanedinitrile derivative was synthesized via Knoevenagel condensation (Patil et al., 2013). In this study, eggshells were dried at 100°C, crushed and calcined at 900°C for three hours, yielding 97% CaO catalysts. Polyhdroquinone was synthesized by means of the Hanzsch reaction over a 45-minute period, providing a yield of 84-96%. Eggshell catalysts were crushed and dried at 100°C for 2 hours and then calcined at (900°C, 3 h) (Borhade et al., 2016; Nooshkam & Madadlou 2016). Borhade et al. (2016) reported the synthesis of unsymmetrical benzothiazoles with eggshell catalysts synthesized in solvent-free conditions using a grinding method at room temperature. The reaction proceeded smoothly and resulted in excellent yields (86-97%), with a shorter reaction time (15-48 min). Catalysts were dried at 100°C for two hours and crushed and calcined at 900°C for three hours (Borhade et al., 2016). Lactulose and lactose were obtained with 70% conversion with eggshell catalysts, washed, dried at 105°C for 4-24 h, and ground (~5 mm). The second set of eggshells was washed, dried (105°C, 12 h), ground and sieved (177 mm) (Nooshkam & Madadlou, 2016) as summarized in Table 1.

5.2 Other applications

The use of eggshell waste as a catalyst has also been investigated in the synthesis of hydrogen; d during hydrogen production, biomass gasification, on wood for example, was carried out in the presence of an eggshell catalyst. Hydrogen production is an alternative, environmentally-friendly source of energy. Oulego et al. (2020) reported the observation of an increase in hydrogen production of up to 23% when CaO from eggshell was utilized as a catalyst. Another green chemistry/environmentally friendly reaction that spiked interest involved the use of eggshell waste to produce dimethyl carbonate. Dimethyl carbonate can replace dimethyl sulfate and methyl halides in methylation reactions and harmful phosgene in polycarbonate and isocyanate synthesis. Eggshell catalysts is used to synthesize dimethyl carbonate through transesterification of propylene carbonate. The conversion of propylene carbonate and the yield of dimethyl carbonate was observed as 75%, and 80%, respectively as reported by Oulego et al. (2020). Additionally, dimethyl carbonate was prepared from propylene carbonate and methanol. The eggshell catalysts used were washed with deionized water, dried at (100°C, for 24 h), and calcined (1000°C, 2 h), resulting in a 12-66% yield (Gao & Xui et al., 2012; Ngayakamo & Onwualu, 2022). Ngayakamo & Onwualu (2022) reported the possibility of using eggshell catalysts as filler in construction materials such as concrete, clay bricks, ceramic tiles, and compressed stabilized blocks (CSBs) with valorization of eggshell catalysts. Solid base catalysts developed from heterogeneous waste solid bases characteristically offer considerably higher rates of transesterification reaction than solid acid catalysts (Boey, et al., 2009). A plethora of literature exists with regard to the application of CaO as transesterification catalysts due to the presence of calcium-rich eggshells, with the majority of these catalysts synthesized at high temperatures. Calcination at 800°C of shells from shellfish and snails has been reported (Boey et al., 2009; Birla et al., 2012; Viriya-empikul et al., 2012). This green method is an economical route for the disposal of maricultural/agricultural wastes and development of value-added products for the conversion of other wastes (Birla et al., 2012; Taufiq-Yap et al., 2013; Laca et al., 2017). Viriya-empikul et al. (2012) highlighted the crucial role played by calcination temperature in the production of efficient solid base catalysts from mollusks/eggshells. These shells mostly possess $CaCO_3$; TGA and XRD characterization studies reveal that the decomposition to CaO at temperatures >700 results in active catalysts for transesterification reaction with the conversion of palm > 90%, soybean and karanja oils in 2-6 hours at 60-70°C, alcohol: oil molar ratios around 10, and catalyst loadings of 2-5 wt.%. (Viriya-empikul et al, 2012). Nurfitri et al. (2013) observed that deactivated waste shell-based catalysts can be regenerated by re-calcination at 900°C. However, regeneration of these catalysts by calcination implies higher energy costs and may not be feasible for industrial processes, as reported by Boro et al. (2012). The major concern with these catalysts is the leaching of Ca^{2+} in the reaction of the alcohol phase, which leads to the loss of activity of catalysts due to formation of a homogeneous solution in the reaction, as well as the saponification method, in which impure or untreated oils are routinely used. Deactivation of CaO catalysts may occur due to poisoning of catalysts upon exposure to air (Bennet et al., 2016). Many calcium-containing catalysts have been developed from waste such as coral, bones, scales, and calcite-containing rocks (Bennet et al., 2016). Bennet et al. (2016) confirmed that corals are marine polyps with calcareous exoskeletons formed from aragonite, a crystalline form of $CaCO_3$. Roschat et al. (2017) studied the conversion of coral fragments to CaO by calcination at 700°C. These catalysts were used in the transesterification of various oils, including palm, soybean, rice bran and waste cooking oils, with methanol (Bennet et al., 2016). Indeed, calcination conducted at temperatures > 650°C under ideal conditions has yielded bone-derived catalysts intended to deliver conversions achieving grades comparable to laboratory grade CaO using high catalyst loadings (8-20 wt.%) with reaction temperature 80°C and alcohol, oil and CaO catalysts synthesized from mollusks or eggshells. Bennet et al. (2016) obtained conversion to fatty acid methyl esters (FAME) of 95%, 97% and 98% from waste cooking oil, safflower oil and soybean oil, respectively, after 6 h at 60 °C with a methanol: oil ratio of 12 and 7.5 wt% catalyst. A similar study was performed by Obadiah et al. (2012), who observed an order of magnitude of increases in the surface area from 680 m^2g^{-1} to 885 m^2g^{-1} of calcined sheep bone with calcination temperature in the range of 800-1000°C and conversions to fatty acid methyl esters (FAME) of 93% (Obadiah et al., 2012). Chakraborty et al. (2010) reported the preparation of calcium-based catalysts from ash scales through calcination at 900°C for 2 hours, yielding a porous material comprised of mainly b-tricalcium phosphate with a surface area of 39 m^2g^{-1}, resulting in 98% FAME in the transesterification of soybean oil for 5 hours at 70°C with merely 1 wt% catalyst and low methanol: oil ratio of 6. However, a series of ecological and sustainability issues are associated with the utilization of calcium obtained from coral and turtle shells for the industrial production of solid base heterogeneous catalysts (Chakraborty et al., 2010). A study by Khan et al. (2020) reported on issues relating to fuel addition and coating with monetite particles using eggshell waste. Karoshi et al. (2015) studied the production of hydrogen through methane oxidation; catalysts were washed with deionized water and calcined at 1000°C for 4 hours, achieving a conversion rate of 26%. Furthermore, Osman et al. (2016) reported the washing of catalysts with deionized water and subsequent calcination at (200-500°C) for use in the oxidation of methane. Eggshell catalysts were washed and dried at 70°C overnight and then crushed, ground to fine powder, and sieved at 210 mm, whilst CaO catalysts were calcined at 800°C, for 2 hours and final catalysts applied to produce important industrial chemicals such as dimethyl-carbonate, oximes, and glycerol oligomers (Osman et al., 2016), thus avoiding the presence of toxic chemicals such as dimethyl sulfate methyl halides in methylation reactions and phosgene in polycarbonate and isocyanate synthesis, dimethyl carbonate. Indeed, CaO catalysts from waste eggshells have proved to act as effective catalysts for use in the development of important pharmaceutical drugs, particularly as they meet a series of requirements established for sustainable and green chemistry techniques (Roschat et al., 2017).

5.3 Wastewater treatment

Eggshell CaO catalysts and membranes have been investigated for use in the removal of iron, cadmium, chromium, Cu (II), reactive dyes, Pb (II), phenol and iodosulphate compound (Xu et al., 2022; Chraibi et al., 2016). Cyanide was removed from wastewater using calcined eggshell catalysts and reached an excellent adsorption efficiency of 84.53% (Eletta et al., 2016; Xu et al., 2022; Chraibi et al., 2016). Another study reported the use of eggshell membranes and modified eggshell membranes as bio-sorbents for the remediation of boron from desalinated water. The eggshell membrane adsorbed 97% of boron, while the modified eggshell membrane adsorbed 95% of boron (Al-Ghouti & Salih, 2018). Adsorption technology has advanced over the decades for wastewater treatment technology. As a result, researchers/scientists have investigated various materials as adsorbents to remove pollutants from wastewater. Activated carbon as an adsorbent is synthesized from coconut husks, date seeds, pistachio seed coat, and macadamia carbonaceous bark (Ateeq et al., 2021). Recently, eggshells have attracted increasing attention for application in the removal of heavy metals and other elements from wastewater (Al-Ghouti & Salih, 2018). Eggshells are potentially capable of removing these pollutants, possessing other advantages such as being environmentally-friendly, readily available and cost-effective. Khan et al. (2020) reported how monetite particles and eggshells facilitated the degradation of Congo red dye, a carcinogenic dye and water decontamination. Monetite is an active catalyst capable of eliciting an increase in kapp value i from 0.0077 to 0.00114 min^{-1} by increasing catalyst dosage from 1.0 to 2.5 mg/ mL. It is also a good additive that results in an increased calorific value from 2.55 to 8.902 MJ/kg at a higher additive dosage of 10 to 40 ppm, as reported by Khan et al. (2020). Xu et al. (2022) prepared eggshell-supported CuO catalysts by washing with deionized water, drying at 80°C, and sieving the powder through 100 mesh; a known amount of eggshell and $CuSO_4$ solution was then added and stirred for 16 h at room temperature and blue solids were washed with ethanol, deionized water several times, and dried in an oven at 80°C. CuO was obtained by calcining the prepared CuO eggshell at 600°C for 2 hours for the degradation of blue dye, with 99.05% yield obtained in 20 min. However, the end-product was unsuitable for use as a catalyst due to the ease of aggregation of CuO nanoparticles. However, this may have been due to excessive loading of Cu solution leading to the aggregation of catalyst particles and reduction of active sites, thus inhibiting catalyst activity (Xu et al., 2022). Oulego et al. (2020) observed degradation of humic acids, analyzing the evolution over time of COD, biodegradability index (BOD5/COD), color number and pH. They achieved excellent results using 15% Cu and 5% Fe catalysts with a COD reduction of 82.3% and 75.1%, respectively, whereas use of non-impregnated eggshell catalysts produced a COD reduction of 58%. This could be attributed to metal loading and excellent metal distribution on the surface of the support (Oulego et al., 2020). Mignardi et al. (2020) reported the removal of metals from water in the range of 70-80% using eggshell catalysts.

5.4 Characterization of eggshell catalysts

Eggshell catalysts were characterized by means of XRD, SEM, DTA-TGA, FT-IR, and BET surface area.

5.4.1 X-ray Diffraction

X-ray Diffraction (XRD) is commonly used to analyze nanoparticle structure, composition and size. The eggshell catalysts were characterized by XRD, which showed a 2θ range from 10° to 90° (Wong & Ang, 2018). Eggshells possess calcium carbonate, which is the major compound. After calcination at 800°C, the concentration of calcium carbonate changed compared to the uncalcined eggshells due to the transformation of calcium carbonate into calcium oxide, leading to the observation of a calcium oxide peak which appeared after calcination at 1000°C under nitrogen gas flow (C) and in open air (D). This was attributed to the high thermal shift that occurred in the eggshells, due to the complete decomposition of calcium carbonate into calcium oxide at 1000°C (Wong & Ang, 2018). On comparing eggshell-derived CaO with commercial CaO catalyst, the peaks of the XRD profile appeared in the vicinity of the 2θ value at 35, confirming the presence of CaO.

5.4.2 Scanning Electron Microscopy

Cheng et al. (2021) reported that the morphologies of CaO, CaDG (calcium diglyceroxide), CaG (calcium glycerolate), and CaG-E (Calcium oxide from eggshell) catalysts were analyzed by Scanning Electron microscopy (SEM). The CaO surface exhibited irregularly shaped particles with considerable gaps and relatively large pores, attributable to the hollow structure of the eggshells. Commercial CaO offers a broader particle size distribution compared to eggshell-derived CaO. The large image shows the presence of cracks on the particle surface, which is ascribed to the overflow of CO_2 gas during the calcination process. The SEM image of CaDG displays particle size in the range of 7 to 10 μm and sharp rock shapes, which is consistently reported by Cheng et al. (2021). The SEM image of the synthesized CaG surface displays the formation of a single layer of irregular sheet structures. Based on these assumptions, it is evident that the large specific surface area enables the dissolution of CaG in glycerol during reuse. Hence, CaG can be utilized as a homogeneous catalyst in the macrolactonisation process. EDX analysis was evaluated to determine the composition of CaGE, with the results revealing the presence of 13 wt% carbon, 32 wt% oxygen, and 55 wt% calcium, as displayed in Figure 1 (Cheng et al., 2021).

5.4.3 Differential Thermal Analysis-Thermogravimetric Analysis

Cheng et al. (2021) reported the findings of TG/DTG analysis of the eggshell powder. The calcination process of waste eggshell powder was characterized by Thermogravimetric/Differential Thermal analysis (TG/DTA). The eggshell powder was heated from 30 to 1000°C at 10°C min−1 in N_2 atmosphere. The reduction of eggshell powder ranged from 30-500°C due to the decomposition of organic matter and removal of moisture from the eggshells. The quality of the eggshell powder had changed dramatically due to the removal of carbon dioxide at 600-800°C, whilst at a temperature of At 850°C the quality of eggshell was found to change repeatedly, likely due to the complete decomposition of $CaCO_3$. These results highlighted the formation of CaO during TG/DTG analysis, further revealing how $CaCO_3$ began to decompose at 600°C, consistent with XRD data analysis reported by Cheng et al. (2021).

5.4.4 Fourier-Transform Infrared

Awogbemi et al., 2020 reported the use of Fourier-Transform Infrared (FTIR) spectroscopy to study the structure and functional group of eggshell CaO catalysts. FTIR analysis was compared for raw, boiled, and calcined chicken eggshell powders, representing similar spectra, with raw and boiled eggshell powders featuring remarkably different spectra versus the calcined eggshell sample. Principal absorption bands of uncalcined raw and boiled chicken eggshell samples appeared at 1394 cm^{-1}, 873 cm^{-1}, and 712 cm^{-1}. This was ascribed to asymmetric stretch, out-of-plane bend and in-plane bend vibration modes for CO_3^{2-} molecules (Awogbemi et al., 2020). The organic matter displayed absorption bands at 2513 cm^{-1} and 1795 cm^{-1} that disappeared after calcination of the raw eggshell samples. However, a slight difference was observed between the absorption band appearing at 1071 cm^{-1} and 1092 cm^{-1} for the raw and boiled eggshell samples. However, calcined chicken eggshell displayed a sharp stretching band at 3642 $cm^{-1,}$ due to presence of the OH^{-1} group (Awogbemi et al., 2020). The absorption peak for calcined eggshell appeared at 3642 cm^{-1}, corresponding to stretching vibration and bending of hydroxyl groups present in $Ca(OH)_2$. However, during the calcination process, carbonate in the chicken eggshells decomposes to CaO and the absorption bands of CO_3^{2-} molecules migrated to higher energy and bands, which were observed at 1403 cm^{-1}, 1065 cm^{-1}, 878

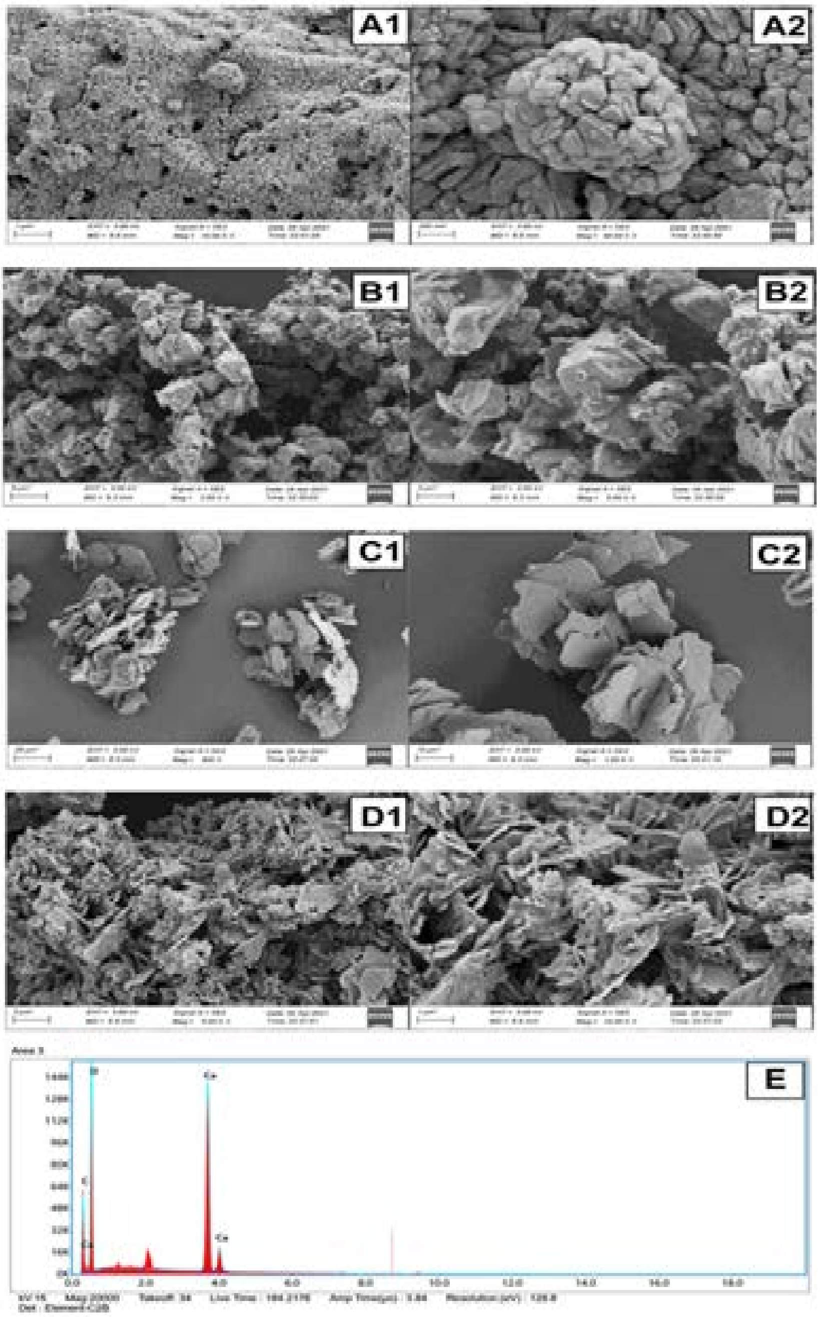

FIGURE 1: SEM images (A) Fresh CaO (B) CaDG (Calcium diglyceroxide) (C) CaG (Calcium glycerolate) and (D) CaO-E Calcium oxide from egg shell samples (E) EDX of CaO-E catalysts Birla et al., 2012.

cm^{-1}, and 529 cm^{-1}. However, a reduction in mass of the functional group attached to CO_3^{2-} ions contributed to the formation of CaO. Commercial CaO showed a good correspondence between spectral bands at 867 cm-1 and 1477 cm-1 and CaO synthesized from waste chicken eggshell due to the vibration modes of mono and bidentate carbonates. The calcined chicken eggshell band peak appeared at 3642 cm^{-1} versus the commercial CaO band at 3460 cm^{-1} (Awogbemi et al, 2020).

5.4.5 Brunauer-Emmett-Teller Analysis

Analysis of surface area and textural properties was performed using Brunauer-Emmett-Teller (BET). Calcined chicken eggshell displayed a higher BET surface area and external surface area compared to uncalcined samples. The external surface area of the eggshell was found to have a greater surface area than that of raw eggshell, likely due to the boiling and calcination of eggshell powders, leading to an increase in both BET and external surface area. The higher surface area promotes a higher catalytic activity of the eggshell powder for transesterification reaction catalyst and wastewater treatment (Tan et al., 2015). However, due to these attributes, CaO derived from eggshells displays better catalytic activity than commercial CaO. Anjana et al. (2016) reported that the high value of the pore radius and particle size of the eggshell powder is further pulverized from the current 75 μm and reduced to nano-size for improved catalytic activity (Anjana et al., 2016). This was credited to better surface exfoliation compared with CaO derived from waste eggshells (Badnore et al., 2018).

5.4.6 Other Characterization Analysis

Hammett method and temperature programmed desorption method (TPD) were used to analyze basicity, strength and site of the prepared catalysts. Hammett indicators (phenolphthalein, dinotroaniline, bromothymol blue, indigo carmine, nitroaniline) were used to measure basicity of eggshell catalysts. The basicity and basic strength of a catalyst play an essential role in the macrolactonisation reaction. Low basicity indicates strong stability of the catalyst; hence, it is assumed that the catalyst is resistant to air moisture. On the other hand, high basic strength and presence of a high number of basic sites indicate that the catalyst is able to moderately function at ambient temperature (Granados et al., 2007). The basic sites of standard CaO can absorb CO_2 and H_2O if exposed to the air (Furuta et al., 2006). Previous studies have proved that modifying different reaction parameters during the synthesis of eggshell catalysts improves the surface area; subsequently yielding larger basic sites (Granados et al., 2007; Liu et al., 2007; Xie & Li, 2006). A total basicity of eggshell catalyst after calcination was recorded as ranging from 2.8 to 3.5mmol/g. At the same time, basic strength ranged between $15.0 < H < 18.4$, similar to a commercial CaO catalyst (Xie & Li, 2006; Kim et al., 2010).

5.4.7 Current Opinion and Future Trends for Eggshell Catalysts

Current opinion and associated future trends relating to the use of eggshell waste in the development of reusable CaO catalysts point to the advantageous reproduction of these green catalysts using an economical process to yield value-added products. A series of different metals have been experimentally incorporated into the process to evaluate the potential use of eggshell waste catalysts as active CaO green catalysts for application in a wide range of scientific fields, including the environment, engineering, medicine, renewable energy, and catalysis. Additional studies should focus on tunable catalysts featuring active metal centers or sites for the development of an environmentally-friendly process based on the concept of green chemistry. Crucial general areas investigated in the literature can be categorized as important variables for the purpose of catalyst synthesis, characterization, and applications for different reactions. Eggshell catalysts have indeed proved effective in fields such as organic transformation, medicine, the environment, and manifold areas of science and engineering. Future studies should be undertaken to assess the feasibility of process commercialization, including physical characterization and optimization studies. The treatment of eggshell CaO catalysts with a series of different reagents and varied parameters should be investigated and pilot-scale studies set up.

6. CONCLUSIONS

This article illustrates the development of CaO catalysts from waste eggshells. The enormous quantities of eggs consumed by humans daily leads to the generation of tons of waste eggshells. CaO and nano CaO derived from eggshell waste are synthesized using a series of different methods and characterized by a range of distinct applications and socioeconomic properties of CaO. The fields indicated for potential applications include biodiesel production, drug synthesis, catalysts for organic transformations such as Knoevenegal reaction, environmental remediation, wastewater treatment, civil engineering (construction), cancer medicine and antimicrobial activity. The physical properties of CaO are studied and further characterized according to XRD, FT-IR, SEM, DSC-TGA and BET surface area. However, studies aimed at developing modified eggshell CaO catalysts should be undertaken to asseess potential novel applications suited to the tunable nature of eggshell CaO catalysts, and exceptional properties represented by their active sites, reusability, and stability for longer reaction times, irrespective of reaction parameters applied. The simple method of synthesis should indeed be further developed for use on an industrial scale as an economic and eco-friendly process. Based on this summary of the most critical and recent achievements obtained in studies of eggshell CaO catalysts, and with the aim of providing a comprehensive overview of the current state-of-the-art, the Authors maintain that this novel use of eggshell wastes may herald the widespread development of a promising heterogeneous catalyst.

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CITRUS WASTE FIBRES FOR NATURAL COSMETIC AND BIOPLASTIC PACKAGING

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ABSTRACT

In this work new biocomposite materials from wasted citrus peels have been validated for value-added packaging and new cosmetic products. Extracted natural fibres and dehydrated pulp from citrus fruit were combined with polylactic acid and other additives as materials for the production of bio-based packaging jars and cosmetic products. Orange powder (dehydrated waste) was found to be the best performing for the composition of packaging, while orange comminute, which results from milling orange peel revealed to be the most suitable for the composition of cosmetic formulations. Polylactic acid was compounded with citrus fibres by up to 25 wt.% by using twin screw extrusion and processed via injection moulding, one of the most widespread processing technologies for the production of rigid packaging containers. Composites were characterized and their mechanical, morphological and thermal properties were defined. Validation of packaging thermostability performance was performed by heat deflection temperature in compliance with ISO-75 and VICAT softening temperature in accordance with ISO-306. Compatibility tests of packaging demonstrators with newly formulated cosmetic products were researched as well. An accelerated compatibility test of the cosmetic jars with water- and oil-based simulants was performed at the ambient temperature, -5°C and 45°C. The results have indicated that new cosmetic packaging is not consistent with water-based cosmetic formulations yet is compatible with products based on natural oils. While the price of these products is generally higher than those using conventional plastics, they are competitive for premium cosmetic brands.

1. INTRODUCTION

Citrus fruits are considered to be the most popular and abundant crops worldwide. However, their industrial process and consumption produce a large amount of waste, especially peels, seeds, and pomace. Citrus processing waste normally occupies up to 60% of the total fresh fruit mass (peels, membranes, pomace, seeds), wastewater and semi-solid residues, such as centrifugation pulp that results from the extraction of juice. Approximately 10 million tons of citrus waste is produced each year (Zema et al., 2018) and, due to its large quantity and perishable nature, citrus by-products are considered as problematic waste. Citrus peels rot quickly, which attracts microbes, flies, mould, and results in the production of mycotoxins. Proper management of these wastes or their conversion into further process streams is, therefore, a necessity and not just an attempt to increase income (Berk, 2016). Proper disposal of citrus waste requires significant investment as unauthorised disposal leads to soil and water pollution, and destruction of the aquatic ecosystem (Zema et al., 2018). However, this type of waste can be also viewed as a potential bio-resource material for several application purposes. Many studies have already demonstrated that citrus processing waste contains numerous phytochemicals and bioactive components, such as polyphenols, vitamins, pectin, essential oils, fibres, terpenoids, etc. (Yun and Liu, 2022). The use of citrus processing waste is, therefore, effective as it aids in recycling valuable compounds and reduces environmental pollution.

In the past few years, citrus processing waste has shown good application potential in the food sector (Kumar et al. 2022, Yun and Liu, 2022, Iqbal et al. 2022, Suri et al. 2022), especially for the preparation of superfoods and nutraceuticals (Zhu et al., 2020). These bioactive components not only serve as functional ingredients, but also prevent the invasion of microbes, allergies, and illnesses (Babbar et al. 2011). Citrus waste usually becomes livestock feed, but it can also be used as a substrate for mushroom

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production, an accelerator for the preparation of compost, certain biofuels and bioethanol (Bernal-Vicente et al., 2008, Casquete et al., 2015). If combined with bioplastics, it can act as an ideal material for packaging applications (Yun and Liu, 2022; Peñalva et al., 2019; Žepič et al., 2016). The world of bioplastic packaging brings about a solution for ecological products by creating a more sustainable and greener world with a smaller environmental footprint. In combination with citrus peel powder, bioplastic materials can be shaped into jars, bottles, dispensing pumps, cosmetic packaging, and other innovative ideas that can be the next selling product on the market. Moreover, due to the presence of high-value compounds, such as essential oils and a unique fragrance, citrus processing waste proves its suitability for incorporation in cosmetic products (Pinto et al. 2021).

This paper investigates the use of citrus peel powder in novel biocomposite materials for value-added packaging and cosmetic products. Polylactic acid (PLA) was compounded with citrus powder in different ratios in order to valorise its application potential for cosmetic packaging. Composites were characterized for mechanical, morphological, and thermal properties, while the validation of packaging demonstrators was carried out via optical 3D scanning measurements, thermostability analyses and compatibility performance tests with cosmetic simulants and newly formulated cosmetic products.

The implementation of this study was possible with the cooperation of the industrial partner OLVETIA and the research institutions of TECOS and AITIIP, within the LIFE CITRUSPACK project (LIFE Citruspack, 2021), co-financed by the LIFE Programme, EU's funding instrument for the environment and climate action.

2. MATERIALS AND METHODS

2.1 Materials

Citrus waste powder (CWP) was supplied by AMC Innova Natural Drinks (Murcia, Spain), a subsidiary of AMC Group, Spain's third-largest international exporter in the food and beverage sector. CWP was extracted from the orange juice production process, blended in a certain percentage (trade secret), milled, and finally dried for further

TABLE 1: Sample composition for cosmetic jars (CJ).

Composite Sample	PLA [wt. %]	CWP [wt. %]	ATBC [wt. %]
CJ10	100	0	--
CJ11	80	15	5
CJ12	70	25	5

processing into composite materials. CWP was added to biodegradable polylactic acid (PLA) with the trade name IngeoTM Biopolymer 3251D by NatureWorks® LLC in different weight ratios, namely 0, 15 and 25 wt.%. Acetyl tributyl citrate (ATBC), supplied by Traquisa S.L. (Barcelona, Spain), was added as a plasticizer to improve the final performance of materials in a conventional extrusion compounding process (Coperion ZSK26 co-rotating twin-screw extruder, semi-industrial) at AITIIP facilities (Zaragoza, Spain). The compounded materials were afterward dried (Mini dryers Moretto X DRY AIR T) to ensure a low water level content that could, otherwise, negatively affect the product properties. The blend composition and sample designations used in this study are presented in Table 1.

2.2 Production of cosmetic packaging jars

Compounded materials were transformed into a three-part set of cosmetic packaging (Figure 1), assembled by an inner insert, an outer case, and a lid, with an injection moulding machine (KraussMaffei 80/380 CX) with newly designed and manufactured injection moulds. For compatibility tests with cosmetic simulants and final products, 50 prototype jars were injected, of which 25 inner inserts were made of pure PLA and 25 were made of CJ11 formulation. Specimens for mechanical testing were obtained by injection moulding with the Fanuc Roboshot ALPHA S50IA/330/IT electric injection machine (Tokyo, Japan) following the ISO 527 standard.

2.3 Characterization analyses

2.3.1 Tensile testing

Tensile tests using Zwick Roell Z 2.5 (Zwick GmbH & Co. KG, Ulm, Germany) were carried out in compliance with ISO 527-1-2 under ambient conditions. The distance be-

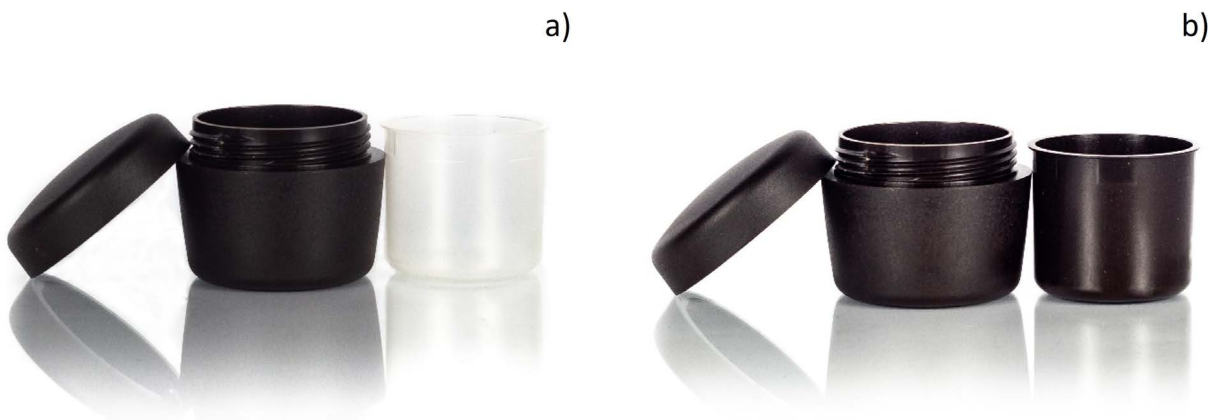

FIGURE 1: Cosmetic Jars with (a) plain PLA insert, and (b) CJ11 composite variation.

tween the grips was set to 60 mm. The presented values of elasticity modulus (E_t), tensile strength (σ_M), and elongation at break (ε_b) are an average from at least six replicates from each tested material.

2.3.2 Scanning electron microscopy

The morphology of the composite fractured surfaces was examined after a tensile test using the Hitachi S3400N (Tokyo, Japan) scanning electron microscope (SEM). An operating voltage of 10 kV and magnifications of 500x and 50x were used.

2.3.3 Thermogravimetric analysis

Thermal characterisation was carried out by using thermogravimetric analysis (Mettler-Toledo TGA/SDTA 851e, Greifensee, Switzerland). The samples were heated from 20 to 1000°C at 20°C/min under N_2 atmosphere (100 mL/min).

2.3.4 Optical 3D measurement

A high-resolution optical 3D scanner (ATOS CS 5M; GOM, Braunschweig, Germany) was used to explore the surface quality of cosmetic jars. The initial step in the optical 3D scanning process involved conducting a calibration procedure to ensure the dimensional consistency of the measuring volume. To minimize the scan noise and surface gloss, a 2 µm thick layer of an anti-reflective TiO_2 coating spray was manually applied on the jars' surface jointly with adhesive reference markers. Each jar was then mounted on the rotating table and scanned at a resolution of 5 megapixels (MP) in 20 positions through 360° with a working distance of 590 mm. A dedicated software (GOM Inspect Suite 2020; GOM, Braunschweig, Germany) was used to automatically obtain a triangulated mesh in STL format from the point cloud.

2.3.5 Heat deflection temperature

Heat deflection temperature (HDT) was determined according to ISO 75-2:2013 with the Zwick/Roell HDT/Vicat 4-300 (Zwick GmbH & Co. KG, Ulm, Germany), operating at a voltage of 0.45 MPa and a heating rate of 120°C h^{-1} (method B). The temperature was determined after the specimen deflecting 0.33 mm. Before testing, specimens were preconditioned for 88 hours at (23 ± 2)°C and (50 ± 10)% rh. The presented results are an average of three conducted experiments.

2.3.6 Vicat softening temperature

Vicat softening temperature (VST) experiment was performed according to ISO 306: 2013. The examination was accomplished according to the test method A50 (10 N; 50°C/h) in silicone oil. The samples were stored in standard climate 23/50-2 (DIN EN ISO 291: 2008-05) for a minimum of 24 hours before the examination.

2.3.7 Compatibility testing

Different cosmetic simulants were tested with cosmetic jars and stored for four months at three different temperature zones: -5°C, 25°C and 45°C. The compatibility tests were conducted with the inserts made of plain PLA and a newly developed composite (formulation CJ11) with five different simulants, namely paraffin oil, caprylic/capric triglyceride, and water samples with 4, 7 and 10 pH values. The water solutions were prepared by adding citric acid to water and adjusting pH by a solution of sodium hydroxide. All the solutions contained 1% phenoxyethanol as a preservative. pH values of the simulant samples were measured again after 14 days and compared to the initial conditions. Appearance, odour, and mass change of the simulated sample were evaluated after four months. Subsequently, compatibility tests of the cosmetic jars with three newly formulated cosmetic products based on citrus waste were carried out to validate their suitability for cosmetic packaging application.

3. RESULTS AND DISCUSSION

3.1 Mechanical properties

The effect of citrus waste powder on the tensile modulus (E_t), tensile strength (σ_M) and elongation at break (ε_b) is shown in Table 2. The results indicate that the addition of a CWP filler has a positive effect on Young's modulus, which is a vital factor for illustrating the stiffness of the materials under tensile loading. The elevated values for Young's modulus of composite formulations C11 and C12 were 3599 MPa and 3819 MPa respectively, which means a 4% and 9% improvement of the E_t value when compared with plain PLA. Nevertheless, these are only marginal enhancements that indicate a good filler distribution across the polymer matrix, but not the desired reinforcement potential.

The last statement coincides with the tensile strength results, showing that CWP addition has a negative effect on this property. The σ_M value falls to 46,15 MPa from 58.13 MPa when the percentage of a filler is 15% and the drop is further pronounced when the percentage of CWP is increased to 25%. It is well stated in the literature that the tensile strength of polymer composites depends on interfacial adhesion (Li et al. 2007). The cause for poor adhesion can also be found in the non-modified surface of CWP, as the surface charge treatment of fillers usually results in improving the final material properties (Fu et al. 2008). On the other hand, CWP particles in the PLA matrix may act as sites for stress concentration, thereby leading to the failure of the biocomposites. Elongation at break was also lower for composite samples compared to plain PLA, indicating extremely brittle materials.

Overall, the mechanical characterisation demonstrated that the filler-matrix interaction in composites produced in the present study followed general trends universally observed for particulate fillers, rather than fibrous elements (Fu et al. 2008). Due to the tendency of citrus residues in water blends to collapse during spray drying protocols,

TABLE 2: Tensile results.

Sample	E_t (MPa)	σ_M (MPa)	ε_b (%)
CJ10	3456,10 ± 145,95	58,13 ± 1,63	2,72 ± 0,12
CJ11	3599,20 ± 209,58	46,15 ± 1,01	1,78 ± 0,06
CJ12	3819,84 ± 172,10	42,81 ± 0,23	1,45 ± 0,03

their geometry is particulate, rather than fibrous (Fig. 2c). As elucidated by Fu et al. (2008), this particulate geometry entails an increase in stiffness, a decrease in strength, and an improvement in toughness.

3.2 Morphology analysis

The fractured surfaces of samples tested under tensile load were observed under SEM. It was found that the voids due to air entrapment are minimum in PLA matrix (Figure 2a), whereas the number of voids kept on increasing with the addition of CWP. Pure PLA sample exhibited large smooth surfaces without any irregularities (Fig. 2a), indicating a brittle fracture behaviour (Todo and Takayama, 2011). The addition of CWP filler dramatically changed the appearance of the fracture surface, which became rougher (Figs. 2b, c and d). Furthermore, the morphological structure of CWP is clearly visible in Figure 2c and can be clearly correleated to the spherical particulates. Higher magnifications also provide an evidence of a weak filler matrix adhesion (Fig. 2c and d) that leads to poor mechanical properties as discussed in the previous section. This lower affinity between the hydrophobic matrix and the hydrophilic filler leads to a decrease in the strength properties of the resulting composites. The agglomeration of CWP particles can also be a reason for poor mechanical properties. Frequently, fibre pull-out and cavities created by debonding of CWP from the surrounding PLA matrix were observed (Fig. 2c). This fracture mechanism is common for fibre reinforced composites under tensile load (Faludi et al. 2014). It is an indication of relatively poor surface adhesion between the CWP and the PLA matrix. As local stress concentration is occurring at these defects, this explains the reduced strength of the PLA/CWP composites.

TABLE 3: Thermogravimetric analysis results.

Sample	T_{onset} [°C]	$T_{50\%}$ [°C]	T_{max} [°C]	FR [%wt]
CJ10	379	433	466	1,26
CJ11	257	402	661	4,23
CJ12	198	391	749	6,47

3.3 Thermogravimetric analysis

The TGA results can be analysed in different phases to recognize the degradation behaviour of the composite materials. The results of the thermogravimetric analysis are presented in Table 3. Initially, the weight loss observed below 285°C corresponds to the thermal decomposition of the polymer chains in plain PLA, where a random scission of ester bonds occurs, resulting in the release of low molecular weight fragments. At approximately 350°C, a significant weight loss indicates a secondary degradation stage, involving further breakdown of residual fragments and the elimination of volatile species, like carbon dioxide, water, and small organic molecules (Peñalva et al. 2019). The presence of CWP particles influences the thermal behaviour of PLA (Table 3). The addition of citrus waste powder led to a lower onset of decomposition temperature (T_{onset}),

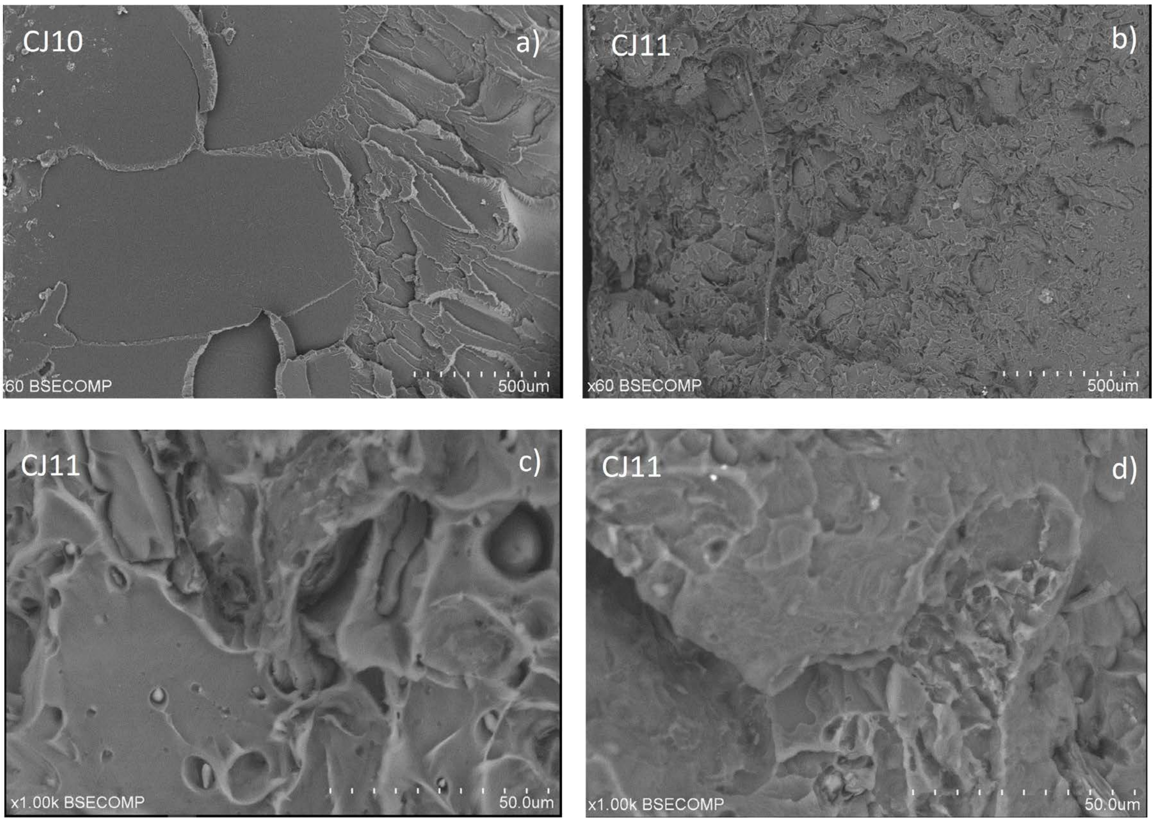

FIGURE 2: SEM images of (a) PLA sample (CJ10) (500 µm), (b) composite sample with 15% CWP (CJ11) at 500 µm magnification, (c) and (d) composite sample with 15% CWP (CJ11) at 50 µm magnification.

which was pronounced with raised CWP contents. This indicates that samples with 25 wt.% CWP show the lowest thermal stability. On the contrary, T_{max}, which is a temperature where the maximum mass loss rate occurs, was found to be increased with elevated CWP contents. The same observation was made for the mineral content and non-volatile fraction of the samples. In addition, the results of the final residue (FR) were consistent with the amount of CWP added to the samples; the higher the amount of CWP, the higher the percentage of FR.

3.4 Optical 3D-scanning and geometrical analysis

3D scanning is one of the important technologies used for designing, inspection, and quality control and it is becoming an essential tool for producers who need an accurate dimensional inspection of their final products (Javaid et al., 2021). This non-contact measuring technology converts a physical model into a digital 3D Computer-Aided Design (CAD) with the help of paired scanning software. To confirm the dimensional compliance, 3D scanning measurements were performed, involving 20 scans (14 on the upper and 6 on the bottom side) captured from various angles (Figure 3a). These scans were aligned using the reference markers and the resulting point cloud was polygonised to create a hole-free mesh in the .stl format for further inspection (Figure 3b). Subsequently, the mesh was refined, including smoothing and thinning, to improve its quality and precision. In the last step, best-fit alignment was used to align the mesh (actual data) to the nominal CAD data (Figure 3c). With the ATOS inspection module, a "surface comparison on actual" analysis was performed, involving the quantifying of positional deviations between the nominal and actual surfaces. By employing the deviation labels (points), multiple positions were validated. The results indicated that the dimensional aspects in all spatial directions (x, y, z) fall within a 0.1 mm tolerance on the labelled surface and a 0.3 mm tolerance on the helix surface (Figure 4). This outcome signifies that the injection mould

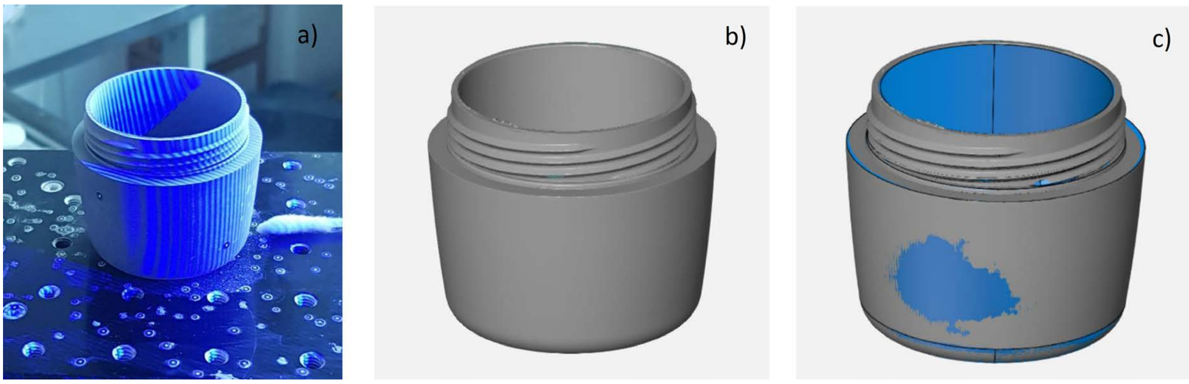

FIGURE 3: Cosmetic jars demonstrators (a) during the 3D scanning procedure; (b) polygonised mesh prepared for measurements (.stl file); and (c) local best-fit alignment between the nominal (blue) and actual (grey) data (c).

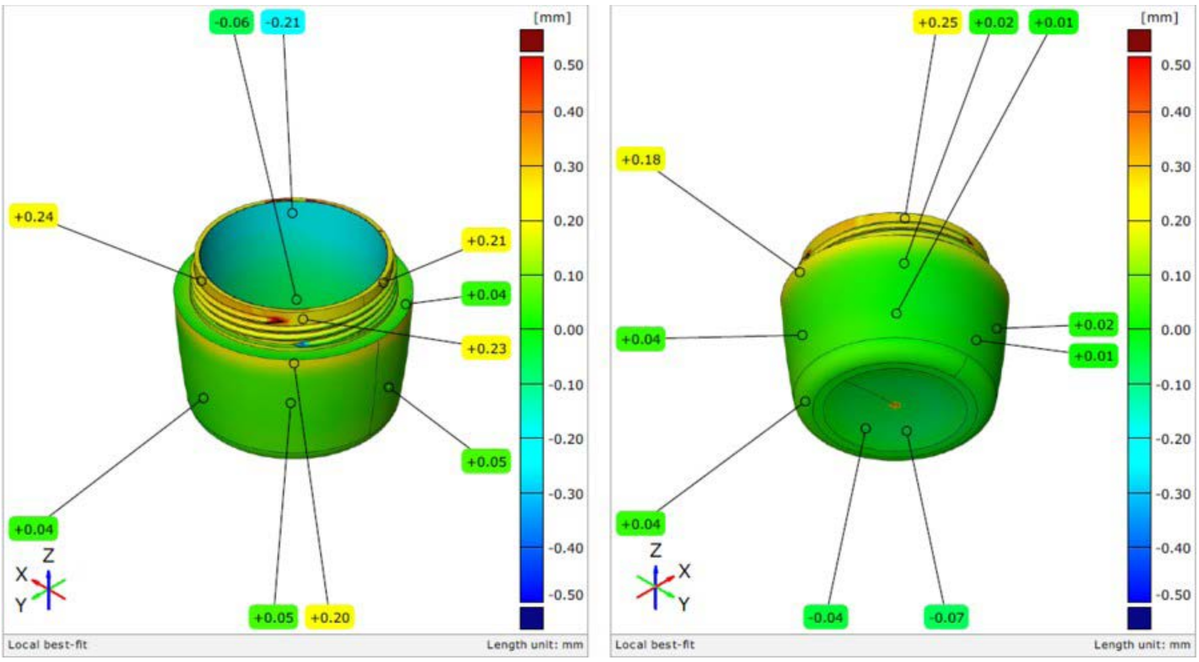

FIGURE 4: Surface comparison on actual CJ specimens with deviation labels.

was manufactured and installed with precision well within the specified tolerances and that the specimens' injection was moulded with appropriate processing parameters, thereby ensuring its operational functionality.

3.5 Thermostability analysis of cosmetic packaging jars

The dimensional stability of PLA is a big concern for its application under high-temperature conditions. Thus, heat deflection temperature (HDT) was employed as a technical requirement by packaging product designers (Narendar et. al., 2018). The HDT measurements of CWP composites are shown in Figure 5. The HDT value of plain PLA is around 55°C. After incorporating the CWP filler, the heat deflection temperature gradually decreased to 45.1°C and 44.8°C for the CJ11 and CJ12 samples respectively.

Vicat softening temperature (VST) is the temperature at which a sample is penetrated to a depth of 1 mm by a tip flat needle with a square or circular cross-section of 1 mm^2. This is an important measure used to characterize a material's thermomechanical strength (He et al. 2015). It can be a useful property for quantifying the filler effects in polymers. Similarly, to HDT, it is important in technical terms during the shaping and projection of industrial applications. The VST of the CWP composites are shown in Figure 6. The VST of plain PLA is around 56.2°C according to the manufacturer data provided (Natureworks LCC). With the CWP addition, the VST of the composites decreased to 51.8°C and 50.9°C for CJ11 and CJ12 respectively. These results are consistent with the TGA results, which also implies the degraded thermal stability of the composites with citrus waste powder. Safe application for the resulting

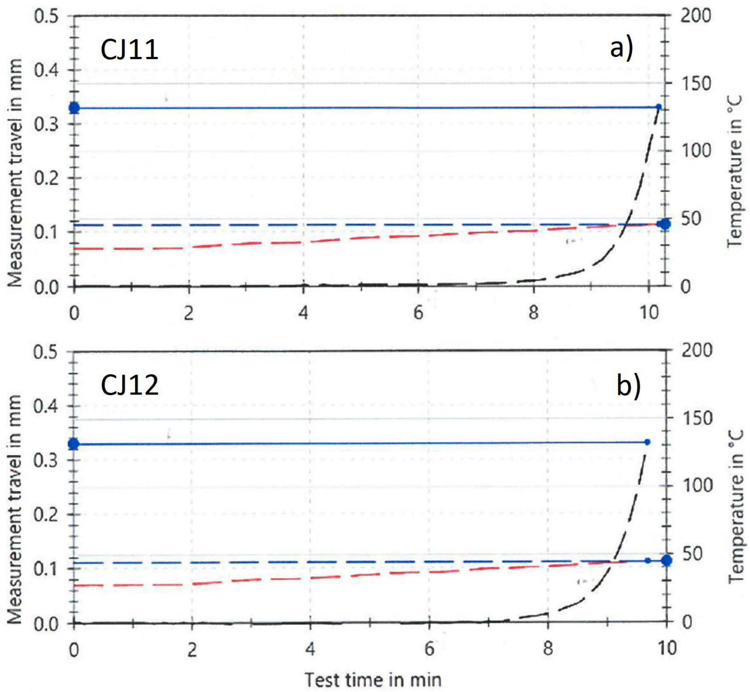

FIGURE 5: HDT diagram as a function of deflection/temperature vs. testing time for (a) composite sample with 15% CWP (CJ11), and (b) composite sample with 25% CWP (CJ12).

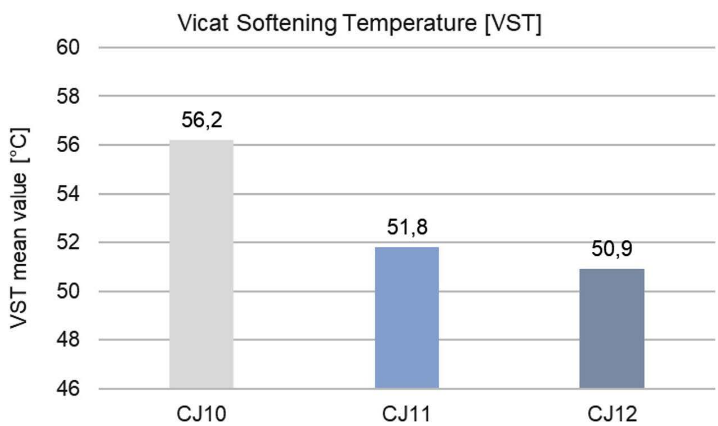

FIGURE 6: Vicat softening temperature for plain PLA and composite samples.

composites without a deformable phenomenon is, therefore, feasible at temperatures below 45°C.

3.6 Compatibility testing of cosmetic packaging

An accelerated (room temperature, freezer, and oven at 45°C) compatibility test of cosmetic jars with different simulants (aqueous solutions of different pH values and oils of different polarity) was performed to evaluate the materials' suitability for cosmetic packaging applications. The performance results of cosmetic jars are shown in Table 4. Compatibility tests were validated with inserts made of plain PLA and the composite sample CJ11.

After monitoring various parameters (pH, weight change, appearance, swelling, odour, colour), compatibility results of cosmetic jars and cosmetic products revealed that newly formulated composites are not compatible with water-based cosmetic simulants, as the brown colour migrates into the cosmetic product, which can be attributed to the hydrolytic effect of the PLA matrix (Dreier et al. 2021). Moreover, composite jars are not resistant to chemical ingredients in cosmetic formulations based on aqueous solutions since the inserts started to show cracks and fractures at low pH values (pH = 4) and elevated temperatures (T = 45°C), The maximum change in mass loss was observed in the composite samples with water-based solutions of pH 7 for all three tested temperature zones (Table 4). On the contrary, the jars made of CWP composites are validated as appropriate for oil-based cosmetic products, which was also indicated by the results of accelerated tests with oil-based media simulants (paraffin oil or caprylic/capric triglyceride) where no or minor changes were noted in terms of appearance, odour, or mass change, even under the most stringent conditions.

Simultaneously, compatibility tests of the cosmetic jars with three newly developed cosmetic formulations based on the citrus waste streams from the AMC's juice production plant were tested. The first product developed was a facial scrub (O/W scrub) presented in Figure 7a, containing 4% pomegranate peel, 0.05% orange essential oil as a perfume, and 1% orange fruit powder for improving the antioxidant and hydrating properties. The second product developed was an eye serum (water-based gel) with 0.5% orange fruit powder and 0.05% mandarin essential oil (Figure 7b). the third cosmetic product developed was a facial mask with kaolin clay for a smooth and clean skin sensation with 0.3% orange essential oil as a perfume and 0.5% orange fruit powder (Figure 7c).

Essential oils of orange, lemon and mandarin citrus waste constituents were found to be suitable for the use of aromatic ingredients in cosmetic products, as they provide a very pleasant citrus fragrance. Orange fruit powder and orange water peel extract were validated as compatible and may be used in various cosmetic formulations. At a high percentage (~ 10 wt.%), they may cause skin irritation depending on the product composition; a measure of caution is, therefore, necessary when using these extracts. Pomegranate seed powder, which was also validated as a potential scrub filler, was found to have the right granulometry for a face scrub, however, due to its strong and unpleasant smell, the use in cosmetic formulations for face masks is not recommended. On the other hand, pomegranate fruit cuts have a pleasant smell and are appropriate for use in body scrubs, which was also exploited in the formulation of the first cosmetic product based on CWP. All products were developed to comply with the COSMOS natural cosmetics standard.

TABLE 4: Compatibility test of cosmetic jars with simulants from water to oily mediums performed at -5°C, ambient temperature, and 45°C. Odour, appearance and mass change evaluated after four months.

Temp. Condition	Simulant	pH value				Appearance		Odour Change		Mass Change	
		C10		CJ11		CJ10	CJ11	CJ10	CJ11	CJ10	CJ11
		0 day	14 days	0 day	14 days						
T= -5°C	Paraffin oil					N	C	N	N	+1.2%	+1.0%
	Caprylic/capric triglyceride					N	C	N	N	+1.2%	+ 0.9%
	Water (pH=4)	4.05	4.36	4.05	4.05	N	S, C	N	S	+1.3%	+1.3%
	Water (pH=7)	6.94	7.02	6.94	4.94	N	S, C	N	S	+1.3%	+ 2.2%
	Water (pH=10)	10.01	10.00	10.01	7.48	N	S, C	N	S	+1.3%	+1.3%
T= 25°C	Paraffin oil					N	N	N	N	+ 0.7%	+ 1.3%
	Caprylic/capric triglyceride					N	N	N	N	+ 1.4%	+ 0.9%
	Water (pH=4)	4.05	3.99	4.05	8.67	M	S, C	N	S	+ 2.6%	+ 4.6%
	Water (pH=7)	6.94	6.33	6.94	7.10	M	S, C	N	S	+ 2.7%	+ 4.9%
	Water (pH=10)	10.01	6.79	10.01	9.10	M	S, C	N	S	+ 3.3%	+ 4.7%
T= 45°C	Paraffin oil					N	N	N	M	+ 1.3%	+ 0.1%
	Caprylic/capric triglyceride					N	N	N	M	+ 1.0%	+ 0.7%
	Water (pH=4)	4.05	3.66	4.05	3.35	S, C	S, C	N	S	+ 5.0%	+ 3.6%
	Water (pH=7)	6.94	6.30	6.94	3.16	S, C	S, C	N	S	+ 2.9%	+ 7.6%
	Water (pH=10)	10.01	4.16	10.01	3.44	S, C	S, C	N	S	+ 3.9%	+ 1.6%

N – no changes; M – minor change; S – substantial change; C – colour change

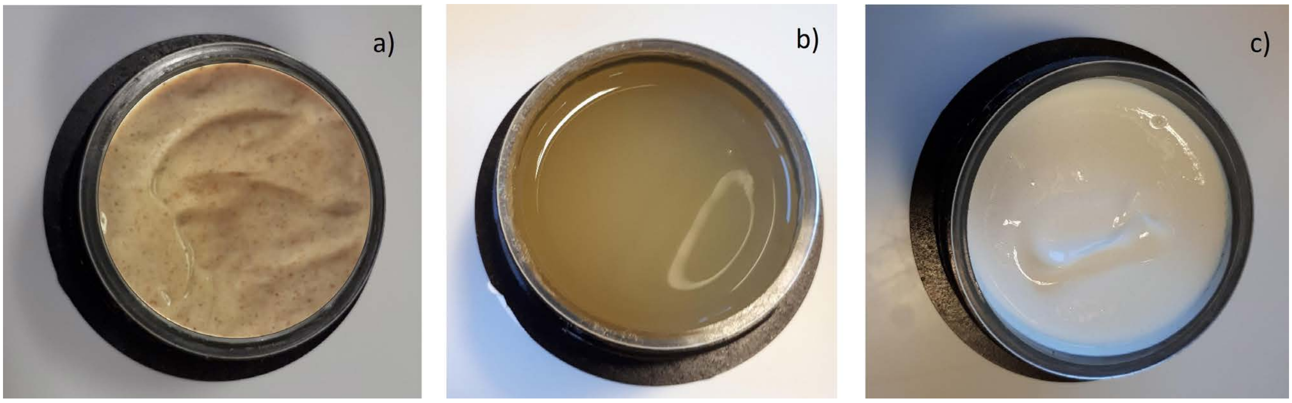

FIGURE 7: Photomacrographs of newly developed cosmetic products based on CWP: a) facial scrub, b) eye serum, and c) facial mask with kaolin clay.

4. CONCLUSIONS

This study has validated the utilization of citrus fruit waste in PLA composites for packaging and cosmetic applications. It has shown that such a secondary resource can be an interesting substitute for a certain plastic percentage in biobased sustainable packaging. Moreover, it can serve as a valuable ingredient in novel cosmetic formulations for antioxidant and hydrating properties. The results presented in this work show a good filler distribution across the biopolymer matrix, but not the desired reinforcement potential. The reinforcing potential is mainly attributed to two effects: (i) the formation of a physically bonded flexible filler network and (ii) strong polymer–filler couplings. Both of these effects arise from a high surface activity and the specific surface of the filler particles. Considering the aggregate morphology and surface structure of CWP fillers after drying the strengthening potential is drastically reduced and such additives serves only as fillers for the natural aesthetics of the final products, or a cheaper phase in the final composite materials. The negative impact on the mechanical properties of the resulting composites could be due to the fact that citrus waste filler was not given any kind of treatment, as a surface modification usually results in improving the final material properties. Poor interfacial adhesion was further proved with morphology results, indicating a weak affinity between the hydrophobic matrix and the hydrophilic filler that leads to decreased strength properties of the resulting composites. The addition of citrus waste powder led to a lower onset of decomposition temperature which was pronounced with increased filler contents. The results of the cosmetic jars' heat deflection and Vicat softening temperature were consistent with the TGA results, which also implies the degraded thermal stability of the composites with citrus waste powder. Safe application of the cosmetic packaging jars made of new composite materials without a deformable phenomenon was, therefore, feasible at temperatures below 45°C. Although newly formulated composites have been found to be incompatible with water-based cosmetic ingredients, they are appropriate for oil-based cosmetic formulations. Overall, the results indicate that the incorporation of citrus waste powder as a filler in PLA composite cosmetic packaging can be a market-appealing advantage. However, in order to meet safety conditions when in contact with cosmetic products, the use of inserts made of pure PLA in operating conditions between -5 and 45°C is strongly advised.

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LIFE CYCLE ASSESSMENT OF SCENARIOS FOR END-OF-LIFE MANAGEMENT OF LITHIUM-ION BATTERIES FROM SMARTPHONES AND LAPTOPS

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ABSTRACT
Recycling lithium-ion batteries (LIBs) is a solution to minimise the environmental problems caused by the consumption of natural resources and the generation of hazardous waste. This paper aims to assess the potential environmental impacts and benefits of four scenarios for recycling LIBs from smartphones and laptops using Life Cycle Assessment (LCA). The methodological approach followed four steps: i) scenario modelling representing the current and future situations of LIBs End-of-Life (EoL) management from smartphones and laptops; ii) estimating smartphones, laptops and respective LIBs waste generation; iii) mapping representative recycling options; and iv) assessment of potential environmental impacts using LCA with 16 ILCD midpoint categories. The results revealed that hydrometallurgical recycling in Brazil could be less harmful than pyrohydrometallurgical recycling in Europe in 12 impact categories. The benefits of recycling are mainly of Co and Ni recovery. Results of scenarios indicate that the more optimistic scenario, which includes expanding Reverse Logistics to 50% of collection, internal recycling to 75%, and reducing of LIBs waste sent to landfills in 44%, had the best environmental performance in all 13 impacts categories. For the Climate change category, scenario 4 presents net environmental benefits of -1.83E+05 kgCO_2eq while scenarios 1, 2 and 3 do not present a net environmental benefit. Scenarios assessment shows that more significant environmental benefits are achieved when the formal collection rate is increased, and the less impactful technology option makes the recovery of materials. These results can help decision-makers promote the management and recycling more sustainable of LIBs waste.

1. INTRODUCTION

The increasing use of Lithium-ion batteries (LIBs) generates thousands of tons of LIBs waste per year (Larouche et al., 2020; Meshram et al., 2020). LIBs waste is one of the fastest-growing segments in global solid waste streams (Song et al., 2017). The flow of Waste Electric and Electronic Equipment (WEEE) contains large amounts of LIBs because portable devices, such as smartphones, tablets and laptops, attach a battery to supply energy (Song et al., 2019; Mejame et al., 2020). Estimates indicate a 30% growth in demand for LIBs annually by 2030 (Hanicke et al., 2023). This increase is driven by rising demand for portable electronics and other applications that utilize LIBs, such as Electric Vehicles (EV) and energy storage (Bauer et al., 2022; Fahimi et al., 2022). As sales increase, larger volumes of LIBs waste will enter the solid waste stream (Boyden et al., 2016; Mejame et al., 2020). This scenario raises concerns about the environmental impacts of consuming natural resources to produce LIBs and and the management of these hazardous wastes at the End-of-Life (EoL) stage (Or et al., 2020; Xu et al., 2020; Lybbert et al., 2021).

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The production of LIBs requires various metallic raw materials and chemicals (Zubi et al., 2018), including cobalt, lithium, nickel and graphite. Improper EoL disposal of LIBs must be avoided to prevent adverse effects on human health and the environment (Winslow et al., 2018; Mrozik et al., 2021). LIBs discarded with Municipal Solid Waste (MSW) can cause fires in collection vehicles and landfills (Herreras-Martínez et al., 2021). Electrolytes can lead to contamination with hazardous substances such as arsenic and phosphorus (Jin et al., 2022). Improper treatment results in the loss of valuable resources (Meshram et al., 2020). Beyond direct environmental impacts, the LIBs supply chain also has serious social consequences. Cobalt mining in the Democratic Republic of Congo involves human rights violations, including forced labor and child labor (Amnesty International, 2017; Harper et al., 2019). Lithium production in Latin America negatively affects local communities' access to drinking water, agriculture, and fishing, leading to human and indigenous rights violations (Henríquez, 2018; Schüler et al., 2018).

LIBs recycling offers significant potential for generating economic, environmental, and social value (Dai et al., 2019). Environmentally, recycling reduces waste and pollution, conserves natural resources and ecosystems while increasing resource efficiency (Harper et al., 2019). Socially and economically, recycling strengthens the economy by creating new jobs, wages, and tax income (EPA, 2020), and it also presents profitable business opportunities (Latini et al., 2022; Lima et al., 2022). The recovery of raw materials from LIBs prevents environmental pollution and resource loss, provides a secondary supply of materials, and contributes to Circular Economy and Sustainable Development (Ducoli et al., 2022; Fahimi et al., 2022).

The technologies for recycling LIBs are classified into three types (Harper et al., 2019; Jin et al., 2022): hydrometallurgical, pyrometallurgical and direct recycling. Pyrometallurgical recycling uses high-temperature thermal processing, including smelting, pyrolysis, incineration or torrefaction/calcination to reduce metal oxides in metal alloys (Or et al., 2020; Makuza et al., 2021). Hydrometallurgical recycling involves a chemical process that uses aqueous solutions with chemical reagents to leach metals from the cathode material (Yun et al., 2018; Raj et al., 2022). Direct, or functional, recycling reconditions and recovers negative and positive electrode materials for reuse in LIBs, avoiding the separation and dissolution of multiple metallic ions in their elemental form (Harper et al., 2019; Kim et al., 2021). While some studies indicate an environmental and economic advantage of hydrometallurgy compared to pyrometallurgy (e.g., Dunn et al., 2022; Jin et al., 2022; Chen et al., 2023), there is still no definitive consensus on the environmentally preferred recycling method (Rey et al., 2021; Rajaeifar et al., 2022).

Recycling technologies for LIBs waste are in the early stages of development and commercialization (Rhee, Jang & Kim, 2021; Rajaeifar et al., 2022). The LIBs recycling chain encounters political, technical, and socioeconomic challenges globally, particularly in developing countries, such as a lack of commercial recycling infrastructure, disposal points, and adequate collection vehicles (Slattery et al., 2021). Specific guidelines and legislation for EoL LIBs management are lacking (Duarte Castro et al., 2021; Cabral-Neto et al., 2022). Many LIBs present in WEEE are poorly managed, improperly disposed of with MSW, or stored in homes (Duarte Castro et al., 2022). There is a need for more assessments of the impacts of LIBs recycling processes, considering a broad set of environmental impact categories, to avoid damage to the environment and human health (Rajaeifar et al., 2022).

Life Cycle Assessment (LCA) is an efficient technique to analysing potential environmental impacts from a system perspective (Souza et al., 2016; Zhao et al., 2021). LCA has been widely used to assess the environmental implications of waste management systems, including WEEE management systems (Torkayesh et al., 2022; Xavier et al., 2023). LCA allows for assessing the level of impact of the waste management system, from the generation stage to final disposal (Wang et al., 2022). By modelling scenarios in LCA, future implications of waste systems can be assessed, describing the impacts of different system options in advance (Villares et al., 2017; Tsoy et al., 2020). LCA plays a crucial role in supporting the development of a Circular Economy by providing a robust assessment of the impacts of circular solutions and enabling the understanding and minimization of impacts from a holistic view (Ingemarsdotter & Dumont, 2022). However, there are limited investigations of the environmental impacts of recycling LIBs using LCA (Ferrara et al., 2021; Arshad et al., 2022; Rajaeifar et al., 2022). Supplementary material 1 provides a summary of LCA studies on LIBs recycling. In summary analysis, existing studies that evaluate the recycling of LIBs through LCA are more focused on LIBs waste from EV, using pyrometallurgical and hydrometallurgical technologies in the context of China and the USA, lacking LCA studies in developing countries. Besides, data from inventories are lacking, and the studies have limited coverage of emissions and potential environmental impacts of LIBs recycling (Rajaeifar et al., 2022).

Recent LCA studies have been conducted to assess the impacts and benefits of various LIBs recycling processes. For instance, there are studies on hydrometallurgical (Dunn et al., 2022; Cao et al., 2023), pyrometallurgical (Rajaeifar et al., 2021; Koroma et al., 2022), and direct recycling (Jiang et al., 2022). However, these studies focus primarily on the direct impacts of unit processes and avoided impacts of materials recovered by each recycling technology, considering only processes for recycling and avoided production within of the system boundary. There is a notable absence of studies evaluating the EoL of LIBs in WEEE from a systemic perspective. Specifically, there is a lack of assessment regarding the impacts of implementing Reverse Logistics (RL) policies with different collection and recycling routes, as well as the effects of disposal in landfills using a functional unit based on generation of LIBs waste per year. There is a lack of LCA studies focused on the number of LIBs present in WEEE streams and the respective impacts of recycling or disposal. Data from the amount of LIBs waste generated per year in WEEE streams and inventories of LIBs EoL beyond of recycling process are lacking. These gaps in research underscore the need for more comprehensive LCA studies that adopt a holistic approach to assess the environmental impacts of various LIBs recycling and disposal methods.

This article aims to evaluate the potential impacts and environmental benefits (avoided impacts) of different scenarios for the recycling of LIBs, adopting the EoL context of smartphones and laptops. For this purpose, this article estimates the generation of LIBs waste from these WEEE streams in the period of 2022-2026; models a potential LIBs recycling process to be implemented in the country; assesses four progressive scenarios of RL implementation, according to annual targets in policies; identifies environmental hotspots to be tackled by the implementation of LIBs recycling capacities. No study has evaluated the EoL of LIBs in WEEE considering a systemic view of the impacts of implementing Reverse Logistics (RL) policies with different collection and recycling routes, disposal in landfills and using a functional unit based on generation of LIBs waste per year. There is a lack of LCA studies focused on the number of LIBs present in WEEE streams and the respective impacts of recycling or disposal. Data from the amount of LIBs waste generated per year in WEEE streams and inventories of LIBs EoL beyond of recycling process are lacking.

2. MATERIALS AND METHODS

Four macro methodological steps were performed in this study:

1) Scenario modelling to represent the current and future situation of LIBs EoL management in WEEE streams. Data from Brazil is used for building scenarios and exemplifiying methodological steps. Scenario modelling is building according to WEEE RL implementation targets defined by Law (Brasil, 2020, which establishes the WEEE Reverse Logistics Sectorial Agreement). This WEEE RL implementation targets policy is used to capture the amount (in the percentage) of WEEE collection per year in the annual implementation plans to verify the potential amount of collection through formal channels. It is important to note that due to a lack of reliable data, internal logistics (transporting the collection and sending it to the pre-treatment recycling plant in Brazil) was not included in the main assessment, but to address possible transport impacts, intermediate transport from each city included in the RL plan to the pretreatment city was analysed through a sensitivity analysis. WEEE RL implementation was used to capture the amount (rates) collected by RL that could be destined for recycling. In addition, data on landfill distances in Brazil and Europe were also not included in the model due to a lack of data.
2) Estimate of smartphones, laptops and LIBs waste generation for 2022 to 2026 using trend analysis in time series and lifespan distribution;
3) Mapping and modelling of a hydrometallurgical recycling process. The hydrometallurgical process was chosen for three main reasons: i) it is the type of process in which pilot tests are being carried out for implementation in Brazil; ii) the hydrometallurgical process has the potential to generate less environmental impact because it uses less energy and recovers a greater number of materials (Dunn et al., 2022; Jin et al., 2022); iii) hydrometallurgical recycling plants are less complex in terms of costs to installation and processing compared to pyrometallurgical recycling (Gaines et al., 2018; Jin et al., 2022; Chen et al., 2023).
4) Assessment of potential environmental impacts of recycling processes and scenarios using LCA.

2.1 Scenario modelling for LIBs recycling in Brazil

Four progressive scenarios for recycling LIBs in Brazil were modelled in line with the Brazilian National Policy on Solid Waste (Brasil, 2010), which determines the progressive increase in WEEE RL (including LIBs) and the reduction in landfilling. The construction of the scenarios aimed to provide a more in-depth view of the paths that WEEE takes to obtain secondary raw materials, quantifying the impacts and benefits (avoided impacts of raw materials production) of different routes and recovery rates in Brazil. The RL collection rates for the first 3 scenarios were derived from the Implementation Schedule of the WEEE RL Sectoral Agreement (Brasil, 2020), which defines percentage targets for collection and destination for each year: 2023-6%, 2024-12% and 2025-17%. As the collection targets for the year 2026 and subsequent years are not defined in the implementation plan, for scenario 4 (2026), which is considered more optimistic, the collection rate by the LR was increased to 50% of the WEEE generated (laptops and smartphones) to verify the environmental implications of a more efficient collection system. The progressive approach increases RL and decreases LIBs waste collected with MSW and informally and destined for landfills, a common practice in WEEE management in Brazil (Souza, 2020; Xavier et al., 2021). The scenarios present variations in the progressive increase in the collection rate by LR and in the recycling that takes place in Brazil, progressively reducing the sending of materials to landfill and the quantity sent to be recycled in Europe.

Scenario 1 - Pessimistic: reflects the baseline (current) scenario, in which the fraction collected by the RL is lower than that collected informally or mixed with MSW. In this scenario, there are no formal recycling operations for LIBs within Brazil, where recycling activities are limited to disassembly and separation, and the more complex components are exported for recycling abroad, mainly in Europe and Asia (Dias et al., 2018; Duarte Castro et al., 2021). Of the fraction of LIBs collected in smartphones and laptops by formal RL in 2023 (6% according to the Brazilian law) 50% are assumed to be destined for pyrohydrometallurgical recycling abroad (Europe), and the other 50% are destined for hazardous waste landfill in Brazil. Reminder LIBs wastes (94%) are collected informally (50%) or mixed with MSW (50%) and are fully sent to an MSW landfill.

Scenario 2 - Intermediate: the formal RL increases the collection rate to 12% of waste LIBs generated in 2024 (Brazil, 2020), reducing the amount collected informally or discarded with MSW. This scenario assumes that there are LIBs recycling operations in Brazil (hydrometallurgical) in their first steps. 75% of LIBs waste formally collected by RL is sent to recycling abroad, 12.5% to recycling in Brazil and 12.5% to hazardous waste landfill. The assumptions for fractions collected by the informal sector or mixed with MSW are the same as in Scenario 1, with 50% collected by each route and a full destination to the MSW landfill.

Scenario 3 - Optimistic: RL increases the collection to 17% of LIBS waste in 2025 (Brasil, 2020) and the fraction recycled in Brazil increases. 50% of LIBs waste formally collected by the RL is destined for recycling abroad, 25% for recycling in Brazil, and 25% for the hazardous waste landfill. The assumptions are the same for fractions collected by the informal sector or mixed with MSW (50% collected by each channel and full destination to the MSW landfill). Table 1 summarises the rates of collection and recycling for the four scenarios. Although the scenarios designed in this study followed a progressive approach, some assumptions can be considered limitations, such as the amount of waste collected in each collection channel, and the amounts sent outside, can be over or underestimated, due to the lack of consolidated data.

2.2 Estimation of LIBs waste generation from laptops and smartphones

The annual flow of LIBs on smartphones and laptops was estimated based on consolidated historical sales data in Brazil and sales forecasting. For the period from 2012 to 2021, historical sales data provided by the Brazilian Electrical and Electronics Industry Association (ABINEE) (ABINEE, 2022) were used for both types of products (smartphones and laptops). From 2022 to 2025, there were no sales forecast data for these products in Brazil. To fill this gap, sales forecasting methods and projected global sales rates were applied to project sales in this period. Sales forecasting for Notebooks from 2022 to 2025 was performed using projected global sales rates provided by Statista (2022); the annual percentage change was calculated and assumed to project sales for this period in Brazil. Sales forecasts for smartphones from 2022 to 2025 were made using trend analysis in time series (Montgomery et al., 2015). Trend analysis uses linear regression to examine patterns and trends in historical data and predict the future values of a serie (Nokeri, 2022). The use of the global rate of change in sales of laptops for the years 2022 to 2025 was necessary because due to the increase in sales in the period 2020 and 2021 (due to the covid-2019 pandemic) the use of the projection using time series could be overestimated - in this way the use of variations considering the global rate of sales each year was considered more appropriate, however, it can be considered an important limitation, as it can also cause underestimation of the quantity sold in a year.

The estimate of yearly smartphones and laptops wastes generation was calculated using the lifespan distribution of the products. The choice of lifespan distribution instead of average lifespan seeks to model the waste generation system as closely as possible to the reality of consumers. The WEEE lifespan distribution in the Brazilian context was extracted from a previous study (Abbondanza & Souza, 2019). The lifespan distribution for smartphones each year is calculated considering 8 years, being: 1 - 35.90% of smartphones discarded; 2 - 52.65%; 3 - 6.09%; 4 - 2.33%; 5 - 2.43%; 6, 7 and 8 years - 0.20% (Abbondanza & Souza, 2019). For laptops, the lifespan distribution is calculated considering 9 years, being: 1 - 7.0% of laptops discarded; 2 - 14.0%; 3- 25.0%; 4 - 12.0%; 5 - 16.0%; 6 - 9.0%; 7 - 3.0%; 8 and 9 years - 7.0% (Abbondanza & Souza, 2019). Supplementary material 3 provides the calculation details for each projection of the quantity of products sold and the waste generated each year.

The mass of waste generated for each product by year was calculated by multiplying the amount (in units) by the average weight values of smartphones and laptops. The average weight of smartphones was 154.54 g by product unit (Ercan, 2016; Babbitt et al., 2020; Cordella et al., 2021). Notebooks' average weight was 2.711 kg by product unit (Abbondanza & Souza, 2019; Babbitt et al., 2020; Kasulaits et al., 2015).

The estimate of the generation of LIBs waste (in mass) was calculated by multiplying the total mass value of the products by the percentage referring to the mass of the LIBs: 27.3% weight for LIBs in smartphones and 9.34% weight for LIBs in laptops (e.g., Wang et al., 2016; Cordella et al., 2021).

2.3 LCA phases

LCA was applied in accordance with the guidelines of ISO 14040 (ISO, 2006a), ISO 14044 (ISO, 2006b) and best practices indicated in the reference manual of the International Reference Life Cycle Data System - Handbook (ILCD) (EC-JRC, 2010).

2.3.1 Definition of the objective, scope and system boundaries

The objective is to evaluate the potential impacts and environmental benefits related to different and progressive recycling scenarios for LIBs from two types of WEEE products widely used and discarded in worldwide, smartphones and laptops. The geographical scope of the study is Bra-

TABLE 1: Collection and recycling rate for scenarios.

Scenario	Collection rate by RL	Collection rate by informal	Collection rate by MSW	Recycling rate in Brazil	Recycling rate ex-ported to Europe
Scenario 1	6%	47%*	47%*	0	50%
Scenario 2	12%	44%*	44%*	12.5%**	75%
Scenario 3	17%	41.5%*	41.5%*	25%**	50%
Scenario 4	50%	25%*	25%*	75%**	25%

** Full LIBs collected in these routes are sent to MSW landfill; **The difference is sent to hazardous waste landfill.*

zil. The intended application of the study is to identify the hotspots and the main environmental indicators related to waste LIBs recycling in the Brazilian context. The structure used for modelling the inventory is attributional.

The evaluated function of the system is the environmentally sound destination of LIBs waste from the mix of laptops and smartphones waste generated in Brazil. The functional unit to assess unit processes of hydrometallurgical recycling carried out in Brazil and pyrohydrometallurgical processes abroad (Europe) was defined as treating 1 t of LIBs waste in Brazil and abroad. To evaluate scenarios, the functional unit was defined as the treatment of LIBs from the mix of smartphones and laptops discarded in Brazil in 1 year. Supplementary material 4 summarises the function, functional unit, and reference flow of the four scenarios.

The hydrometallurgical process was chosen for Brazil because the hydrometallurgical process is one of the most promising methods for recycling of LIBs waste (Zhou et al., 2021).The hydrometallurgical process is highly effective for all types of battery chemical recovery (Zhang et al., 2022) and has the potential to generate less environmental impact because it uses less energy and recovers a greater number of materials (Dunn et al., 2022; Jin et al., 2022). Hydrometallurgical recycling plants are less complex in terms of costs to install and process compared to pyrometallurgical recycling, which seems more suitable for developing country contexts (Gaines et al., 2018; Jin et al., 2022; Chen et al., 2023). Finally, a company that has been operating the recycling of LIBs in Brazil (Energy Source) uses a hydrometallurgical process (Energy Source, 2022).

The pyrohydrometallurgical process was chosen for Europe because evidence states that this is the type of process most commonly used in recycling plants in European countries (Chen et al., 2019; Mayyas et al., 2019; Yu et al., 2022; Jin et al., 2022). There are well-established LIB recycling plants in Europe that use pyrometallurgy and hydrometallurgy processes together (Mayyas et al., 2019). Pyrometallurgy in conjunction with hydrometallurgy is the dominant recycling method, widely prevalent in European recyclers such as Umicore, Batrec, Accurec, Glencore (Jin et al., 2022).

Multifunctionality is addressed with the substitution approach, including the avoided production of recovered metals and graphite in the system boundary. Assumptions for substitution, such as the amount of each material present in the LIBs waste stream, the recovery efficiency materials in each recycling process, the quality of the recovered material, and the types of processes avoided in the upstream chain were derived from literature (Supplementary material 5 details assumptions for substitution). The substitution was modeled considering open-loop recycling. There are two ways to model the substitution of materials by recycling, Open loop or Closed loop. Closed-loop recycling considers that recycled material can replace the original virgin material and be used in the same type of products. Open-loop recycling, the properties of the recycled material differ from those of the material used in the original product, which is why it is used in other applications of the product, replacing other materials (Larrain et al., 2020). In this LIBs recycling case, recovered metals displace equivalent amounts of raw materials (assumed recycling substitution ratio 1:1) in the respective primary supply chains (mining and ore beneficiation processes). Closed-loop recycling, in which the recovered materials would return to the supply chain to produce new LIBs, cannot be evaluated because of uncertainty about the purity of the recovered metallic salts and their suitability for manufacturing new LIBs (Rajaeifar et al., 2021).

The system is modelled considering the limit of the grave-to-gate system, which includes the steps from the waste entry in the recycling process, recovery of materials and avoided processes (Hao et al., 2017; Bobba et al., 2018; Qiao et al., 2019). LIBs waste is modelled as post-consumer waste, entering the system without environmental charges from the production and use of LIBs (Allacker et al., 2014; Rinne et al., 2021). Due to the lack of data on WEEE collection within Brazil, the transport for collection, sorting and disassembly of smartphones and laptops was excluded from the system boundary. It can be considered a limitation because significant environmental burdens can arise from these steps. Infrastructure is also not considered in the model.

Recycling of LIBs through hydrometallurgy in Brazil begins with the reception of LIBs after the smartphones and laptops products have been disassembled. Then, a resistor discharged LIBs cells to avoid combustion and explosions. Next, three sequential mechanical processes (crushing, shredding and sieving) are used to separate fine fractions rich in metals (black mass) of the medium and coarse fractions, including plastics, current collectors and papers from LIBs. The medium and coarse fractions are sent for final disposal in landfills according to the characteristics of each material. The fine fraction (approximately 50% of the weight of the starting material) is sent to the hydrometallurgical processing, where the black mass undergoes a leaching step with sulfuric acid and glucose. After the leaching, sequential processes separate the graphite and precipitate the metals, which are filtered and oven-dried. The product system and system boundary for recycling technology in Brazil are shown in Figure 1.

The pyrohydrometallurgical process for recycling in Europe is modelled using Ecoinvent inventories and updated data from the literature. The European pyrohydrometallurgical process begins with the national and international transports of sorted cells and the receipt of LIBs at the recycling plant. Next, the cells are heat-treated through a blast furnace casting process. Furnace products consist of alloys and slag that are sent for further treatment. The alloys are sent to a hydrometallurgical process that uses inorganic chemical reagents. Supplementary material S6 provide flowchart the pyrohydrometallurgical recycling in Europe.

Figure 2, Figure 3, Figure 4 and Figure 5 shows flowcharts of scenarios 1 at scenario 4.

2.3.2 Inventory analysis

Data for the recycling process of LIBs in Brazil (hydrometallurgical technology) were collected from the literature and a research institute (Center for Information Technolo-

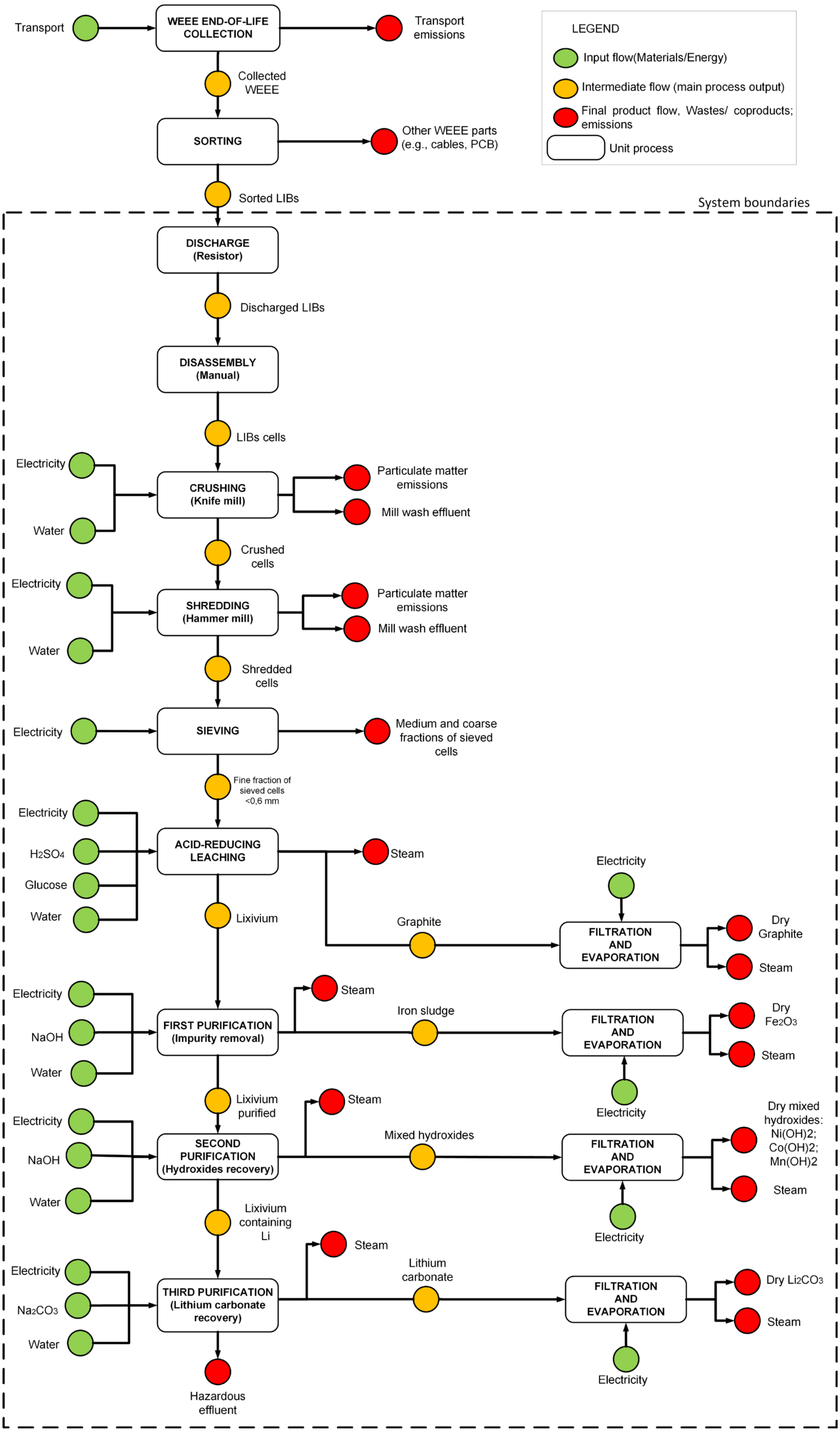

FIGURE 1: Product system and system boundary for hydrometallurgical LIBs recycling in Brazil.

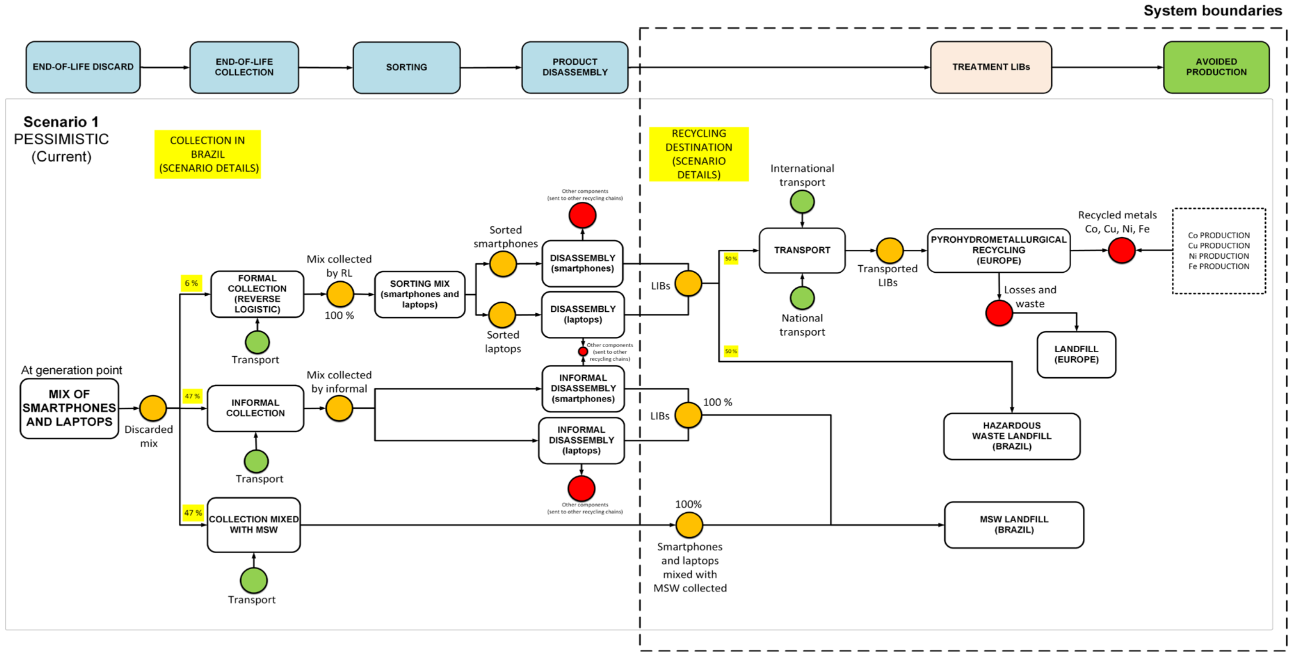

FIGURE 2: Flowchart Scenario 1 for LIBs recycling.

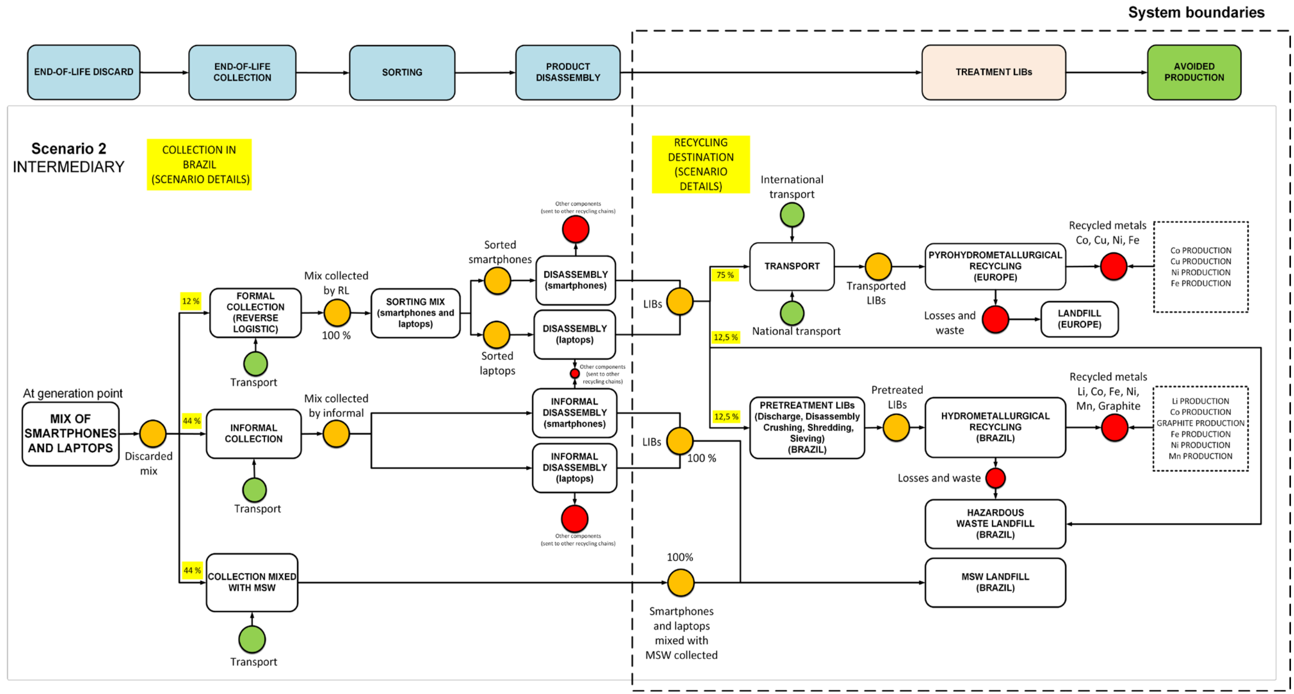

FIGURE 3: Flowchart Scenario 2 for LIBs recycling.

gy Renato Archer - CTI) located in the Campinas//Brazil. This institute provided experimental data and a patent containing the main data on mass and energy flows of unit processes (European Patent EP 2 450 991 B1-2011). The characterisation of the electrolytic powder and recovery rate of metals from leaching using glucose was obtained from the literature (Granata et al., 2012; Pagnanelli et al., 2017), as well as air emissions of fine particulate material from crushing and shredding processes (Costa, 2010). Emissions to water were estimated based on stoichiometric calculations (Supplementary material 7 provides stoichiometric details).

Data for LIBs waste recycling processes abroad were collected from the Ecoinvent v.3.6 database and literature. The process of recycling LIBs abroad was modelled with the processes available in the Ecoinvent database; the available inventories use data from Fisher et al. (2006). The pyrometallurgical process has been edited to include the

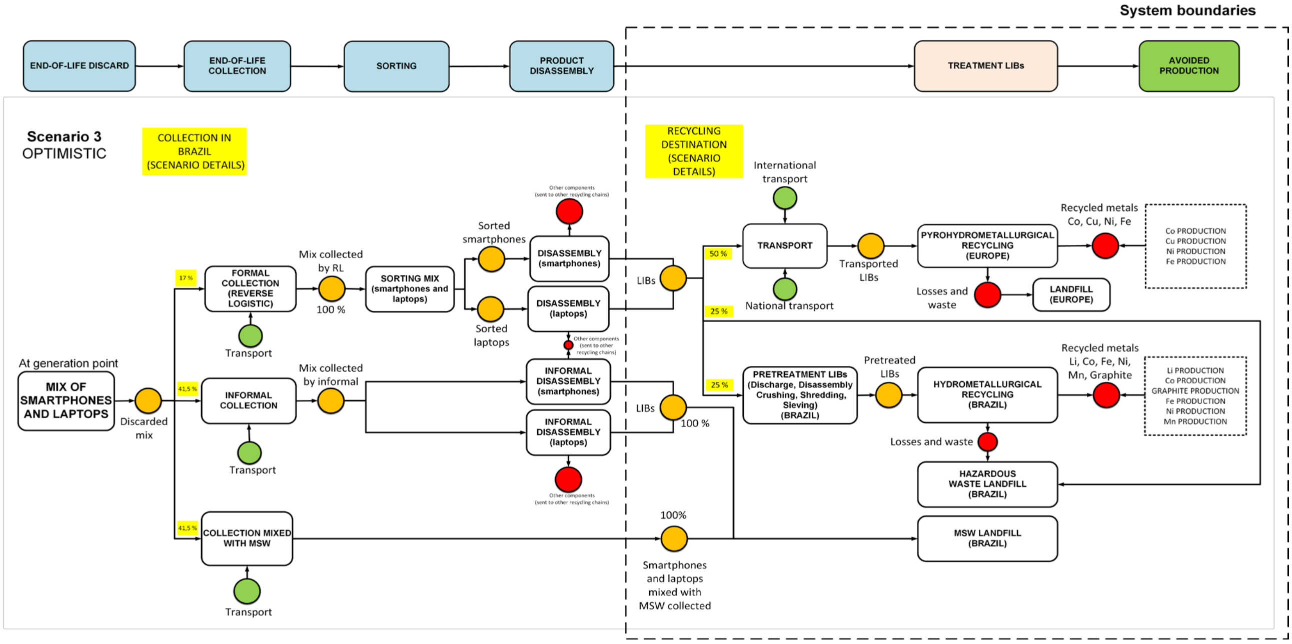

FIGURE 4: Flowchart Scenario 3 for LIBs recycling.

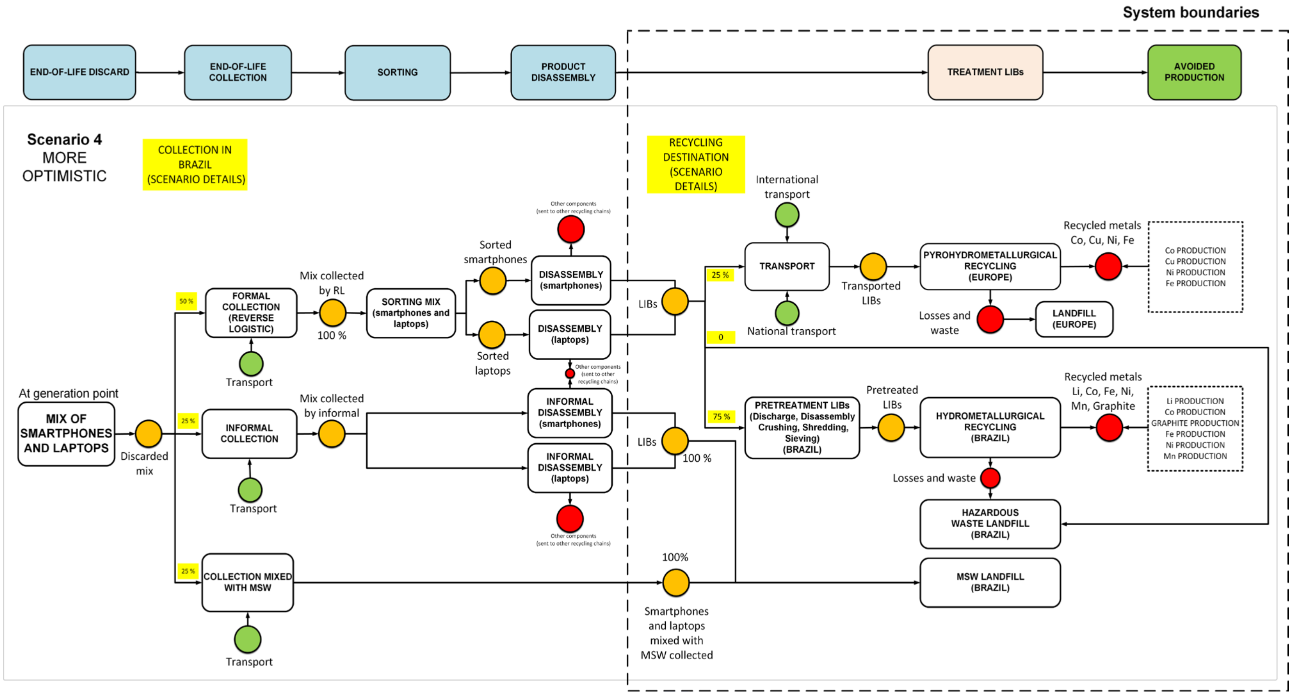

FIGURE 5: Flowchart Scenario 4 for LIBs recycling.

subsequent hydrometallurgy step. In addition, materials recovered, and recovery rates were adapted from Rajaeifar et al. (2021). The adaptation was necessary, as the process available in Ecoinvent is modelled with older data (2006), where manganese, instead of nickel, was recovered, which does not reflect current operations (Harper et al., 2019; Jin et al. al., 2022).

Data on the production of chemical inputs, energy generation, landfilling, water treatment and transport are collected in the Ecoinvent v.3.6 database. It is important to note that medium landfill inventories used within the system boundary of each recycling process to model the shipment of fractions that are destined for final disposal, such as plastics and other materials, to both Brazil and Europe. However, due to the lack of specific local landfill data, the landfills used may not be representative enough to capture the differences between Brazilian and European landfills, which is a limitation of this study.

International transport distances (maritime transport) are estimated using geospatial tools available on the internet (9.361 km – distance from the port of Santos – Brazil for the port of Le Havre - France) (Google Maps, 2022). The average road transport distance from cities to the main port in Brazil (200 km) is collected from the literature (Rocha & Penteado, 2021). This average transport distance considers a pretreatment centre in the state of São Paulo to the Port of Santos. This distance was considered due to the scope of the study considering the transport from the consolidation of WEEE collected in Brazil. Supplementary material 8 provides the life cycle inventory normalised to 1 t and sources data types to hydrometallurgical recycling in Brazil and pyrohydrometallurgical recycling in Europe.

SimaPro Faculty software v. 9.1.1.1 was used for modelling and assessing environmental impacts.

LIBs landfill limitations

Regarding the landfill inventory used to model the disposal of LIBs in the scenarios, one limitation should be considered the use of an average landfill inventory available on Ecoinvent for the disposal of LIBs, which may not adequately reflect the direct emissions of LIBs. Although this strategy of using average landfills is widely used in the literature, there are clear limitations on the representation of specific wastes. To try to address the uncertainty involved in this limitation, we followed guidelines from the literature (Doka, 2023) and inventoried the substances at the elemental level of each constituent material of the LIBs. This is a suitable strategy to make transparent the limitations of landfill modelling for specific wastes where no data is available (Doka, 2023). The inventory at the elemental level shows that LIBs have the potential to emit different substances, such as CH_4, CO_2 and hydrofluorocarbon (HFC), due to the presence of hydrogen, fluorine, and carbon in LIB materials. In addition, the amount of different metals present in LIBs can pose risks of potential impacts, mainly related to toxicity.

As LIBs have a complex composition, with different materials that can interact in different ways in landfill environments (Kilgo et al., 2022), it was not possible to model all the possible emissions that can occur. However, we performed a simulation using parameters from some recent studies that determine the daily emission of CH_4, CO_2 and C_2H_4 from discarded PP, PE and PET plastics due to photodegradation (Royer et al., 2018; Shaw et al., 2023) to model possible emissions of these gases from these plastic fractions that are present in LIBs sent to landfill. Following guidelines from the literature on landfill modelling, we considered a time horizon of 100 years (Kirkeby et al., 2007). In addition, we calculated the potential for metals to be leached from the fraction of LIBs sent to landfill by considering parameters from a study that carried out laboratory tests on metal leaching from LIBs under landfill conditions (Kilgo et al., 2022). The results of this landfill-specific assessment were not considered in the evaluation of the results but were used to compare the sensitivity of how possible emissions may change due to waste specific modelling. The inventory at the elemental level and the details for the calculations of possible emissions can be found in supplementary material S16.

2.3.3 Impact assessment

The assessment of potential environmental impacts was carried out using the impact assessment method ILCD 2011 Midpoint + version 1.0.9 (EC-JRC, 2012), including the 16 impact categories: Climate change (kgCO_2eq), Ozone depletion (kgCFC-11 eq), Human toxicity – non-cancer effects (CTUh), Human toxicity-cancer effects (CTUh), Particulate matter (kgPM2.5eq), Ionising radiation HH (kBqU235eq), Ionising radiation E -interim (CTUe), Photochemical ozone formation (kgNMVOCeq), Acidification (molcH+eq), Terrestrial eutrophication (molcNeq), Freshwater eutrophication (kgPeq), Marine eutrophication (kgNeq), Freshwater ecotoxicity (CTUe), Land Use (kgCdeficit), Water resource depletion (m3 water eq), Mineral, fossil & renewable resources (kgSbeq). The midpoint level assessment was chosen so that the cause-and-effect chains of environmental impacts could be better understood (EC-JRC, 2010). The assessment of impacts at the midpoint level has fewer uncertainties associated with environmental mechanisms when compared to the endpoint analysis (Huijbregts et al., 2016; Mulya et al., 2022).

2.3.4 Interpretation

The interpretation phase presents a critical evaluation of the results, drawing conclusions and recommendations on the main points of environmental impacts per process, the benefits arising from the avoided production, and the identification of the best and worst scenarios for managing the LIBs waste generated in Brazil. Possible alternatives for improving the environmental profile are recommended, and a sensitivity analysis was conducted.

Sensitivity analysis

A perturbation analysis was performed to assess the robustness of the results: first, one perturbation of one input at a time and then a global perturbation analysis, with all inputs being varied together (Laurent et al., 2020). A range of -/+5% and -/+ 10% were used for the evaluation. A second sensitivity analysis was conducted to assess how sensitive the results are to the recycling efficiency ratio. The recycling efficiency rate is an important parameter, as different technologies, input material quality and subsequent use of recovered materials interfere with the result of the recycling system (Rocha & Penteado, 2021). A Third sensitivity analysis of the results was performed using another impact assessment method, the Recipe Midpoint H (Huijbregts et al., 2016), to verify whether the results could change due to the impact assessment method.

Due to the lack of complete data on the stages of RL, such as transport distances for collection and to dismantling and recycling facilities within Brazil, a simplified RL model that only considers transport from consolidation sites in different cities, after collection, to pre-treatment in São Paulo was tested to check the sensitivity in evaluating the scenarios due to the incorporation of intermediate transport in the model. The calculation took into account the coverage of cities served each year as defined in the sector agreement, the population of the cities and the distances to São Paulo. As the WEEE RL implementation targets defined in the Law (Brazil, 2020) establish the gradual implementation of RL in the states, but do not provide the specific cities for each year, only the number of cities served each year, the cities were added to the RL according to population, with the most populous cities being served first until a total of 400 cities were served. The cities with the most inhabitants are served first until the 400 cities are reached in 2025. For 2026, as there was no data on the cities served, the same number of cities was assumed as in 2025 (400 cities). Point-to-point collection transport within the cities was not included due to the lack of data on the average distances travelled in each of the cities. Therefore, what is included in the calculations is the intermediate transport, after collection, from each generating city to the pre-treatment centre (here considered to be the city of São Paulo). All the data and details of the calculations are provided in supplementary material S15.

3. RESULTS AND DISCUSSION

This section presents the results and discussion of the assessment of unit processes and scenarios, and potential revenues of LIBs hydrometallurgical recycling in Brazil.

3.1 Contribution analysis of recycling technologies

Figure 6 presents the results of the contribution analysis for the overall and unit process of hydrometallurgical recycling in Brazil. Results are presented for a functional unit of 1 t of LIBs waste treated. Results are detailed for the supply of key inputs and direct impacts on processes. Direct impact refers to emissions released directly by a unit process to the foreground system (EC-JRC, 2010), for example, particulate matter emissions from crushing.

The overall contribution analysis for the characterised impacts of the hydrometallurgical technology shows that the main contributors to the impacts are the leaching and material recovery (UP 4) and sieving (UP 3) unit processes. The UP 4, referring to the leaching, precipitation and final recovery of metals, is the main contributor to 13 of the 16 impact categories evaluated (Climate Change, Ozone depletion, Human Toxicity - cancer effects, Ionising radiation HH, Ionising radiation E (interim), Photochemical ozone formation, Acidification, Terrestrial eutrophication, Freshwater eutrophication, Marine eutrophication, Land Use, Water resource depletion, Mineral, fossil & ren resource depletion). The UP 3 is the main contributor to 2 categories (Human Toxicity – non-cancer, Freshwater ecotoxicity). Due to direct process emissions, crushing (UP 1) and shredding (UP 2) unit processes are the main contributors to the Particulate Matter category. The leaching and recovery of materials in UP 4 is the step that consumes a larger amount of chemicals and glucose, with significant impacts on its production. The production of sulfuric acid emits sulphur dioxide and nitrogen oxide, contributing to terrestrial acidification, ozone formation and particulate matter emissions. Using glucose as a reducing agent implies emissions of ammonia, nitrate and nitrous oxide derived from maise cultivation to produce maise starch. These substances contribute to terrestrial acidification, marine eutrophication and stratospheric ozone depletion. Sending materials such as plastics and ferrous metals for final disposal in landfills contributes mainly to freshwater ecotoxicity, Marine eutrophication, Human toxicity- non cancer and Climate change categories due to landfill emissions. The use of sodium carbonate, sulfuric acid and glucose contributes to the reduction of mineral resources. CO_2, methane and nitrous oxide emissions from electricity production, maise production and direct process emissions are the main contributors to the Climate change category. These results align with previous studies that found that the impacts of the hydrometallurgical process are mainly derived from indirect emissions from the production of chemical inputs and electricity (Cusenza et al., 2019; Mohr et al., 2020; Rinne et al., 2021). For the unitary process UP 1, the production of the electricity mix used for crushing is the largest contributor in 14 of the 16 categories. Water production, used to wash away particles that adhere to the mill surface, is the second largest contributor to 12 of the 16 impact categories. For the Particulate matter category, the crushing process is the largest contributor, with 100% of the impact derived from direct emissions of fine particles (<2.5 microns). Water production impacts are mainly for the Mineral, fossil & renewable resources category, due to the use of copper and aluminium sulphate in water treatment at the municipal plant. The water production process provides credits towards the Water resource depletion category due to the return of unpolluted water to the environment during water treatment.

The profile is similar to the previous process for the shredding process (UP 2). Electricity consumption for hammer mill operation is the main contributor to impacts in 15 out of 16 categories. The total Particulate matter category emission is the direct emissions of fine particulates <2.5 microns from the process. The impacts of the crushing and shredding process are mainly derived from the auxiliary systems, which supply energy and water to the system. The control of Particulate matter emissions is a priority for developing the industrial recycling plants for LIBs. Using mechanisms that filter particulate emissions and equipment with increased energy efficiency and higher t/h yield should improve the environmental profile of the crushing operation.

The sieving process contribution analysis (UP 3) shows that the destination of the medium and coarse fractions to the landfill (paper, plastics, and metals of current collectors) is mainly responsible for the impacts in the 16 impact categories. Recovery of medium and coarse fraction materials, such as plastics of casing and metals of current collectors (steel, copper, aluminium) can reduce the environmental impacts associated with the sieving process in the future, as it will reduce the amount of material sent to landfills and production of virgin materials (Hua et al., 2020; Sun et al., 2020; Mohr et al., 2020, Windisch-Kern et al., 2022). For example, recycling of aluminium from waste LIBs has the potential to reduce emissions of 0.03 kg of C_2H_4 eq and 41.6 kg of 1,4-DCB equivalent, avoiding impacts on the Photochemical oxidant formation e Human toxicity categories (Sun et al., 2020). Furthermore, aluminium and copper are easily recovered in the pretreatment and do not require further hydrometallurgical processes (Cao et al., 2023).

The contribution analysis for the inputs of the leaching and material recovery process (UP 4) shows that the main contributors are the production of chemical inputs: sulfuric acid, sodium hydroxide, sodium carbonate and glucose. Chemical inputs were also the main contributors to impacts in the literature, mainly sulfuric acid and sodium hydroxide (Rinne et al., 2021). The use of glucose reduces the general impacts on the Climate change categories, as in the cultivation of maise for production of maise, there is CO_2 sequestration. However, using glucose increases direct CO_2 emissions from the leaching process, as glucose breaks down into water and CO_2 by oxidation. The oxidation of glucose into CO_2 is responsible for 22.5% of CO_2 emissions from hydrometallurgy. Although CO_2 emitted from glucose is considered biogenic, in this study it was treated with a characterisation factor of 1 in the emissions count, since there is still no consensus among scientists on the neutrality of biogenic CO_2 (Cusenza et al., 2021). Glucose has a negative impact reduction in the Human Toxicity non-cancer category, as maise cultivation also captures metals, such as zinc and copper, from the soil, providing an environmental credit. Regarding the credits provides from absorption of metals in crop of maise, it is important to point out that the ILCD method does not include the consumption of these metals within the system boundaries and so the plants serve as a sink (Nessi et al., 2021). The fate of heavy metal emissions taken up by crops are not further considered within the system boundary is a limitation of the potential environmental impacts and should be viewed with caution. Supplementary material 9 provides detailed numerical results of a contribution analysis for the hydrometallurgical technology.

The contribution analysis of the impacts of the modelled pyrohydrometallurgical process for recycling in Europe shows that the impacts of the pyrohydrometallurgical process derive mainly from indirect emissions from producing electricity and chemical inputs and from direct emissions from burning materials in the smelting stage. The main emissions are CO_2, methane, nitrous oxide, sulphur dioxide and heavy metals. The pyrometallurgical smelting process generates many greenhouse gas emissions due to the combustion of materials. The water consumption category is also significant, as both the pyrometallurgical and hydrometallurgical processes consume high volumes of water. It induces the highest amount of water consumption because it uses limestone, which requires a large amount of indirect water consumption (Yu et al., 2021). Transport, road and maritime contribute up to 30% of the impacts for the Photochemical ozone formation and Terrestrial eutrophication categories and 20% for Marine eutrophication and Acidification. Detailed results for contribution analysis of pyrohydrometallurgical recycling to Europe are presented in the supplementary material 10. Additionally, it is important to point out that the energy mix used to evaluate the process in Europe refers to a European mix that considers a proportion of energy generated by different European countries, therefore a comparison with an energy mix from a specific country, such as France, which has an energy matrix with more nuclear sources can change the result of the amount of emissions derived from energy.

A direct comparison of the categories for the two technologies is presented in Figure 7. Recycling 1t of LIBs with the pyrohydrometallurgical option of the Europe has greater environmental impacts in 12 categories. For example, for the Climate Change category, the external option emits 1240.00 $kgCO_2eq$/ton of recycled LIBs, which is 143% more impactful than the hydrometallurgical technology in Brazil, emits 510.1 $kgCO_2eq$/ton of recycled LIBs. This result is explained by the fact that recycling with pyrohydrometallurgical technologies consumes more resources to be performed, mainly energy. Electricity production emits large amounts of CO_2 and methane, the main greenhouse gases. The difference can also be explained by the fact that the Brazilian energy mix is composed of more renewable energy than the global average (Lima et al., 2019). Contributions to the Human toxicity – non-cancer category are derived from zinc, arsenic and lead emissions from the treatment of waste coal used in European electricity production. Recycling in Brazil has greater impacts in 4 categories (Particulate matter, Acidification, Land Use, and Water resource depletion). For the Particulate matter, Acidification, and Water resource depletion categories, the greatest impacts are derived from using sulfuric acid and the direct emission of particulates from crushing and shredding processes. The production of sulfuric acid consumes large amounts of water and emits substances that contribute to terrestrial acidification and the formation of particulates, such as SO_2 and NOx. For the Land use category, maise cultivation for glucose production is the main cause of impacts.

3.1.1 Avoided impacts of material recovery

The results shown that the greatest environmental benefits (avoided impacts) are derived from the avoided production of Co and Ni. The recovery of Co and Ni is responsible for more than 93% of the avoided impacts, mainly in the categories of Climate Change, Freshwater ecotoxicity and Land Use. The production of Co and Ni is intensive in energy resources and CO_2 emissions (Chen et al., 2023). The production of Cobalt emits 8.8 kg of CO_2 eq/kg and Nickel production 13.2 kg of CO_2 eq/kg. In addition, they occupy large areas of soil and release toxic pollutants into the water. Nickel production incurs high toxicity impacts, and its recovery avoids them (Cao et al., 2023). Although Graphite recovered is significant per ton of LIBs recycled (252 kg/ton of LIBs waste in this study), the recovery benefit is small compared to other metals because the considered process of mining natural graphite is less energy and emissions intensive than metals, around 0.037 $kgCO_2eq$/kg of natural graphite production. Comparatively, even though 44% more graphite is recovered by mass than cobalt, the avoided impact of graphite is less than 1% of the avoided impact of cobalt. Greater benefits would be captured if the quality of the recovered graphite was battery grade. A recent study shows that battery-grade graphite production emits 9.66 kg of CO_2eq/kg of battery-grade graphite produced (Engels et al., 2022), but further studies on additional treatments

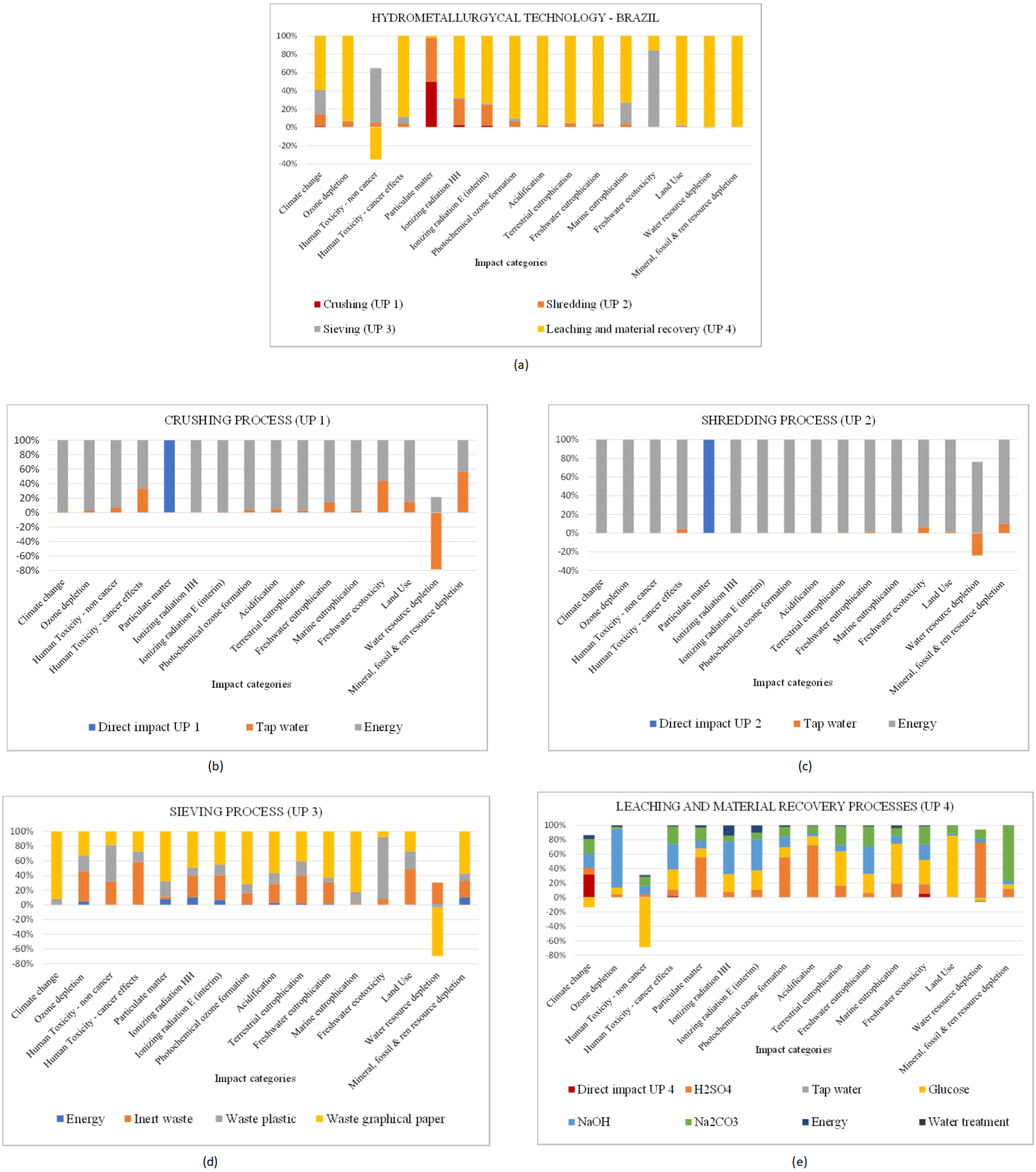

FIGURE 6: Contribution analysis of the unit processes of LIBs hydrometallurgical recycling in Brazil (and supply of key inputs). Note: (a) Overall contribution analysis of the hydrometallurgical technology – Brazil; (b) Contribution analysis of the crushing process (UP 1); (c) Contribution analysis of the shredding process (UP 2); (d) Contribution analysis of the sieving process (UP 3); (e) Contribution analysis of the leaching and material recovery process (UP 4).

are needed to verify the actual benefit. The detailed results for avoided impacts of material recovery to all impact categories are shown in Supplementary material 11.

3.1.2 Comparison with other LCAs

Direct comparison of LCA results from LIBs is challenging, as modelling choices, such as functional unit, system boundary, types of the treatment process, chemical inputs, number of evaluated impact categories and recovered materials interfere with study results and cannot provide a basis for direct comparison (Arshad et al., 2022). However, in order to provide a comparison of the modelled hydrometallurgical recycling process, Table 2 presents a general comparison with other LCA studies of the recycling of LIBs

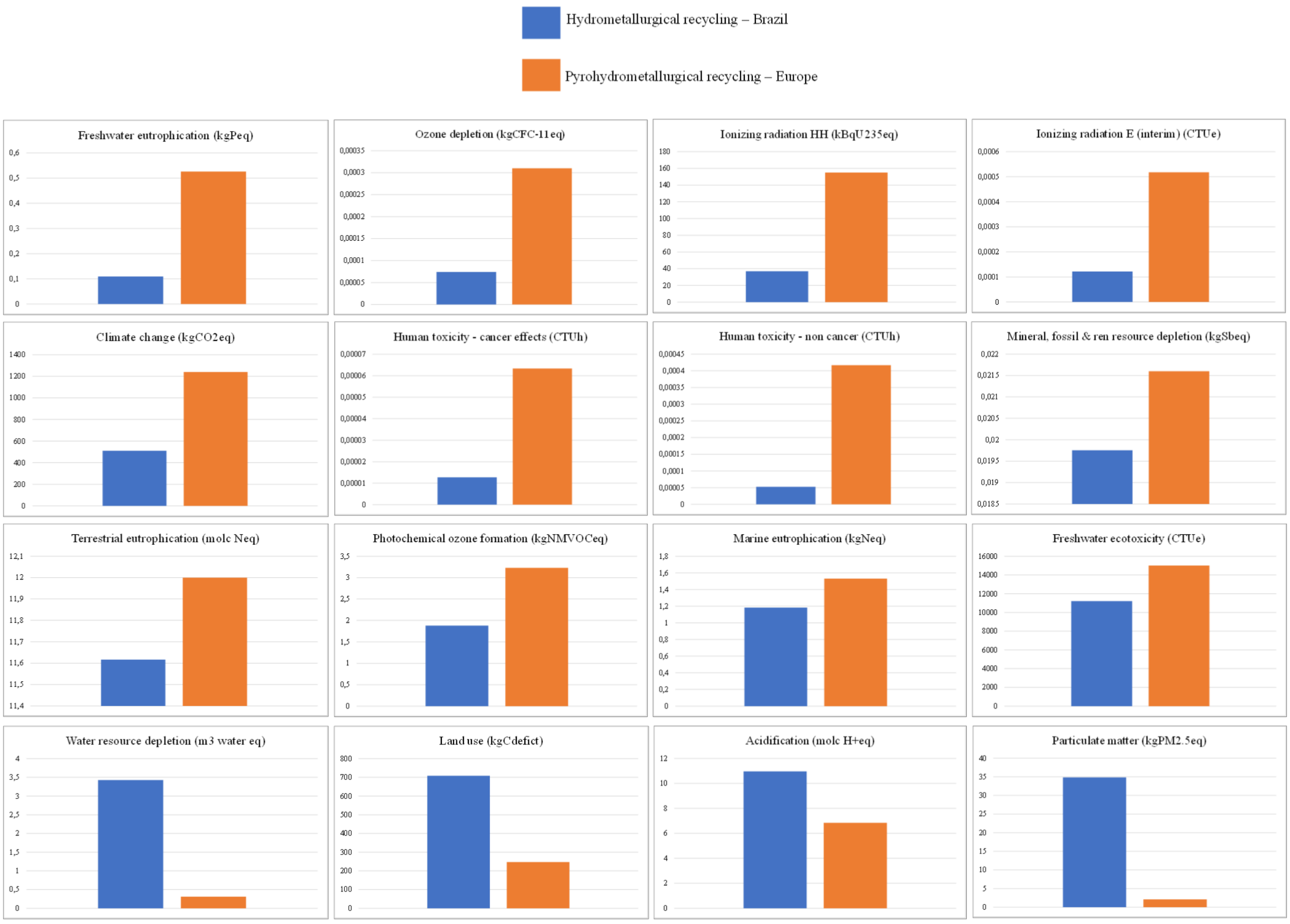

FIGURE 7: Comparison between recycling of 1t of LIBs waste with hydrometallurgical process in Brazil and pyrohydrometallurgical in Europe.

in the Global Warming category (kg of CO_2 eq), which is the most evaluated category in studies of recycling on LIBs (Rajaeifar et al., 2022). Analysing the normalised results for the treatment of 1 kg and 1 ton of LIBs, the impacts of the hydrometallurgical treatment proposed for implementation in Brazil have less impact in the Global Warming category than all other LCAs analysed.

3.2 Scenario assessment

The estimation of LIBs waste generation indicates that more than 3.000.00 tons of LIBs waste can be discarded each year (from 2023 to 2026) in Brazil's smartphones and laptops waste streams. Considering that these two WEEE products have a wide market penetration worldwide and a short lifespan, this result confirms the great potential of LIBs waste generation and resource recovery in the WEEE streams in all countries. Supplementary material 3 provides the results of estimating the generation of LIBs waste.

The results of the environmental assessment of the four scenarios with the functional unit of annual LIBs waste generation are summarised in Figure 8 and Figure 9. Scenario 1 shows the worst performance due to greater disposal of LIBs waste in landfills (2.925.00 t), lower collection rate by RL (6%) and recycling in Europe, which is more impactful. Scenario 4 presents the best performance due to a reduction of 44% in the disposal of LIBs waste in landfills (compared to the baseline scenario), an increase in RL collection rate to 50% and 75% of LIBs waste recycled in Brazil, using the technology less impactful. The results indicate that RL progression and landfill rate reduction significantly improve the environmental profile of LIBs management in Brazil. In previous evaluations of scenarios for waste management, scenarios in which the largest share of waste is destined for landfills result in greater environmental impacts (Turner et al., 2016; Souza et al., 2016; Coelho & Lange, 2018). Landfilling mainly contributes to large greenhouse gas emissions and has a greater impact on global warming (Turner et al., 2016; Jaafarzadeh et al., 2021). To detailed results of each scenario, see supplementary material 12. These landfill results should be interpreted with caution due to the use of an medium landfill to calculate the potential impacts of the fraction of LIBs destined for landfill. However, the analysis of the inventory at the elemental level of LIBs, the modelling of CO_2 and CH_4 emissions from the plastic fraction of LIBs shows that LIBs have the potential for GHG emissions directly through plastics but also through other interactions, such as the formation of Hydrofluorocarbon (HFC), since hydrogen, fluorine, and carbon are present in the composition of LIBs.

TABLE 2: Comparison with Other LCAs.

Wet methods			
Authors	**Treatment type**	**kg CO_2eq / kg of battery**	**kg CO_2eq / ton of battery**
This study	Hydrometallurgical	0.51	510.00
Souza et al. (2022)	Hydrometallurgical	1.13 to 1.99	1130.00 to 1990.00
Dunn et al. (2022)	Hydrometallurgical	1.8	1800.00
Xu et al. (2021)	Hydrometallurgical	2.27	2270.00
Xu et al. (2020)	Hydrometallurgical	2.3	2300.00
Jiang et al. (2022)	Hydrometallurgical	2.7	2700.00
Rinne et al. (2021)	Hydrometallurgical	3.8	3800.00
Yu et al. (2021)	Hydrometallurgical	10	10000.00
Dry methods			
Authors	**Treatment type**	**kg CO_2eq / kg of battery**	**kg CO_2eq / ton of battery**
This study	Pyrometallurgical	1.24	1240.00
Rajaeifar et al. (2021)	Pyrometallurgical	1.4	1400.00
Dunn et al. (2022)	Pyrometallurgical	2.0	2000.00
Xu et al. (2021)	Pyrometallurgical	2.16	2160.00
Xu et al. (2020)	Pyrometallurgical	2.4	2400.00
Yu et al. (2021)	Pyrometallurgical	10.9	10900.00

In addition, the metal leaching potential of LIBs can also contribute significantly to the toxicity categories. The comparison between the results of the fraction sent to landfill using the limited simulated emissions data shows that modelling the sending of LIBs with limited data is not adequate, as several emissions could not be accounted for, as well as the destination of such emissions. Figure S1 in supplementary material S16 shows that compared to the medium landfill used with the limited LIBs landfill that was simulated, the results for the climate change category is much lower than the medium landfill. In contrast, for the Freshwater ecotoxicity and Human toxicity - cancer effects categories, the amount of metals leached from the LIBs causes much more impact compared to the medium landfill. Therefore, the landfill results in the evaluation of LIB waste management scenarios should be interpreted with caution, since not all the potential toxicity or GHG emissions can be modelled, and the average landfill does not capture all the potential toxicity from metals. Further studies into the disposal of LIBs in landfill is urgent.

The results of net environmental impacts indicate that recycling more LIBs waste by Brazil's hydrometallurgical technology results in net environmental benefits across 13 impact categories. For example, in Climate change, scenario 4 presents net environmental benefits of -183.355.00 kgCO_2eq. In contrast, scenarios 1, 2 and 3 do not present net benefits because recycling credits do not outweigh the burdens derived from the greater amount of LIBs waste sent to landfills and a higher recycling rate with pyrohydrometallurgical technology.

The impact categories in which scenario 4 does not present the best net impact results, that is, the material recovery credits do not exceed the total impacts of the scenario, are Ozone depletion, Particulate matter and Water resource depletion categories. This result is explained by the increase in the consumption of inputs such as NaOH, glucose, Na_2CO_3, limestone and particulate matter emissions due to the increase in the number of waste LIBs that are sent for recycling. Therefore, improvements in particulate matter emissions in the crushing and shredding processes and the increase in yield of the leaching processes and material recovery can help reduce the overall impacts and result in net environmental benefits (avoided impacts) far superior to the impacts generated by recycling.

In summary, the results indicate that it is beneficial for the environment to recycle LIBs in Brazil, avoiding flows to landfills. Suppose additional improvements are made to the hydrometallurgical process, such as improving recycling efficiency, reducing the use of chemicals, controlling particulate matter, and using glucose sources from agro-industrial waste, such as molasses or whey (Pagnanelli et al., 2017). In that case, the environmental profile of recycling can significantly improve.

3.3 Sensitivity analysis

The sensitivity analysis considering a variation in inputs in the hydrometallurgical process, both individually and globally, indicates that a variation of 5% to 10% in the inputs results in insignificant changes in the results. The sensitivity analysis to the recycling efficiency ratio indicated that it would take a reduction of more than 70% in material recovery rates or in the replacement rate of all materials for the impacts of the recycling system to outweigh the benefits. The results of this sensitivity analysis indicate that for LIBs recycling systems that allow high efficiency and purity of metals is a critical factor in achieving net benefits. Previous studies have also highlighted that the yield of metal recovery and the consumption of chemical inputs are sensitive parameters for the environmental and economic viability of recycling LIBs (Cao et al., 2023). The nu-

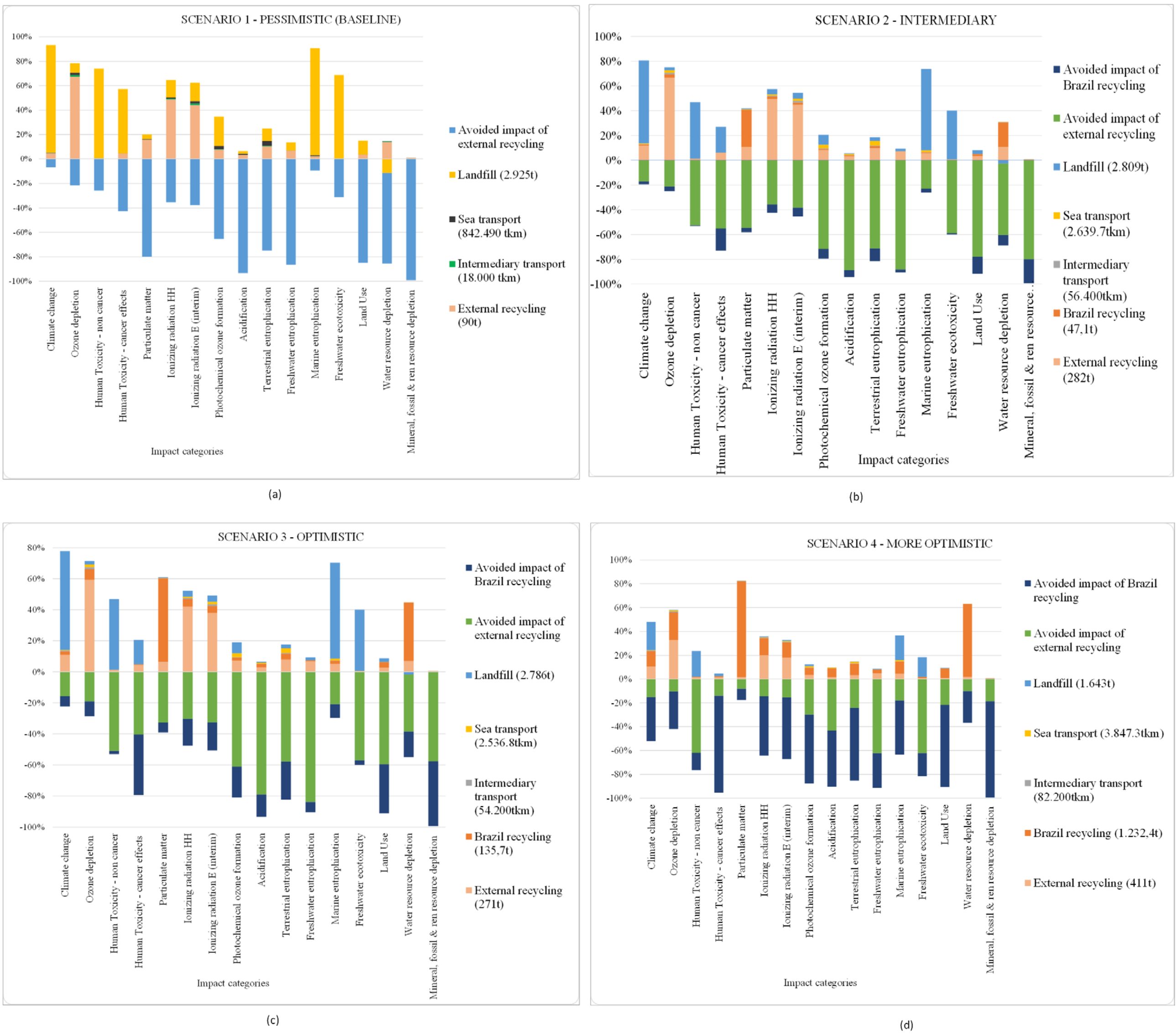

FIGURE 8: Scenarios assessment based on the amount of LIBs waste generated per year. Note: Scenarios assumptions: (a) Scenario 1 - Pessimistic (baseline): Collection: 6% collected by RL and 94% collected by informal and MSW / Recycling: 100% collected by RL sent for pyrohydrometallurgical recycling in Europe; (b) Scenario 2 - Intermediate: Collection: 12% collected by RL and 88% collected by informal and MSW/Recycling: 75% collected by RL sent for pyrohydrometallurgical recycling in Europe and 12,5% sent for hydrometallurgical recycling in Brazil; (c) Scenario 3 - Optimistic: Collection: 17% collected by RL and 83% collected by informal and MSW /Recycling: 50% collected by RL sent for pyrohydrometallurgical recycling in Europe and 25% sent for hydrometallurgical recycling in Brazil; (d) Scenario 4 - More Optimistic: Collection: 50% collected by RL and 50% collected by informal and MSW/Recycling: 25% collected by RL sent for pyrohydrometallurgical recycling in Europe and 75% sent for hydrometallurgical recycling in Brazil.

merical results of these sensitivity analyses can be found in Supplementary material 13.

Figure 10 shows the results of sensitivity analysis using another impact assessment method (ReCiPe midpoint H). Net performance analysis of the scenarios in the six comparable categories revealed that there are significant differences in the results. In this study, the difference is seen mainly in the Global Warming category due to the use of glucose. In the Recipe 2016 method, maise cultivation causes additional impacts on CO_2 emissions. Unlike the ILCD 2011 method, maise cultivation has negative impacts because it sequesters CO_2 from the atmosphere. In the Recipe method, carbon sequestration is not included. Therefore, depending on the method used, the results of the recycling process may differ slightly. Other studies that compare the Recipe 2016 midpoint and ILCD 2011 midpoint + also observed differences in results, mainly in the Global warming category (Borghesi et al., 2022). For numerical results of the comparison in each method, see Supplementary material 14.

As the methods have different measurement units for some categories, it is impossible to directly compare the results for all categories. It was only possible to directly compare the results for 6 categories in which the measurement units were the same in both methods: Climate Change (kgCO_2eq), Ozone Depletion (kgCFC-11eq), Marine Eutrophication (kgNeq), Freshwater Eutrophication (kgPeq), Particulate Matter (kgPM2.5 eq), Water Depletion

FIGURE 9: Numerical results of total impact, total avoided impact and total net impact for each scenario.

(m3). The net results of the scenarios, that is, the total impact of the scenario minus the total recycling credits in the two methods, are comparable in the Freshwater Eutrophication (kgPeq) category, presenting practically matching results. However, in the other 5 categories, there were discrepancies in the net result of the scenario. For example, for the Ozone Depletion category (kgCFC-11eq), the Recipe method has a negative net benefit, while the impacts of the ILCD method are still positive. Recipe 2016 results have lower impacts in all scenarios in the Particulate matter category. However, even with the discrepancies in the final net performance of the categories, the methods agreed with the classification of the scenarios' performance: scenario 1 was the worst scenario, and scenario 4 was the best in both. Therefore, the sensitivity analysis results indicate that the results of the LCA conducted for recycling scenarios in Brazil are robust.

The sensitivity analysis showed that the recovery rates of the materials and the impact assessment method used mainly influenced the results of net impacts. Greater recovery of materials in the hydrometallurgical process is essential to obtain net negative environmental benefits because it uses many processes and chemical inputs that bring an environmental burden to its production. In addition, the disposal of materials such as plastics, paper and nonferrous metals by the cases and current collectors can influence the overall impacts. For Brazil, which still does not have LIBs collection networks and recycling systems developed, the results of this sensitivity analysis help to clarify the importance of implementing efficient systems for material recovery, to increase the benefits from avoiding the production of virgin raw materials and the disposal of large amounts of valuable materials in landfills.

3.3.1 Sensitivity analysis from inclusion of RL transport

The sensitivity analysis that considers the inclusion of intermediate transport in the evaluation of the scenarios shows that the inclusion of intermediate transport to the pretreatment point in each scenario can increase the overall impacts of each scenario. The generation of impacts is directly related to the increase in the amount collected by RL and the number of cities served, as more weight is collected and transported over more distances once more cities are included in the plan. In scenario 1 where less waste is collected (6%) and fewer cities are served (186 cities) the impacts of transport for the Climate change category increase the total impacts by only 1.48% while in scenarios 2, 3 and 4 the impacts increase by 13.67%, 10.44% and 32.21% respectively. If intermediate transport is included, the magnitude of the impacts can increase from less than 0.01% to 1000% depending on the impact category analysed and the scenario analysed. For details of the results of the scenario evaluation considering RL, see supplementary material S15.

The results show that the categories Terrestrial eutrophication, Water resource depletion, Land Use, Acidification, Photochemical ozone formation, Ionising radiation and Ozone depletion are the most sensitive when transport

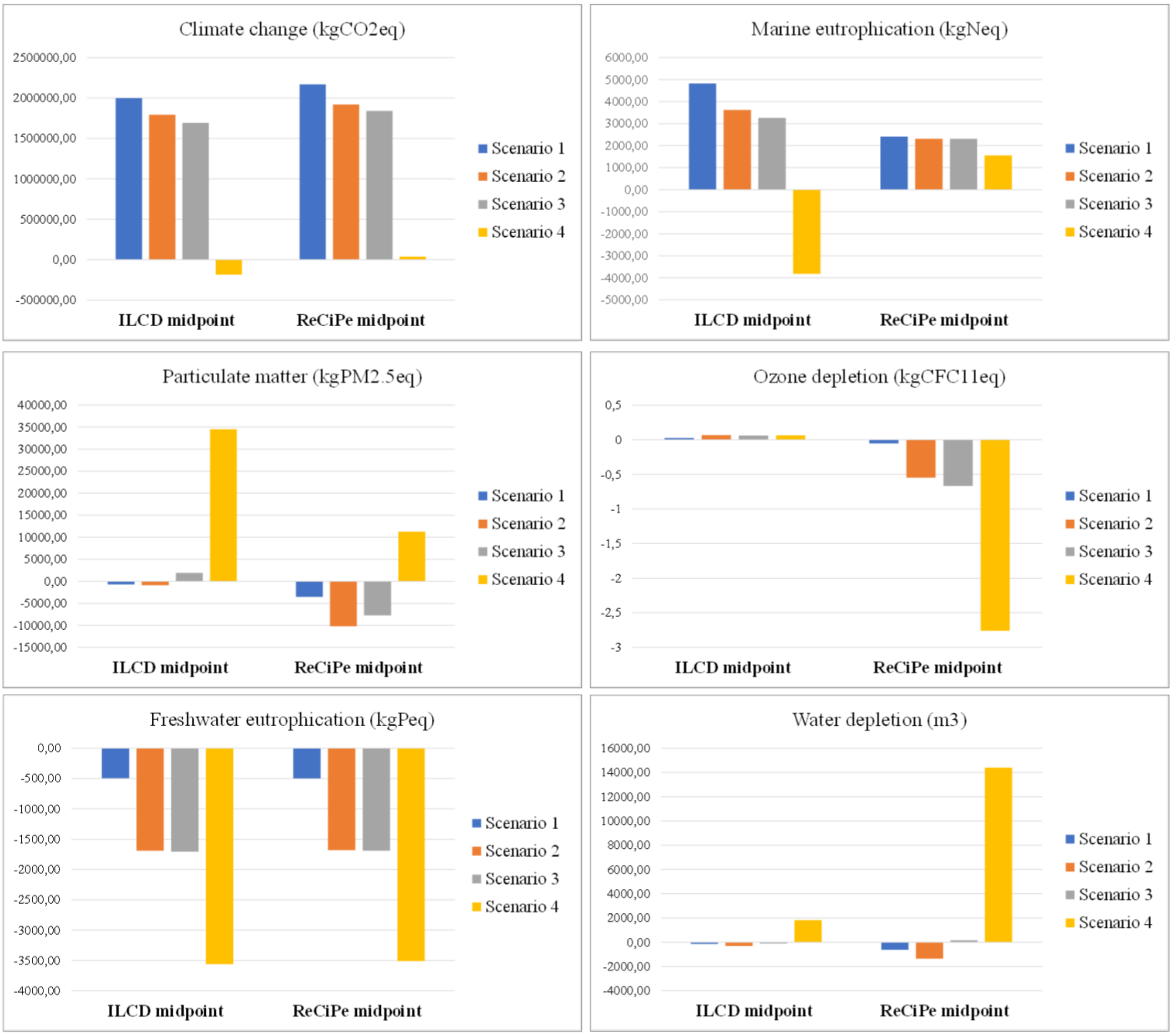

FIGURE 10: Sensitivity analysis comparing results of ILCD midpoint and ReCiPe midpoint methods in net result of scenarios.

is included. This makes sense, since road transport LCAs point out that transport exhaust emissions such as Nox, SOx and NMVOC are the main contributors to the acidification, eutrophication, photochemical ozone formation and ionising radiation categories, the use of diesel fuel contributes to increased impacts in the ozone depletion category (Merchan et al., 2020) and the use of roads contributes mainly to increased impacts in the land use categories (Osorio-Tejada et al., 2023). For the water depletion category, water consumption in the production of diesel and trucks is the main contributor to the increase in impacts.

However, due to the progressive approach of the scenarios, even with the inclusion of transport the ranking of the scenarios does not change, although the environmental benefits arising from material recovery are no longer sufficient to reduce emissions to a net negative state in scenario 4, even so the total net impacts are largely reduced. Therefore, the results of the sensitivity analysis show that depending on the amount of transport required for RL operations, the net impact results can be largely affected.

Finally, it should be noted that the lack of data on the average distance of collection routes in each city and to landfills in Brazil and Europe meant that the transport of these operations could not be included in the system limit, which is a limitation and can interfere with the total amount of impacts generated by the RL of LIBs.

4. CONCLUSIONS

This study assesses potential environmental impacts for prospective LIBs waste recycling from smartphones and laptops. The estimate of LIBS waste generation revealed that a large amount of LIBs waste can be discarded each year in the smartphones and laptops waste stream in Brazil (3,000 tons/year from 2023 to 2026), confirming the great potential of LIBs waste generation and resource recovery in the WEEE streams.

Through a hydrometallurgical route, recycling brings net environmental benefits, reducing environmental impacts in all assessed categories. Ni and Co recovery provides the highest environmental benefits across all impact categories. The pretreatment processes implemented enriched the electrolytic material fraction of LIBs can be enriched, increasing the potential material recovery and providing greater environmental benefits. The use of an organic reducing agent (glucose) resulted in negative impacts (benefits) due to carbon sequestration accounted in the maise cultivation phase. However, the glucose produced from maise can still be replaced by agro-industrial waste, such as molasses and whey, which can further improve the overall impacts of the process. In addition, recovery of other types of materials, such as plastics, steel, aluminium and copper, from casings and current collectors can improve the performance of the hydrometallurgical process.

The LCA results for scenario assessment show that more significant environmental benefits are achieved when the RL collection rate is increased, and the less impactful technology option makes the recovery of materials. The results of net environmental impacts indicate that recycling a greater amount of LIBs waste by Brazil's hydrometallurgical technology results in net environmental benefits across 13 impact categories. The scenario 4 presents the best performance, with an increase in the collection by RL to 50% of the waste generated and a reduction in the landfill rate by 44%. Three impact categories (Ozone depletion category, Particulate matter category and Water resource depletion category) in scenario 4 do not present the best net impact results due to the increase in the consumption of inputs such as NaOH, glucose, Na_2CO_3, limestone and particulate matter emissions for recycling a greater amount of LIBs waste. Therefore, further improvements in the hydrometallurgical process, such as recycling efficiency, reducing the use of chemicals, controlling particulate matter, and using glucose sources from agro-industrial waste, can improve the environmental profile of hydrometallurgical recycling and LIBs waste management in general.

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ELECTRONIC WASTE TREATMENT FLOWS IN NORWAY: INVESTIGATING RECYCLING RATES AND EMBODIED EMISSIONS

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ABSTRACT
Norway is one of the countries in Europe generating the most waste from electrical and electronic equipment (WEEE) per capita. Extended producer responsibility schemes are incorporated as part of the national waste policy, with clear goals towards recovery of materials from the waste fraction. Investigating the WEEE flows in Norway, we observe clear improvements needed in the transparency of the sector, and based on the information gathered, we estimate lower recycling of materials than provided through official statistics based on reporting. 68% of WEEE sent to recycling treatments are recycled into reusable material. Accounting for WEEE occurring outside of the treatment system, only 58% is recovered for recycling. We also estimate the CO_2-equivalent emissions of different End-of-Life treatments of WEEE, and the embodied CO_2-eq emissions of each WEEE category, illustrating 1) what category carry the largest environmental burden with respect to its embedded materials, and 2) the environmental impact of specific treatment options within the system. We show how the recycling rate of precious metals have significant influence over the environmental impact recovery potential of the system. It is not just the amount of material that is recycled that is important, including a proxy for expended emissions effectively illustrates the need for more precise policy implementation to ensure a functional circular economy.

1. INTRODUCTION

Collection and safe treatment of Waste from Electrical and Electronic Equipment (WEEE) has been established through regulations, both in Norway and in the EU, with a strong focus on avoiding emissions of hazardous compounds embedded in the waste products (Oguchi et al., 2013). In recent years the focus has expanded to include material recycling of WEEE, with a recent strategic policy shift to recovering critical raw materials (CRMs). The European Commission has introduced the Critical Raw Materials Act, with a clear push towards improving sustainability and circularity within a commodity market with strategic importance for future renewable energy and technology sector growth in Europe (European Commission. Joint Research Centre., 2021).

Waste data from Eurostat and the Norwegian statistical agency (SSB) shows that Norway and the other Scandinavian countries are generating and collecting significant amounts of WEEE per capita compared to the rest of the EU. In 2018 the consumer WEEE per capita collection was 14.5 kg for Norway and 7.8 kg for the EU (Constantinescu et al., 2022).With an average recycling rate of 80.5% between 2015 and 2021 reported by official statistics (SSB, table 10513), one could naturally assume that the system for WEEE management is a significant contributor to the circular economy.

Multiple studies have investigated both the economic and the environmental effects of collecting WEEE for recycling (Messmann et al., 2019; Peng & Shehabi, 2022; Rodriguez-Garcia, n.d.). Ismail & Hanafiah (2019) provide a review of 61 Life Cycle Assessment (LCA) studies performed between 2005 and 2018 on WEEE collection and treatment. Although the quality of the studies was high, one apparent weakness is a general lack of holistic or full-system perspective when analyzing WEEE treatment. Most studies focus their environmental analysis on a limited number of EEE products and/or potential treatments, these studies do generally show significant environmental benefits of recycling WEEE (Ismail & Hanafiah, 2019). A study by De Meester et al., (2019) provide a mixed Material Flow Analysis (MFA) and LCA assessment of WEEE treatment in Belgium, also a country with highly developed collection and treatment systems for waste. The study shows that there is significant material loss in the treatment system due to sorting activity, and significant functional loss due to downcycling of materials, effectively embedding precious

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metals and other critical materials in lower value recycled materials (De Meester et al., 2019). Multiple studies have highlighted the same issue with sorting and separation of critical/precious metals from WEEE in Europe (Bigum et al., 2012; Parajuly et al., 2017; Ueberschaar et al., 2017).

Recently the complexity of modern electric and electronic equipment (EEE) has gained more attention in the literature, highlighting the complex composition of the resulting WEEE (Babbitt et al., 2020, 2021; Graedel et al., 2022), and the various treatments available for extended recovery of CRMs (Chen et al., 2021). As this brief review of the literature shows, collection and treatment of WEEE is a significant challenge in terms of separating and valorizing CRMs embedded in the waste stream. However, due to large metal fractions embedded in the waste, recycling generally provides significant benefits compared to landfilling or incineration (Ismail & Hanafiah, 2019).

In Norway few studies have been conducted on the WEEE management system specifically. One study by Baxter et al., (2016) performed an LCA of the downstream treatment of three different kinds of WEEE (refrigerator, mobile phone, and LCD TV). The study shows benefits of recycling metals and highlight the importance of proper handling of smaller trace elements like refrigerator gases and recovery of precious metals as important drivers for GHG emissions (Baxter et al., 2016). Others have analyzed trends in WEEE generation, collection, and alternative routes of disposal outside the Extended Producer Responsibility (EPR) scheme adopted in Norwegian legislation for the collection system (NORSUS, 2021). No full system assessment of the WEEE management has been performed for Norway using LCA or MFA methodology.

Within the Norwegian waste regulation framework, waste policy towards recycling and reuse is clearly formulated and provided as requirements for EPR companies. The regulations set specific requirements for the recovery of eight specific WEEE product categories. These requirements are set as a percentage of the total mass of each product category to be either recovered in the form of reuse, material recovery, or energy recovery (incineration). The recovery target can vary from 75% to 85% when including energy recovery, or between 55% and 80% when excluding energy recovery of products put on market (Avfallsforskriften, 2004). These regulations are clear indications of waste policy, providing a push for increased material recovery. However, as pointed out by Bigum et al., (2012) in the case of Denmark, and Baxter et al., (2016) in the case of Norway, policy focused on optimizing aggregated and simple quantitative measures, like bulk recovery rates and recycling rates, might fail to address significant challenges within the system, or fail to incentivize recovery of small, but important materials.

In this paper we first focus on identifying the flows of consumer WEEE in Norway, addressing both the collection within the EPR schemes, and the potentially missed waste occurring outside regulated collection. We then focus on the flows of materials, and the embedded emissions of these materials. This approach is taken to assess the flow of already expended emissions within the system, and to investigate how much of our GHG emissions are being kept within the economy with the potential to substitute raw materials. Does the current system adequately recover and maintain the environmental load associated with producing consumer electronics? We address these questions by first investigating the reported flows and interviewing EPR companies, and use this data to produce a static material flow analysis model for WEEE flows in Norway. Material flow indicators are supplemented with a greenhouse gas footprint approach using LCA calculation software.

2. METHOD AND APPROACH

In this research work we gathered information from official databases, literature, and direct communication with WEEE management industry on the collection and treatment of WEEE. We follow the material flow of consumer WEEE from collection to end treatment, assessing the waste flow composition. We characterize the content of WEEE using reference products, covering iron/steel containing metals, non-ferrous metals, plastic, glass, and some precious metals. To assess the embodied emissions of different materials embedded in the waste, we use a life cycle assessment screening method to calculate the CO_2-eqvivalent (CO_2-eq) emissions associated with production, and the life cycle emissions generated by recycling, incineration, or landfilling of the respective materials. The term CO_2-eq emissions encompasses "..the amount of carbon dioxide (CO_2) emissions that would cause the same integrated radiative forcing or temperature change, over a given time horizon, as an emitted amount of a greenhouse gas (GHG) or a mixture of GHGs" (IPCC, 2018).

2.1 Waste generation and composition

The generation of WEEE is taken from the Environmental Agency's online database for producer responsibility reporting on electric and electronic waste. The data is generated by approved EPR companies, and report yearly quantity of different categories of waste collected, and the respective aggregate downstream treatments for each category (Table 1).

The amount of WEEE within each category is gathered from the database itself, however, the database has no information on the material composition of each category. To assess this, we gathered composition data from one product within each category, using this as a proxy for material composition. Table 2 shows the composition assumptions used and the reference product assumed for each category.

2.2 WEEE Flows

The total mass flows of generated e-waste are based on data from the Norwegian Environmental Agency reported by the return companies (Miljødirektoratet, 2023). In total, 142,000 tons of e-waste is collected annually (based on a four-year average of 2019-2022) in Norway, which contains all eight product categories including the large industrial equipment and large industrial cables which are not considered in this paper. The large industrial equipment is approximately 13% of the collected e-waste and the large industrial cables are approximately 12%. Excluding the in-

TABLE 1: Average share of WEEE category going to different waste treatments 2019 – 2022 according to the Norwegian Environmental Agency.

WEEE Category	Product mix	Total share (%)	Reuse	Landfilling	Energy recovery	Material recovery
1	Heating and cooling equipment	20%	0%	0%	3%	16%
2	Screens, monitors and equipment containing screens with a surface of more than 100 cm²	6%	1%	0%	0%	4%
3	Light sources	1%	0%	0%	0%	1%
4	Other large products where one of the outer dimensions is over 50 cm other large	36%	0%	2%	4%	30%
5	Other small products where the longest outer dimension is less than 50 cm	30%	0%	1%	4%	24%
6	Smaller IT and telecommunications equipment where the longest outer dimension is less than 50 cm	7%	1%	0%	1%	5%
Total		100%	2%	4%	12%	82%

TABLE 2: Material composition for reference products within each WEEE category based on Baxter et al., 2016 (a), Filimonau et al., 2021 (b), Fulvio et al., 2015 (c), Gallego-Schmid et al., 2018 (d).

Product	Steel	Aluminum	Plastic	Glass	Copper	Precious metals	Source
Refrigerator	73.9%	0.7%	24.7%	0.0%	0.60%	0.0%	(a)
Television (LCD screen)	34.7%	8.1%	44.5%	5.8%	1.7%	5.2%	(a)
CFL Lamp (Compact fluorescent lamp production)	12.2%	7.8%	23.1%	45.7%	6.8%	4.4%	(b)
Dishwasher	78.8%	0.9%	16.9%	0.0%	2.9%	0.4%	(c)
Microwave	66.4%	5.8%	7.4%	11.4%	6.1%	2.9%	(d)
Smartphone	3.0%	2.0%	59.0%	16.0%	15.0%	5.0%	(a)

dustrial waste, the amount of collected e-waste that will be considered in this study is approximately 107,232 tons per year.

The collection and first-hand treatment of WEEE we consider rather transparent, however, the recycling downstream treatment chain is significantly lacking in transparency with low reporting standards, thus leading to certain model assumptions being necessary. Assessing the treatments within the system is limited to available general treatment options in the latest Ecoinvent database (3.9.1).

We added estimates on WEEE being generated but unavailable for the collection and treatment system due to scavenging and illegal export, leading to uncontrolled disposal and highly speculative end-use of the waste. This was estimated by NORSUS (2021) at approximately 10% of EEE put on market. Based on this we estimate that 10,921 tons of WEEE is generated outside of the collection system and illegally exported with no consequent control of the material flow. We assume this category of WEEE is a mix of all categories usually collected within the EPR system.

Using waste composition analysis from residual waste generation in the households (CIVAC, 2021), we can estimate the likely collection of 1-2% of WEEE going directly to municipal solid waste incineration. We assume that this category is heavily represented by smaller WEEE like phones, lamps, electric toys, smaller electrical equipment etc. (NORSUS, 2021). We estimate at that at least 7,500 tons of WEEE are directly sent to incineration.

The total amount of consumer WEEE collected for treatment and other illegitimate End-of-Life (EOL) options are estimated at 125,653 tons per year. Table 1 in the supplementary information shows the amount of each material within each category.

2.3 Treatment assumptions

Collection and manual sorting: Manual sorting is assumed to occur at collection and sorting facilities operated by EPR-companies or affiliated companies with permission from the Environmental Agency to store different kinds of electronic and hazardous waste. The reuse fraction is assumed to be sorted manually according to Table 1. Parts of category 2 and 6 is assumed to be sorted manually due to having high composition of precious metals (phones, laptops, etc.). Based on the WEEE products accepted at UMICORE plants for recycling, we assume that most EPR companies and first order collectors of WEEE would avoid shredding these categories. However, we assume that 20% of these fractions could be shredded together with other electronic waste.

Collection and shredding: Most categories are assumed to undergo shredding to efficiently sort out metallic fractions from the WEEE (Bigum et al., 2012; Baxter et al., 2016). Most of the ABS plastic fraction is assumed to be sorted out for incineration due to hazardous compounds embedded in the plastic (Baxter et al., 2014). A mix of shredder residue from all fractions is assumed to provide the remaining fraction going to incineration. The amount going to landfill from sorting operations is assumed to mainly be glass or other inert masses contained in WEEE.

Recycling yields: Table 3 summarizes the recycling yields assumed for each material fraction making it to material recovery. We assume that all loss is landfilled from recycling plants, except for plastic recycling, where the loss fraction is sent for incineration.

The underlying assumption of waste generated and the specific treatment options within the EPR scheme is based on the assertion that the treatment statistics show fractions sorted for specific treatments and does not include any estimate of recycling loss. Effectively not reporting recycling rates, but the level of sorting for recycling, excluding actual yields. Recycling loss is introduced in the model from the review of the scientific literature referenced in table 3. Communication with industry failed to get a definitive clarification on this assumption (personal communication, 2023), the Environmental Agency is also unclear regarding the nature of the reported numbers (personal communication, 2022).

Incineration and landfilling: These treatments are based on Ecoinvent life cycle inventory models for average treatments and are referenced in section 2.5.

2.4 Material Flow Analysis model

MFA has been widely used within the research field of waste management in order to represent complex systems within an analytic framework, providing a systematic and quantitative overview of any given system over space and time (Allesch & Brunner, 2015; Hendriks et al., 2000). Following the data gathered in section 2.1 to 2.3, we construct a simple static system, representing an average flow of WEEE being generated in Norway, and sent to specific treatments. The system is mass balanced and provides a full account of mass and substance flows within the system boundary.

Figure 1 illustrates the MFA system definition, showing inputs to the system (I) in terms of WEEE collected/delivered to waste facilities of different kinds, and electronics found in the residual bin of households. The system is mass balanced by subtracting the flow of outputs (O) from the inflows. The outputs are further specified into A) Recycling yield, B) Incineration, C) Landfilling, and D) Illegal export.

The recycling rate of the total WEEE system is then calculated by dividing the recycling yield (A), over the sum of all outputs of the system:

$$Recycling\ rate = \frac{A}{O}$$

The transfer coefficients of each process (collection and sorting, material recovery, recycling, incineration etc.) is based on section 2.3, while the input to the system is based on section 2.2. Material flowing through the system are based on section 2.1.

2.5 Embodied category and EOL CO_2-eq emissions

Categories of embodied emissions are calculated based on the material composition of the WEEE reference products and the respective Ecoinvent Life-Cycle-Inventory (LCI) data. Further information on Ecoinvent inventories used in the CO_2-eq footprint calculations is available in the supplementary information. The impacts of production and treatment was scaled up by the respective demand for each material within each category and treatment option and multiplied with environmental impact indicators taken from the IPCC2021 GWP100 method made available by ecoinvent. This quantifies the embodied impacts of production in terms of CO_2-eq emissions over the time-horizon of 100 years (Forster et al., 2021), and include the CO_2-eq emissions due to EOL recycling, incineration or landfilling of the composite materials. The calculations were performed using the Brightway open-source framework for LCA calculations in python (C. Mutel, 2017). This approach does not constitute a full life cycle assessment, but an LCA screening approach similar to other footprint calculation approaches within the environmental assessment literature (Hung et al., 2020; Kaufman et al., 2010; Mora-Sojo et al., 2022; Song et al., 2017).

TABLE 3: Recycling yield based on literature assessing the mass balance of recycling activities, yield indicate the share of material output from a recycling process that can substitute a primary material.

Material	Recycling Yield	Source
Steel	87%	(NORSUS, 2022)
Aluminium	75%	(Bigum et al., 2012)
Plastic mix	70%	(Rigamonti et al., 2020)
Glass	90%	(Haupt et al., 2018)
Copper	60%	(Pfaff et al., 2018)
Palladium	90%	(Bigum et al., 2012)
Gold	90%	(Bigum et al., 2012)
Silver	90%	(Bigum et al., 2012)
Nickel	90%	(Bigum et al., 2012)

3. RESULTS

The following section show the results of the identified WEEE flows and the embedded materials, and the embedded CO_2-eq emissions of materials and the added emissions of electronic waste EOL treatment. The recycling rate of materials is estimated, and consequently the reintroduction of embedded emissions is also estimated and illustrated with the use of Sankey diagrams.

3.1 Material flows

Combining the data on WEEE collection and treatment with the literature data on the material composition of WEEE products, in Figure 2 we can observe the distribution of materials between the main categories included in our system.

Categories 1, 4 and 5 are dominated by steel, with some plastic mix, glass and copper making up the rest of the main mass. Categories 2, 3, and 6 are mainly dominated by plastic mixes. We also observe significant amounts of steel and plastics not entering the EPR-scheme via residuals and illegal exports.

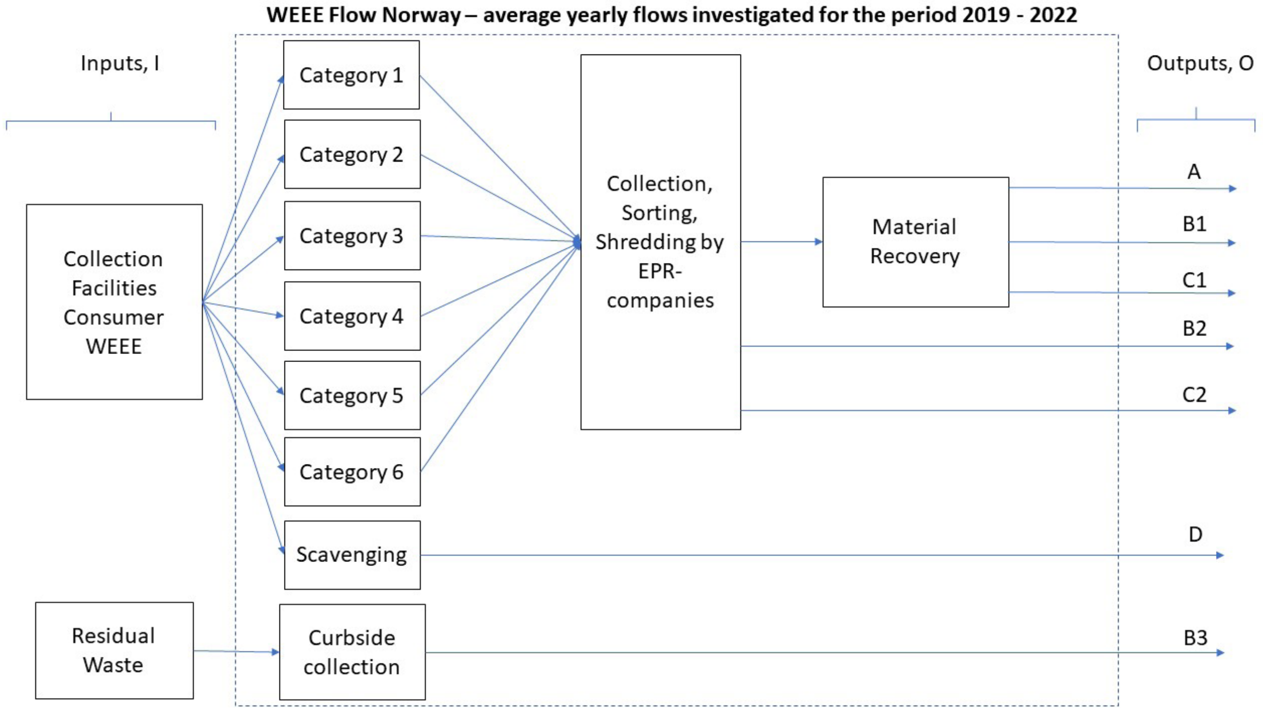

FIGURE 1: Material Flow Analysis system description for calculating the material flows and the recycling rate of the WEEE system. The flows labeled A) for recycling, B) for incineration, C) for landfilling and D) for illegal exports.

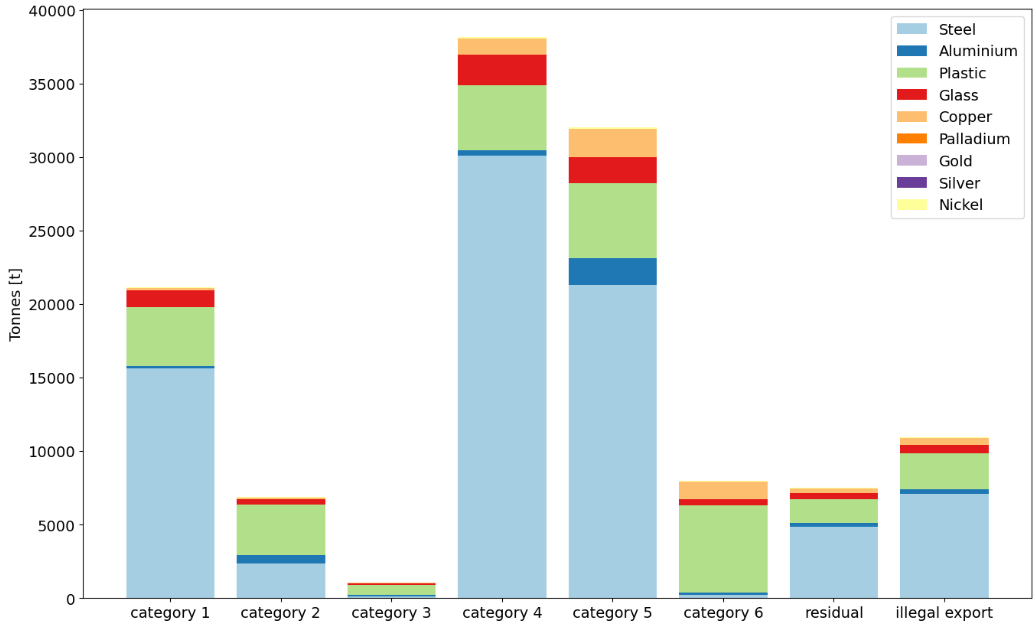

FIGURE 2: Bar graph showing the total yearly mass per WEEE category and its subsequent share of embodied materials.

Figure 3 shows the mass flow of the WEEE management system in Norway, including consumer electronics discarded as part of household residual waste, and electronics scavenged from collection facilities. Based on the assumption that the official statistics are not including losses occurring at the recycling plants, we can observe that 68% of category 1-6 is recycled. 2% is reused. If we include the amounts generated outside the EPR scheme, 58% of generated WEEE is recycled into materials.

Here we denote the recycling and reuse of materials into valuable goods "Circular Material Use", meaning that the system provides a flow of materials that can potentially be used, and contribute to a circular economy. We denote the end treatments that destroy, store, or in other ways lose materials from the downstream treatment chains, making them inaccessible for further valorization, as "Linear Material Loss". This highlights that the loss of material in terms of export, the incineration of materials, and the landfilling of ma-

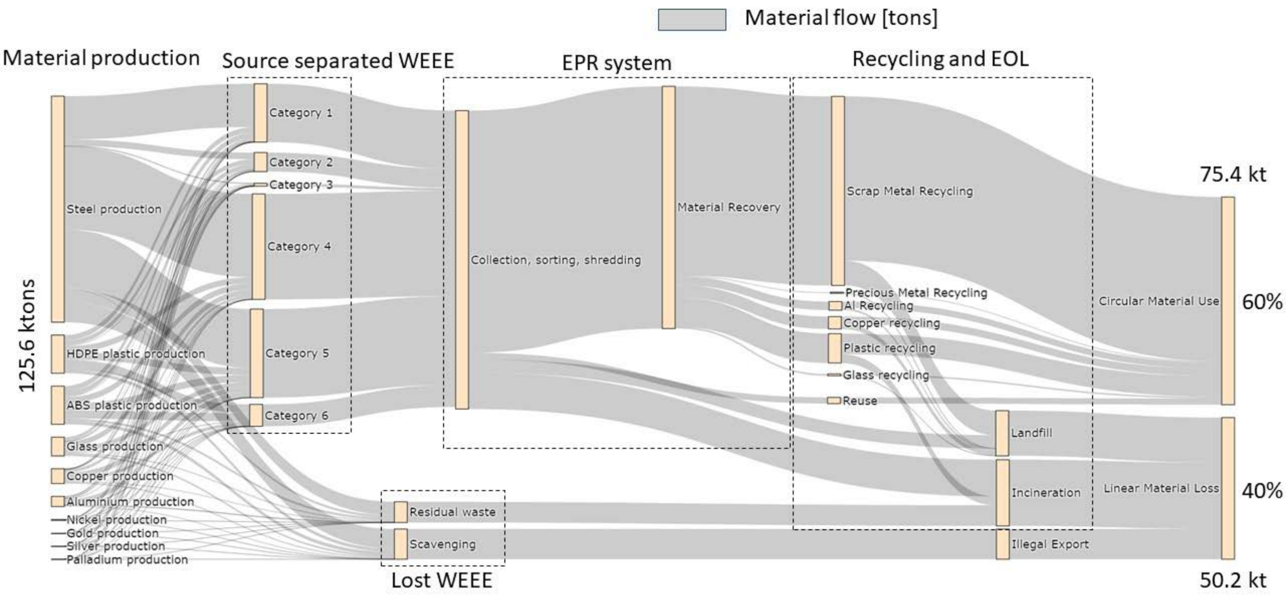

FIGURE 3: Mass flow diagram of material production, collection within and outside the EPR scheme, and materials going to Circular Material Use, or Linear Material Loss based on collection raters and recycling yields.

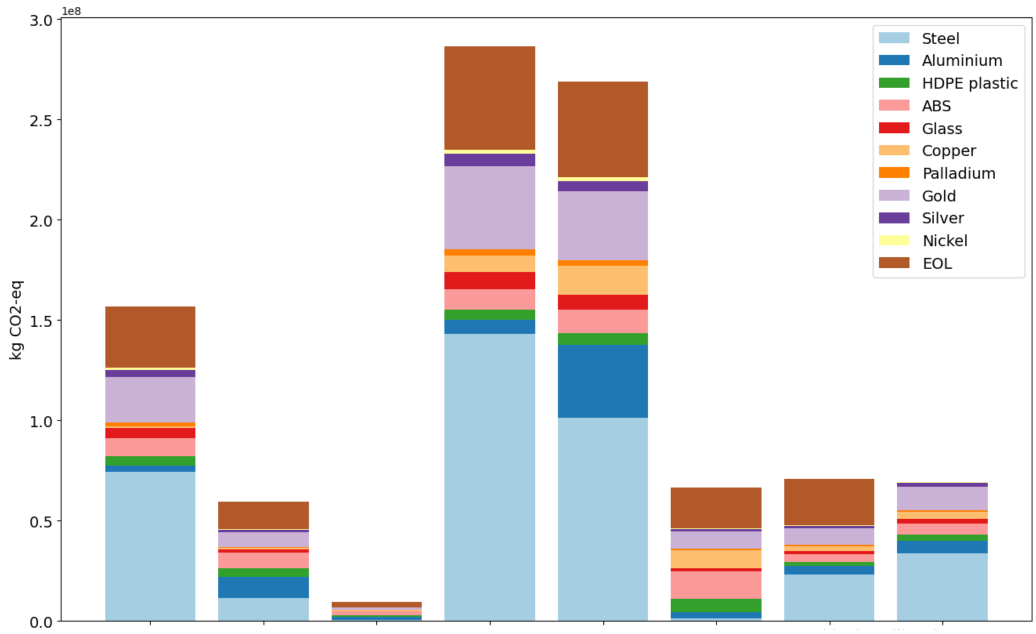

FIGURE 4: Kg CO_2-eq emissions embedded in material production within each waste category, and End-of-Life (EOL) treatment emissions due to incineration, landfilling or recycling processes.

terials, is directly linked to a linear system model, where we extract materials and dispose them with no significant value added in the process (Romero-Hernández & Romero, 2018).

3.2 Embedded carbon footprint and EOL GHG emissions

Figure 4 include the embodied CO_2-eq emissions of the materials within each category and the added aggregate EOL emission. Steel is dominant for the larger categories, however, we see the significant contribution of gold, silver, and nickel, as well as aluminium for smaller fractions. Materials that occupy a tiny fraction in terms of the total mass of a product, still contribute a significant share of the emissions associated with the production of the product. We can also observe that the EOL emissions are comparatively small to the emissions of the embodied materials within the discarded electronic products.

Figure 5 further disaggregates the emissions from EOL into specific treatments. In terms of CO_2-eq emissions from downstream waste treatment, we see that incineration of plastic and shredder debris contribute significantly to the emissions within each WEEE category.

We see that sanitary landfilling of waste have low emissions of CO_2-eq, emissions are mainly from operating vehicles and maintaining landfill operations and infrastructure. We can observe that the treatment of precious metals dominates much of the emissions from EOL recycling treatments. Following a contribution analysis of the results for gold recycling, we can observe that the main emission drivers are high energy use. This being the case for all precious metal recycling processes. Energy use, sorting and pressing, and quicklime production are the main contributors to CO_2-eq emissions within the steel recycling process. In terms of incineration, the process-specific emissions are the most important component, especially since the main waste type sent for incineration is plastic, with high carbon content. (Fiore et al., 2019; Stubbings et al., 2021)

We add the embodied CO_2-eq emissions of materials in WEEE to the material flows shown in Figure 3 and add the EOL CO_2-eq emissions (orange flow) from each EOL treatment, showing the flow of emissions through the WEEE treatment system in Figure 4.

This helps us to illustrate the impacts associated with both production and EOL treatment and the large amounts of embodied emissions being directed back to the economy, or to Linear Material Loss. Figure 6 is based on the official information provided by the Norwegian Environmental Agency, assuming valorization of all material content. In Figure 7, instead of assuming precious metals (PM) going to Circular Material Use, we assume that 50% of these metals are downcycled into steel, other metallic fractions, incinerated or landfilled, which in either case means that the PM is not actually reintroduced in a functional way to the circular economy, they are "functionally lost" (Ortego et al. 2018).

We see from Figure 7 a 50/50 split of total emissions between Linear Material Loss and Circular Material Use. Redirecting a tiny fraction of specific materials contribute to a 9% shift towards loss of materials. Indicating that a significant amounts of expended CO_2-eq emissions are directly lost and permanently unavailable for reintroduction to the circular economy, requiring continued extraction of primary raw material, and consequently a substantial potential for increasing GHG emissions if we are to sustain or grow our consumption of consumer electronics.

4. DISCUSSION

This material flow and life cycle emissions screening approach shows that the embodied emissions of EEE products is mainly coming from metals, with significant shares from precious metals. The CO_2-eq emissions from production of materials embedded in the WEEE categories is estimated at approximately 320 kg CO_2-eq / household in Norway. These results where comparable to previous studies on GHG emissions from EEE in Norwegian households (Hertwich & Roux, 2011). Emissions from EOL is reported by Hertwich & Roux (2011) at 43 kg CO_2-eq per household, where our results indicate approximately 75 kg CO_2-eq per household. Our assessment likely overestimates the amount of precious metals going to recycling, which is the main contributing processes to EOL emissions within our system. It is quite likely that larger shares of precious metal containing category waste is downcycled.

The material composition of each category of WEEE is a significant source of uncertainty in this analysis, however, the data considered does give some useful indication of the potential composition and embodied emissions due to primary production. The lack of transparency into downstream treatment chains within the WEEE management sector is also a significant challenge for assessing both the resource recovery efficiency of different recycling paths, and the environmental impacts generated for these treatments. These issues have been highlighted by other

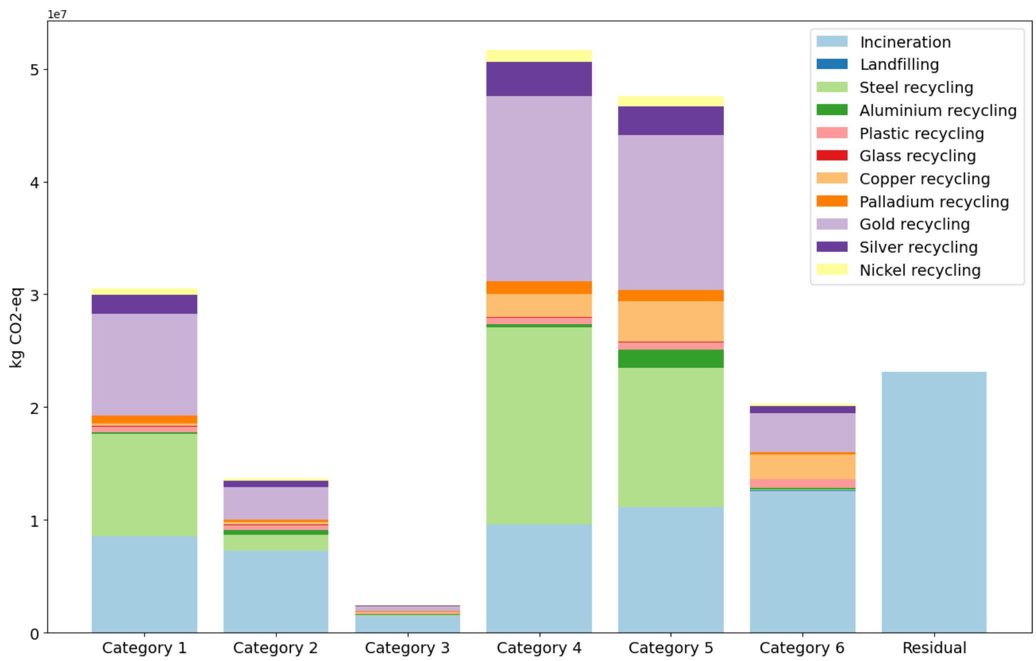

FIGURE 5: Breakdown of CO_2-eq emissions for each End-of-Life treatment, per waste category.

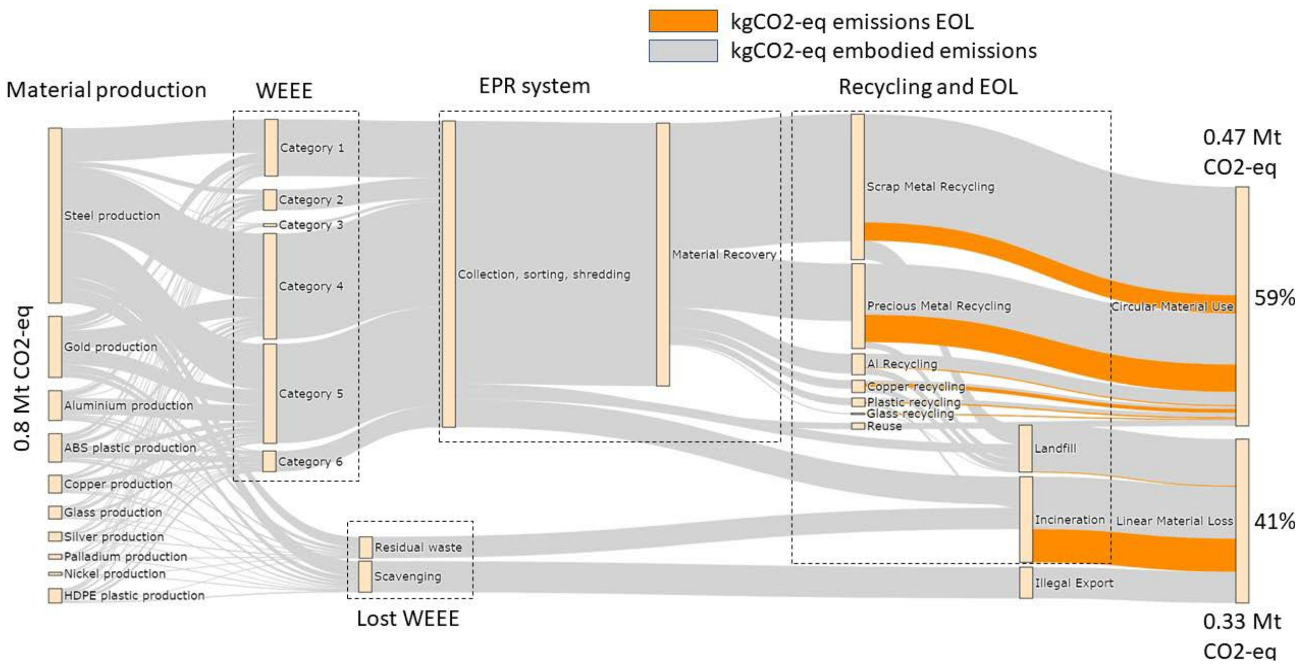

FIGURE 6: Emission flow layer showing embodied CO_2-eq emissions from material production, and emissions occuring at EoL treatments, baseline assumption is recycling of precious metals.

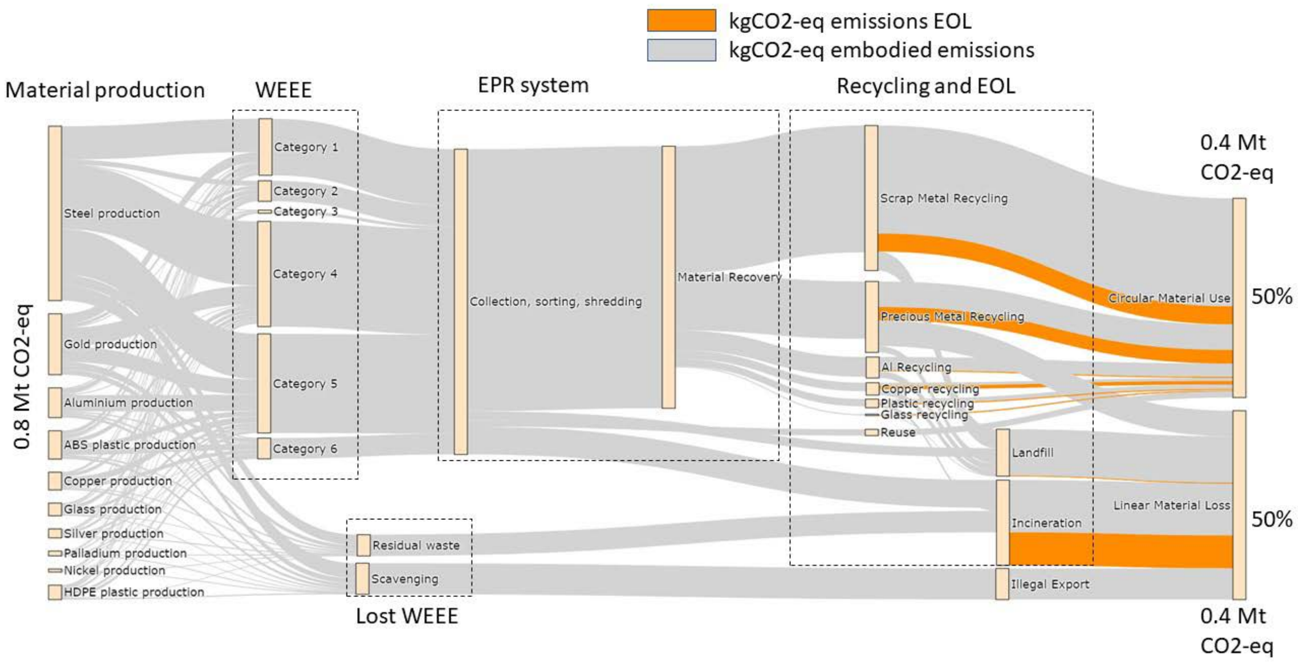

FIGURE 7: Emission flow layer showing embodied CO_2-eq emissions from material production, and emissions occuring at EoL treatments, Scenario with 50% less recycling of precious metals.

studies assessing the environmental impact of WEEE in specific countries or Europe in general (Di Maio et al., 2015; De Meester et al., 2019; Horta Arduin et al., 2019). A recent comprehensive comparative LCA study on gold recycling from WEEE, highlights the variability of human health impacts and energy requirements for the most conventional gold recycling techniques (He et al., 2023). Better resolution on the specific treatments used for recycling is of crucial importance when indicating environmental performance. Amending existing gold refining techniques (Fernando et al., 2019) and optimizing gold refinery operations (Mahyapour & Mohammadnejad, 2022) has a significant part to play when it comes to lowering the environmental burden of continued gold use.

Using waste treatment life cycle inventories available in Ecoinvent, we have added an emissions layer to our material flows, showing that there exists a significant advantage to recycling materials compared to primary production of materials. However, the model approach is simplistic, and as pointed out by Fortier et al., (2022) in the 2021 annual review of critical minerals by the United States Geological Survey; existing LCI data is highly variable and based on information on few mining sites. The LCI would not be able to account for capacity issues due to increased demand, or spatially explicit emissions (Fortier et al., 2022). Only focusing on the climate change indicator in our case, does reduce some of the uncertainty due to GHG emissions representing a global impact. In this study we do not include the environmental impacts of dismantling and shredding operations, this should be included when assessing the direct impacts on human health from WEEE treatment (Lan et al., 2023). Impacts does naturally also occur during collection, dismantling, shredding, and transport operations, for a full LCA study of WEEE, these aspects should be included in the system.

This initial study does not look carefully into the issue of downcycling and substitution rates of the recycled materials, which has been highlighted by multiple previous studies (Rigamonti et al., 2020; Vadenbo et al., 2017; Viau et al., 2020). It is unclear to what extent the output of the recycling processes could be used directly in the production of complex electronic equipment. To assess these issues, plant level studies would be needed in order to model these complex material processes.

In the results we present a Sankey flow chart of embedded CO_2-eq emissions and include the emissions being produced in the treatment of WEEE. This helps us see where the treatment emissions are occurring, what categories of WEEE have the largest footprints, and it also indicates how well the system can avoid spending emissions (in terms of incineration and landfilling) and wasting forgone emissions (embodied emissions from production) in a linear "one-time-use" system. The idea of Linear Material Loss indicate that the emissions expended while producing a certain product, permanently goes to waste. This is contrasted with the Circular Material Use, indicating a renewed life for past emissions, albeit at the cost of some recycling emissions and recycling loss. It is clear from the result that if the current system fails to incentives the delivery of critical or precious materials back into the production system for consumer EEE, we are in dire need of better policy, or radically different use-patterns for electronics.

Adding the layer of emissions to the material flow analysis helps us illustrate that materials that represent a very small fraction of the total weight of embedded materials, has a disproportional level of environmental impact. Seeing the problem in this way, exemplifies the necessity for policies that target these small but critical materials.

This concept could be extended further, looking at more aggregated impacts in terms of human health, ecosystem damage, and resource depletion. For instance using the ReCiPe2016 method for life cycle impact assessment (Hauschild & Huijbregts, 2015; Huijbregts et al., 2017; Woods et al., 2018). Making it possible to assess the impact of multiple environmental emissions, not just the once relevant for climate change. However, in order to capture these impacts to a higher degree of certainty one would need to address spatio-temporal characteristics of production and treatments of WEEE categories (Beloin-Saint-Pierre et al., 2020; C. L. Mutel & Hellweg, 2009; Pfister et al., 2020). This could be done by a combining dynamic-Material Flow Analysis with a regionalized Life Cycle Assessment, making it possible to look at multiple policy options over time, and the potential environmental effects with greater spatial resolution (Barkhausen et al., 2023). Further development of these environmental systems analysis methods is important for developing more concrete policies for resource management, especially for the advanced material demand from technology and energy sectors.

Another aspect of the waste management system that is not assessed in this paper, is the importance of consumer sorting behavior and engagement with the recycling system (Flygansvær et al., 2021). One aspect is the lifetime of the EEE products (Bridgens et al., 2019), another is the willingness to engage with the recycling system by citizens and businesses (Jabbour et al., 2023). The referenced papers highlight the importance of information on the value and benefit of recycling, which is hard to adopt in a regulatory framework. However, national policies should reflect that convenience of recycling (Wang et al., 2019) and information on recycling benefits (Jabbour et al., 2023) could promote better practice. Having transparent information on the WEEE treatment system becomes an important part of any information campaign.

Multiple review studies have highlighted the importance of improved and enforceable waste legislation for WEEE (Constantinescu et al., 2022; Patil & Ramakrishna, 2020; Shahabuddin et al., 2023). In the case of Norway, the type of strict legislation suggested by Patil and Ramakrishna (2020) does exist, but it is clearly not enough to sufficiently incentivize the recycling of smaller and more complex materials (i.e., gold, palladium, nickel etc.). Claims of recycling rates and EPR schemes successful integration of circular economy principles should be scrutinized and assessed critically. Researching the downstream treatments of the different WEEE categories proved difficult, especially due to limited information and proof regarding countries actual recovery rates. It would be advisable for policymakers to engage more directly with the existing EPR structure, and tailor regulations towards value and emission-based recycling goals, in contrast to bulk recycling requirements which is the current practice.

5. CONCLUSIONS

We show that the effectiveness of the Norwegian WEEE management system in terms of resource management is highly dependent on how well the precious metals are recovered at the EOL. There is a significant lack of transparency within the WEEE sector, and there are few indications of an effective valorization of precious and/or critical materials embedded in electronic consumer products. We argue that resource management should include embedded emissions of production, to effectively include the high environmental costs of producing as well as managing our waste. If we fail to reintroduce materials into the economy within the same product niche they initially where part of, we are at a significant risk of not curbing future demand for materials that will require substantial amounts of energy and resources to acquire. Its apparent that current policy enacted in Norway is not sufficient to safeguard the recovery of critical raw materials from WEEE. The first step to improve and incentivize for increased Circular Material Use would be to set recycling goals for specific material fractions (precious metals, aluminium, copper etc.), and introducing stricter reporting requirements, providing clear information on both the real recycling yield, and the limitations of current systems and technologies.

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COMPARATIVE LIFE CYCLE ASSESSMENT OF DIFFERENT RECYCLED CONCRETE AGGREGATES

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ABSTRACT
Concrete is recognized as the second most consumed materials in the whole world. Therefore, applying circular solutions to concrete, like recycling or reusing can guarantee a considerable benefits in terms of environmental impacts. In this paper, a comparative life cycle assessment is done for different recipes of recycled concrete aggregates in comparison to a recipe of virgin concrete aggregate, which are used in a case study called “Musterbude”. The recycling rate of aggregates used in the recipes are 45%, 60%, and 100% and they are supplied from different resources. For environmental impacts calculation, each recipe is defined as a scenario and their impacts are compared to each. The life cycle assessment results show that, despite low performance in water depletion indicator, the recipes with 100% recycled aggregates shows the best performance from environmental point of view.

1. INTRODUCTION

In the past few years, the member countries of the European Union have made several efforts to move from an economical system described as a linear economy, based on a “produce-use-dispose” model, to a circular economy, focused on the reuse and recycling of waste. Although it could provide several advantages from both an environmental, economic and even social point of view, there are still a lot of barriers to the implementation of this type of economy.

The European Directive 2008/98/EC established a target for reuse, recycling, and other types of material recovery of inert waste up to 70% by weight, and the present paper focused on one of the most voluminous waste flows generated from the EU: concrete and demolition waste. There are different solutions that would allow recycling it, but presently the most commonly used is backfilling, which is a form of down cycling.

Although the construction industry plays an essential role in a country’s economic growth, it is also responsible for activities and processes that generate high volumes of waste, which are constantly on the rise with little effort to minimize them. Indeed, the natural environment is constantly being polluted by the activities of the construction industry and the built environment as a whole. Likewise, finite natural resources that are used as raw materials for construction are increasingly being depleted due to continuous extraction (Ogunmakinde et al., 2022).

Construction activities are classified as input and output, with the former being resources that are channeled into a construction project and the latter being products generated from the project. It can be argued that all phases and activities of construction generate waste. The volume of waste generated from each phase and/or activity may vary. However, there is a need to ensure that as much waste as possible is minimized. Construction waste minimization is meant to reduce waste by determining its causes, reducing it from the source and applying ameliorating techniques (Ogunmakinde et al., 2022).

Recycling of demolished concrete and using them as recycled aggregates in concrete production can be considered as a promising solution that can guarantee a considerable decrease in environmental impacts like natural resource use, because 70-80% of concrete is made of aggregates (Alexander & Mindess, 2005). Recycling of concrete aggregates means reduction in excavation of virgin aggregates which results in natural resource preservation. On the other hand, extraction and manufacturing of concrete aggregates is responsible for 7% of total global energy, which can be saved by recycling of concrete (Hossain et al., 2016).

For this reason, research in the field of the construction industry is now focusing on the possibility to use recycled aggregates for generating new concrete, but it is necessary to assess the actual sustainability of this option, especially in comparison to producing from virgin aggregates. In

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order to reach this goal, there are several instruments that could be used, including the Life Cycle Assessment (LCA), a scientific approach that allows taking into account all the aspects of creating a product and evaluating the environmental impact associated with it (ISO 14040:2006). According to ISO 14040, LCA can model, quantify and analyze the environmental impacts of a product over its entire life cycle. ISO 14044 defined a step-by-step environmental assessment methodology to secure the reliable results for further interpretation of environmental footprint of products or services (ISO 14040:2006). In "Methodology" section, each step of LCA will be defined based on ISO 14040 and ISO 14044.

2. CASE STUDY - MUSTERBUDE

Musterbude is a small cabin built in Hamburg, Germany, using seven different recipes of recycled concrete and one recipe of standard concrete, in order to test their different technical and environmental qualities not only on a laboratory scale, but also in a scenario closer to reality.

This showcase is part of the CIRCuIT project, which aims to identify how circular construction approaches can be scaled and replicated across Europe to build more sustainably and promote the transition to a circular economy for the built environment, by delivering a series of demonstrations, case studies, events and other dissemination activities (CIRCuIT). CIRCuIT is led by university and industry partners with an interest in construction and the environment from Copenhagen, Hamburg, the Helsinki region and Greater London. Hamburg partners involved in CIRCuIT and consequently Musterbude project are Otto Dörner Kies und Deponien GmbH & Co.KG, the EGGERS Tiefbau GmbH Group, the Technical University of Hamburg, the Free and Hanseatic City of Hamburg, e-hoch-3 eco impact experts GmbH & Co.KG and OTTO WULFF Bauunternehmung, a family-run company that works in the field of constructions (Figure 1).

It was possible to conduct a life cycle assessment in order to describe the environmental impact of recycled concrete under different recipes because the OTTO WULFF Bauunternehmung GmbH in collaboration with EGGERS Tiefbau GmbH, and Otto Dörner Kies und Deponien GmbH & Co.KG, provided the data of the recipes used to obtain the different concrete mixture.

3. METHODOLOGY

Based on the framework provided by the ISO 14040 and ISO 14044 standards, it is possible to report the information gathered during a Life Cycle Assessment study, presenting the results obtained in an organic form and making it possible for different audiences to understand how the study was conducted. Goal definition, scope definition, inventory analysis, impact assessment, and interpretation are presented in the following sections.

3.1 Goal and Scope Definition

The OTTO WULFF Company, as part of the CIRCuIT project, commissioned the study and it was performed at the Technical University of Hamburg, with the use of OpenLCA, a free software purposely created to conduct Life Cycle Assessments, with the support of sophisticated information databases.

The database used in this study is Ecoinvent, version 3.7.1, which has sufficient information to perform LCA in the construction field. It is a Swiss database that contains approximately 12,500 processes organized under different themes like transport, energy, material production, agriculture, etc. All processes are available as unit- and system-processes, all of which are documented in detail (Bjørn et al., 2018).

The present study aims at establishing the environmental impact of seven different recipes of concrete (indicated as R1, R3, R4, R5, R6, R7, R8), containing recycled aggregates from demolition waste, in comparison to a reference

FIGURE 1: Musterbude.

concrete (RB), produced in accordance with the standard mixture of materials and only using virgin resources (Table 1).

The purpose of the study is to support research in the field of circular economy applied to the construction industry.

The study did not apply the standard "cradle-to-grave" approach of LCA, but it was limited to the final production of the concrete, without considering the impacts determined by the use and disposal at the end-of-life. Therefore, the study can be defined as a "cradle-to-gate" study and this approach was chosen due to the lack of information about the use and disposal phases, as the "Musterbude" was only recently constructed.

The chosen functional unit for this study is 1m^3 of concrete produced, however it should be mentioned that the stress resistance of the different recipes resulted to be extremely variable, making them fit for different use, and none of them showed physical characteristics actually comparable to those of the reference concrete.

The entire study was conducted after defining the foreground system (Figure 2).

The study is performed in Attributional LCI modelling, whose overall aim is to represent a product system in isolation from the rest of the Technosphere or economy. In fact, the main purpose of the study was to identify the best mixture in a set of options, not to consider how the environmental impact will change in accordance with variations in the technology or in the market demand.

As for the geographical boundaries, it was taken in consideration that the entire study was located in Germany, so, where possible, the providers for the different inputs and outputs processes were associated to Germany itself. However, when the information were not present for this country, they were associated to average European values, always provided by the Ecoinvent v3.7.1 database. At the same time, also the temporal settings needed to be established. The construction of the Musterbude took place between April and June 2022, so all the technologies considered were referred to this period. Moreover, as the study does not take in consideration the environmental impact associated to the use and disposal phases, possible future technological development was not taken into consideration.

3.2 Inventory Analysis

The data collected for the comparison between the recipes were relative to the quantity and type of material used in each of them, including the standard one, and they have been provided directly from the OTTO WULFF company, while all the information relative to the energy, emissions and heat used or wasted, were collected from the Ecoinvent v3.7.1 database.

The information relative to the distance covered by the trucks for the material transport, were calculated by Google Maps, based on the information about where the different activities were taking place. Transportation activity is modelled by the pre-defined flow "transport, freight, lorry >32 metric ton, EURO6" in Ecoinvent v3.7.1 database.

TABLE 1: Concrete Recipes (OTTO WULFF).

Parameter	R1	R3	R4	R5	R6	R7	R8	RB	Unit
Recycled Aggregates	299.8	400.1	296.4	233.4	607.1	606.9	233.4	0	dm^3
Virgin sand	625	715	300	661	0	0	661	787	kg
Virgin Gravel	346	0	650	484	0	0	484	1180	kg
Total Water	240	274	258	231	332	343	243	170	kg
Cement	300	300	300	300	300	300	300	300	kg
Fly Ash	60	60	60	60	60	60	60	60	kg

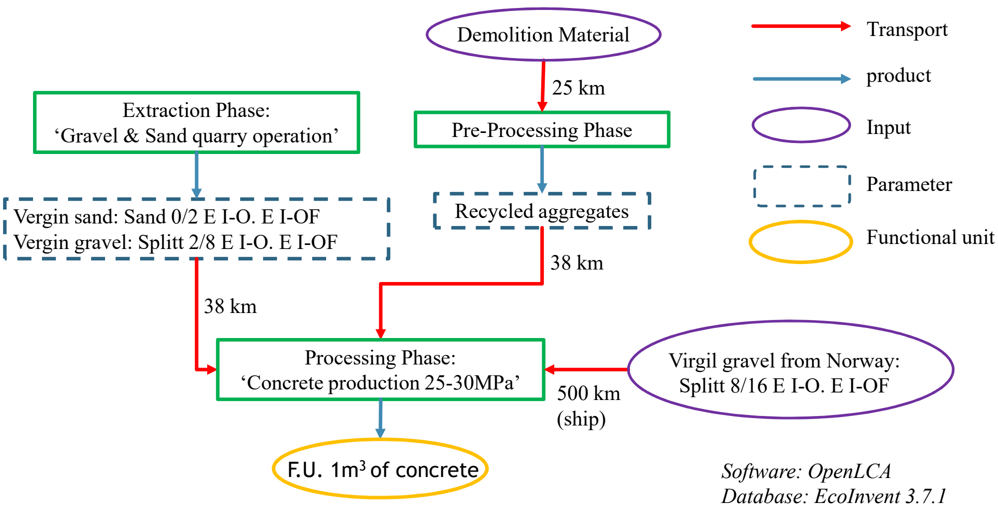

FIGURE 2: Foreground system LCA.

To compare the environmental impacts of different recipes, seven scenarios are defined:

- Reference Scenario (RB): Production of 1 m^3 of concrete with virgin aggregate (0% recycled aggregates);
- Scenario 1 (R1): Production of 1 m^3 of recycled concrete with 45% recycled aggregate;
- Scenario 2 (R3): Production of 1 m^3 of recycled concrete with 60% recycled aggregate;
- Scenario 3 (R4): Production of 1 m^3 of recycled concrete with 45% recycled aggregate (different supplier);
- Scenario 4 (R5): Production of 1 m^3 of recycled concrete with 35% recycled aggregate (different supplier);
- Scenario 5 R(6): Production of 1 m^3 of recycled concrete with 100% recycled aggregate;
- Scenario 6 (R7): Production of 1 m^3 of recycled concrete with 100% recycled aggregate (different supplier);
- Scenario 7 (R8): Production of 1 m^3 of recycled concrete with 45% recycled aggregate (different supplier).

3.3 Life Cycle Impact Assessment

The impact assessment method used in this LCA is the ReCiPe Midpoint H, which is part of the ReCiPe method, one of the most recent and updated impact assessment methods available to LCA practitioners. It addresses a number of environmental concerns and expresses them through 18 midpoint level categories, which are then weighed and aggregated into a set of three endpoint categories. In this study, it was decided to present the results only at midpoint level (Table 2).

3.4 Interpretation of Results

In this section, the significant issues from the LCA phases that are not considered in the system boundaries are mentioned and their influence on the overall LCA results will be evaluated. The interpretation of results deals with the identification of the significant issues from the other phases of the LCA as well as with the evaluation of their influence on the overall results of the LCA and the completeness and consistency with which they have been handled in the study. Then, the results of the evaluation are used in the formulation of conclusions and recommendations from the study.

The present study was conducted with the information provided by the OTTO WULFF construction company, which was directly responsible for the production of the eight types of concrete. Such information included the composition of each recipe, with a high level of detail in regard to the amount of consumed water, the quantity of cement and fly ashes, and the amount of recycled and virgin aggregates used, also grouped according to their dimensions and origin.

During the production of the recycled aggregates, some information regarding the type of instruments were provided as well, however, there was a lack of information about the amount of energy they used. For this reason, the study was conducted using as inputs some engines that belonged to the same category and had similar technical characteristics, but above everything, the only parameter needed to calculate their impact was the amount of material that they processed. Nonetheless, it is logical to suppose that the production of material with different granulomeres dimensions could determine different amounts of energy consumed by the machines, and, consequently, generate different impacts. Unfortunately, this was impossible to determine, but it is supposed to have little influence on the results, as the impact associated exclusively to the production of the recycled aggregates represents only a small fraction of the total impact described by the LCA.

One more element that should be taken into account is that at the beginning of the LCA it was assumed that the demolition activities were the same in all scenarios, so both in the case of the recycled and the standard concrete; this is far from reality, as, in order to create the recycled aggregates, it is necessary to conduct a selective demolition, which implies the use of different instruments from a standard demolition and takes more time. Generally, a selective demolition has a higher influence on the costs than on the environment and, for this reason, it was decided that the energy used for the demolition in any scenario was the same. Nonetheless, the important effects of demolition technique and treatment equipment of concrete waste on the performance of the recycled aggregates should be emphasized, since they lead to different mechanical properties and durability of recycled aggregates, and influence the energy consumption and environmental emissions in producing them. Hence, the life cycle inventory of 72 recycled aggregate should include its production modes, the effects of demolition techniques, treatment equipment and size of aggregate (Zhang et al., 2019).

In addition, it was assumed that, according to the law in Germany, no waste was generated from both the demolition and all of the other activities, as concrete waste is always used for backfilling. One additional choice was not

TABLE 2: ReCiPe Midpoint H (Ecoinvent 3.7, 2021).

Indicator	Acronym	Unit
Agricultural land occupation	ALOP	m^2a
Climate change	GWP	kg CO_2- Eq
Fossil depletion	FDP	kg oil-Eq
Freshwater ecotoxicity	FETPinf	kg 1,4-DCB-Eq
Freshwater eutrophication	FEP	kg P-Eq
Human toxicity	HTPinf	kg 1,4-DCB-Eq
Ionising radiation	IRP_HE	kg U_{235}- Eq
Marine ecotoxicity	METPinf	kg 1,4-DCB-Eq
Marine eutrophication	MEP	kg N-Eq
Metal depletion	MDP	kg Fe-Eq
Natural land transformation	NLTP	m^2
Ozone depletion	ODPinf	kg CFC-11-Eq
Particulate matter formation	PMFP	kg PM_{10}- Eq
Photochemical oxidant formation	POFP	kg NMVOC
Terrestrial acidification	TAP100	kg SO_2- Eq
Terrestrial ecotoxicity	TETPinf	kg 1,4-DCB-Eq
Urban land occupation	ULOP	m^2a
Water depletion	WDP	m^3

considering the recycling of the steel incorporated in the concrete to be recycled, mainly because the aim of the study was to define the environmental impact of recycling the concrete waste exclusively and also to avoid dealing with the problem of allocation, even though the energy needed for separating the steel from the concrete in the different scenarios could have slightly altered the impact results if taken in account.

4. RESULTS AND DISCUSSION

The results obtained by this LCA study provide useful information for establishing the mixture, which has the lowest environmental impact. This information can be useful for further studies on other recycled aggregate recipes or as a supporting tool for decisions on future construction.

Climate change indicator, which shows the equivalent amount of CO_2 emissions into the air, indicates the lowest amount in R6 and R7 compared to standard recipe (RB) (Figure 3). This reduction can be attributed to elimination of virgin material extraction phase, and part of transportation phase, because no virgin aggregate is being consumed in R6 and R7.

The spider grams shown in Figure 4 and Figure 5 present the impact associated to each recipe by broken lines that connect their standardised values for each impact category. In both figures, there is a red broken line, which indicates the environmental impact of the average concrete, used as a reference for normalizing the impact values of the other recipes. All the points outside of its confined area correspond to higher values of impact, while the points inside its area indicate a lower impact.

The points of the broken lines of both recipes R6 and R7 fall closer to the center of such area, in comparison to those of the other mixtures, indicating that they have the lowest impact values, which are identical for almost all of the considered impact categories, due to the fact that R6 and R7 have extremely alike compositions and characteristics, as shown in Table 1.

These two recipes are made exclusively of recycled aggregates, which is the reason behind the extremely low values of impact in certain categories such as "Natural land transformation" (NLTP), with an impact of -41%, followed by the recipe R3 with -30%, or "Urban land occupation"(U-

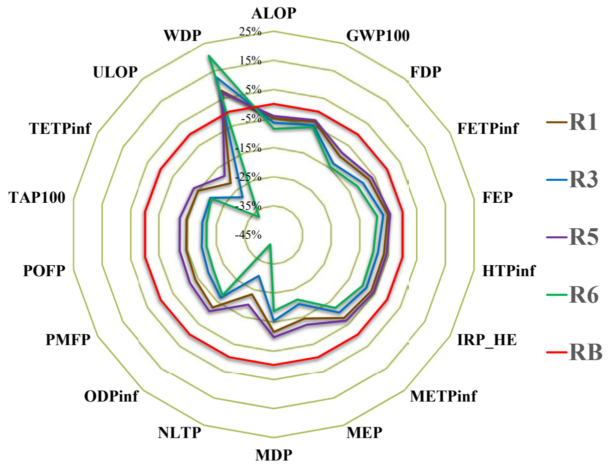

FIGURE 4: Standardised Life Cycle Impact Assessment results, part 1.

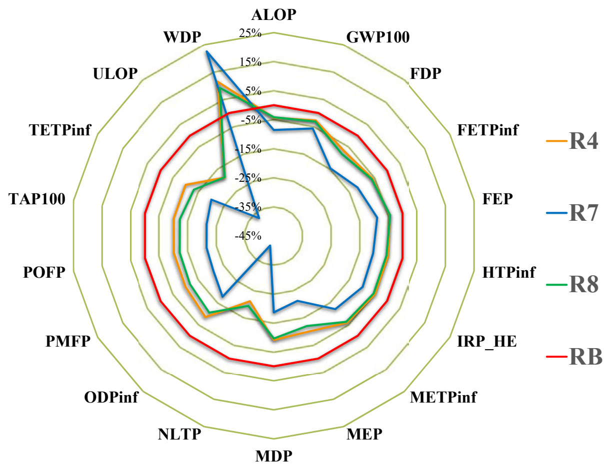

FIGURE 5: Standardised Life Cycle Impact Assessment results, part 2.

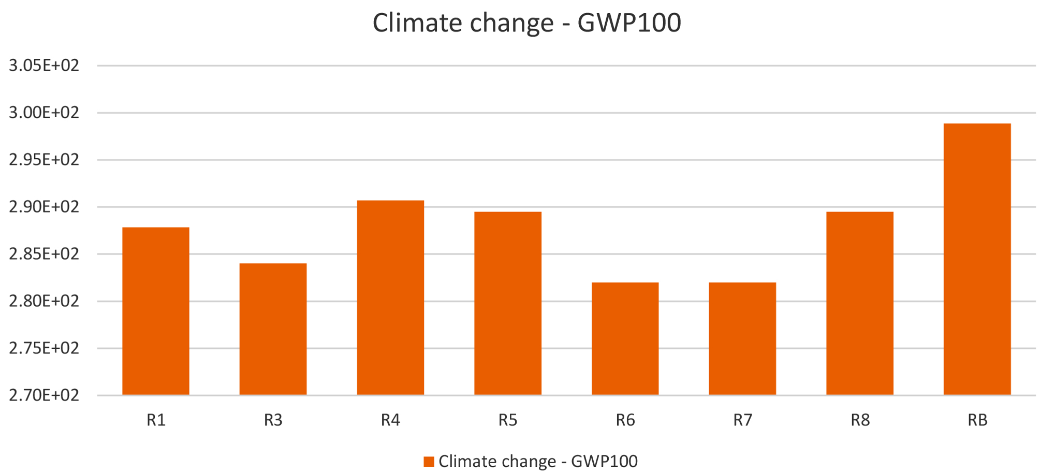

FIGURE 3: Environmental Impacts of different recipes in terms of Climate Change indicator.

LOP), with a value of -38%, while the next best result is for R3. The impact in these categories is mostly connected to the extraction phase but considering that, the two mixtures use no virgin aggregates; this phase does not take place for them, which has a positive influence on the reduction of the overall environmental impact of concrete production. For other impact categories, it is still possible to notice a reduced impact if compared to an average concrete, but the gap is more similar to that of the other recipes.

On the contrary, in the case of the "Water Depletion" (WDP) category the impact is higher in comparison to the standard recipe. Indeed, the water consumption was found to be 21% (R6) and 23% (R7) higher than the reference, due to the fact that recycled aggregates have a larger surface area and are usually drier than natural aggregates, causing a need for additional water (Knoeri et al., 2013; Ding et al., 2016).

5. CONCLUSIONS

In this paper, an LCA study of different types of recycled and virgin concrete was reported, to investigate the opportunities, but also the flaws, in the execution of this approach to establish the sustainability of concrete itself. The present LCA shows that recipes R6 and R7 are the options with the best performance from an environmental point of view, but because R6 has, the lowest impact on Water Depletion among the two, it can be considered the most environmental-friendly mixture of the present study.

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HOSPITAL WASTE – LEGISLATIVE FRAMEWORK AND MANAGEMENT DIFFERENTIATIONS IN AUSTRIA, BELGIUM, GERMANY AND GREECE

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ABSTRACT
In this paper, the differences regarding hospital waste management, specifically regarding the legal and technical situation and the definition of hazardous hospital waste in four member states (Austria, Belgium, Germany and Greece) are presented. All countries follow the legal guidelines of the European Union, but the technical situation, especially the possibilities for thermal waste disposal in incineration plants are different. Also the definition and categories of hazardous medical wastes are different in the aforementioned four EU member states. The annual production amounts of medical waste in selected countries are presented, as well as criteria that lead to their classification as hazardous medical waste.

1. INTRODUCTION

Hospitals focus on the protection of public health. Therefore, no negative environmental effects or any hazards to the employees and in the environment should be caused by the provided services. The developments in medicine, which require highest medical and care services, have produced growing waste amounts in the last years. Different types of hospital waste have to be collected, transported, sorted and prepared for disposal. The different waste fractions range from waste similar to households like packaging material and paper through hospital hazardous waste like chemicals or infectious materials.

Solid medical waste management has gained considerable importance during the COVID-19 pandemic. The situation in crisis conditions during the pandemic is evidently not representative for the overall management of medical waste beyond pandemics, but nevertheless, a number of fundamental questions were asked, in particular, the question of the uniformization of waste management across countries. The aim of this position paper is to contribute to this request by comparing in some detail the management of solid medical waste from legal and technical point of view in 4 European countries. To the best of our knowledge, a similar comparison was made already 20 years ago (Mühlich, Scherrer, & Daschner, 2003) and in view of the significant development of health care and interest in waste management in last decades, a new review was deemed necessary albeit with a limited number of EU member states.

On a European scale, many legal constraints on the management of waste in general and on medical waste are available. Hospital waste is classified according to the European waste catalogue 2000/532/EC (European Waste Catalogue 2000/532/EC). Applying the European directives is always an obligation of the EU members and differences in interpretation leads to differences in waste management among countries. Even within each member state, the application on the floor, i.e. among hospitals is on occasion different. The aim of this paper is to illustrate this diversity.

It is clear that this diversity is contra productive to contribute to the optimalisation of the Sustainable Development Goals (SDG, Ranjbari et al., 2022) and the European Green deal. Adapting waste in this respect has been done for house hold waste (López-Portillo, Martinez-Jiménez,

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Ropero-Moriones, & Saavedra-Serrano, 2021), any attempt to reconsider medical waste from these perspectives is welcome. The wish of the EU to enforce the circular economy model in all branches of economic activity is appreciated, but it is clear that legal constraints are inhibitory in this respect.

This paper shows the current differences (in 2023) in Austria, Germany, Belgium and Greece for hospital waste management regarding:

- The legal situation,
- The technical disposal situation, and
- The definition of hospital (or medical) hazardous waste.

2. LEGAL SITUATION

2.1 Europe

A number of European regulations and directives govern waste disposal. The basis is the Waste Framework Directive 2008/98/EC (Directive 2008/98/EC), which defines the main waste-related terms. The Waste Framework Directive (2008/98/EC) must be separately transposed into the national law by each European member state.

2.2 Austria

In Austria, the legal basis for waste management is the "Austrian Law on Waste" (Abfallwirtschaftsgesetz 2002). A national law is complemented by 9 different state laws. The Austrian Law on waste contains basic regulations concerning waste management, waste separation, handling of hazardous waste, official data sheets for hazardous waste and regulations about the waste manager. It has been updated and revised several times to include the legal requirements of the European Union, e.g. waste framework directive.

The legal regulations are amended by various special waste regulations and the "Austrian Standardization Guideline for the Medical Field" (ÖNORM S2104, 2020). Waste from Medical institutions, Austrian Standards International, 2020). It contains exact regulations regarding types of waste, waste definition, waste collection and disposal and is authored by a board of experts from waste management, disposal engineers, hygiene experts, medical doctors, users, etc. it also contains the Austrian waste catalogue numbers, which are relevant for medical institutions.

The Federal Ministry of Austria Climate Action, Environment, Energy, Mobility, Innovation and Technology assume the legal responsibility for medical waste management. The ministry is supported by the Federal Environmental Agency. Waste data is reported to the Federal Ministry of Austria Climate Action, Environment, Energy, Mobility, Innovation and Technology.

2.3 Germany

Germany supports sustainable waste management concepts for obtaining raw materials or energy from waste with the Waste Management Act (Kreislaufwirtschaftsgesetz – KrWG); it entered into force on 1 June 2012. A waste hierarchy that lays down a fundamental series of steps comprising waste prevention, reuse, recycling, and other elements besides, including energy recovery, and finally waste disposal. The Waste Management Act supplemented is fleshed out by a number of other regulations in Germany and is further differentiated by the waste management acts of the 16 federal states.

The European waste catalogue classified healthcare and hospital waste in chapter 18. Furthermore, classifications and categorization of those waste streams are laid down in the German guideline for disposal of wastes from healthcare facilities established by Federation/Länder Working Group on Waste (LAGA M18, 2021).

2.4 Belgium

Belgium is a country with three different governments depending on the geographical location: Flanders (northern part), Wallonia (southern part) and Brussels (in between). Each of the governments is responsible for managing waste including medical waste. Hence, three different law-systems have to be considered. Hereafter, medical waste is discussed, excluding radioactive waste. This type of waste is still the responsibility of the Belgian federal government overruling the 3 separate governments.

The legal basis for the management of medical waste in Flanders is described in the "Flemisch law on waste of 17/2/2012 subdivision 5.2.3 (VLAREMA, 2012). Medical waste is produced during medical treatment or diagnosis of humans and animals in hospitals or in private practices, during pharmaceutical and biological research in laboratories and during routine home-treatment of patients by specialized personal. The law distinguishes household and industrial waste; medical waste is considered as industrial waste. Within the category industrial waste, the distinction is made between special waste and others. Medical waste is considered as special waste.

The legal basis in Wallonia is the decree of 30/6/1994, which has been modified on several occasions e.g. 4/7/2002, 3/6/2010 and 13/7/2017 (Decree 30/6/1994). The follow-up is done by the "Directorat Général de l'Agriculture, des Resourses naturel et Environment which is founded in 2008. Waste from clinical activities is classified according to the risks for the community. The legal basis for medical waste in Brussels is the decree of 23 March 1994 (Decree 23/3/1994). Recent laws related to waste in general are implied in 13/1/2017 ("BRUDALEX") with emphasis on the responsibility of the owner of the waste, i.e. the head of the hospital (BRUDALEX 13/1/2017). At the same time, 11 older decrees are suspended with this new law. Within this law, both specific items (i.e. use of reusable plastic bags) and more general rules (i.e. on the traceability of medical waste, transport…) are mentioned.

2.5 Greece

The main Greek legal regulation concerning medical waste management was first issued on 2003 and was then updated on 2012 (HCMD, 2012) with the title "Measures and terms for the management of wastes from health-care facilities". The circular of 2012 and the circular of 2013 provide more details on the management of the medical wastes in Greece. A minor update of the principal 2012 decree was issued on 2017 (HCMD, 2017) that modified operational issues (temperature, volume) related to the on-site storage of medical waste prior to their transport for further treatment.

The legal regulation applies to healthcare facilities, dental practices and veterinary clinics for the entire country. It covers furthermore definitions, classification, source separation, storage, collection and transport, treatment, disposal and obligations of producers.

The Hellenic Ministry of the Environment and Energy and the Hellenic Ministry of Health have the legal responsibility for medical waste management. Waste generation and composition data is reported currently to the Hellenic Ministry of the Environment and Energy.

3. DEFINITION OF HAZARDOUS MEDICAL WASTE

3.1 Austria

In Austria, the legal basis for the definition of waste from medical institutions is the ON S2104 (ÖNORM S2104, 2020). Here, the types of waste are listed, that are possibly produced in hospitals and medical institutions.

The official definition of hazardous medical waste is waste that poses a risk inside and outside of medical institutions and therefore requires special treatment in both areas. Following this classification "Waste with certain infectious virus" is collected, e.g. black death, anthrax, rabies, tuberculosis, dengue fever etc., not included are HIV and hepatitis. For the list of virus that are important for the definition of hazardous medical waste see Table 1.

According to the "Austrian Waste Catalogue Number" the waste is classified with the number "97101", following the European Waste Catalogue it is classified as "180103", wastes whose collection and disposal is subject to special requirements in order to prevent infection.

This waste is collected and disposed in compliance with certain conditions:

- Collection in special one-way containers;
- Containers cannot be re-opened;
- Carriage as hazardous materials transportation (following ADR law in UN class 6.2-compliant containers).

Disposal is dedicated to be carried out in a special incineration plant for hazardous waste without opening. There is only one plant in Austria, where all the hazardous medical waste is incinerated. The disposal costs are summed up to approximately 1000 €/t (without transport etc.).

The other types of waste produced in hospitals have to be collected separately as well following the classification criteria and waste catalogue numbers in the ON S2104 (ÖNORM S2104, 2020), e.g.:

- Pharmaceutical and cytotoxic waste (Austrian waste catalogue number 53510, EWC 180108);
- Body parts and organs (Austrian waste catalogue number 97103, EWC 180102);
- Sharp waste (Austrian waste catalogue number 97105, EWC 180101);
- Disinfectants, chemical waste, mercury waste, batteries, electronic waste.

In Austria, approximately 1.070 t of hazardous medical waste was produced in 2020. The amount of sharp waste (Austrian waste catalogue number 97105 and EWC 180101 respectively) ranged from 1.140 in 2018 to 1.170 t in 2020, which was in the same magnitude as disinfected waste (1.050 t in 2018 – 1.030 t in 2020). In 2020, the amount of body parts was 50 t. The amount of non-hazardous hospital residual waste varied from 43.830 t in 2018 to 42.420 t in 2020 (Austrian waste catalogue number 97104 and EWC 180104 respectively). The total amount of medical waste ranged from 46.960 t in 2018 to 45.740 t in 2020. Table 2 shows the medical waste data of Austria in 2018 – 2020 (Bundesabfallwirtschaftsplan, 2023) considering the impact of Covid 19 pandemic (Fraeyman N., et al., 2022).

3.2 Germany

The term "hazardous waste" refers to various types of waste with defined hazardous properties that are harmful for the environment and/or human health such as:

- infectious (EWC 180103*, EWC 180202*),
- cytotoxic and cytostatic medicines (EWC 180108*),
- chemicals consisting of or containing dangerous substances and
- amalgam waste from dental care (EWC 180110*).

The needs for waste management EWC 180103* describes the following. While defining EWC 180103* wastes (wastes whose collection and disposal is subject to special requirements in order to prevent infection) it is assumed from knowledge or to be expected from medical experi-

TABLE 1: List of virus for the definition of hazardous medical waste (EWC 180103*) in Austria (ÖNORM S2104, 2020) and Germany (LAGA M18, 2021).

Types of virus - Austria	Types of virus - Germany
Anthrax	AIDS/HIV
Aphthous fever	Brucellosis
Brucellosis	Cholera
Cholera	Diphtheria, Leprosy
Dengue Fever	Dysentery
Glanders	Glanders
Haemorrhagic fever	Haemorrhagic fever
Jakob-Creutzfeld disease	Hepatitis virus
Leprosy	Meningitis/encephalitis
Paratyphoid A, B, C	Poliomyelitis
Pest	Psittacosis
Polio	Q-Fever, Rabies
Psittacosis	Rabbitfever
Q-fever	Smallpox
Rabies	Spleen, Plague
Tuberculosis	TSE – (Transmissible spongiforme encephalopathy); Creutzfeldt-Jakob disease
Tularemia	Tuberculosis
Typhus abdominalis	Typhus/Paratyphoid

TABLE 2: Medical waste data of Austria in 2018-2020 (Bundesabfallwirtschaftsplan, 2023).

Waste	Mass (t) 2018	Mass [t] 2019	Mass [t] 2020
SN 97101 (Hazardous medical waste) – EWC 180103	890	890	1.070
SN 97102 (disinfected Waste)	1.050	930	1.030
SN 97103 (Body parts and organs) EWC 180102	50	60	50
SN 97104 (Hospital residual waste) EWC 180104	43.830	43.960	42.420
SN 97105 (sharp waste) EWC 180101	1.140	1.190	1.170
SN 97105 g (hazardous sharp waste)	<1	<1	<1
Total	46.960	47.030	45.740
% hazardous amount	1,9%	1,9%	2,3%

ence that these wastes are contaminated with germs of specific diseases mentioned in the guidelines and that it can be feared that these wastes will cause the diseases to spread.

With reference to the Infection Protection Act and LAGA M 18 a list of diseases / germs, seen in table 1, is given in the guidelines which have to be considered for:

- the danger of infection (contagiousness, infections dosage, epidemic potential),
- the survival capacity of the germ (incubation period),
- the mode of transmission,
- the extent and type of potential contamination,
- the quantity of contaminated waste and
- the gravity of the disease possibly caused and the possibility of treatment,
- special requirements for their prevention (LAGA M18, 2021).

The collection of infectious and cytotoxic waste operates separately with type-approved waste bins. Thus, wastes go directly to incineration (without any compression) for transporting hazardous goods according to ADR (Accord européen relatif au transport international des marchandises Dangereuses par Route).

Wastes that are not in accordance to the EWC 180103* criteria are listed in EWC 180104 (wastes whose collection and disposal is not subject to special requirements in order to prevent infection). Wastes listed in EWC 180104 that contain facultative pathogenous germs, could cause nosocomial infections in patients. They are only dangerous for immune supprimized patients and only if a direct or indirect contact via a medium takes place. A clinical case study at Jena University Hospital for example aimed to investigate the influence of waste container construction on the release of microorganisms in a hospital setting (Büchner et al., 2021).

Table 3 shows medical waste data about selected waste types in 2019 and 2020 in consideration of the circumstances corona/covid pandemic (Datenbank des Statistischen Bundesamtes, 2022). The amount of hazardous hospital waste (EWC 180103) ranged from 8.900-10.800 t in 2019-2020, the amount of sharp waste (EWC 180101) was 800-900 t in 2019 and 2020. In 2019 386.200 t of non hazardous hospital waste (EWC 180104) was produced, in 2020 382.000 t.

3.3 Belgium

There are some differences in identifying the waste depending on the region: In Flanders, medical waste is further categorized as either “risk waste” or “non-risk waste”. Medical risk waste has risks for infection or can cause wounds to those involved in the management of the waste. Waste that poses specific ethical problems (recognizable body parts, placenta…) is also considered as risk waste.

In addition to the law, several extended documents are available; the most recent is the “code of good practice” (2014) written by OVAM which is the managing organization of medical waste in Flanders (Code of good practice, 2014). This is a joint effort of the government, medical institutions, federation of industries active in environmental issues and federations of hospitals. It also takes into consideration the law on the transport of dangerous goods. The content comprises waste types, hazards, very detailed regulations, and scientific support from VITO (Flemish Institute for Technological Research). In Wallonia, medical waste is either class A: comparable to household waste or class B2: infectious waste. Any other type of waste is class B1. Within class B2, medical waste further contains (1) needles, bistoury and alike, (2) infected, contaminated material including anatomical waste, (3) waste from medication, cytotoxic waste, heavy metals, chemical waste, (4) containers under pressure and (5) radioactive material. A specific regulation for anatomical waste was set up. For each class, specific regulations on handling, transport and elimination are given. Medical waste of class B2 might be transformed into class A of B1 provided the transformation is certified

TABLE 3: Medical waste data in 2019 and 2020 at Germany (Datenbank des Statistischen Bundesamtes, 2022).

Waste	Mass [t] 2019	Mass [t] 2020
EWC 180101	800	900
EWC 180102	3.100	2.800
EWC 180103*	8.900	10.800
EWC 180104	386.200	382.000
EWC 180108*	2.300	2.300
EWC 180109	7.900	16.000
EWC 180110	100	100
EWC 180202*	700	600

(e.g. treatment in an autoclave). In all cases, the incineration is controlled as far as temperature and the exhaust of environmentally burdensome compounds like dioxins, some gases such as sulfur dioxide, hydrogen chloride, nitrogen oxides, hydrogen fluorides, total organic carbon etc.

Identification and categorization of Medical waste in Brussels are nearly identical to the waste in Wallonia. Two different kinds of medical waste are considered: special waste, which is equivalent to the B2 type of waste in Wallonia and includes needles, body parts, blood, infectious waste from laboratories, and cultures of a specimen from patients infected with specified diseases (e.g. anthrax, cholera, tuberculosis, Creutzfeld Jacob). Non-special waste is waste comparable to household waste but produced during medical activities. Non-risk waste which is comparable to the B1 type of waste in Wallonia is collected in grey plastic bags with specifications; specific waste is collected in yellow containers. Both are incinerated. Risk waste (Flanders) or B2 waste (Brussels and Wallonia) that has been decontaminated by autoclaving or any other method for sterilization is considered non-risk waste or B1 type of waste provided the decontamination process is validated. The collection of risk waste is identical in the three regions. The waste with risks is collected in yellow 60 or 30-liter one-time-use containers with self-closing covers. Solid waste but also fluids of different kinds (blood, perfusion liquid, urine, gastric fluid after lavage etc…) and mixtures can be stored in these containers. As an alternative, non-fluid or semi-fluid risk medical waste can be collected in plastic bags, which are inside a cardboard box, both with technical specifications. Non-risk waste is collected in blue plastic bags again with technical specifications. Waste classified as risk waste and fluid or semi-fluid non risk waste must be incinerated. Sharp waste is collected in specific small yellow plastic boxes, which are put into the large yellow containers. Waste with risks can be decontaminated (e.g. steam sterilization) provided evidence for the decontamination process is delivered.

3.4 Greece

According to the principal Greek legislation on medical waste management of 2012 (HCMD, 2012), hospital or medical waste belongs to chapter 18 of the EWC with the two main categories being the human-related healthcare waste (18 01) and the animal related healthcare waste (18 02). However, the categorization below, as described in HCMD (2012), diverts a little from the codes of chapter 18 of the EWC, as the three main fractions of the medical waste are:

- municipal type medical waste,
- hazardous medical waste (HMW), which is comprised of the following three sub-fractions:
- infectious waste,
- mixed waste (infectious and other hazardous waste),
- other hazardous waste,
- special waste streams (e.g. radioactive waste, batteries, pressurized cans, WEEE etc.) may or may not contain hazardous materials.

In particular, the three sub-fractions of the hazardous medical waste stream are described as follows:

- medical waste with infectious properties that exhibit only the HP9 (infectious) hazardous property (e.g. body tissues, blood, fecal and urine samples from patients with an infectious disease, needles etc.), Infectious waste should be either sterilized and then landfilled, or be incinerated.
- mixed hazardous medical waste that exhibit both the HP9 property as well as at least one additional hazardous property (e.g. chemotherapy related waste, waste from bio-pathology and histology laboratories etc.). Mixed hazardous medical waste should be only incinerated.
- other hazardous medical waste that exhibit one or more hazardous properties other than HP9 to which incineration or material recovery can be applied.

According to the Greek Hazardous Waste Management Plan of 2016 (HCMD, 2016), the total annual amount of solid hazardous medical waste (estimates are for year 2011) was 16.300 t, from which around 80% was infectious waste and 20% was mixed hazardous and other hazardous waste. The liquid (wastewater) hazardous medical waste was estimated to be around 41.300 L/d, from which 40% was the infectious liquid waste and 60% was the mixed and other hazardous liquid waste.

Hazardous medical waste (HMW) in Greece is treated by either incineration or steam sterilization. Sorting within the healthcare facilities is realized successfully and efficiently for the past 10 years. There is one medical waste incineration unit in Greece (Athens) which receives hazardous medical waste from public and private hospitals all over Greece. Apart from the above incineration unit, there are also six (6) sterilization units, which receive only infectious medical waste. Five of the above sterilization units can also store medical waste, if needed, for long term. On site (within the hospitals) sterilization, that used to exist in some public hospitals in the past, is not applied anymore as the requirements for sterilization are now quite strict.

The hazardous medical waste (HMW) is source separated, stored, collected, transported, treated and disposed of, according to technical specifications described in the HCMD (2012). Special color-coded containers are used in all healthcare units (public and private) to separate the HMW in the aforementioned three categories, with sharps sorted separately in rigid plastic containers (but still treated as infectious waste usually via autoclaving in the above six sterilization facilities). Medical waste can be stored for up to 30 days at low temperature rooms within the hospitals. The transport of HMW is conducted according to "European Agreement Concerning the International Carriage of Dangerous Goods by Road (ADR)". All types of hazardous medical waste (infectious, mixed, other) can be incinerated; however, the medical waste that is solely infectious can be alternatively sterilized (via autoclaving with the use of steam), which is a technology less expensive than incineration. After sterilization, infectious waste is rendered non-hazardous and can be disposed of into a MSW (non-hazardous waste) sanitary landfill, which is the dominant type of solid waste landfill now in Greece. Tests are sometimes performed on sterilized infectious HMW, typically at the landfill gate, to validate the effectiveness of sterilization. Medical waste incinerator ash, if rendered hazardous, is currently exported to other European countries, but plans are under way to develop an ash treatment facility and a hazardous waste landfill in Greece.

The HMW treatment costs in Greece are currently up to 700 €/t for sterilization, up to 1.700 €/t for incineration, with an overall cost that is roughly estimated to range from 700 to 3.500 €/t, when collection and transportation costs are included.

4. TECHNICAL DISPOSAL SITUATION FOR HEALTHCARE WASTE

4.1 Austria

In Austria, medical waste has to be incinerated (ÖNORM S2104, 2020). Therefore, one incineration plant for hazardous hospital waste and about 10 for non-hazardous hospital waste are available in the whole country. Furthermore, several treatment plants for chemical waste, other hazardous waste and recycling waste are used for the correct disposal of the different waste types (Bundesabfallwirtschaftsplan, 2023).

4.2 Germany

In Germany most noninfectious healthcare waste can be disposed of or recycled at thermal waste treatment plants. Currently there are 68 municipal waste incineration facilities (Umweltbundesamt, 2015).

In accordance with the LAGA guideline M 18, infectious healthcare waste is to be collected separately and thermally treated as hazardous waste using the mandated Robert-Koch-Institute procedures such as sterilization, steam disinfection, or incineration (Robert-Koch Institut, 2017). Infectious waste for incineration is thermally disposed of separately in purpose-built hazardous waste incineration facilities, after being transported in UN class 6.2-compliant containers.

TABLE 4: Summarizing table – legislation, classification and management of hazardous hospital waste in Austria, Belgium, Germany and Greece.

	Austria	Germany	Belgium	Greece
Legislation	• Waste Framework Directive 2008/98/EC • Austrian Law on Waste (2002) • Austrian Standardization Guideline for the medical field (2020)	• Waste Framework Directive 2008/98/EC • Kreislaufwirtschaftsgesetz (2012) • German guideline for disposal of healthcare wastes from healthcare facilities (2021)	• Waste Framework Directive 2008/98/EC • Flemish Law on Waste (2012) for Flanders • Decree 30/6/1994 for Wallonia • Decree 23/3/1994 for Brussels	• Waste Framework Directive 2008/98/EC • Hellenic Common Ministerial Decision (2012)
Classification of (Hazardous) Medical Waste – Main hospital Waste Types	A. Hazardous medical waste - waste that poses a risk inside and outside of medical institutions and therefore requires special treatment in both areas (equivalent to EWC 180103) B. Cytotoxic waste (equivalent to EWC 180108), C. Body parts and organs (equivalent to EWC 180102), D. Sharp waste (equivalent to EWC 180101), E. Chemical and other wastes, e.g. disinfectants, chemical waste, mercury waste, batteries, electronic waste	A. Infectious wastes whose collection and disposal is subject to special requirements in order to prevent infection (EWC 180103) B. Cytotoxic and cytostatic medicines (EWC 180108*), C. Chemicals consisting of or containing dangerous substances and D. Amalgam waste from dental care (EWC 180110*).	<u>Flanders:</u> A. Risk Waste (risks for infection, ethical problems) B. Non Risk Waste <u>Wallonia:</u> A. class A: comparable to household waste B. class B2: infectious waste - (1) needles, bistoury and alike, (2) infected, contaminated material including anatomical waste, (3) waste from medication, cytotoxic waste, heavy metals, chemical waste, (4) containers under pressure and (5) radioactive material. C. class B1: any other type of waste. <u>Brussels (identical to Wallonia):</u> A. Special waste (needles, body parts, blood, cultures etc.) B. Non special waste (comparable to household)	A. Municipal type medical waste, B. Hazardous medical waste (HMW), 1. infectious waste, 2. mixed waste (infectious and other hazardous waste), 3. other hazardous waste, 4. special waste streams (e.g. radioactive waste, batteries, pressurized cans, WEEE etc.)
Management of hazardous medical waste	Medical waste hast to be incinerated. • 1 Incineration plant for hazardous hospital waste • 10 Incineration plants for non hazardous hospital waste	Medical waste has to be thermally treated. • 68 municipal waste incineration facilities for non-infectious healthcare waste • Thermal treatment for infectious healthcare waste (Robert-Koch Institute Procedures sterilization, steam disinfection or incineration	Hazardous waste has to be incinerated in a plant with permission: • 2 incineration plants for hazardous medical waste	Hazardous medical waste has to be thermally treated: • 1 incineration plant • 6 sterilization facilities

4.3 Belgium

Hazardous waste has to be legally incinerated in a plant with a permission to treat this type of waste. In Belgium two incineration plants are available for hazardous medical waste, one in Flanders, one in Wallonia.

4.4 Greece

In Greece, there exists one incineration plant dedicated to thermally treat all types of hazardous medical wastes generated by the public and private health care facilities in the country. It is located in the Prefecture of Attica (near Athens). Also, there exist six (6) sterilization facilities (Volos, Rhodes, Salonica, Larisa, Crete, Tripoli) that can treat only the infectious fraction of the hazardous medical waste.

5. CONCLUSIONS

In this paper, the legal and technical bases for hospital waste management have been presented for Austria, Germany, Belgium and Greece. As you can see in Table 4, which gives an overview about legislation, classification and management of hazardous hospital waste, even though all the countries are members of the European Union and follow European regulations, the national specifications are different due to variable legal requirements, existing technologies and technical facilities, and options for waste disposal. In addition, even though the members of the European Union seek to harmonize concepts and modes of action, in this context, this harmonization is not totally achieved. But irrespective of the different waste definition, hazardous medical waste in all four member states under study is sorted and collected separately and then is either thermally treated (principal method) or sterilized.

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EVALUATION OF DUST EMISSION RATE FROM LANDFILL MINING ACTIVITIES

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ABSTRACT
Mining operations are one of the most significant sources of particulate emissions in the atmosphere. Landfill mining (LFM) process activities, including excavation, screening, shredding, and equipment handling, have the potential to emit particulate matter into the environment as short-term episodic emissions during operational periods. Previous investigations show that LFM activities can potentially cause human health and environmental impacts through exposure of these emissions. This paper evaluates the dust emission rate of such activities to understand factors responsible for higher emissions rate and determine where any pressure points exist in order to mitigate risk. Nine empirical formulas were adopted from surface mining activities, including point, line, and area sources of activity. Parameters identified in the equations were adjusted to LFM application conditions. From emission results, it is observed that point source activities were the major sources of emission. The study area was divided into multiple phases and one phase cumulative for the maximum/average/minimum point sources emissions over the lifetime of the landfill mining operation calculated in this study are approximately 5.04 tonnes (t) / 3.23 (t) / 1.61 (t), respectively. However, the one phase cumulative for the maximum/average/minimum line sources emissions over the lifetime of the landfill mining operation are approximately 100.8 (kg/m) / 40.32 (kg/m) / 20.16 (kg/m), respectively. Mitigation measures to control high emission rate of LFM related activity, such as utilising tankers or bowsers to spray water around the LFM area, to control airborne emissions, should be considered. The results of this research are expected to inform air dispersion modelling for environmental impact assessment studies of air pollution.

1. INTRODUCTION

Particulate matter (PM) is one of the principal pollutants resulting from surface mining operations (here also termed 'dust') (Luo et al., 2021; W. Wang et al., 2022). Landfill mining (LFM) practices also potentially contribute to the production of dust by a series of on-site mechanical processes into the environment (Zari et al., 2022). Dust released during these activities is directly introduced to the ambient atmosphere (Patra et al., 2016), which includes potentially toxic materials in a diverse size range (Entwistle et al., 2019; Guo et al., 2021). These materials can be transported into the surroundings and affect local communities if not controlled properly (Pandey et al., 2014).

Atmospheric particles (dust), with a sizes range of <60 μm can play an important role in the transfer of pollutants into the environment. (Csavina et al., 2012; Omrani et al., 2017). Due to the potential distance, speed, and aerial extent with which contaminants can be transported in the environment, atmospheric transport is likely to be the most significant transport pathway relating to potential impact on the environment and human health (Csavina et al., 2012; Elmes & Gasparon, 2017; Kim et al., 2015; Liu et al., 2023). Furthermore, in the very short-term (hours to days), airborne particles have the greatest ability to transport contaminants within the environment (Csavina et al., 2012; Manisalidis et al., 2020).

Health studies show a strong association between airborne PM and adverse effects such as obstructed airways, cancer, decreased lung function and capacity, pneumoconiosis, and increased cardiovascular disease. (Cristaldi et al., 2022; W. Wang et al., 2022). Smaller particles, especially those <10 μm, are more likely to be inhaled through the respiratory system (Gao et al., 2017). Particles with diameters ranging from 30 to 10 μm remain suspended in the air for

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a short time before getting trapped in the nostrils or mouth and being ingested. (Luo et al., 2021; Patra et al., 2016).

Previous findings by Dino et al. (2016) and Zari et al. (2022) that assessed the suitability of historic landfill sites for LFM, showed that some potentially toxic elements within landfilled waste materials have a potential risk to the human health and the environment. In addition, Adenuga et al. (2022), Fang et al. (2022) and Houessionon et al. (2021) demonstrated that potential risk exposure above an acceptable range exist from recycling activities worldwide. Air dispersion modelling enables the Environmental Impact Assessment of air pollution from such activities to be considered. In order to assess the air quality impact, equations for calculating predicted emission rates (also called emission inventory), are necessary in the absence of actual monitoring data. Only equations for calculating emission rates associated with LFM activities are considered in this study. Generally, there are four methods of particulates emission estimation: mass balance calculations; engineering calculations; sampling or direct measurements; and application of emission factors. For this study, an engineering calculations approach was adopted as the most applicable.

For each source of activity in the mining sector, an equation for estimating emissions has been established (Neshuku, 2012). The United States Environmental Protection Agency (USEPA) and Central Mining Research Institute in India has implemented extensive work to develop emission estimating formulas for these activities (Chakraborty et al., 2002; Chaulya et al., 2003; Chaulya et al., 2019; Lal & Tripathy, 2012; Neshuku, 2012; Richardson et al., 2019). A number of field observations of dust formation from all individual activities was validated against these equations (Lal & Tripathy, 2012). The formulas used in this study were derived from surface mining activities by Chaulya (2006) since the focus of those studies was based on surface mining, which allowed for a more precise prediction of dust production (Triantafyllou et al., 2021). Chaulya (2006) has worked extensively on the formulation and validation of a series of Gaussian dispersion equations for determining the emission rate from various surface mining operations, taking into account ground level, vertical, and horizontal dispersion factors. A field investigation of three surface mines and an average of three data from each monitoring station serve as the foundation for each activity's calculation. It was discovered that the validation study's accuracy ranged from 92% to 97% of the actual field measurement data (Lilic et al., 2018; Patra et al., 2016). These equations have taken into account the key determining factors for each specific activity (Chaulya, 2006).

The purpose of this study is to evaluate the dust that can become airborne during LFM activities. This evaluation was achieved by adjusting parameters identified in the equations to LFM applications and to represent local weather conditions. Nine empirical formulas were adopted to account for nine different LFM processes: cover removal loading, recovered waste loading, cover removal unloading, recovered waste unloading, haul road, transport road, exposed cover removal waste, exposed pit surface and screening plant.

2. MATERIALS AND METHODS

2.1 Site selection and description

Landfilling was mainly implemented by local authorities and private sector companies in the UK prior to the introduction of regulatory standards and modern engineering, implementation of the Landfill Directive, and modern regulation via the environmental permitting regulations. The landfill site selected for this study was based on the identification of a site that is representative of the many hundreds of landfill sites in the UK of this type which contain historic deposits >30 years old, primarily being filled during the 1970s, 1980s, and early 1990s. The waste material used in this research originates from a closed landfill located in Norfolk, eastern England. Norfolk County Council utilised the site as a Local Authority landfill, filling in a mineral working that extracted sand and gravel. The site is about 10 m deep, has a surface area of 3.12 hectares (about 7.71 acres), and was operational from 1978 until 1986. The landfill had no liner or leachate collection system (non-engineered). In 1998, an engineered geo-synthetic clay liner cap was installed to allow for intermittent landfill gas extraction whilst also reducing rainwater infiltration and leachate generation. Municipal solid waste (MSW) with commercial and industrial waste were the dominant wastes accepted in the site. The site was operated and regulated on a co-disposal basis in the early days of waste regulation under the Control of Pollution Act (1974). The landfill is composed of a complex mixture of organic and inorganic waste.

2.2 Excavation and sample processing

Using a rotary drilling rig at a depth of 7-8 m, approximately 40 kg of MSW was collected from four individual wells, numbered 1901, 1904, 1906, and 1907 (10 kg each) (Figure 1 and 2). Although ASTM D5231-92 (2016) specifies a minimum quantity of 91-136 kg for compositional analysis of unprocessed MSW, the collected volume of wastes was limited due to sample storage constraints. To prevent any loss, the samples were placed in thick- labelled plastic bags that were tightly sealed. After being delivered to the laboratory in a sturdy plastic box, the samples were kept in a refrigerator at 4°C until preparation for moisture content measurement. Coning and quartering were used to produce representative samples. Manual sorting was used to remove bulky non-biodegradable components such as metal, plastic, textile, and paper from the obtained MSW samples in order to homogenize the samples for analytical purposes. Coning and quartering yielded approximately 20 kg of representative samples (5 kg per well). These 5 kg samples from each well were further subsampled to roughly 1 kg each using a riffle sample splitter. Following the moisture content test, the samples were oven-dried at 105°C for 3 days to enable the mechanical sieving analysis to divide the samples into selected size fractions.

2.3 Characterization of landfill mining material

2.3.1 Moisture Content (MC)

For the determination of MC, 60 representative subsamples (15 from each well) of 50 g were dried in a drying oven at 105°C for 24 hours, given that the samples were

FIGURE 1: Aerial photograph of the study area marked with a red boundary and sample locations illustrated in purple squares with corresponding well numbers.

well stabilised in terms of organic content and not expected to interfere with the moisture content results. This information is presented in a previous investigation of the same landfill site (Zari et al., 2022).

2.3.2 Particle size distribution (PSD)

For the PSD test, approximately 800 g of representative samples from each well were used. For each well, mechanical sieving was employed for 15 minutes to divide the materials into size fractions. Each sieve's weight and % were recorded.

2.4 Emission Estimation

In this study, a range of each formula component was provided for modelling emission rate variability in order to estimate maximum and minimum emission values. Table 1 (please see the Supplementary material) shows the mathematical expressions used for calculating emission rates for different LFM activities. Equation parameters were modified to account for local meteorological conditions and LFM applications.

The MC and silt content of the recovered waste examined in this study were used to calculate the emission rate. To account for both the best- and worst-case scenarios, wind speed was calculated for a range of 0 to 30 m/s. The required equation components for heavy equipment dimensions and average vehicle speed range were determined using literature from LFM case studies (Jain et al., 2013; Parrodi et al., 2019) and landfill operating reports. Loader sizes range from 1-2.5 m^3 was used for loading and unloading activities (e.g., Caterpillar 320, Hitachi 250LC, and Liebherr 934). The range for unloading activities is also 1-2.5 m^3 if waste is directly unloaded to the screening plant or 22-35 t if waste is delivered to the screening plant using an unloader truck in wide areas. Because direct waste delivery to the screening plant was the method considered in this study, and the capacity of the unloader in the equation was given per t, a range of 0.5-1.5 tonnes was used in the emission rate calculation for unloading activities, which can roughly fill the bucket size of 1-2.5 m^3 used. This is consistent with most published LFM investigations and projects that have employed the direct loading of waste materials into screening plants (Hogland et al., 2014; Jain et al., 2013; LLC, 2009; Lopez et al., 2019; Lucas et al., 2019; Parrodi et al., 2020).

Area units from the current landfill were utilized in the equations whose source type is area (1 phase = 0.0044 km^2). The estimate of the emission rate used average vehicle speeds of 2.6 m/s for haul roads and 11 m/s for transport roads. The silt content of haul and transport road dust range were estimated a little higher than the silt content of waste materials (15-25,10-20%), respectively. The screening plant operational area was estimated as 0.0009 km^2.

FIGURE 2: Photographs of the recovered wastes from drilling (a) well 1901 (b) well 1904 (c) well 1906 (d) well 1907.

For loading and unloading activities, the drop height values employed in the equations ranged from 1-3 m. The frequency of loading values in the equations were approximated using the study's calculated volume of landfill waste.

2.5 LFM Assumed Process Plan

The study area was divided into five phases based on its unit area and the landfill is assumed to be mined on a phased basis as shown in Figure 3. The LFM process plan was carried out on the basis of one phase out of five, which accounts for approximately 0.48 hectares. Area and line source associated activity geometry were approximated using X and Y coordinates over a horizontal convex polygon with four vertices and a straight line joining two vertices, respectively.

The total assumed waste volume range of one phase was obtained by multiplying the length*width*depth of the investigated landfill site by 6 and 7 m, respectively. In addition, an equation was established (1) to be used for the determination of time taken for a LFM project to be com-

FIGURE 3: Assumed plan of LFM phases.

pleted to enable the estimation of emissions cumulative over the lifetime of a LFM operation.

$$T = \frac{\text{Waste volume } (m^3)}{\text{Loader dimension } (m^3) * \text{Total loading frequencies (no./hour)} * \text{Working hours (no.hour/day)}} \quad (1)$$

where T is the timescale of a LFM project. The loading frequencies refer to the total number of repetitions per day in a LFM project. Based on the empirical formulas used in the emission rate calculation and the waste volume range that was estimated, a LFM assumed processes plan flowsheet was developed to visualize the processes and specify volumetric throughput ranges for each unit operation. The road line source is not considered in the LFM assumed processes plan.

3. RESULTS AND DISCUSSION

3.1 Moisture Content

The mean MC results across the 4 wells revealed consistent values. The highest level of MC was relatively represented by well number 1901, reaching 25.9% (Figure 4). Due to the visual appearance during sampling, a higher MC of well no. 1901 was expected.

The average moisture content of all wells in this study was 23.5%, which is consistent with that of Jani et al. (2016) at Högbytorp landfill under comparable conditions. The findings were also comparable to another study by Frank et al. (2017) across seven UK landfill sites and that of Scott et al. (2019) for similar conditions. The landfill site was non-engineered for 13 years after its closure. An engineered Geosynthetic Clay Liner cap was installed in 1998, hence a greater level of infiltration and possibility of contaminant flushing is expected within the landfill site.

3.2 Particle size distribution

One of the key considerations in defining the physical characteristics of landfill waste is the PSD test. (Jani et al., 2016). Analysing this parameter provides a measure of the amount of fine fractions that could be dispersed as dust during LFM activities in conjunction with the MC parameter. The specific waste surface area has an influence on particle size (Dino et al., 2018). The finer a particle is, the higher its specific surface area.

According to earlier research, fine fractions of waste (soil-like material) account for a significant proportion of the overall volume of MSW disposed of in landfills (60%-70%), which is attributable to the application of daily soil cover, construction and demolition waste, and the humification of organic materials (Parrodi et al., 2018; Somani et al., 2018). In addition, there is a noticeable steady decline of waste fractions over time, according to a study by Francois

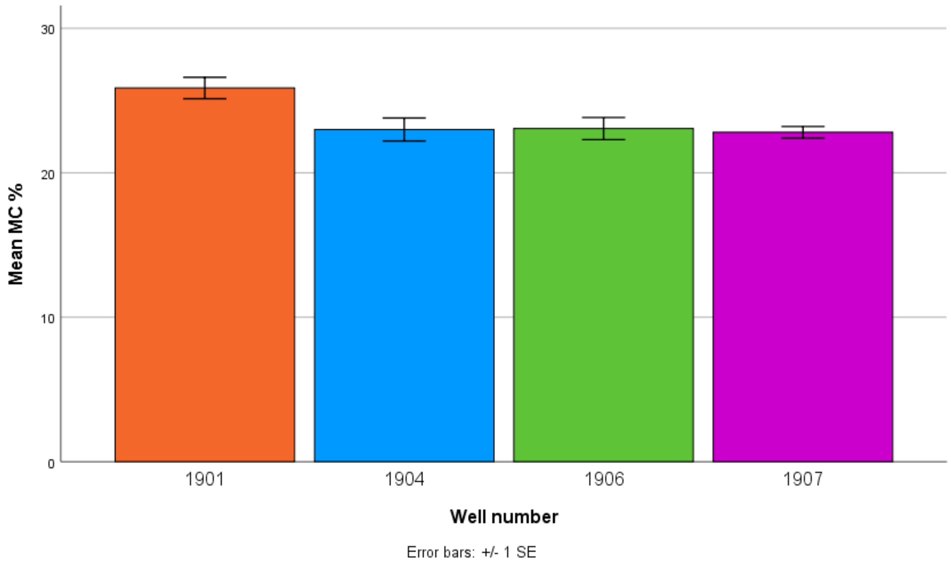

FIGURE 4: Mean moisture content of landfilled waste recovered from the four wells.

et al. (2006) that examined landfilled MSW of various ages (3, 8, 20, and 30 years old) at four different sites (Wang et al., 2021). A study by Singh & Chandel (2022) showed a similar trend of decreasing particle size. As a result, considering the age of the landfill being studied, a significant amount of fine fractions was expected.

Figure 5 shows the results of dry sieving waste fractions of the four wells to the following particle sizes: (>26.5, > 22.4, >19, >16, >12.5, >9.5, >6.7, >4.75, >3.35, >2.36, >1.7, >1.18, >0.85, >0.6, >0.425, >0.3, >0.212, >0.15, >0.106, >0.075, >0.053, >0.038, and ≤0.038 mm). In all wells, PSD revealed similar behaviour. Almost 70% of the total excavated waste fractions from the four pits passed through a sieve with a diameter of 4.75 mm, while 56% of the particles ≤2.3 mm in size. These fractions predominantly consisted of soil-like materials that had soil-like consistency. On average, waste fractions of 0.106 mm account for 11% of the sieve passings within all wells. Within the four wells, well no. 1901 had the greatest proportion of fines (2.5% of the analysed waste fractions < 0.075 mm and ≥ 0.053 mm). Photographs of different size fractions from all wells are shown in Figure 6 (a-d).

Visual observation revealed that soil fraction was notably prevalent within the waste particle size fraction <4 mm. During sample processing and screening, a diverse combination of waste components was detected, as was expected by the heterogeneity and degradation processes associated with landfilled waste. PSD results were comparable to

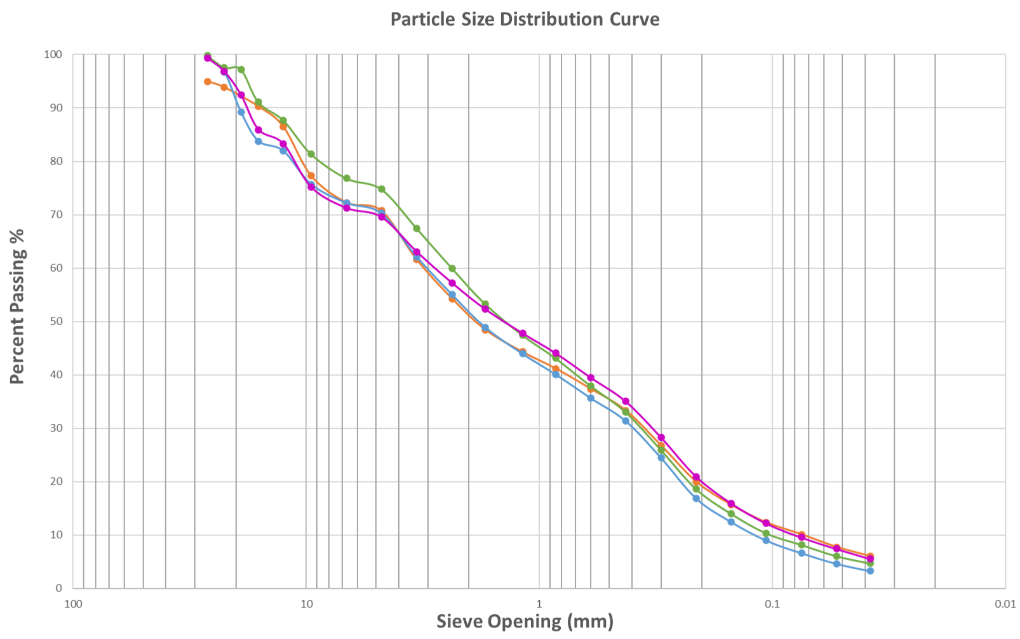

FIGURE 5: Particle size distribution cumulative passing curve for different sieve sizes of all wells.

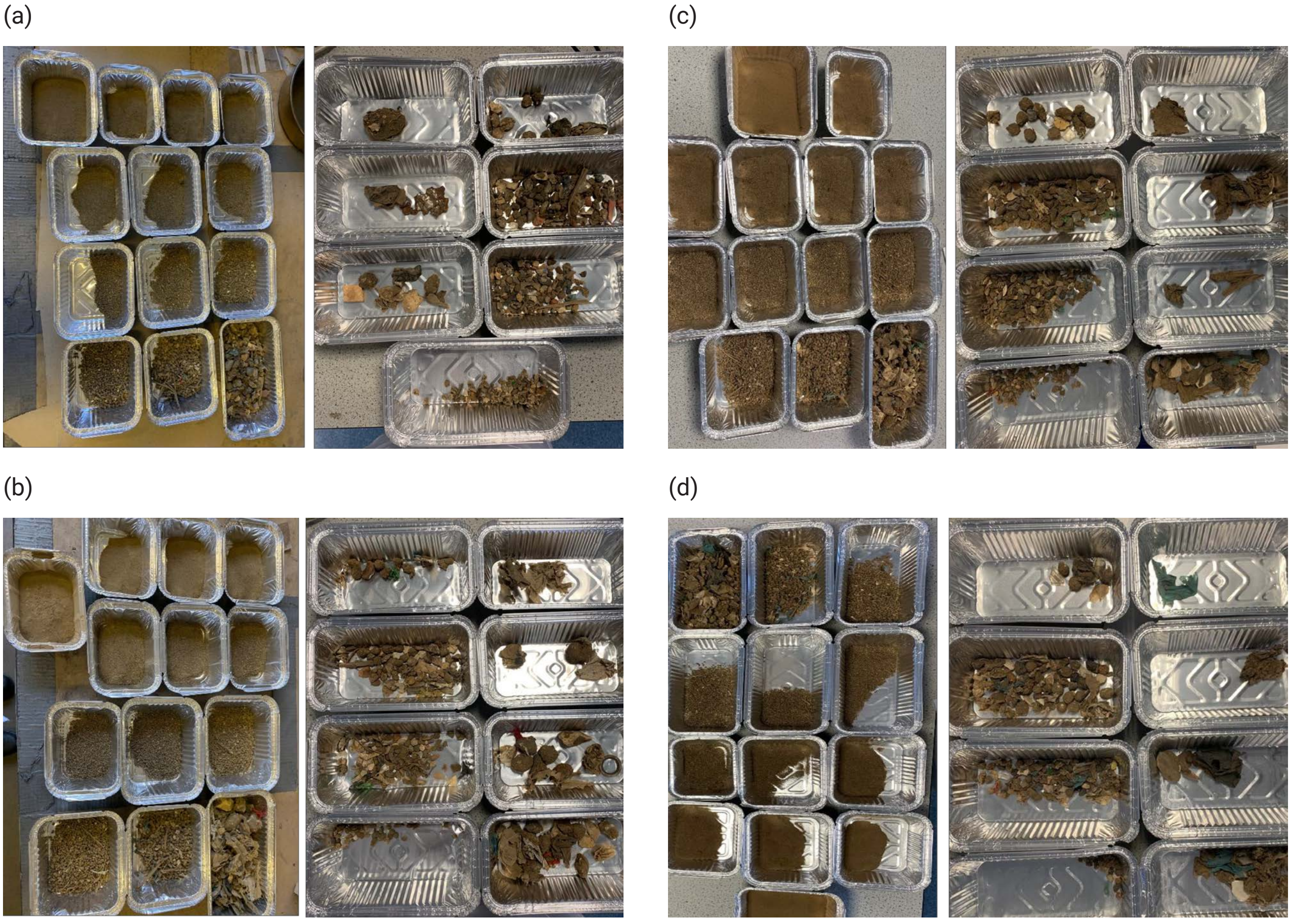

FIGURE 6: Size fractions of wells (a) 1901, (b) 1904, (c) 1906 and (d) 1907.

those of Mönkäre et al. (2016) with 70% of the MSW waste fraction (24-40 years old) having a particle size <5.6 mm. They agreed also with (Masi et al., 2014), with around 65% of the waste fine fractions (30-60 years old) <4 mm. Conversely, (Prechthai et al., 2008), reported that 70% of the MSW fractions of (3-5 years old) were >50 mm, while only 18% were <25 mm. The discrepancy in results among different studies is primarily due to the age of landfilled MSW.

3.3 Emission Estimation

For estimating the suspended emission rate for each LFM activity, nine empirical formulas from surface mining activities were adopted. These activities are: cover removal loading, recovered waste loading, cover removal unloading, recovered waste unloading, haul road, transport road, exposed cover removal waste, exposed pit surface, and screening plant. Table 2 (please see the Supplementary material) displays the calculated range of emission rates, along with descriptive statistics.

The MC and silt content of the recovered waste examined in this study were utilized in the emission rate calculation. MC values varied from 22.8% to 25.8% across all sampling locations. Because fractions between 0.002 and 0.06 mm are defined as silt content (Abril et al., 2016; Neshuku, 2012; Pimolthai & Wagner, 2014) and airborne particular matter (referred to as TSP) (Patra et al., 2016; Ronowijoyo et al., 2020; Sahu et al., 2018), the passing waste size fractions of 0.053 mm sieve obtained from the entire recovered MSW in this study were treated as the silt content range (4.6%-7.7%) in the emission rate calculation. Additionally, it is thought that particles >50 μm cannot penetrate the respiratory tract (Araújo et al., 2014), this range being considered a pollutant criteria in air quality standards (Huertas et al., 2014).

Wind speed, moisture content, and silt content are some of the formula components that have the greatest influence on emission rate (Chakraborty et al., 2002; Chaulya, 2006; Kim et al., 2020; Richardson et al., 2019). This is also valid in this study for all LFM activity sources with the exclusion of wind speed and the inclusion of the drop height parameter as a main driver, especially in the recovered waste loading and unloading activities' emissions.

Point source activities account for the majority of the emission sources in this study, as shown in Figure 7. This finding is consistent with other studies (Chakraborty et al., 2002; Chaulya, 2006). The order of the factors influencing dust emission rate of point source activities was determined as the significant emissions. For cover removal loading, the order of the factors influencing dust emission rate was as follows: moisture content > drop height > silt content > size loader > wind speed > loading frequency.

For recovered waste loading, the order of the factors influencing dust emission rate was as follows: drop height > silt content > size loader > moisture content > wind speed > loading frequency. For cover removal unloading, the order of the factors influencing dust emission rate was as follows: silt content > unloading frequency > drop height > moisture content > capacity of unloader > wind speed. For recovered waste unloading, the order of the factors influencing dust emission rate was as follows: drop height > silt content > moisture content > unloading frequency > capacity of unloader > wind speed.

Overall, the results of the calculated emission rates revealed a higher emission rate with an increase in wind speed, silt content, drop height, size loader/unloader, loading/unloading frequency, and a decrease in moisture content. This is consistent with comparisons of emission rates in previous research (Kim et al., 2020; Richardson et al., 2019; Wang et al., 2022). The rate of increment or decrement of emission rate of all factors is different for each LFM activity source as influencing factor rates differ between LFM activities. They all followed the same trend of increasing or decreasing with emission rate. The overall minimum and maximum emission rate of loading activity sources varied between 0.08 to 1.3 g/s, while unloading activity sources emitted 0.76 to 1.18 g/s, respectively.

Figure 7 demonstrates that the dust emission rate of cover removal loading is significantly higher than recovered waste loading, whilst the inverse is true for the unloading process. This can be explained by the recovered waste unloading being assumed to be unloaded on a screening plant in the calculation of emission rate, while the cover removal unloading was assumed to be unloaded on the ground as a stockpile using lower drop height range as opposed to the recovered waste unloading. The interpretation of the significantly higher emission rate of the cover removal loading over the recovered waste loading might be due to the field measured values of the validation studies used in developing these equations, assuming that the overburden materials contain a higher quantity of fine fractions, thereby producing a higher emission rate, which is comparable with the case of the landfill cover (geosynthetics clay liner) that contained fine materials. The cover removal loading activity source had the highest calculated emission rate of 1.11 g/s. Moisture content was the most influential driver of high and low emission rate for this source. This emphasizes the significance of low moisture content enabling dust formation during LFM activities. Consequently, against high emission rate related activity, dust mitigation or suppression methods such as applying tankers or bowsers to spray water around the mining area should be considered. In order to reduce dust emissions in the LFM area further, a lower drop height and smaller size loader should be considered.

The overall maximum/average/minimum point sources emission rate is 9/ 5.76/ 2.88 kg/hour, respectively. However, line sources maximum/average/minimum emission rates were only responsible for producing 0.18/ 0.072/ 0.036 kg/hour/m, respectively. The total emissions rate of area sources demonstrated a minimal rate of emissions.

Previous studies by Holnicki & Nahorski (2015), Joseph et al. (2018), Ni et al. (2018) and Srivastava et al. (2021), report that atmospheric particle dispersion models are affected by the emission inventory input data. The formulas used in this study were originally developed based on winter season emission inventory to predict the worst-case scenario (maximum concentration) of total suspended particular matter around mining activities in a different region with a larger leasehold mining area (Chaulya, 2006; Chaulya et al., 2019). Consequently, these factors may result in values that are overestimated (Huertas et al., 2012). Additionally, the formulas used for emissions estimation may carry large uncertainties because of variations in site operations, nature of mining, mitigation measures used, and climatic and geological conditions (Chakraborty et al.,

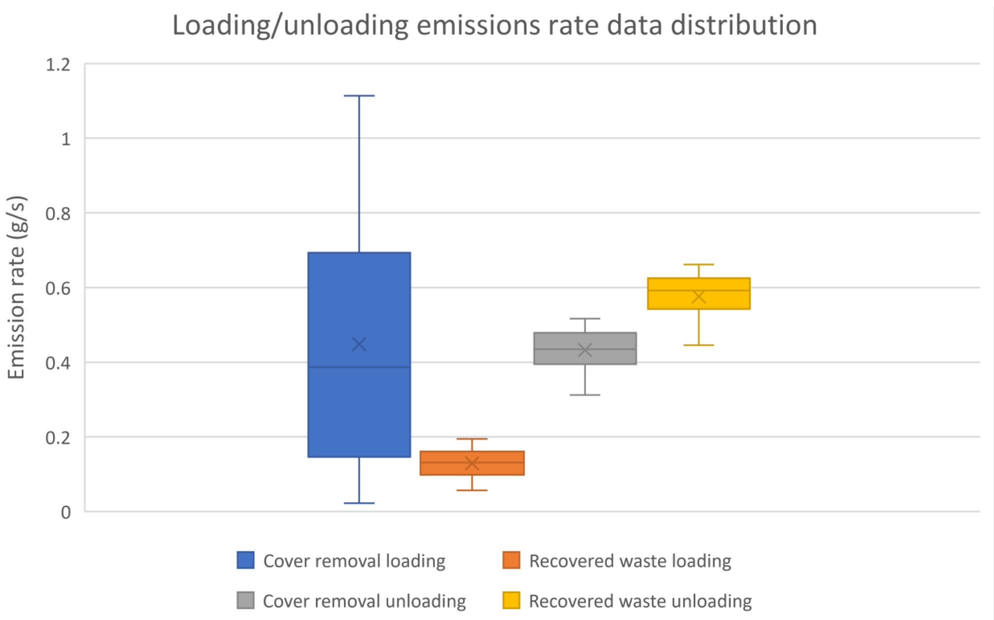

FIGURE 7: Box and whisker plot showing point sources activity emission rates.

2002; Holnicki & Nahorski, 2015; Richardson et al., 2019). This was evidenced in the use of U.S. emission estimation methods to other regions (Triantafyllou et al., 2021). Therefore, to minimise over- or under-estimation of emissions, experimentally based derivation of emission factors pertinent to the region and activities under study is required for a reliable calculation of fugitive dust emissions (Chaulya et al., 2022; Richardson et al., 2019). This can be clearly seen in the screening plant activity results of this study, which is very low, necessitating further research.

3.4 LFM Assumed Process Plan

The cover removal loading is the first unit operation in LFM activities (López et al., 2018; Parrodi et al., 2019; Parrodi et al., 2020). During this process, overlying soils (capping) are removed to expose the waste (Scott et al., 2019). Air pollution is a major concern in surface mining during the removal and transportation of overburden materials (Patra et al., 2016). Similarly, the final cover (capping) of a landfill site might include materials that aid in the removal of heavy metals, thereby increasing the risk of dust exposure and air pollution during LFM projects. Depending on the air velocity, particulate diameter, and drop height, these particles become airborne (Patra et al., 2016). Subsequent mining operation practices have also been identified as substantial causes of airborne particle pollution (Chaulya, 2006; Ghose & Majee, 2000; Onder & Yigit, 2009). The total assumed waste volume for 1 phase out of 5 is approximated as 22,400 m^3 (assuming landfill depth is 6 m) to 26,880 m^3 (assuming landfill depth is 7 m). The LFM project is estimated in this study to take about three months to complete each phase, considering one day of regular working hours (8 hours) and excluding weekends. For the purpose of visualizing the LFM processes and defining volumetric throughput, number of loading frequencies, and temporal ranges for each unit operation, a LFM flowsheet processes plan was produced based on a typical working day of 8 hours. The flowsheet diagram of LFM processes plan is presented in Figure 8 with respective emission rates.

The total number of loading frequencies ranged from 20 to 24 per hour in the flowsheet, based on the estimated volume range of landfilled waste under the study using a size loader of 2 m^3 operating for 8 hours per day. As illustrated in the flowsheet diagram (Figure 8), these numbers were distributed to cover and waste removal loadings and unloadings and adapted to the nature of LFM activities. Because a landfill cover (capping) is typically 1-2 m thick (Hölzle, 2017), the majority of loading numbers were in favour of waste recovery. The diagram provided a timescale for each individual activity dependent on the number of loading frequencies.

According to research by Lopez et al. (2019) and Parrodi et al. (2019), only recovered waste was supposed to be introduced to the screening plant operation that divides the fractions into various sizes. Therefore, it is the scenario considered in this study and thereby the volumetric throughput was in the range of 36-42 m^3/hour. The screening plant heavy machinery intended for use in the proposed plan has a capacity of 49.7 m^3/hour (TR510). Stockpiled exposed cover removal of waste volumetric throughput ranges were assumed to include the reclaimed soil from the screening plant activity, which is estimated to be 100 m^3/day out of the 288-336 range m^3/day. The volumetric throughput range estimates for exposed pit surface were calculated using both cover and waste removal volume per day. Screened waste is assumed to be loaded on to an articulated dump truck for off-site transportation (Parrodi et al., 2019). The screened waste volumetric throughput ranges meant to be transferred off-site were assumed to be 23.5-29.5 m^3/hour since the reclaimed soil from the screening plant was estimated to be 12.5 m^3/hour. Reclaimed soil is assumed to remain on site until completion of a LFM project. Based on articulated dump truck capacity of 13.7 m^3 (Caterpillar 725), the haul road number of loading frequencies per hour was estimated.

The one phase cumulative for the maximum/average/minimum point sources emissions over the lifetime of the LFM operation are approximately 5.04 tonnes (t) / 3.23 (t) / 1.61 (t), respectively. However, the one phase cumulative for the maximum/average/minimum line sources emissions over the lifetime of the LFM operation are approximately 100.8 (kg/m) / 40.32 (kg/m) / 20.16 (kg/m), respectively. These cumulative emissions are based on the assumed waste volume range of 22,400 m^3 to 26,880 m^3.

4. CONCLUSIONS

Predicting particulate matter pollution prior to the initiation of any landfill mining project is now possible with the adoption of surface mining activities using equations to calculate the suspended emission rate for each individual LFM related-activity. An equation was established to be used for the determination of time taken for a LFM project to be completed to enable the estimation of cumulative emissions during the lifetime of a LFM operation. Cumulative maximum, average and minimum emission rates of different sources over the lifetime of the landfill mining operation based on one phase were determined. Results show that the estimated emission rate is higher with an increase in wind speed, silt content, drop height, size loader/unloader, loading/unloading frequency, and a decrease in moisture content for all LFM activities. It was also observed from the emission results that point source activities were the major sources of estimated emissions in this study. The use of fog cannons or water spraying, as well as the method of road sprinkling should be considered to reduce PM emissions. The screening plant activity should be explored in more detail since the results showed a relatively low emission rate.

The one phase out of five cumulative for the maximum, average and minimum point sources emissions over the lifetime of the landfill mining operation calculated in this study are approximately 5.04 t, 3.23 t and 1.61 t, respectively. However, the one phase cumulative for the maximum/average/minimum line sources emissions over the lifetime of the landfill mining operation are approximately 100.8 kg/m, 40.32 kg/m and 20.16 kg/m, respectively.

The formulas used in this study were not validated against LFM activities. Therefore, experimentally based derivation of emission factors reported here should be validated against actual LFM site activities for use in air

FIGURE 8: Flowsheet illustrating LFM processes and defining volumetric throughput, number of loading frequencies and timescales ranges for each unit operation with their respective emission rate.

quality decision-making frameworks that are informed by emission inventories. Equations should be validated using monitoring stations for each LFM activity separately to minimise overlapping of emission rates for the different activities. Validation studies should be conducted within a small scale to limit dust emissions into the atmosphere. They also should be performed without using mitigation measures between the source of emission and monitoring stations, as well as avoiding obstructive objects where monitoring equipment is placed around a LFM area in order to obtain an optimum level of accuracy in comparison with formulas used in this research methodology.

Due to the increase in development activities and projected climate change impacts (rainfall intensity, warmer temperatures), transport of atmospheric particulates is potentially becoming more significant.

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COLUMNS AND SPECIAL CONTENTS

This section comprises columns and special contents not subjected to peer-review

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CISA

BOOKS REVIEW

WASTE MANAGEMENT IN DEVELOPING COUNTRIES

Edited by: Hassan El Bari, Cristina Trois

The book is aimed at a broad readership, from sector experts to readers without consolidated experience in this field. Indeed, through its 12 chapters, the book provides a detailed analysis of the state-of-the-art of solid waste management (SWM) in developing countries (DCs), appropriate technologies to be adopted, and the impact of inadequate SWM on the environment and human health, also considering the greenhouse gas (GHG) emissions. Social aspects are also discussed. World-renowned experts were involved in writing the book.

The first Chapter summarises crucial waste characterisation, collection, and transportation elements. It is stressed that the world population will continue to grow significantly over the next few decades, with a prevailing push coming from DCs. Consequently, solid waste generation will increase in those regions, and appropriate strategies will be crucial to avoid making the situation more troublesome. Furthermore, it is highlighted that waste management planning, design, and implementation depend on using reliable data. The authors emphasize that waste generation per capita, waste composition and density are essential elements to consider for developing adequate strategies, and they are strongly influenced by factors such as the regional income level and seasonal variation. All these aspects are essential, although sometimes underestimated.

Chapter 2 focuses on landfill disposal in DCs. The authors highlight that, in these areas, dumpsites often represent the prevalent SWM method. An innovative approach is presented based on the 3S idea (Sanitisation, Subsistence economic, Sustainable landfilling). Then, discussing the sustainable landfill, the authors introduce the concept of the bioreactor landfill, and among the possibilities, they identify the semi-aerobic landfill as probably the most appropriate in DCs. It is based on natural air ventilation, without high-technological tools, to favour oxygen in the waste mass by natural advection, speeding up waste stabilization and leachate nitrogen removal. Thus, considering that, at least in the short-medium term, in several DCs, a considerable amount of waste will continue to be disposed of instead of recycled or composted, the sustainable landfill appears strategic.

In Chapter 3, an innovative model (i.e., the WROSE) is presented and applied in a case study in South Africa. It aims to assess GHG emissions associated with SWM practices, considering alternative waste diversion pathways. Although similar models have been used internationally, the WROSE represents a reliable alternative with potential use in countries from the Global South. Additional interesting aspects concern the estimate of the landfill space savings associated with waste diversion.

In Chapter 4, an intriguing approach is discussed. It aims to identify the best leachate treatments for cost efficiency based on site-specific conditions. Thus, although it focuses on Latin American areas, it could also be implemented in other DCs (and probably even in industrialised countries). It could appear provocative to claim that leachate qualitative standards are often inefficient when only based on tabular limits; however, the author indirectly refers to the concept of health risk analysis that has spread over the last decades in Europe and the USA. Consequently, considering site-specific conditions, it is highlighted that the so-called fate and transport models should be applied to avoid resource squandering.

Chapter 5 implements the WROSE model in another South African case study, i.e., the Garden Route District

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Municipality; thus, it has elements in common with Chapter 3. The authors compared three Scenarios:

(1) the disposal of unsorted, untreated MSW to landfill;
(2) unsorted and untreated MSW are mechanically separated, and then the organic fraction undergoes anaerobic digestion;
(3) similar to the second scenario but with the organic fraction that undergoes composting.

Within a 50-year projection, the landfilling of all MSW represented the worst scenario in terms of GHG emissions, highlighting additional reasons for minimising the waste to be disposed of in such sites.

In Chapter 6, the legislative aspects of solid waste management are discussed. The gaps often affecting DCs are highlighted, particularly the lack of reliable data, financial availability, awareness, and regulations. Concerning regulations, the issues interest their adoption and enforcement; the latter is sometimes even more challenging as it requires well-established governance that allows efficient monitoring and control. Thus, solutions are discussed for E-waste, plastic waste, and legislation in general. Finally, the Waste Management Law 28-00 issued in 2006 in Morocco is presented as a case study offering engaging insights.

Chapter 7 deals with the waste informal sector, focusing on the MENA region. It is not easy to find such a detailed document on this topic; indeed, the organizational structure of the informal waste sector is described with great accuracy. Thus, the chapter is of undoubted value. However, the disorder that sometimes emerges in the text can make some parts of the Chapter challenging to follow. Experiences from three North African countries are presented, i.e., Algeria, Morocco, and Tunisia. It is emphasised that the informal sector may play a crucial role in DCs to improve waste separation and recycling, reducing waste disposed of in landfills and dumpsites, aiming at the circular economy. Issues affecting waste pickers are discussed, and several reasonable improvements are proposed, taking previous research and successful experiences as a reference.

Chapter 8 discusses the Max-Neef's Fundamental Human Needs (FHNs). They are used as social indicators of sustainability in the context of waste management, taking eight South African townships as case studies. The nine FHNs through which the authors assess waste management are the following: subsistence; affection; participation; creation; understanding; idleness; protection; freedom; identity. It is unique to find such analysis in technical manuals about solid waste, making the chapter a characterising element of the book. Additionally, the methodology used in the quantitative and qualitative data collection appears strong and can be taken as a reference for other studies. The only discordant note: the results could have been presented more clearly.

Chapter 9 introduces the concept of urban mining, i.e., actions and technologies aiming to recover materials and energy from products of the urban environment. Thus, focusing on South Africa, some of the high-priority waste streams identified by the South African Research, Development and Innovation waste roadmap are considered: glass from MSW, waste tyres, paper mill sludge, and bone from organic waste. They are discussed, aiming at their reuse as building materials. Indeed, the massive amount of natural resources exploited worldwide in the construction and demolition sector and the waste produced must be kept in mind. Therefore, using waste as a resource to generate new hybrid materials represents a great opportunity.

In Chapter 10, composting is discussed. It must be kept in mind that the organic fraction represents the higher percentage of MSW generated worldwide. Thus, its sustainable management is paramount, and composting presents multiple advantages. Indeed, it extends landfill lifetime, reduces GHG emissions, and produces an organic fertiliser and soil improver. The authors stress that DCs need to improve composting regulations and show examples from industrialised countries. Some technical aspects are presented. Furthermore, the need for adequate compost quality standards is discussed, highlighting that they are rarely considered in DCs.

Chapter 11 introduces the biochemical conversion of waste with the primary purpose of producing biogas. Thus, anaerobic digestion (AD) is discussed. The authors point out that many kinds of organic waste can be used. However, such feedstocks can have different characteristics to be considered, including the pathogenicity and presence of chemical pollutants, mainly in the cases of manure and sludges, less for food and green waste; thus, pre- and post-treatments can be required. AD products are the biogas and the digestate, obtaining energy and material from waste, respectively. In DCs, the digestate can be used as a fertiliser, while the biogas is mainly for cooking or lighting. It must be considered that AD has already been used at the industrial level in industrialised countries; however, appropriate forms of such technology can also be implemented in DCs and are described in the Chapter. It is crucial to keep in mind that the biogas from AD represents a form of renewable energy strategic in fighting climate change; indeed, on the one hand, no fossil fuels are used; on the other hand, during biogas (mainly methane) combustion, CO2 is generated. It is important to note that when landfills and dumpsites receive organic waste, methane is generated in anaerobic conditions, and it is a GHG several times more potent than CO2. The Chapter gives essential elements regarding challenges, benefits, and strategies to consider for making AD in DCs successful.

The last Chapter is dedicated to the thermochemical conversion of organic waste. Although it could appear a high-tech topic, it deserves to be discussed also for DCs. With this in mind, the authors investigate many technologies whose potential implementation depends on site-specific conditions. Thus, it is highlighted that the waste characterisation plays a crucial role. Among the technologies discussed, the one that could probably find more straightforward implementation in DCs is hydrothermal carbonisation. Indeed, temperatures between 180°C and 260°C are required, and the main product is hydrochar that can be used as solid fuel, activated carbon for gas and water purification, or soil amendment.

Overall, the book discusses a broad topic that would appear challenging to present exhaustively in a monograph. Notwithstanding, it succeeds and adequately examines technical, governance, social, legislative, and environmental aspects of SWM in DCs. It has all the potential to represent an excellent scientific reference for anyone wanting to deal with the topic or merely understand more about it.

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DETRITUS & ART / A personal point of view on Environment and Art

by Rainer Stegmann

When I am in Venice I always visit the Guggenheim Museum Venice. It is a small beautiful museum at the Canale Grande with some extraordinary pieces of Artwork. During my last visit I came across the sculpture from the Italian artist Umberto Boccioni (19.10.1882- 17.8.1916). He is one of the co-founders of futurism in Italy. The Futurism, is an early 20th-century artistic movement centered in Italy that emphasized the dynamism, speed, energy, and power of the machine and the vitality, change, and restlessness of modern life (Britannica). The Futurist aesthetic platform advocates the use of various materials in a single work, the rejection of closed form, and the suggestion of the interpenetration of form and the environment through the device of intersecting planes. (Guggenheim Museum , Venice, Italy, https://www.guggenheim.org/artwork/580).

Umberto Boccioni: Dynamism of a Speeding Horse + houses (Guggenheim Museum, Venice, Italy)

The here presented sculpture is called "Dynamism of a Speeding Horse + Houses". Boccioni assembled wood, cardboard, and metal, with painted areas. His work is influenced by the Cubism of Pablo Picasso and Georges Braque. In his sculpture he used the horse to demonstrate his observation that the nature of vision produces the illusion of a fusing of forms. When the distance between a galloping horse and a stationary house is visually imperceptible, horse and house appear to merge into a single changing form (based on text from Lucy Flint, Guggenheim Museum, Venice, Italy) https://www.guggenheim.org/artwork/580).

This is the interpretation of the expert Lucy Flint from Guggenheim Museum, but of course everybody may have his own. I find the sculpture very esthetic; the different used materials fit well together and form a homogeneous sculpture. The used materials are very different like wood, cardboard and metal. Due to the similar colour the different materials combine to form a unit.

I do not think that Boccioni wanted arouse our interest for recycling, because during his lifetime used materials and products were not easily thrown away. It was in general the goal to reuse used materials for other purposes. Waste composition during those times consisted mainly ashes and dust, 2.7% paper, 3.4% slags and coal residues (Waste Composition in 1895 in Berlin, Germany from Breer et. al. 2010). The waste actually did not contain the materials used by Boccioni. But artwork can be interpreted differently in different times. I see more the recycling aspect of the used materials, the potential for reuse. Once more an artist shows us the value of these materials when they are completely or partly reused. I presented already the bull

Umberto Boccioni: Dynamism of a Speeding Horse + houses (Guggenheim Museum, Venice, Italy).

head created from bicycle parts by Pablo Picasso, which is a very prominent example. I see the use of the materials in some cases as upcycling, to increase the value of the materials compared to their past use. Of course, looking at the sculpture created by Boccioni I did not relate it immediately to waste, to recycling. But on the second look the different used materials caught my attention also because they fit so well to each other although they are so different. How would Boccioni react if he would know about my interpretation? Maybe he would turn over in his grave – a German saying.

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In the next edition I will present a work of art by the contemporary artist Eddy Ekete (1978) from Kinshasa, Democratic Republic of Kongo, Africa. He makes installations on persons in their environment in Kinshasa highlighting specific waste problems.*

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