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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

CIRCULAR ECONOMY BETWEEN BUSINESS AND BIODIVERSITY

The increasing loss of biodiversity is one of the most dramatic consequences of climate change and human impact. Biodiversity provides ecosystem services essential for human life such as food, materials, clean water, climate regulation, and many others. According to IPBES (Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services), the main drivers of biodiversity loss can be found in five main causes: (i) changes in land and sea use (human actions have significantly altered 77% of land and 87% of the ocean); (ii) Overexploitation of species and natural resources (in 2022, humanity was using nature and its resources at rates 1.75 times faster than the planet's ecosystems can regenerate); (iii) Climate change (human actions have warmed the globe by more than 1°C compared to pre-industrial levels); (iv) Pollution (approximately 11 million tonnes of plastic are dumped into the world's oceans each year, affecting more than 250 animal species); (v) Invasion of alien species (since 1980, cumulative records of alien species have increased by 40%, negatively affecting native species) (Ellen Mac Arthur Foundation, 2021).

The circular economy aims at changing the production and consumption systems eliminating waste and pollution to reduce threats to biodiversity, circulating products and materials to give space for the restoration of biodiversity, and regenerating nature to help biodiversity.

Biodiversity and ecosystem services

The definition of biodiversity universally accepted could be the following: "Any kind of variability among living organisms, including, inter alia, terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are part; it includes diversity within species, between species and between ecosystems" (Secretariat of the Convention on Biological Diversity, 2011).

This definition puts the focus on the concept of variability, which can be considered at three different levels: (i) within species, when considering genetic and population-level measures; (ii) between species, when considering the variation of species level; and (iii) within ecosystems, when considering the regional and landscape level.

The concept of biodiversity, hence, is a multifaceted and encompassing idea that goes beyond just the diversity of species in an ecosystem. It encompasses the entire range of living organisms, including all species of plants, animals, fungi, and microorganisms, as well as the genetic differences within these species, the interactions between them, and the ecosystems they form a part of. The concept

of biodiversity is crucial because it underpins the health and functioning of ecosystems, which in turn sustains human well-being. Biodiversity supports ecosystem services that are essential for food production, medicine, climate regulation, and cultural values.

Since ecosystem services are services provided by nature to humans they can be simply defined as: "The benefits people obtain from ecosystems" (Millennium Ecosystem Assessment-MEA, 2005). MEA provided a classification in four categories of ecosystem services, which is still considered as universal:

- **Provisioning.** The goods or products obtained from ecosystems such as food, freshwater, timber, and fiber.
- **Regulating.** The benefits obtained from an ecosystem's control of natural processes such as climate, disease, erosion, water flows, and pollination, as well as protection from natural hazards. "Regulating" in this context is a natural phenomenon and is not to be confused with government policies or regulations.
- **Cultural.** The non-material benefits obtained from ecosystems such as recreation, spiritual values, and aesthetic enjoyment.
- **Supporting.** The natural processes such as nutrient cycling and primary production that maintain the other services.

Business and biodiversity

Biodiversity could be seen as an integral component of natural capital. Nowadays, the relationship between economic activities and the environment is universally known, meaning that business can generate positive and negative impacts on the environment and at the same time, it relies on the resources that the environment provides to it. Therefore, businesses both impact and depend on biodiversity and the ecosystem services it provides.

The concept of dependency refers to the organization's reliance on the use of biodiversity, meaning the biological resources of species and their interaction with ecosystems and ecosystems services, such as pollination, water flow regulation, water filtration. These dependencies are directly and indirectly interlinked with biodiversity impacts, which can be defined as "a change in the diversity of ecosystems and/or species that may take place because of business activities". Such impacts can be direct, where business directly exploit biodiversity resources (agriculture, forestry, fishing) or indirect, where business affect indirectly biodiversity and natural resources (fashion, finance, manufacturing) (Mace et. al. 2012). However, the relationship

between business and biodiversity is not yet well-known at business strategy level, even though the link between economic activities and biodiversity/ecosystem services is extremely strong. In this sense valuing biodiversity and ecosystem services can facilitate business in understanding the real value of biodiversity and the economic benefits that conservation practices could bring to them. Economic valuation of ecosystem services and biodiversity can help society and business in let them understand that biodiversity and ecosystem services are scarce and that their depreciation or degradation is bringing costs to society. Therefore, valuing ecosystem services and biodiversity is a needed step for businesses for increasing their awareness concerning the importance of these services for the efficiency of such economic activities in the long term.

Biodiversity and circular economy

As it has been previously shown, one of the main drivers of biodiversity loss is the wasteful and unsustainable economy characterized by a linear flow. Circular economy, as well known, is a closed system focused on the values of materials, resources, and products, which can bring benefits in different sectors such as reduction in GHG emissions, materials, reduction in primary material consumption, in sustainable agriculture (it has been estimated that designing food products, such as wheat, potatoes, dairy, etc., for nature using diverse, upcycled, lower-impact, and regeneratively grown ingredients can bring to a 50% reduction in the negative impact on farm-level biodiversity), in economy, in the form of net material cost savings and reductions in externalities, and in the environment (e.g. in the global plastic sector CE can bring to a strong reduction in plastic leakage).

Biodiversity presents different features. It is composed by ecosystems, genes and species and therefore, when it comes to measure the status and/or the impact of biodiversity, it can be tricky. Therefore, for each component, different aspects must be measured (UNEP-WCMC et al., 2022):

- a) Ecosystems - two aspects must be considered: the ecosystem extent (spatial features of the ecosystem) and the ecosystem condition (composition of ecological communities);
- b) Species - the population size must be measured (MSA - Mean Species Abundance is one of the most utilized indicators in this framework), and the extinction risk. So, it provides an idea on the health of a species in an ecosystem;
- c) Genes - only the genetic diversity must be considered, which describes the variability in genetic characteristics within a given species, or within an ecosystem. It can provide an indication of its resilience to change.

Hundreds of different indicators, indices, or metrics that measure the status of one of each component exist and the challenge relies in finding the more holistic one.

Final remarks

In conclusion, biodiversity is indispensable for the functioning of ecosystems and the provision of ecosystem ser-

vices. The loss of biodiversity is a pressing global concern with far-reaching consequences, and addressing it is paramount for sustainable development. The circular economy offers to business and human activities an option to mitigate the main drivers of biodiversity loss.

Understanding this relationship and valuing biodiversity and ecosystem services can enhance business awareness of their long-term benefits and the costs associated with their degradation.

The crucial point remains the full enforcement of the circularity of the circular economy, which even today encounters various implementation challenges and therefore still has a long way to go before it can be considered a business system that is neutral towards biodiversity.

Technical, regulatory, and authorization barriers (e.g., End of Waste) in recycling waste, the difficulty in producing pollutant-free goods and thus fully reintegrating them into the secondary production cycle, the need to economically support the more fragile and less aware countries of the impacts of uncontrolled waste management, are all aspects that slow down the positive impact of the circular economy on biodiversity conservation. If biodiversity can truly be protected and promoted through the conscious use of resources and their conservation, the new economic paradigm that bases its material circularity on the recovery of materials through waste recycling, does not, however, address the issue of uncontrolled consumption of goods and services in an equally important way, and, indeed, emphasizes the need to always find new sources, alternatives to virgin raw materials, but not to drastically reduce their consumption.

If we will be able to define and quantify the ecosystem values on which various corporate businesses depend and impact, the circular economy could still make progress in preserving and promoting biodiversity.

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APPLICATION OF NANOCELLULOSE IN FOOD PACKAGING - A SWOT ANALYSIS

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ABSTRACT

Environmental concerns due to the wide use of plastic in food packaging have become one of the most significant challenges in the world. Consequently, the research in developing sustainable materials for food packaging has accelerated. Nanocellulose-based packaging is a biodegradable, renewable, and antimicrobial material with some competitive physicochemical characteristics when compared to plastic packaging. However, there has been insufficient research on a holistic discussion of the potentials and drawbacks of nanocellulose as well as its production, applications and disposal regarding sustainability. This study aims to evaluate the application of nanocellulose in food packaging in the view of environmental sustainability. It gives an exhaustive overview of the essential aspects from the production to disposal of nanocellulose through a literature review. Then, a SWOT (Strengths, Weaknesses, Opportunities, Threats) analysis is used to evaluate the potentials and drawbacks of applying nanocellulose in food packaging. It has been observed that the physicochemical properties of nanocellulose materials have the potential to be used in food packaging with fewer negative impacts on the environment. Furthermore, it supports the top tiers of the waste hierarchy and a circular economy. However, some challenges need to be addressed to ensure the safe and effective use of nanocellulose in food packaging, including high expenses, a lack of guidelines, and potential hazards to people and the environment. To eliminate these uncertainties, more studies need to be performed on applying nanocellulose in food packaging.

1. INTRODUCTION

Packaging is essential for preventing food items from spoilage, damage, contamination, or deterioration and shielding sensitive bioactive compounds from damaging physical and environmental forces (Ahankari et al., 2021; Ashfaq et al., 2022). With 85%, food packaging comprises the largest part of the whole packaging industry (Silva et al., 2020). Therefore, it is the key lever when aiming to make changes in the packaging industry. The most popular packaging materials are plastics (ibid.). Their characteristics show significant advantages in comparison to other materials and thus make them especially suitable for application in food packaging. Plastics are easy to use due to their light weight and flexibility, have low production costs making them accessible for a wide range of producers and users, and show excellent mechanical and barrier qualities which support the main goal of packaging - protecting the food items (Sangroniz et al., 2019).

In addition, Silva et al. (2020) state that the market for plastic packaging has been expanding by 20 to 25 per cent

annually. In Europe, 23 million tonnes of plastic packaging are generated annually which is estimated to reach 92 million tonnes by 2050 (Ncube et al., 2021). Furthermore, since 1950, the amount of plastic waste in the world has been estimated at 6300 million tons, of which 4977 million tons have been accumulated in landfills or waterways (Silva et al., 2020). There is a growing concern about the widespread usage of petroleum-based plastics which are non-renewable, non-biodegradable, and contribute to an environmental crisis (Azeredo et al., 2017). Plastic waste that starts to decompose results in microplastic particles that intrude the food chain, carrying pollutants and hence impacting animal and human health (Azeredo et al., 2017; Chai et al., 2023). Furthermore, microplastic and global warming supplement each other and, as a result, contribute to worsening climate change (Bergmann et al., 2022).

Consequently, the need for packaging materials that are renewable (not petroleum-based), biodegradable, and cause minimal environmental impacts has increased as stated by Ahankari et al. (2021). One such substance is nanocellulose which has demonstrated exceptional promise



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for food packaging due to its availability, renewability, degradability, and physical-chemical characteristics (Silva et al., 2020). ISO/TS 20477:2017 defines nanocellulose as a structure with at least one dimension equal to or less than 100 nm, which can be subdivided into cellulose nanocrystals and cellulose nanofibrils formed chemically and mechanically, respectively (Fotie et al., 2020).

Nanocellulose is made from natural cellulose that has been fermented by plants, seaweeds, and microorganisms via chemical (Zhou et al., 2021), biological and mechanical processes like pressure refiner and grinder treatment (Sarwar et al., 2018). The reduction of the particle size improves the properties of commercial cellulose fibres (Siddiqui et al., 2023). Depending on the sources and techniques used for extraction, the types of nanocellulose have a variety of structures and sizes (Amara et al., 2021). Nanocellulose products, such as cellulose nanocrystals, nanofibrillated cellulose, and bacterial nanocellulose, have outstanding physicochemical properties that have the potential to significantly alter a variety of commercial applications like the packaging industry (Ferrer et al., 2017). Therefore, different methods of applying nanocellulose for packaging, like coating and casting have been developed (Ferrer et al., 2017).

Although most of the studies considering the application of nanocellulose in food packaging (e.g., Ma et al., 2022; Mu et al., 2019) include both positive aspects and challenges, a holistic overview and a linkage of the material's features to its production, uses and disposal are still missing in this research field. Therefore, due to the importance of a thorough discussion, this study aims to systematically evaluate to what extent the application of nanocellulose in food packaging can potentially alter and minimise food packaging waste. To achieve this, two objectives are addressed, including (1) to examine the production processes, current uses of nanocellulose in food packaging and its disposal, and (2) to identify and discuss potentials and drawbacks of nanocellulose in food packaging in the view of sustainability and circular economy.

2. METHODS

The methodology of the study consisted of two parts. The first objective was met by a literature review. Moreover, a SWOT analysis was conducted to meet the second objective mentioned above. The main area of study was limited to a circularity and sustainability perspective, especially environmental sustainability, which creates the system boundaries of the analysis. Therefore, technical aspects are included to an extent where they support answering the objectives but not described in detail. The period of the literature considered is 10 years, therefore only studies completed and published after 2013 are analysed. This was brought on by how quickly the field of technology research was developing. Furthermore, a study conducted by Garrido-Romero et al. (2022) has shown a rapid abundance of publications focusing on nanocellulose in food packaging within the past ten years, supporting the chosen timeframe. Since the study was technology-focused, there was no geographical restriction. In addition, as the title im-

plies, nanocellulose's application in food packaging was the study's sole emphasis.

The first objective comprises a literature review of the production processes, applications and disposal of nanocellulose. To gather information on those areas, the following keywords were applied: "nanocellulose" OR "cellulose" OR "cellulose nanofibres" OR "cellulose nanocrystal" AND "production" OR "sources" OR "processes" OR "applications" AND "food packaging" OR "disposal". To find full-text articles that meet the aforementioned requirements, the databases Scopus, ScienceDirect, Google Scholar and Wiley Online Library were used.

The second objective was additionally accomplished by conducting a SWOT (Strengths, Weaknesses, Opportunities, Threats) analysis using the information extracted from the literature. This well-established approach was chosen because it facilitates decision-making through evaluating and comparing potentials and downsides as emphasised by Puyt et al. (2023) and Yuan (2013). Previous SWOT assessments in the context of environmental and waste management have evidently demonstrated potentials of improving the systems in question (Yuan, 2013).

In order to choose matching keywords that would facilitate finding appropriate literature for the SWOT analysis, the previously mentioned study conducted by Garrido-Romero et al. (2022) was consulted. The most common keywords in the context of nanocellulose in food packaging were analysed. Based on these results, the following keywords were applied in the literature review: "nanocellulose" OR "cellulose" OR "cellulose nanofibres" OR "cellulose nanocrystal" AND "food packaging" OR "waste" OR "plastic waste". The databases used for research were the same as mentioned in the first objective.

3. RESULTS

3.1 From production to disposal of nanocellulose

3.1.1 Production processes

Salimi et al. (2019) state that the main steps in producing nanocellulose are (1) selecting proper cellulose sources, (2) pre-treatment, and (3) extraction. Firstly, choosing appropriate cellulose sources, like those with a high cellulose content, is one of the most significant factors in the production of nanocellulose (Salimi et al., 2019). With regard to this, Ferrer et al. (2017) emphasise that the source of cellulose influences the characteristics of the produced nanocellulose, such as morphology, size and crystallinity. For instance, when materials like wood, sugar beets, or potato tubers are used as the source, the diameter of the nanocellulose produced ranges from 5 to 60 nm (Klemm et al., 2011). Wood has a porous structure that combines high strength and low density (Kargarzadeh et al., 2017). On the other hand, when low-molecular-weight sugars and alcohols are used for nanocellulose production, the diameter of the resulting nanocellulose ranges from 20 to 100 nm (Klemm et al., 2011). Cellulose nanocrystals, which are a type of nanocellulose, are mainly sourced from materials like wood, cotton, tunicin, algae, and bacteria with a diameter ranging from 5 to 70 nanometers (ibid.). Cellulose nanocrystals have excellent mechanical properties (Kargarzadeh et al., 2017).

Secondly, according to Palacios Hinestroza et al. (2019), pre-treatment is required to eliminate impurities and lower energy consumption during the mechanical breakdown of cellulose. Each type of biomass source consists primarily of cellulose, lignin, hemicellulose, pectin, and other smaller components that have various physical and chemical properties which hence require different pre-treatment methods (Lunardi et al., 2021). These can be broadly divided into two groups, non-biological and biological, depending on the type of treatment process applied. Physical, chemical, and physico-chemical pre-treatment methods are the three main categories of non-biological pre-treatment techniques, which do not use any microbiological treatments. Kumar and Sharma (2017) have reviewed the characteristics and applicability of the most common methods. According to Ko et al. (2020), alkaline pre-treatment, which is an eco-friendly method, can increase the production of nanocellulose by removing 73% of hemicellulose and 50% of lignin from wood fibre. So, pre-treatment makes the cellulose accessible for the procedure.

Thirdly, there are three main extraction processes for producing nanocellulose: mechanical, chemical, and biological (Salimi et al., 2019). The different processes lead to variations in the types and characteristics of the produced nanocellulose (Phanthong et al., 2018). Mechanical processes utilise mechanical forces that cause the cellulose to become defibrillated and create a crucial tension centre in the fibrous substance (Salimi et al., 2019). According to Phanthong et al. (2018), the most common mechanical techniques are ultrasonication (using ultrasound for defibrillation) and high-pressure homogenization (by injecting cellulose slurry into a container at high pressure and speed). Chemical processes employ chemical substances (such as acid hydrolysis and metal salts) that break down the cellulose. The most common chemical process is acid hydrolysis, which uses sulfuric acid to isolate nanocellulose from cellulosic materials and break down the disordered regions (Phanthong et al., 2018). Moreover, different factors, including reaction temperature, time, acid type, and acid concentration, have a substantial influence on the process (Tiwari and Sanjog, 2023). Biological processes are applied when cellulose is degraded by biological reactions (Salimi et al., 2019). Enzymatic hydrolysis is one example of a biological process that uses enzymes to break down or change cellulose fibres (Phanthong et al., 2018). This process often requires moderate conditions. Since this process takes a lot of time, it is always combined with another technique (ibid.).

To optimise the production process, researchers have also combined several of these extraction strategies (Ferrer et al., 2017). The introduced processes highlight the variety of methods available for producing nanocellulose. This helps understand how the environmental impacts of nanocellulose can be minimised starting from the early stages of the material's lifecycle.

3.1.2 Uses in food packaging

Thermoplastics are a major conventional plastic category commonly used in food packaging since they become soft when heat is applied (Marsh and Bugusu, 2007). Ther-

moplastics most widely used in food packaging are polypropylene (PP), polystyrene (PS), polyvinyl chloride (PVC), polyethylene (ET), and polyethylene terephthalate (PET). The properties of the different thermoplastics are mainly influenced by the composition of the raw materials and physical and environmental conditions (such as linearity and molecular weight) (Teck Kim, Min and Won Kim, 2014). They have shown good performance in terms of thermal and mechanical behavior, water vapor transfer rate, gas permeability, and chemical resistance (ibid.). They are also lightweight and low-cost, which makes them suitable for food packaging applications (Ncube et al., 2021). However, they are nonbiodegradable, derived from fossil fuels, and responsible for different environmental issues like the emission of greenhouse gases (ibid.), underlining the need of more environmentally friendly alternatives, such as nanocellulose.

Nanocellulose has a variety of uses in the food packaging industry. According to Ma et al. (2022), the most common application of nanocellulose is as a reinforcing phase for food packaging materials, although it may also perform as matrices for other materials. Moreover, Mu et al. (2019) emphasise that nanocellulose is added to traditional polymers to improve mechanical and biodegradable characteristics. Nanocellulose can be incorporated in two different ways. It can be coated to traditional food packaging, in the form of films, for instance, or by fabricating multi-layer materials, through casting and evaporation, extrusion, and layer-by-layer (Ashfaq et al., 2022; Ferrer et al., 2017).

It is well-recognized that coatings are an efficient and sustainable technique for integrating nanocellulose into food packaging (Fotie et al., 2020; Li et al., 2015). The coating improves several qualities of food packaging materials (Li et al., 2015), including antibacterial capabilities, moisture retention, control of food oxidation, and providing the substrate with different types of barriers (like water, oxygen, and grease) (Fotie et al., 2020). Furthermore, there is the possibility of using edible and environmentally friendly polymers that are either dipped or coated directly onto the food surface (Li et al., 2022; Zhang et al., 2021).

According to Li et al. (2022), casting is one of the simplest methods for producing nanocellulose packaging materials. It is a process that involves drying films of neat or modified nanocellulose by evaporating water or solvent from the nanocellulose at a specific temperature (Ferrer et al., 2017). It enhances mechanical and water resistance characteristics, increases tensile strength, decreases water solubility, and leads to high UV resistance (Ferrer et al., 2017; Li et al., 2022). Extrusion is an economical and productive method to fabricate nanocellulose films, creating good oxygen and water vapour barrier qualities (Li et al., 2022). In addition, the material is extruded via an extruder fitted with a metal die in the appropriate pattern to make the required shape (ibid.).

In the layer-by-layer method, a skinny layer of functional substances deposits onto surfaces. Fotie et al. (2020) emphasise that it takes numerous coats of nanocrystal cellulose dispersions on solid layers to achieve the necessary thickness to meet the demands for wet strength, mechanical qualities, and gas barrier capabilities. In conclusion, the variety of nanocellulose applications provides

a widespread base for working with nanocellulose in food packaging. In particular, coating and casting options hold positive aspects that will be elaborated on in the SWOT analysis.

3.1.3 Disposal

The end-of-life stage is disposal, a significant phase in circular economy. A vital part of circular economy is looping the waste flow into producing new goods and services (Johansen et al., 2022). Nanocellulose packaging can facilitate the transition from a linear economy to a circular economy in different ways. Firstly, recycling and composting are two main methods for the disposal of nanocellulose (Shahzad et al., 2023; Shanmugam et al., 2019), offering a promising approach to closing the resource cycle, improving the environmental situation and retaining value (Meister et al., 2022). In contrast, using conventional plastic, which most often follows a linear economy (production, use, and disposal), can pose severe threats to the environment and human health (Johansen et al., 2022). Secondly, the production of nanocellulose from wastes (such as food or agricultural waste) aligns with the principles of the circular economy (Mateo et al., 2021).

Since nanocellulose is biodegradable and non-toxic, composting is an affordable and environmentally friendly way to dispose of the material along with other organic waste, which also contributes to lowering the quantity of waste that ends up in landfills (Shahzad et al., 2023). The packaging will decompose into its natural compounds, such as water and carbon dioxide, and can then be used as a nutrient-rich soil fertiliser (Fotie et al., 2020).

Some types of nanocellulose-based materials can be fully recycled several times or converted into other valuable materials like carbon black, a material mainly used as a pigment and reinforcement in vehicle tyres (Uttaravalli et al., 2022), in the recovery process (Shahzad et al., 2023; Shanmugam et al., 2019). Shanmugam et al. (2019) recycled nanocellulose films into films with approximately 70 percent of the primary film's strength. The process of recycling leads to a lower density, less homogeneity, and poorer strength. However, the recycled nanocellulose materials were still able to deliver acceptable performance for most packaging applications. The disposal options of nanocellulose and its recycling ability have an impact on its use and will be a factor in the following SWOT analysis.

3.2 SWOT analysis of nanocellulose in food packaging

The following SWOT analysis determines potentials and drawbacks of nanocellulose in food packaging. All aspects are chosen with regard to nanocellulose's role in and contributions to sustainability, namely environmental impacts, and its potential of circularity according to the system boundaries aforementioned. This includes chemical and physical details that impact the mentioned perspectives but excludes a deep dive into technical details.

3.2.1 Strengths

Nanocellulose incorporated into packaging is biodegradable, compostable and easily recyclable, without caus-

ing any environmental harm (Arrieta et al., 2016; Fotie et al., 2020). By-products from recycling nanocellulose can be used in a different context, as mentioned before (Shahzad et al., 2023). It can be obtained from a wide range of biological sources (Silva et al., 2020). Depending on the source, nanocellulose is relatively low in price, compared to other biodegradable plastics alternatives (Arrieta et al., 2016; Siddiqui et al., 2023). According to Fotie et al. (2020) and Silva et al. (2020), nanocellulose is biocompatible and non-toxic. Regarding chemical properties, the material has high crystallinity and high polymerisation (Perera et al., 2022; Silva et al., 2020). Resulting of the decreased particle size, cellulose nanofibres show improved properties with regard to an increased surface-to-volume ratio and improved mechanical properties, such as thermal stability or tensile strength (Siddiqui et al., 2023; Silva et al., 2020). Furthermore, this leads to light weight and low density (ibid.). Using nanocellulose as a coating for food packaging made of paper, the material's gas barrier qualities are improved (Fotie et al., 2020; Garrido-Romero et al., 2022).

Nanocellulose can make the nanomaterial-based films more tortuous, which causes diffusion processes to move more slowly and lowers permeability (Yu et al., 2014). Moreover, nanocellulose and other bio-nanocomposites usually show strong antimicrobial, oxygen-scavenging, and ethylene-absorption activities (Perera et al., 2022). Oxygen-scavenging refers to the process of reducing oxidative damage in a wide range of food constituents, such as oils and fats, to prevent rancidity, discolouration, and loss of taste and nutritional components (Vermeiren et al., 2003). Absorbing ethylene in food items helps extend their shelf life (Ayhan, 2019). Moreover, nanocellulose has a strong ability to inhibit lipid oxidation and high moisture control (Machado et al., 2022; Siddiqui et al., 2023). These characteristics are essential for reducing microbial growth, and improving the quality and safety of food products during storage and transport (ibid.) which assists in extending the food items' shelf-life (Ashfaq et al., 2022) and, consequently, reduces the amount of food going to waste. Besides, these properties reduce the need for fossil fuels when using nanocellulose (Souza et al., 2020).

3.2.2 Weaknesses

When compared to plastics, nanocellulose has some drawbacks. It is not as affordable and accessible (Perera et al., 2022). Compared to conventional coating plastics, the production costs of nanocellulose are higher since it requires more machinery (Fotie et al., 2020). While plastics can be utilised in all food packaging applications, the present bio-nanocomposites are particular to the type of food (Perera et al., 2022). Since nanocellulose is most often obtained from plants in which water is an integral part, the material itself has weak water resistance, making it inappropriate for environments with high humidity (Ahankari et al., 2021; Solhi et al., 2023). It further lacks the ability to be sealed by heat, which is the most frequently used method to form and close plastic packaging (Fotie et al., 2020; Ilhan et al., 2021). Because biodegradable packaging is novel, research must be conducted to develop it into a global standard for food packaging (Perera et al., 2022). Also, there are

not sufficient regulations and legislation regarding the use of bio-nanocomposites in food packaging (ibid.).

3.2.3 Opportunities

Nanocellulose can be obtained from a variety of wastes, including barley waste, tomato peels, garlic straw residues, wood waste and paper waste, corncob residue, exoskeleton of prawns and crabs, and industrial waste cotton (Noor et al., 2020; Silva et al., 2020). Consequently, by being produced of waste, the nanocomposite supports a circular economy and is recognized as more sustainable than conventional plastics (Siddiqui et al., 2023). This makes nanocellulose a great alternative for large-scale applications, such as packaging for beef meat (Machado et al., 2022; Siddiqui et al., 2023). The characteristics of polymers, including their thermal, mechanical, barrier, and surface wettability properties, could all be improved with the addition of nanocellulose to the existing material (Arrieta et al., 2016). As a result, biobased and biodegradable polymers perform better thermomechanically (ibid.).

The combination of nanocellulose with biodegradable additives can strengthen its characteristics or mitigate some of its flaws: By combining nanocellulose with gelatin, starch, gelatin-based biopolymers, or other bioplastics, the above-mentioned lack of water resistance can be improved (Garrido-Romero et al., 2022). Adding essential oils, such as thyme, gives the material an antioxidant activity, which restricts the oxidation of proteins, lipids, DNA, and other molecules by preventing the growth stage in oxidative reaction chains (Atasoy et al., 2019; Machado et al., 2022). With further nanocellulose applications, waste disposal and greenhouse gas balances will be reduced, promoting environmental sustainability (Abdul Khalil et al., 2016). Moreover, research into the use of nanocellulose in food packaging is a hot topic in the sciences right now (Garrido-Romero et al., 2022).

3.2.4 Threats

Although using nanocellulose in food packaging has many benefits, there are some concerns as well. One is that the adoption of nanotechnology in society and the economy is still unclear because of consumer and manufacturing worries about worker safety and health due to potential environmental harm, and human health effects like cancer and gastrointestinal tract inflammation (Souza et al., 2020). Also, given the absence of guidelines and assurances for the effectiveness of its implementation, the package manufacturers have not yet demonstrated a strong interest in the applicability and processability of nanocellulose (Fotie et al., 2020). Another risk that might affect the usage of nanocellulose in food packaging is its potential migration to foods which has not yet been studied (Ashfaq et al., 2022; Azeredo et al., 2017; Souza et al., 2020). Given the novelty of bio-nanocomposites in food packaging, the migration levels and the cytotoxicity effects of nanomaterials are still poorly understood (Perera et al., 2022). Furthermore, advantages of conventional plastics pose a threat to an extended integration of nanocellulose in food packaging. Their longevity, low production price and broad possibilities of application make it difficult for

many alternatives to prevail on a large scale (Fotie et al., 2020; Perera et al., 2022; Sangroniz et al., 2019). Other than conventional plastics, nanocellulose's particularity to a type of food, as mentioned in Perera et al. (2022) limits its applicability in food packaging implementations. Additionally, despite the awareness of problems with plastics, current human behaviour and habits often impede the change to other solutions (Heidbreder et al., 2019). Due to these reasons, nanocellulose encounters difficulties outcompeting conventional plastics and establishing on the markets.

Table 1 summarises this SWOT analysis of nanocellulose in food packaging. Overall, nanocellulose used in food packaging offers numerous strengths, such as being easily recyclable and extractable from a variety of biological sources. It does, however, have some weaknesses, e.g., not being as accessible and affordable as plastics, and not being covered in enough laws and regulations. Opportunities that prevail for nanocellulose include enhancing the properties of polymers and promoting a circular economy. There are also threats, though, such as the potential migration to food and the connected unclear adoption of nanocellulose by consumers and manufacturers.

4. DISCUSSION

In the results, both the life cycle of nanocellulose (from production to disposal) and the SWOT analysis showed positive and negative aspects of nanocellulose in food packaging with regard to circularity and environmental sustainability. Other perspectives such as social aspects and technical details were not in focus of this study which has to be kept in mind when critically discussing potentials of nanocellulose.

4.1 Production, application, and disposal of nanocellulose

The current production processes, applications and disposal options of nanocellulose in food packaging are already promising. According to Ferrer et al. (2017), its potential will probably increase with further research. This is due to the fact that more knowledge about nanocellulose can reduce insecurities with regard to environmental and health issues and improve production and application procedures. Further development of the processes and their combination could simplify production, and also reduce costs and resource use as underpinned by Salimi et al. (2019). Nevertheless, current processes already allow for a more individually designable implementation as producers can choose between three different approaches; mechanical, chemical and biological. Additionally, the diverse possibilities for applying nanocellulose in food packaging, such as coating and the layer-by-layer method, are connected to different characteristics and advantages and thus make its use customisable.

However, concerning the production, use and disposal of nanocellulose in food packaging, from a circularity and environmental sustainability perspective, several restricting aspects have to be taken into account. Firstly, in addition to wisely choosing the extraction process, both an appropriate cellulose source and pre-treatment play a ma-

TABLE 1: SWOT table for the application of nanocellulose in food packaging.

Strengths	Weaknesses
• Biodegradable • Compostable • Recyclable • Can be obtained from a wide range of biological sources • Relatively low price compared to other biodegradable plastic alternatives • Biocompatible • Non-toxic • High crystallinity and polymerisation • High thermal stability and tensile strength • Low weight • Low density • Lowers permeability in applications • Improves gas barrier properties in paper packaging • Antimicrobial • Oxygen-scavenging • Ethylene-absorbing • Inhibits lipid oxidation • High moisture control • Supports high-quality and safe food storage and transport, increasing the items' shelf-life • Reduced dependency on fossil fuels	• Less affordable and accessible than conventional plastics • Higher production costs than conventional plastics • Particular to a type of food • Low water resistance in itself • No thermosealability • Lack in research • No global standards and sufficient regulations/legislation
Opportunities	Threats
• Variety of waste can be used to obtain nanocellulose • More sustainable than conventional plastics • Supports circular economy • Applicable on a larger scale • Combination with other components improves their properties • Biodegradable additives can improve the lack of water resistance • Addition of essential oils gives the material antioxidant activity • Helps reduce waste disposal and greenhouse gas emission • Important current research topic	• Potentially harming the environment and human health causing insecurities for consumers and manufacturers • No guidelines and assurances for manufacturers • Potential migration to foods • Potential cytotoxicity • Advantages of conventional plastics: longevity, low production price, applicability • Human behaviour and habits (social resistance to change)

major role as supported by Ferrer et al. (2017) and Salimi et al. (2019). Regarding use and disposal, for instance, when using nanocellulose as a coating, the substrate material has to show disposal properties similar to nanocellulose to ensure that the coatings' positive characteristics are not overshadowed. In addition, although recycling nanocellulose food packaging is possible, its limitation due to the downgrading of several characteristics, as explored by Shanmugam et al. (2019), should not be forgotten.

4.2 Problems with nanocellulose and potential solutions

The SWOT analysis shows several positive aspects of nanocellulose in food packaging. Nevertheless, the severity of the elaborated weaknesses and threats is partly still uncertain. For example, the potential harm to the environment and human health must be further examined as stated by Li et al. (2015). In particular, there is research and data missing on the migration of toxic compounds to food and its consequences, among others specified by Ashfaq et al. (2022), Machado et al. (2022) and Perera et al. (2022). This threat could be remediated by only using nanomaterials whose risks are studied and known, such as non-toxic and allergy-free components. However, this might limit the number of usable nanomaterials not because they are not suitable but due to the lack of data. Another weakness is nanocellulose's inability to be heat sealed which might make nanocellulose in food packaging incompatible with current packaging processes and calls for a combination with sealable materials as mentioned by Fotie et al. (2020). Moreover, the more expensive and complex production in comparison to conventional plastics as well as difficulties

to change human behaviour might hinder further implementation of nanocellulose in food packaging significantly.

A solution to reduce or eliminate several drawbacks could be the involvement of authorities as supported by Siddiqui et al. (2023) and Souza et al. (2020). In particular, missing guidelines, standards, and assurances for manufacturers could be reduced by working towards new legislation and regulations. Clear guidelines support both the approach of research and the later usage of nanocellulose in food packaging. Although nanocellulose in food packaging is still in research and far from being fully implemented, starting to develop and define a standard would improve the later uniformity of this technology. Additionally, assurances would reduce risks for manufacturers and encourage them to invest in this approach. Furthermore, subsidies on research and production of nanocellulose would reduce their costs and make the technology more attractive.

4.3 Possibilities offered by nanocellulose

Realising those suggestions would strengthen the positive aspects found in the SWOT analysis which comprise a large part of the found data as can be taken from Table 1. An example is the combination of nanocellulose with other components which increases its flexibility in applications and is thus a promising research field as emphasised by both Ashfaq et al. (2022) and Ferrer et al. (2017). The outcome of this research is still open as the variety and number of possible composites are broad. Furthermore, nanocellulose outcompetes conventional plastics in terms of disposal by contributing both to a decrease in waste amounts and waste types. This is due to the reduction of plastic packaging or coating thickness, the possibility of

fully replacing plastic coatings, its good compostability and recyclability and the several applications of composted or recycled nanocellulose food packaging.

Therefore, nanocellulose supports a circular economy and promotes two of the upper levels of the waste hierarchy: waste prevention and waste recycling. The highest level, waste prevention, is achieved by reducing waste amounts and types as mentioned above. Additionally, nanocellulose's capability to decrease microbial growth can extend the freshness and quality of stored food which, in return, may avoid food waste since less food has to be disposed of due to spoilage, as stated by Siddiqui et al. (2023). The third highest waste hierarchy level, waste recycling, is supported both by using a variety of waste sources to produce nanocellulose and by nanocellulose's own beneficial characteristics with regard to its recycling.

4.4 Limitations and uncertainties

As the conducted literature review used peer-reviewed academic publications, the quality and reliability of the results are assumed to be high. However, the study's focus on environmental aspects and possibilities of circularity limited the analysis and discussion, and excluded not only a technological, but also a social perspective. A more socially oriented study could, for example, focus on environmental awareness and a shift in society's mindset towards more environmentally friendly and circular packaging options. In addition, the data availability restricted a further, more detailed study which requires more research in this still-very-new area. A more systematic and broader literature review delivering more SWOT features for future research is desirable. Moreover, SWOT analysis has some limitations. It accords equal weight to all factors and does not offer any solutions or plans of action. Therefore, the SWOT analysis conducted could be supplemented by quantitative or semi-quantitative approaches that include prioritisation of the found aspects to remediate issues by evaluating their severity and significance for the implementation of nanocellulose in food packaging.

5. CONCLUSIONS

Given the fact that plastics are the most common food packaging material, yet cause various harmful environmental effects, finding biodegradable alternatives to them has become one of the main research fields in waste prevention and recycling. Nanocellulose is one that has drawn the attention of researchers. Its characteristics have been studied in the past. However, an extensive analysis of potentials and drawbacks with regards to environmental sustainability and circularity has not been conducted yet. By performing a literature review, it was evaluated how nanocellulose is produced, currently used in food packaging and disposed of. The potentials and drawbacks of nanocellulose as an alternative to conventional plastics were analysed through a SWOT analysis.

Three steps of producing nanocellulose have been identified and described, namely selecting cellulose sources, pre-treatment, and extraction. As manufacturers can select from three alternative approaches - mechanical,

chemical, and biological -, current production procedures allow for an individually designable implementation. The study revealed a variety of applications which were divided into coating for conventional food packaging and multi-layer materials. The different possibilities for using nanocellulose in food packaging were linked to its characteristics and benefits, making the use customisable. The safe disposal of nanocellulose is not much hampered; in fact, it outperforms the disposal of plastics. The main potential of nanocellulose is found in its features, which are overall beneficial for use in food packaging. Also, it supports the top tiers of the waste hierarchy. The novelty of the study field, as well as high expenses, a lack of norms or guidelines, and potential hazards to people and the environment, all contribute to the weaknesses.

Given the novelty of the research field, some threats of nanocellulose are not yet fully understood, most notably the potential migration to food. Further research is required on this topic to evaluate the harm nanocellulose may have in food packaging and whether it is applicable on a larger scale. Overall, as the demand for sustainable packaging materials continues to grow, the potential of nanocellulose in food packaging on a larger scale could rise, contributing to a more sustainable future for all.

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PROPORTIONAL EVALUATION OF WATER HYACINTH AS A RAW MATERIAL FOR BIOGAS, BIOMANURE AND SOIL CONDITIONER GENERATION VIA ANAEROBIC PROCESSES AT LAB AND PILOT SCALE

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ABSTRACT

Water hyacinth is capable of inexplicably concealing an entire water body within two weeks by forming thick mats due to its rapid proliferation intimidating the existence of aquatic organisms. Therefore, this study is aimed to investigate the conversion of WH to biogas, biomanure and soil conditioner (SC) through anaerobic digestion (AD) and composting process. While composting was studied at the lab and pilot scale using leaves and roots, the AD of WH at various food to inoculum (F/I) ratios of 1, 2, and 5 was carried out with leaves, roots, and a mixture of leaves and roots. The study found that leaves can efficiently produce biogas at a 2:1 F/I ratio with 350 mL/g VS reduced methane and 63% CH₄. The qualitative evaluation of biomanure and SC from AD and composting processes revealed that the SC produced from leaves is suitable for agriculture farming (Fenugreek and Coriandrum Sativum) while that of roots for horticulture (Hibiscus and Papaya). About 1200 tons of WH was converted to 120 tons of soil conditioner to control the WH growth in Kapra Lake in Hyderabad city.

1. INTRODUCTION

Water hyacinth (WH), also known as Eichhornia Crassipes, is a promising feedstock for anaerobic digestion due to its high proliferation rate and ability to withstand harsh environmental conditions (Patil et al., 2014; Rezanian et al., 2015). The rapid reproduction of WH forms a floating cover on water surfaces, obstructing light and oxygen, leading to aquatic fauna death and significant impacts on sustainability, economic growth, biodiversity, and human health. The methods of control used to curb the extensive growth of WH are found either to be not sustainable (chemical), not cost-effective (mechanical), or may require a longer time to degrade and stabilize (biological) (Barua and Kalamdhad, 2017; Bhui et al., 2018; Li et al., 2021; Mathew et al., 2015). However, the use of either of these methods is inevitable because the WH growing in water bodies if left unattended would lead to the extinction of aquatic life.

WH can be utilized efficiently for phytoremediation, compost production, enzyme extraction and bioenergy generation via biological pathways (Barua and Kalamdhad, 2017; Ilo et al., 2020). In India, most of the domestic waste-

water is released into the water bodies resulting in the accumulation of micro and macro nutrients along with heavy metals leading to eutrophication. Moreover, heavy metals do not decompose throughout the composting process since they are inert components. Therefore, repeated long-term applications of manures and compost may cause heavy metal components to accumulate in agricultural soil to dangerous levels, which may have an impact on crop yield and health (Rashid et al., 2023). The corresponding detection limits for lead, zinc, cobalt, copper, nickel, and cadmium were 0.4 µg L⁻¹, 0.012, 0.2, 0.17, 0.11, and 0.2 µg L⁻¹ (Smirnova et al., 2021). Heavy metals such as phosphorus and nitrogen in water cause algal overgrowth, causing fish and aquatic species to die due to reduced oxygen levels caused by algae death and decomposition.

However, heavy metal accumulation in roots may hinder biological processes (Sanmuga Priya and Senthamil Selvan, 2017). Biological stabilization of waste material is found to be efficient with substrates having high moisture content. WH is an aquatic weed which has more than 90% of moisture which by itself declares the suitability of the substrate for biological valorization (Islam et al., 2021). Further,

the WH consists of abundant organic matter in the form of volatile organic and chemical oxygen demand which can be efficiently utilized for the generation of bioenergy or its conversion to biofertilizer. Although the WH contains biodegradable organic matter, it contains lignocellulosic material, heavy metals which may substantially hinder the anaerobic digestion process for biogas generation (Koyama et al., 2015; Pelleria and Gidarakos, 2018).

Nevertheless, optimizing the critical operational parameters may facilitate efficient biogas production from WH despite its inherent disadvantages. One of the critical parameters in AD is the feed to inoculum (F/I) ratio, that is defined as the ratio between VS of feed and VS of inoculum used. Determination of ideal F/I ratio is necessary to prevent process imbalance due to depletion of food or stressed organic loading resulting in stocked up intermediate products that hinder the activity of microorganisms (Castro and Agblevor, 2020). A study reported by (Nugraha et al., 2018) illustrates that Food to Microorganism (F/M) ratio of 0.2 resulted in the maximum methane of 164.7 g/g COD, while the substrates such as food waste under different operating conditions results in the generation of broad range of cumulative biogas yields between 300 – 600 mL/g VS (Begum et al., 2016; Kuruti et al., 2017). The low amount of biogas produced by WH (53-58% methane) and 45.18% methane can be ascribed to the presence of lignocellulosic complex, heavy metals and bacteria which impede the microorganisms' ability to produce biogas. The available literature on the AD or composting of WH is demonstrated with the complete plant but the novelty of this study lies in the AD of WH with leaves, roots, and a combination of both (Barua and Kalamdhad, 2017; Unpaprom et al., 2021).

Further, in the recent past, application of organic manure has shown a significant interest and impact on farming in comparison to chemical fertilizers and the rising demand of health-conscious food consumers, preferring food which is cultivated with bio manure application rather than chemical fertilizers (Vidya and Girish, 2014). The digestate or the compost derived from AD and composting can be used as biomanure/biofertilizer. An added benefit of biomanure is that it encourages the growth of nitrogen-fixing bacteria, whereas artificial fertilisers inhibit this growth due to their high acid content. Accelerated Anaerobic Composting (AAC) specified as the anaerobic composting process that yields finished compost in less time with the addition of cow dung and Anaerobic mixed microbial consortia (AMMC) in large piles for adequate treatment of WH. Furthermore, it is interesting to note that WH can sequester large amounts of CO₂ from the environment which creates a positive impact in the reduction of GHGs and stabilizing the WH in anaerobic environment would curb the release of GHGs. WH is rich in organic matter, making it a suitable feedstock for anaerobic digestion and generates methane, which can be used as a renewable energy source for electricity generation or as a fuel. The implementation of biogas plant or a composting unit for the valorization of WH depends on the utilization of product at the source of waste treatment facility. Besides, WH plays a crucial role in absorbing carbon dioxide (CO₂) from the atmosphere and storing carbon within its biomass. This weed acts as

a carbon sink, aiding in the mitigation of climate change by reducing atmospheric CO₂ levels and consequently decreasing global warming. For instance, in Assam, India, the density of WH was 10.37 plants/m² during the pre-monsoon season, which increased to 21.56 plants/m² in the monsoon and post-monsoon seasons. However, it decreased to 25.91 plants/m² in winter (Lahon et al., 2023). Reports on tropical lakes suggest that the rapid growth of water hyacinth results in a substantial intake of carbon dioxide (CO₂) at rates ranging from 3.4 to 5.4 g C-CO₂ m⁻² day⁻¹ (Junior et al., n.d.).

This study is aimed to valorize WH in two ways via anaerobic processes. In the first method, AD of WH has been investigated to study the impact of substrate composition (roots, leaves and mixture of roots and leaves) and F/I ratio on biogas and biomanure generation potential. The performance of AD is studied in terms of biogas yield, methane (%), VS reduction (%), volatile fatty acids and pH profile. In the second approach, the WH is converted to soil conditioners with leaves and roots separately via AAC process at lab as well as pilot scale. As the digestate obtained from AD process can also be utilized as fertilizer, this study has made a comparative quantitative and qualitative analysis of biomanure, and soil conditioner generated via AD and ACC process respectively for leaves, roots and combination of both. The performance of composting process is assessed based on the response of plants growth (*Fenugreek* and *Coriandrum Sativum*). The pilot scale demonstrated for the cleaning of Kapra lake is the first of its kind project in the country where 1200 tons of WH is removed to produce soil conditioner.

2. MATERIALS AND METHODS

2.1 Water hyacinth collection and sample preparation for laboratory scale experiments

About 50 kg of freshwater hyacinth weed was pulled out from Kapra Lake in Hyderabad which was washed to remove mud tangled with WH roots and leaves. The WH was then cut into three parts namely leaves, stalks and roots as shown in Figure 1. The chopped leaves, roots and stalks were coarsely ground into a paste separately with particles of approximately 0.5-1 cm. In addition to both these substrates, a third substrate was prepared with both the pastes mixed in 1:1 ratio. All the three substrates were analysed for total and volatile solids (TS and VS) and stored at 4°C for further experimental studies on anaerobic digestion (AD). The same substrate is used for AAC with a composition of 30% inoculum (AMMC + Cow dung) and 70% of substrate.

2.2 Inoculum

Anaerobic mixed microbial consortia (AMMC) was collected from the inhouse biogas plant facility operated with food and vegetable waste generated in the departmental canteen. The AMMC was used as seed inoculum for AD experiments while a mixture of cow dung and AMMC in the ratio of 1:1 was used for composting experiments. About 50 L of AMMC was collected from an existing biogas plant operated at mesophilic temperature with cooked



FIGURE 1: The whole WH and the chopped leaves, stalks and roots.

and uncooked (vegetable waste and peels) food waste as feedstocks in Hyderabad while the cow dung was procured from the nearby farm. Both the AMMC and cow dung were analysed for TS and VS to design the experiments.

2.3 Characteristics of substrate and inoculum

The characteristics of substrates, AMMC as well as cow dung in terms of pH, TS and VS is as follows. The pH of Leaves (L), Roots (R), L + R in 1:1 ratio, AMMC and Cow dung are 6.8, 8.4, 6.9, 7.0 and 7.1 respectively. The TS (%) of L, R, L + R, AMMC and Cow dung are 6.6, 8.3, 6.9, 3.8 and 11 respectively. Similarly the VS (%) of L, R, L + R, AMMC and Cow dung are 6.4, 6.9, 6.2, 2.7 and 7 respectively. The VS of substrate and inoculum was used to design the experiments.

2.4 Experimental setup and operational strategy

The potential of WH as an alternative raw material for biogas, biomanure and soil conditioner generation was assessed via two anaerobic processes namely, anaerobic digestion (AD)/biomethanation and composting.

2.4.1 Configuration of anaerobic digestion experiments based on food to inoculum (F/I) ratio

A set of nine anaerobic batch reactors of 500 mL with an effective volume less than 380 mL with required arrangements for feed inlet, gas collection and sampling were operated in parallel along with their duplicates for 30 days. The substrate composition in reactors R1, R2 and R3 was Leaves, Roots and mixture of leaves and roots in 1:1 ratio which remained same in all the reactors. The detailed

composition of reactors designed based on the VS content in substrate and AMMC is summarized in Table 1. As can be seen from Table 1, that the reactor R1-L, R2-R and R3-L+R were designed with 30% AMMC and 70% substrate of the effective volume which was found to be operated at an F/I ratio of ~5 while the other reactors were designed to achieve a desired F/I ratio of 1:1 and 2:1 respectively. In addition to these reactors, a set of two blank reactors only with inoculum were set to determine the biogas generation potential of the inoculum. The biogas generated from the inoculum was deducted from the biogas produced from the other reactors to determine the biogas generation potential of the substrate. All the reactors were operated at mesophilic temperature ($37 \pm 2^\circ\text{C}$) placed in a water bath shaker at an rpm of 100. The pH of the reactors was not controlled as the substrate as well as the inoculum was at neutral or slightly alkaline pH. Reactor samples were withdrawn on 10th day to check the variation in performance indicators such as VFA and pH and compared to that of initial and final day samples. There is a limit to the ability of bacteria to feed on substrate, or the substrate has entirely broken down and no gas is being produced. Thus, in order to produce gas, more bacteria or substrate must be supplied. The biogas generated was measured every day and analysed for its composition through GC.

2.4.2 Composting of Water hyacinth

Two anaerobic composting reactors (ACR) of 50 L volume were used to perform composting experiments with leaves and stalks (ACR1) and roots (ACR2) as shown in Figure 2. The effective volume of reactors was 35 L which

TABLE 1: Composition of reactor contents based on definite inoculum volume and F/I ratio.

Reactor label	Substrate (g)	AMMC (mL)	Equivalent gVS _{substrate}	Equivalent gVS _{AMMC}	F/I
R1 - L	245	105	15.7	2.8	5.60
R2 - R	245	105	16.9	2.8	6.03
R3 - L+R	245	105	15.2	2.8	5.42
R1 (1:1)	110	240	7.04	6.5	1.08
R2 (1:1)	110	260	7.59	7.1	1.07
R3 (1:1)	110	240	6.82	6.5	1.05
R1 (2:1)	155	180	9.92	5.0	2.04
R2 (2:1)	160	200	11.04	5.4	2.04
R3 (2:1)	170	190	10.54	5.1	2.05

were fed with 30% inoculum (AMMC + Cow dung) and 70% of substrate. Each reactor was fed with 24.5 kg of respective substrate, 5.25 L of AMMC and 5.25 kg of cow dung. Further, the reactors were supplemented with micro and macronutrients solutions of 10 mL each to ensure better activity of the microorganisms. The reactors were operated at ambient temperature and ensure proper mixing of inoculum and substrate, then tightly sealed with the lid to ensure perfect anaerobic environment. The possibility of biogas generation in these reactors is less due to ambient temperature conditions. This is because the process of AD for biogas generation occurs efficiently either at mesophilic or thermophilic temperature but not psychrophilic. The product after 40 days of composting has been spread and sun dried and the dried material referred to as soil conditioner was analysed for N, P₂O₅ and K₂O as mentioned earlier.

2.5 Analytical methodology

The three substrates (leaves + stalks, roots and mixture of leaves + stalks + roots) along with AMMC and cow dung were initially characterized individually for pH, TS, VS, moisture content (MC) and chemical oxygen demand (COD) as per the standard chemical procedures (American Public Health Association, Andrew D., American Water Works Association., Water Environment Federation., 2005). Further the substrates were also analyzed for carbon (C), nitrogen (N) and sulphur (S) using CHNS analyser (Vario micro cube elemental, Germany), heavy metals, Phosphates (P₂O₅), potash (K₂O) using inductively coupled plasma - optical emission spectrometry (Thermo scientific, model: iCAP6500DUO). The reactor samples from AD and composting experiments were analyzed for pH and VFA on initial day, 10th and 30th day. VFAs were analyzed using High Performance Liquid Chromatography (HPLC) (LC- 10 AT, VP, Shimadzu, Japan) equipped with ultra-violet detector (C18 chromasil column was used under mobile phase conditions using 0.001 N, 90% H₂SO₄ in aceto nitrile). The flow rate and the temperature of the column were maintained at 0.6 mL/min and 30°C respectively. The volume and composition of biogas was analyzed using wet gas flow meter



FIGURE 2: Composting of WH leaves & stalks and roots in 50 L containers.

(ISI 068; INSCIN private ltd) and Gas Chromatography (GC) respectively which was equipped with TCD detector operated at 250°C at a flowrate of 1 L/min and a plot capillary column at 38°C, injector temperature at 200°C (TCD, GC 2010, shimadzu, Japan).

2.6 Evaluation of Kinetic potential parameters of AD of WH

The modified Gompertz model is an empirical non-linear regression model that is used to analyze the lag phase (λ) in days of AD of WH as shown in equation (1) (Begum et al., 2020). All the data obtained from AD experiments were analyzed using IBM SPSS software version 24.0.

$$Y = M * \text{Exp} \{ -\text{Exp} [((R_m * 2.718) / M) * (\lambda - t) + 1] \} \quad (1)$$

where:

Y= Cumulative biogas generation at time't' (L)

λ = Lag phase (day)

M = Highest biogas generation potential (mL)

R_m = Maximum biogas generation rate (mL/day)

3. RESULTS AND DISCUSSION

3.1 Physio-chemical characterization of WH

The extent of biological conversion of any substrate to value-added products is determined based on the presence of quantity of volatile organic matter and the recalcitrant substances. Therefore, the physico-chemical characteristics of WH were analyzed for roots and leaves separately and the average values are presented in Table 2. The BOD/COD, C/N and the V/S ratio of WH leaves illustrate that the leaves and stalk portion are biodegradable and suitable for AD process as the required C/N ratio for AD falls between 20 and 30 (Barua and Kalamdhad, 2019). The pH, C/N and the BOD/COD ratio for roots of WH seem not favourable for AD process which means that roots have to be co-digested with other substrates for better outputs. Further, the concentration of heavy metals like Hg, Pb, As₂O₃, Cr, Cu and Cd are higher in roots that hinders AD compared to leaves. According to World Health Organisation guidelines, the allowable limits of some heavy metal ions in drinking water are as follows: chromium: 5.69 ppm, copper: 11.31 ppm, zinc: 22.8 ppm, etc. Besides, the cumulative generation of biogas and methane volumetric concentration in an AD using olive mill waste as the substrate were 64.6% and 0.426 Nm³/kg VS, respectively. While the concentration of CH₄ was higher, the biogas generation with the addition of nickel dropped to 0.083 Nm³/kg VS as compared to no addition at all. As a result of the addition of copper and chromium, the overall production of biogas dropped by 0.233 Nm³/kgVS (Alrawashdeh, K.A.b.; Gul, E.; Yang, Q.; Yang, H.; Bartocci, P.; Fantozzi, 2020). Therefore, the leaves and stalk portion of WH plant are more amenable for AD or composting than roots. Zinc and Copper were the highest recorded metals in roots at 22 and 12 ppm of dry weight, respectively. In accordance with literature, it was observed that WH can be used as indicator for heavy metal water contamination. Since this plant grows extensively on water bodies, it can be utilized for phytoremediation due to its high metal uptake capacity and later can be converted to biogas com-

TABLE 2: Physico-chemical characteristics of WH leaves and roots.

Parameter	WH leaves	WH roots
C (%)	39.26	38.27
H (%)	5.72	5.46
N (%)	1.5	2.0
S (%)	0.32	1.27
C/N	26	19
TS (%)	6.6	8.3
VS (%)	6.4	6.9
VS/TS	0.96	0.83
COD (mg/L)	56,000	48,000
BOD (mg/L)	32,000	21,200
BOD/COD	0.57	0.43
pH	7.1	8.89
Arsenic (As ₂ O ₃) (mg/kg)	BD*	2.87
Cadmium (Cd) (mg/kg)	BD*	0.19
Chromium (Cr) (mg/kg)	3.69	5.69
Copper (Cu) (mg/kg)	1.64	11.31
Mercury (Hg) (mg/kg)	BD*	0.47
Nickel (Ni) (mg/kg)	0.12	1.54
Lead (Pb) (mg/kg)	0.54	2.94
Zinc (Zn) (mg/kg)	8.93	22.8

BD* Below detection limit

bined heat and power applications and biomanure and soil conditioner for soil nourishment. Therefore, bioconversion of WH via AD and composting process can be regarded as a means of eradication by re-utilization.

3.2 Effect of substrate composition and F/I ratio on the biogas generation profile and yield

The daily biogas generation profile and the cumulative biogas yield obtained from AD of WH with different substrate composition and varied F/I ratio is graphically presented in Figure 3. The gas production of inoculum was measured and subtracted from the gas production of feedstock and the final amount of gas was included in the data. As can be seen from Figure 3 (a) that the daily biogas generation was < 100 mL/day in reactors R2 operated with roots which is relatively lower than R1 and R3 which were operated with leaves and mixture of leaves and roots of WH respectively which could be attributed to the presence of higher concentration of heavy metals like Pb, Cu, Hg, Cr and As₂O₃ in roots. Presence of other metals such as Zn and Ni might have favoured biogas generation but couldn't eliminate the inhibitory effects of other heavy metals that proven to be toxic to micro-organisms (Guo et al., 2019). The critical factors that affect the performance of AD are pH, organic loading rate (OLR), hydraulic retention time (HRT), temperature, solids consistency, food to inoculum and carbon to nitrogen ratio, availability of nutrients, microbial community, presence of inhibitors, etc. If the maximum daily biogas production from reactors R1 is compared, the biogas generation sharply increased from 60-140 mL/day

from Day 2 to 12 at an F/I ratio of 2:1 while it took longer time in R1-L (day 14-17) which was operated with 30% inoculum and 70% substrate i.e., F/I ratio of 5 to produce maximum biogas of 150 mL/day. This indicates that the balance of food and micro-organism number is essential to prevent organic loading shock and facilitate biogas production from even hardly biodegradable substrates. As the biogas generation process is a biological process dependant on the activity of biomass, the maintenance of optimal F/I ratio in the reactor is important to ensure availability of the active biomass to the amount of food, balance in their nutrient availability, maintenance of pH etc., are some of the critical performance indicators resulting in efficient biogas production. In the present study, at an F/I ratio of 2:1, the biogas generation increased from day 2 to 12 indicating the optimal F/I and C/N ratio. In contrast to that, the reactor R1-L with only leaves exhibited lagphase as the reactor did not had adequate nitrogen to maintain optimal C/N ratio. Therefore, the study concludes that the optimal F/I ratio varies depending on the specific characteristics of the feedstock and the microbial community. Generally, a balanced F/I ratio is crucial for efficient biogas production. Too high of a substrate concentration (high F/I ratio) may lead to process inhibition, while too low of a substrate concentration may result in underutilization of the microbial capacity.

Similarly, from Figure 3 (b) it can be observed that the maximum cumulative biogas yield that can be achieved from roots of WH (R2) is between 120 to 160 mL/g VS reduced, while it is between 220 – 352 mL/ g VS reduced from leaves (R1) and 189 to 301 mL/ g VS reduced from mixture of leaves and roots (R3). The maintenance of F/I ratio played a role in biogas generation which is evident from the maximum yield obtained at F/I ratio of 2:1 than 1:1 or 5:1. A study reported by (Raposo et al., 2009) demonstrated the increase in F/I ratio may lead to the accumulation of VFA during the process. Accumulation of VFA in higher F/I ratio reveals the kinetic uncoupling between acid producing and acid consuming microorganisms. It was reported that the F/I ratio of 1:2 was found to be ideal for substrates like food waste to prevent accumulation of acids, but in the present study, the higher F/I ratio of 5 is favoured but at 2, the biogas generation was better compared to 1 and 5 (Begum et al., 2021). The maintenance of F/I ratio and proportional mixing of roots and leaves have favoured the process because the maximum biogas generation was observed in reactors R1 and R3. This indicates that the bi-methanation of roots alone is not a feasible option but could be made suitable for biogas generation when the carbon rich stalk and leaves are co-digested.

3.3 Performance evaluation of anaerobic digestion of water hyacinth: Impact of F/I ratio and substrate composition on methane yield, methane concentration and volatile solids reduction

The biogas generation potential from WH investigated at different F/I ratios and substrate composition is illustrated in terms of VS reduction (%), methane yield and concentration as shown in Figure 4. As can be seen from Fig-

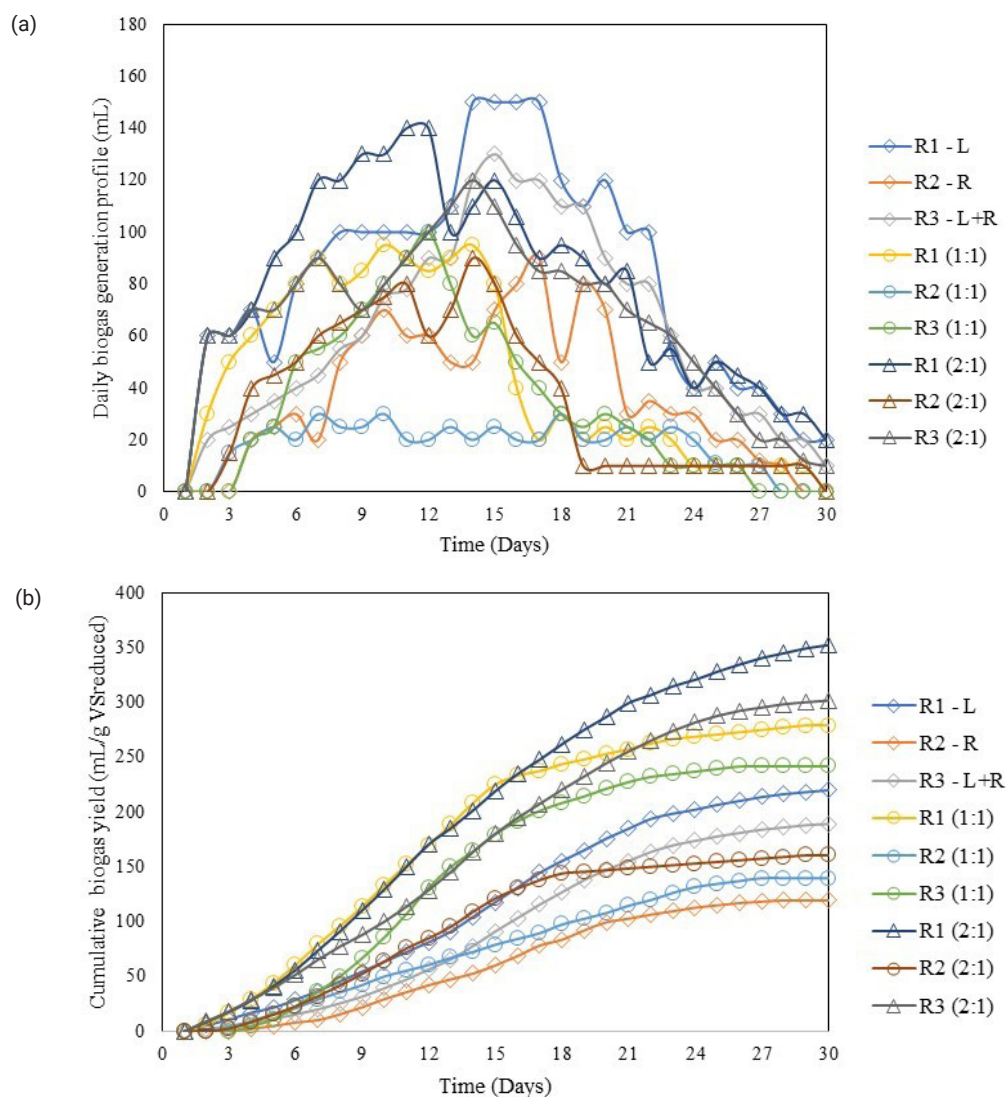


FIGURE 3: Influence of substrate composition and F/I ratio on the biogas generation (a) Daily biogas generation profile, (b) Cumulative biogas yield.

ure 4 that the maximum biogas yield that can be achieved is 350 mL/g VS_{reduced} from leaves of WH at an F/I ratio of 2:1 followed by a combination of roots and leaves i.e., 300 mL/g VS_{reduced} in R3 with a corresponding CH₄ concentration of 63% and 64% and VS reduction of 69% and 63% respectively. According to a study, the methane concentration is 60% for 40 days despite the macrophytes *Brachiaria*, *Oxycariumcubense*, *Eichhornia crassipes*, *Pistiastratiotes*, and *Salvinia* being processed at 120°C, with *Brachiaria* having the greatest lignin content (Paulo et al., 2021). A study reported by (Barua and Kalamdhad, 2017) stated that highest biogas yield that can be achieved from untreated WH at an F/I ratio of 2 is 250±14 mL/g VS (57% CH₄) on the 32nd day whereas in the present study, the maximum biogas yield of 350 mL/g VS with 63% CH₄ was obtained at the same F/I ratio of 2:1 which is 40% higher, however the study is in agreement with the optimized F/I ratio of 2:1. Different studies yielded different amount of biogas and methane in certain period of time with varying F/I ratios. In a study reported by Simbayi, the biogas yield

was 185 mL/g Vs with an F/I ratio of 2:1, generating 65% methane in 45 days (Simbayi et al., 2023). At the same F/I ratio, Wembe reported biogas yield of 175 mL/g Vs with 51% methane in 42 days (Wembe et al., 2023). At an F/I ratio of 1:1, three studies by Ali, Bote and Pachaiyappan yielded 226, 170 and 200-300 mL/g Vs of biogas with 50, 45 and 24% methane during 35, 32 and 30 days residence time respectively (Ali and Sun, 2019), (Bote et al., 2020) and (Pachaiyappan et al., 2014). It is interesting to note that the present study demonstrated superior biogas yields amongst all the reported literature at an F/I ratio of 2:1 with an yield of 350 mL/g VS_{reduced}. The methane content in biogas is also notable with 63% CH₄.

From roots of WH, the maximum biogas yield that can be obtained is 160 mL/g VS_{reduced} at an F/I ratio of 2:1 at a corresponding VS reduction and CH₄ % of 60% and 62% respectively. Another interesting observation that can be made from Figure 4 is that concentration of CH₄ in biogas can be achieved above 60% if the F/I ratio is maintained at 1:1 or 2:1 while it drops down to 56% if the F/I ratio is

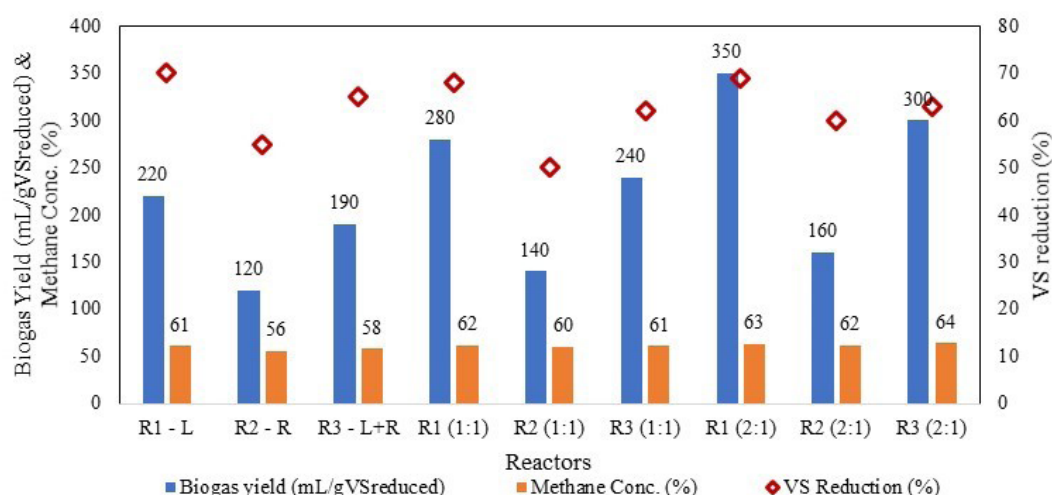


FIGURE 4: Performance evaluation of AD of WH at different F/I ratios and substrate composition in terms of VS reduction (%), methane yield and concentration.

increased to 5 which is evident from the numerical shown in R2-R and R3-L+R. In summary, the results presented in Figure 4 illustrate the AD of WH results in minimum yield of 220 mL/g VSreduced and maximum reaching up to 350 mL/g VSreduced upon maintaining suitable F/I ratio and substrate composition. The results obtained in this study agree with the study carried out by O'Sullivan et al., (O'Sullivan et al., 2010) who has demonstrated the biomethane potential of WH to be 292 ± 43 g/g VS respectively. The substrates such as food waste under different operating conditions results in the generation of broad range of cumulative biogas yields between 300-600 mL/g VS.

3.4 Effect of major performance indicators on the biogas generation potential of water hyacinth

The concentration of VFA in the reactors and its impact on pH was regarded as the major performance indicators in AD process. To assess these performance indicators,

samples were withdrawn on 10th day and analysed for VFA and pH. The VFA, pH and biogas profile on 0th, 10th and 30th day along with % increase in VFA and biogas is presented in Table 3. It can be interpreted from Table 3, that the concentration of VFA increased substantially from 0 to 10th day in all the reactors with % increase in VFA ranging between 80 to 208%, which is indicative of dominance of hydrolysis in the reactors by the conversion of complex organic material to bio accessible organic acids than the methanogenesis, because the biogas generation from the reactors was less than 50% from day 0 to 10 while it increased substantially from day 11 to 30 (i.e., 52 to 208%), because the consumption of acids had taken place during that period of operation except for reactor R1(1:1) (Table 3) (Ganesh et al., 2005). The reactors R1, R2 and R3 operated with L, R and L+R and reactors at an F/I ratio of 2:1 showed relatively higher increase in VFAs compared to reactors at F/I ratio of 1:1 which could be attributed to the abundant availability

TABLE 3: Effect of performance indicators such as VFA and pH profile on biogas generation potential of WH.

Reactor	VFA (mg/L)					pH			Biogas generation, (mL/g VS)			
	0th Day	10th Day	30th Day	% Increase in VFA	% Utilization of VFA	0th Day	10th Day	30th Day	0th Day	10th Day	30th Day	% Increase in cumulative biogas
R1 - L	1.2	3.7	1.8	208	51	6.9	6.4	7.2	0	710	1754	147
R2- R	0.7	1.6	1.1	129	31	7.1	6.8	8.1	0	275	847	208
R3- L+R	0.8	2.4	1.2	200	50	7	6.6	7.6	0	385	1468	281
R1 (1:1)	1.1	2.9	1.4	164	52	6.8	6.6	7.1	0	640	700	9.4
R2 (1:1)	0.9	1.7	1.1	89	35	6.9	6.5	7.8	0	190	341	79
R3 (1:1)	1	1.8	1.2	80	33	6.9	6.7	7.5	0	360	655	82
R1 (2:1)	0.9	2.3	0.6	156	74	7	6.5	7.3	0	880	1516	72
R2 (2:1)	0.8	1.8	1.1	125	39	7.1	6.7	7.6	0	420	640	52
R3 (2:1)	0.8	2.1	0.7	163	67	7	6.6	7.2	0	660	1332	102

0th, 10th, and 30th day VFA is measured in 'mg/L'

0th, 10th, and 30th day Biogas generation is measured in 'L'

of food to micro-organisms for conversion while opposite is the case in reactors at an F/I ratio of 1:1. The increase in VFAs from day 0 to 10 resulted in the reduction of pH considerably to less than 6.8; therefore, the anaerobic chamber is purged with an inert gas, generally nitrogen, to remove most oxygen. However, the consumption of VFAs began after day 10 which in turn improved the biogas production due to the predominant activity of acetoclastic and hydrogenotrophic methanogens during acidogenic and methanogenic process of AD. This observation agrees with the observation made by (Barua and Kalamdhad, 2019) who had demonstrated the AD of pre-treated, untreated, and co-digested WH. These observations indicate that, to anaerobically digest the WH, a longer residence is needed which should be more than 30 days or a pre-treatment step is required to improve the hydrolysis rate and % increase in VFAs within 5 days of operation. On 30th day, the pH levels in the reactors varied from 7.2 to 8.1, indicating that the reactors were operating in anaerobic conditions with low levels of methane production.

3.5 Kinetic potential evaluation of biogas generation from water hyacinth

The kinetic parameters of AD of WH were evaluated using modified Gompertz model equation is presented in Table 4. It can be observed from Table 4 that the substrate composition and F/I ratio had its effect on the lag phase of the process. This is due to certain cells that exhibit growth and metabolic lag, particularly those with more stringent dietary needs. This time frame might align with the duration required for the concentration on essential intermediaries to build up (Bertranda, 2019). In particular, the lag phase in the reactors operated with roots alone and mixture of roots and leaves of WH showed longer lag phases ranging between 3.5 and 6.6 days while the reactors operated with leaves alone showed shorter lag phases ranging between 3.5 to 4.7 days. Furthermore, the rate of biogas generation from the reactors R2-R, R2 (1:1) and R3 (2:1) was 73, 27 and 83 mL/day respectively which is much lower compared to that of R1-L, R1 (1:1) and R1 (2:1).

The experimental data was found to be a good fit to the modified Gompertz model equation which can be ascertained from the R^2 values as shown in Table 4. The experimental and predicted biogas generation obtained from WH is plotted and shown in Figure 5. The difference in the experimental and predicted biogas is not substantial which again affirms the good fit of the experimental data to the Gompertz model equation.

3.6 Quantitative and qualitative evaluation of nutrients in biomanure and soil conditioner generated from water hyacinth in lab scale experiments

The sun dried anaerobic digestate i.e., biomanure and soil conditioner generated at the end of the AD and composting experiments respectively were further analysed for macronutrients, namely nitrogen (N), phosphorous (P_2O_5), and potash (K_2O). The quantity of biomanure and soil conditioner produced along with their nutrient concentration is tabulated and shown in Table 5. The quantity of bioma-

TABLE 4: Kinetic potential parameters evaluated using modified Gompertz model.

Reactor Label	M (mL)	Rm (mL/day)	λ (Days)	R^2
R1-L	2464	133	4.7	0.998
R2-R	1122	73	6.6	0.998
R3-L+R	1853	112	6.7	0.998
R1 (1:1)	1340	99	3.3	0.999
R2 (1:1)	531	27	3.6	0.996
R3 (1:1)	1015	87	5.6	1.0
R1 (2:1)	2396	133	3.5	1.0
R1 (2:1)	1060	83	4.5	0.998
R1 (2:1)	1992	105	3.5	0.999

nure produced after AD and composting was estimated by mass balance in terms of TS, VS and fixed solids (FS) as shown in Table 7. The nutrient values obtained from bioma-

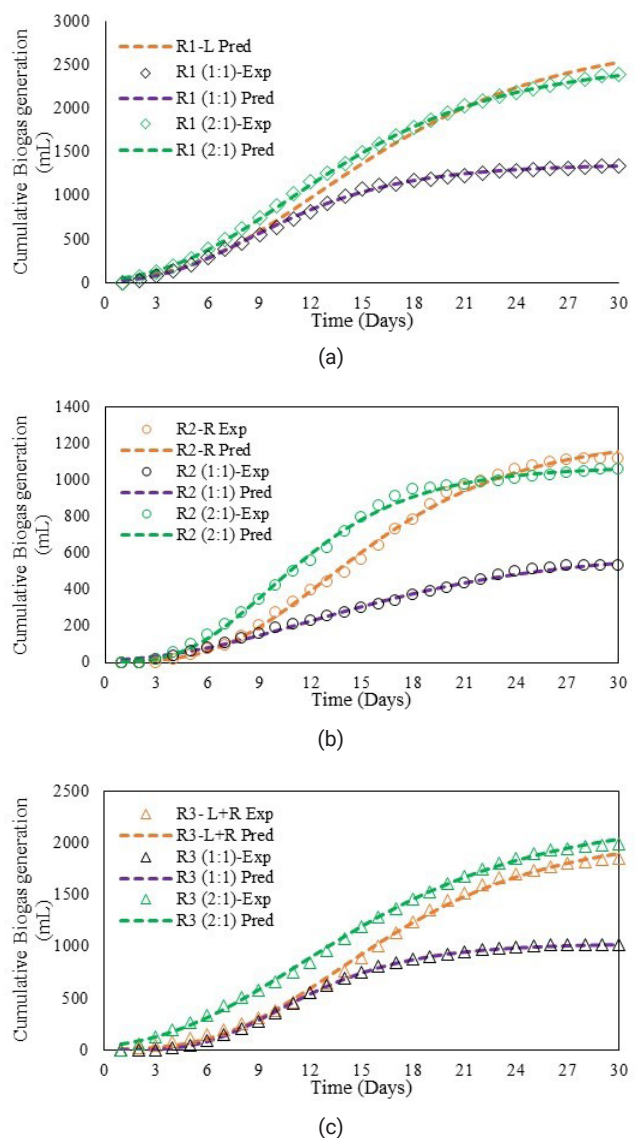


FIGURE 5: The experimental and predicted cumulative biogas generation profiles calculated at different F/I ratio using modified Gompertz model (a) R1; (b) R2 and (c) R3.

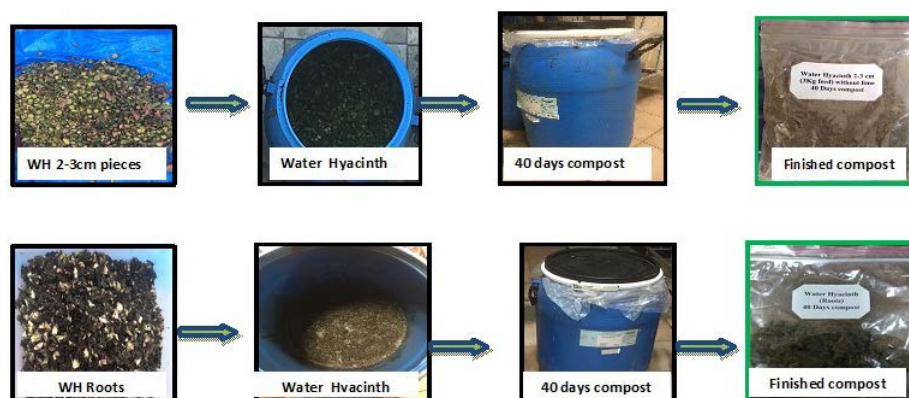


FIGURE 6: The whole WH and the chopped leaves, stalks and roots.

nure, and soil conditioner were compared with the standards given by the FCO (Fertilizer Control Order) for organic manure (FCO, 1985). The end product obtained from roots and leaves of WH is shown in Figure 6. From Table 5, a substantial difference in the quantity of biomanure and soil conditioner generation is observed i.e., 0.14 and 0.13 kg of soil conditioner is produced from 1 kg of substrate via composting process while in AD it was found to be 0.44, 0.62 and 0.53 g for leaves, roots and leaves + roots per g of total solids. For leaves and roots, WH determined that a C/N ratio of 25-32 was suitable to reduce or prevent ammonia inhibition at thermophilic AD (Krakat et al., 2017).

Further, the nutrient value in terms of N, P_2O_5 and K_2O for organic manure as per the standards of FCO is 0.5 each while in our study, the nutrients in leaves, roots and mixture of leaves and roots varied greatly in both AD and composting processes. The N concentration was relatively higher manure and conditioner produced from roots as feed either directly or in mixture with leaves. This could be due to natural assimilation of N from soil by roots compared to plant leaves and stem. Nevertheless, the low N in anaerobic digestate compared to anaerobic compost could be fact that, most of the N content would have been consumed by the methanogens for the generation of biogas as they require an optimum C/N ratio between 25 and 30. In the same way, P_2O_5 and K_2O concentrations found to be low in biomanure generated from both the reactors operated with leaves as feed than other two reactors. The results suggested that it is feasible to produce good quality fertilizer with desired macro nutrients from WH plant waste that is superior to the standards prescribed FCO with less cost.

3.7 Demonstration of accelerated anaerobic composting of WH to clean the KapraLake in Hyderabad city: A pilot scale societal project

The experimental results from the bench scale experiments of WH indicated that composting could be one of the best options to be implemented at pilot or full scale to create wealth out of waste weed. Composting the leaves and roots of WH is done separately because the concentration of heavy metals is high in roots with more absorption capacity. One of the lakes in Hyderabad city called Kapra lake or Oora Cheruvu which spreads around 1254 m near Sainikpuri in the north-east part of Greater Hyderabad has become hazardous to the aquatic fauna ("Kapra Lake," n.d.). Therefore, in association with the local policy makers, an initiative has been taken to install the pilot plant at Kapra lake, as it is filled with WH. The composting process proven successful in the laboratory experiments and has been implemented at the pilot facility where, 1200 tons of WH is pulled out using JCB and boats followed by washing and chopping into roots, stalks and leaves separately as shown in Figure 7. The leaves and roots were chopped into small pieces using a chaffer machine. About 30 compost pits have been constructed for composting, 15 each for roots and leaves of WH and about 4 tons of chopped WH was fed into each pit. The compost pits were initially added with equal quantity of cattle dung and AMMC developed in the laboratory to maintain the methanogenic population and subsequently the WH was thoroughly mixed with the inoculum in each pit maintaining appropriate moisture. Daily samples were collected through sampling ports were made to ensure

TABLE 5: Comparison of quantity and quality of biomanure and soil conditioner to that of organic manure by FCO.

Parameter/ Label	Biomanure (kg/kg substrate)			Soil Conditioner (kg/kg substrate)		FCO standard (min)	Standard deviation (s)
	R1-L	R2-R	R3-L+R	ACR1	ACR2	Organic manure	
Kg product/ kg substrate	0.44	0.62	0.53	0.14	0.13	-	0.225
N (%)	0.32	0.85	0.64	0.46	10.09	0.5	4.263
P_2O_5 (%)	0.16	0.42	0.38	0.48	0.46	0.5	0.151
K_2O (%)	0.18	0.34	0.36	0.5	0.57	0.5	0.151

TABLE 6: Detailed compositional analysis of soil conditioner produced from leaves, roots and organic manure by FCO.

Parameters	Water hyacinth roots	Water hyacinth leaves	According to FCO, Schedule-iv, specification of organic matter Standard values - Organic manure	Standard deviation (S)
MC by weight (%)	32.33	12.18	25	14.248
Colour	Brown	Green	-	-
Odor	Ammonia smell	Wet grass smell	-	-
Bulk Density (g/cm ³)	0.21	0.31	<1.0	0.070
Total Organic Carbon, wt. (%), minimum	16.23	0.83	14	10.889
Total Nitrogen (as N), wt. (%), minimum	10.09	0.46	0.5	6.809
Total Phosphates (as P ₂ O ₅), wt. (%), minimum	0.46	0.48	0.5	0.014
Total Potash (as K ₂ O), wt. (%), minimum	0.57	0.50	0.5	0.049
C:N ratio	19:1	39:1	<20	-
pH	8.72	9.10	6.5-7.5	0.268
Conductivity (as dsm-1), Not more than	2.16	1.21	Not more than-4.0	0.671
Arsenic (as As ₂ O ₃) (mg/kg)	2.87	Below detection	Maximum- 10	-
Cadmium (as Cd) (mg/kg)	0.19	Below detection	Maximum-5	-
Chromium (as Cr) (mg/kg)	5.69	3.69	Maximum-50	1.414
Copper (as Cu) (mg/kg)	11.31	1.64	Maximum-300	6.837
Mercury (as Hg) (mg/kg)	0.47	Below detection	Maximum-0.15	-
Nickel (as Ni) (mg/kg)	1.54	0.12	Maximum-50	1.004
Lead (as Pb) (mg/kg)	2.94	0.59	-	1.661
Zinc (as Zn) (mg/kg)	22.80	8.93	Maximum 1000	9.807

TABLE 7: Mass balance to estimate the generation of biomanure from AD of WH.

Anaerobic digestion (AD) of WH											
	TS _{Substrate} (g)	TS _{AMMC} (g)	VS _{Substrate} (g)	VS _{AMMC} (g)	TS _{Total} (g)	VS _{Total} (g)	FS (g)	VS reduction (g)	VS remaining (g)	VS+FS (g)	g manure/g of TS or kg/kg of TS
R1- L	16	4	16	3	20	19	1	11	8	9	0.4
R2- R	20	4	17	3	24	20	4	9	11	15	0.6
R3-L+R	17	4	15	3	21	18	3	10	8	11	0.5
Accelerated Anaerobic Composting (AAC) of WH											
	TS _{Substrate} (kg)	TS _{CD} (kg)	TS _{AMMC} (kg)	VS _{Substrate} (kg)	VS _{CD} (kg)	VS _{AMMC} (kg)	FS (kg)	VS increment (kg)	VS remaining (kg)	VS+FS (kg)	kg soil conditioner/kg of substrate
ACR1	1.6	0.6	0.2	1.6	2	0.1	1	0.1	3.7	5	0.1
ACR2	2.1	0.6	0.2	1.7	2	0.1	1	0.2	3.9	5	0.1

that the proper environmental and operational conditions were being maintained.

Temperature, pH, alkalinity and toxicity, mixing, nutrient requirements, sludge retention time (SRT), and microbial capacity are some of the factors that influence the AAC at different stages. The final compost after 40 days was removed, and sun dried to get the solid compost. The final product obtained from roots and leaves was evaluated for detailed compositional analysis as shown in Table 6. Although the nutrients N P and K value of soil conditioner

agreed with the organic manure by FCO but the concentration of heavy metals in roots was relatively higher which could indirectly impact the human health. Therefore, the application of soil conditioner produced from leaves was found to be suitable for agriculture field application while that of roots was found suitable for horticulture. In light of this, both the soil conditioners from leaves and roots were tested in real field for the growth of different plants and differences were observed and explained in subsequent section. The dried product named as "Soil Conditioner" was



FIGURE 7: Composting process implemented for the removal and treatment of WH from Kapra Lake, Hyderabad, India.

packed and sealed for distributing into the outlets for sale. As anaerobic inoculum and cow dung were added which was kept under anaerobic conditions, biogas including methane will be produced. The biogas thus produced was in turn used as carbon source of energy for the compost microbes in the production of microbial cells (Hamer et al., 1967). The product obtained was stable and the odors were also resistible with wet grass smell in its vicinity.

3.8 Field application of biomanure and soil conditioner: Response of Fenugreek and Coriandrum sativum plants

The soil conditioner produced from leaves was tested in real field for the growth of Fenugreek and Coriandrum Sativum plants and the periodic growth of the plants from seeding to complete plant i.e., 0 to 30 days is shown in Figure 8 and 9, while the soil conditioner from roots was applied for horticulture and the impact of it on the growth and freshness of plants and flowers is shown in Figure 10 respectively. To compare the growth of Fenugreek and Coriandrum upon application of soil conditioner from WH leaves, commercially available general compost was also administered and the control plants were grown without the application of any fertilizer for proportional evaluation. The response of the plant growth was measured on every 5th day from seeding and the height of the plants was measured in 'cm', as plant height is the key parameter to determine the impact of any fertilizer. One of the interesting observations that can be made from Figure 8 and 9 is that the germination of Coriandrum seeds was delayed by 5 days compared to Fenugreek plants which grew about 2

cm by the end of 5th day in all the cases. Another intriguing and common observation from Figure 8 and 9 is that the height of Fenugreek plants is relatively higher than Coriandrum and the plant growth with soil conditioner and general compost administered plants was relatively higher than control which illustrates that the application of soil conditioner and compost has promoted the growth of plants rapidly. Therefore, the soil conditioner produced from leaves of WH was found suitable for application as a soil conditioner in the agriculture fields.

Similarly, the soil conditioner developed from roots of WH was administered to Hibiscus rosa-sinensis and Carica papaya plants to check the growth of the plants. As can be seen from Figure 10 that the soil conditioner administered hibiscus flower and papaya plant was bright and fresh compared to control. This is indicative of positive impact of soil conditioner on the growth horticulture plants.

4. CONCLUSIONS

This study investigated the feasible ways to valorize WH by two methods namely, anaerobic digestion for the generation of biogas and biomanure and secondly, accelerated anaerobic composting for the conversion of WH to soil conditioner at bench and pilot scale. The highest biogas yield of 350 mL/g VS reduced with 63% CH₄ can be achieved from leaves of WH at an F/I ratio of 2:1 followed by roots in combination with leaves. While no substantial biogas generation was observed from roots which could be attributed to high concentration of heavy metals. Furthermore, the results illustrate the positive impact on agricul-

Fenugreek (Methi)



FIGURE 8: Comparison of Fenugreek plant growth estimation from 0 to 30th day upon application of general compost and soil conditioner over the control.

Coriandrum Sativum

**Seeding
(0-Day)**



General compost (GC)

Soil Conditioner (WH)

Control (C)

5th Day



No growth (NG)



No growth (NG)



No growth (NG)

10th Day



2.0-2.3 cm



2.1-2.5 cm



2.1-2.5 cm

15th Day



5.0-5.6 cm



5.8-6.9 cm



5.2-5.8 cm

20th Day



8.6-9.4 cm



11.6-12.3 cm



8.0-8.6 cm

25th Day



13.1-13.9 cm



16.5-17.2 cm



12.2-12.9 cm

30th Day



18.1-18.9 cm



20.6-21.7 cm



17.0-18.7 cm

FIGURE 9: Comparison of *Coriandrum Sativum* plant growth estimation from 0 to 30th day upon application of general compost and soil conditioner over the control.



FIGURE 10: Application of soil conditioner produced from roots of WH on *Hibiscus rosa-sinensis* and *Carica papaya* plants.

ture farming (*Fenugreek* and *Coriandrum Sativum*) using soil conditioner produced from leaves of WH while that of roots was suitable for horticulture (*Hibiscus rosa-sinensis* and *Carica papaya* plants). Heavy metal presence in roots inhibits AD whereas whole plant material used for composting towards biofertilizer/ soil conditioner production made composting of WH more viable than AD. The pilot scale demonstration was found to be success with 1200 tons of WH removed from Kapra lake and converted to 120 tons of soil conditioner.

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OCCUPATIONAL EXPOSURE TO AMMONIA DURING INDUSTRIAL-SCALE ANAEROBIC DIGESTION COMPOSTING OF HOUSEHOLD WASTE

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ABSTRACT

Biodegradable waste material can be exploited through anaerobic digestion, an oxygen-free multi-stage process often completed by a composting step, which harnesses the power of microorganisms to break down organic matter. This waste-treatment sector is growing rapidly in France as it allows the production of valuable digestates and biogas (biomethane). One of the main occupational health concerns in this sector is exposure to ammonia. The present work focused on ammonia emitted over the course of several stages in the anaerobic digestion and composting process, and assessed employee exposure to this pollutant. Exposure was measured for distinct worker activities and work areas, as well as during the various steps of the process. The results revealed non-negligible ammonia exposure for workers operating in the dehydration (until 55 ppm_v in the workplace and 17 ppm_v on the worker) and composting (until 45 ppm_v in the workplace and 17 ppm_v on the loader driver) areas on several sites. A classical monitoring approach was supplemented by the identification of explanatory parameters contributing to exposure: material emissivity, amounts stored or handled, and ventilation systems in use. Based on these elements, recommendations were made on prevention choices, including reducing emissions at source namely by process improvement and management of stored quantities, improving existing preventive systems such as capture at source or general ventilation, and employee education to raise awareness of waste management risks and improve practices.

1. INTRODUCTION

Anaerobic digestion is considered to be one of the most viable ways to exploit the organic fraction of solid waste (sewage sludge, animal waste, and industrial wastewater), it is therefore the focus of continuous study to improve its effectiveness (Khalid et al., 2011; Lin et al., 2018; RIS International Ltd., 2005). This natural biological process harnesses the capacity of microorganisms to decompose organic material in oxygen-free conditions. At industrial-scale, organic waste digestion takes place in a reactor, or enclosed chamber, where critical environmental conditions such as moisture content, temperature, and pH can be controlled to maximize microorganism proliferation, gas generation, and waste decomposition rates.

Industrial facilities use several reactor types that differ by their structural (infinitely mixed digester, reactors with stationary or moving support, etc.) and operating param-

eters (dry matter content in the reactor, operating temperature, etc.). In the field of domestic waste-treatment, due to the composition of the waste, dry digestion is often used. The technologies available for this process can be classified based on three main modes of operation: vertical reactors with digestate or biogas recirculation, horizontal reactors, and batch reactors (see supplementary Figure S1, online edition) (Damien, 2016; Dirrenberger, 2020; Zeshan, 2012).

To assess occupational exposure for workers in anaerobic digestion plants, it is therefore important to consider the whole waste-treatment line and the associated tasks. Indeed, upstream and downstream of the anaerobic digestion technology deployed, organic waste management requires significant pre-treatment and post-treatment. Thus, waste may be sorted at source to allow separate collection of biowaste, in some situations mixed waste may be collected, which would require waste sorting systems to

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be implemented. After fermentation, digestates can be exploited by direct application as fertilizer or by composting. A combination of anaerobic digestion and composting is often used to treat domestic waste. The steps involved in the process implemented in anaerobic digestion composting plants are presented in the Supplementary Material (see supplementary Figure S2, online edition).

Ammonium cations (NH_4^+) are produced from waste and organic effluent following biodegradation of organic matter as part of the ammonification stage (Moletta, 2015). NH_4^+ is subsequently transformed into ammonia (NH_3), volatilization of which is a physicochemical phenomenon boosted by basic pH, high temperatures, and air velocity at the solid-air interface (Beck-Friis et al., 2003; de Guardia et al., 2010a, 2010b; de Guardia et al., 2008; Fukumoto et al., 2003; Ni, 1999). Compared to the initial waste, anaerobic digestates have a significantly higher relative humidity, lower biodegradable matter content, and higher concentrations of ammoniacal nitrogen (de Guardia et al., 2010a). Consequently, compared to fresh waste, digestates have a reduced self-heating capacity – diminishing the sanitizing power of the composting process – and emit higher levels of ammonia due to their high ammoniacal nitrogen content. If the process is completed by a composting step, ammonia emissions continue to occur as the temperature increases and during windrow turning (Peigné & Girardin, 2001). In anaerobic digestion composting plants, the confined nature of treatment buildings could lead to the accumulation of gases in the workplace atmosphere (ADEME et al., 2019; Lavoie & Marchand, 1997).

High ammonia concentrations can cause irritation to the eyes and respiratory system (INRS, 2021) as well as presenting an odour issue. Ammonia can cause extensive bronchoconstriction (likened to an asthma attack), which can become permanent through the development of bronchiectasis. Chronic exposure to low levels of ammonia can also lead to sinusitis, irritation of the mucosa and conjunctiva, chronic bronchitis, or asthma-like symptoms (Sundblad et al., 2004). Consequently, in workplaces where ammonia is produced, effective collective preventive solutions must be implemented (adequate ventilation, appropriate use of loaders, filtering boxes, etc.) with or without individual solutions (respiratory protective equipment or RPE) (ADEME et al., 2019). Exposure levels can be compared to Occupational Exposure Limits or OELs (Table S1, as Supplementary Material) (INRS, 2021, 2022a, 2022b).

In composting plants, occupational exposure to biological and chemical pollutants is well documented (Lavoie & Marchand, 1997; Nadal et al., 2009; Park et al., 2020; Poirot et al., 2010; Robertson et al., 2019). These reports highlight major pollutants such as bioaerosols, dusts, endotoxins, VOCs and ammonia in particular. In anaerobic digestion plants – where waste treatment is often completed by a composting step – workers are potentially exposed to similar pollutants, including a number of biological (culturable microorganisms, endotoxins, etc.) and chemical pollutants (e.g., methane, nitrous oxide, hydrogen sulfide, VOCs, carbon oxides). In most cases, the major chemical contaminant is ammonia (NH_3), which is mainly produced downstream of the anaerobic digestion step. Ammonia has

an inhibitory effect in digestates, and consequently most studies have focused on improving the efficiency of the anaerobic digestion processes (Chen et al., 2016; Krakat et al., 2017; Rocamora et al., 2023). Some papers deal with ammonia emissions from matter (Bell et al., 2016; Lemes et al., 2023; Preble et al., 2020) but only a few recent papers assess ambient concentrations and worker exposure to ammonia in this sector (Bakhiyi et al., 2018; Pöther et al., 2021).

In this context, the purposes of this study were (i) to characterize the ammonia levels in distinct areas and in loaders, depending on the process considered and any pre-treatment and post-treatment methods; (ii) to assess worker exposure for the various activities and tasks performed based on individual ammonia samples in breathed air; (iii) to characterize the explanatory variables – such as the inventory of materials present/manipulated, their overall emissivity and surface emissivity – as an additional exploratory work in addition to highlighting occupational exposure to ammonia. From this characterization, it was possible to prioritize the search for preventive solutions to reduce emissions at source (process improvement, management of stored volumes, etc.) and/or to improve preventive systems (confinement, local exhaust ventilation, etc.).

Direct reading instruments were used in addition to conventional sampling to objectify worker exposure to ammonia; these measurements are the subject of a companion article (Battais et al., 2022).

2. MATERIAL AND METHODS

2.1 Characteristics of anaerobic digestion composting plants

Measurement campaigns were carried out in six anaerobic digestion composting plants located in France (Battais et al., 2022). Plants were selected based on the variety of the processes encountered and the operation methods applied, representing all the types of dry anaerobic digestion processes currently used in France to treat domestic waste (see supplementary Figure S1, online edition; Dranco®, OWS, Belgium; Valorga®, Urbaser, Spain; Linde-BRV®, Linde-KCA-Dresden GmbH, Germany; Kompogas®, Vinci Environnement, Switzerland; BAL-HYBRID®, BAL Biogas, France) (Damien, 2016; Dirrenberger, 2020). Table 1 summarizes the characteristics of the plants studied, and the climate conditions prevailing during the measurement campaigns, depending on the period when measurements were taken and on the geographical location of the plants. Campaigns were held in plants not located in tourist areas, or in the off-season, so that tonnages handled were quite usual. Working conditions were typical (no major maintenance operations), including any short technical stops or failures which are usual in this kind of plant. Campaigns D1 and D2 were performed in the same site (D) on two dates, hence temperature and hygrometry were quite different (Battais et al., 2022).

Due to the diversity of the industrial sites monitored, data exploitation was simplified by grouping the activities into five major zones:

TABLE 1: Summary of the main characteristics of the anaerobic digestion composting plants investigated and climatic conditions measured during the measurement campaigns.

Plant	Type of waste (tonnage)	AD ⁵ process	Sampling period	Mean T°C*	Mean HR %*
A	70,000 t MSW ¹ 30,000 t varied BW ²	Valorga®	October 2017	18°C	74%
B	30,000 t MSW ¹ 6,000 t varied BW ²	BAL-HYBRID®	March 2017	16°C	75%
C	70,000 t MSW ¹	Dranco®	May 2017	25°C	59%
D	30,000 t domestic BW ² 5,000 t varied BW ² 10,000 t GW ³	Kompogas®	December 2016 (D1)	8°C	78%
			September 2017 (D2)	19°C	87%
E	20,000 t OFMSW ⁴ 20,000 t GW ³ 4,000 t varied BW ²	Linde-BRV®	April 2017	15°C	72%
F	85,000 t MSW ¹	Valorga®	November 2018	21°C	70%

¹ Municipal Solid Waste - ² BioWaste - ³ Green Waste - ⁴ Organic Fraction of Municipal Solid Waste - ⁵ Anaerobic Digestion

* Measured in ambient air on site during the campaigns

- The reception area with its unloading docks for the main materials treated, apart from green waste, which is most often unloaded in the “digestate-structuring area”;
- The separation area, where waste is sorted and treated to select the finest possible organic fraction (usually less than 10 mm) for input into the digester;
- The digestate-structuring area where waste is dehydrated by centrifuging or pressing, or where outgoing digestate is mixed with structuring material (green waste, fresh waste, etc.). Green waste is generally shredded in this area;
- The composting tunnels area where “solid” digestates are composted;
- The refinement area where the last screen is performed to obtain marketable compost.

An additional fictitious “loaders” area was defined to deal separately with exposure for employees driving loaders.

2.2 Exposure measurement methods

Personal and ambient samples were collected over a continuous period lasting between 3 h and 8 h. Workers were fitted with a pump (Gilian 5000, Sensidyne, USA) and a sampling system mounted in their breathing zone. Static samples were taken at airway level. Ammonia was sampled on two-stage 37-mm diameter closed-face cassettes (CFC), fitted with sulfuric acid-impregnated quartz filters. The flow rate for sampling was 1 L/min, and its duration was 8 h maximum. Ammonia trapped on filters was desorbed with Ultra-Pure Water, and after filtration (MIL-LEX filters 0.22 µm) and 10-fold dilution, subjected to ion

chromatography (METROHM 881 COMPACT IC PRO; METROSE C3 column, without suppressor device and with conductivity detector), applying the INRS Metropol M-13 method (INRS, 2020).

The overall numbers of filter samples collected to monitor ammonia levels on each site are summarized in Table 2.

In some cases (namely on sites B and D), a subset of individual samples was collected over 4 h at a flow rate of 2 L/min. These samples were either used to assess ammonia exposure for a specific task or to allow other individual measurements to be taken to assess exposure to other pollutants.

2.3 Explanatory variables for exposure

Employee exposure to pollutants depends on many factors, including the activity, in particular when performed close to pollutant sources, and when it involves more general parameters depending on installations or processes (intensity of polluting sources, efficiency of localized capture, ventilation of premises, etc.).

Acquiring activity-related determinants, by recording the employee’s work to identify tasks leading to exposure involves monitoring over a full working day. This approach, which may require video-type observation resources coupled with real-time sensors (Martin et al., 1999), is rarely extended to all employees on an industrial site. For this study, only a few workstations were assessed and are not described here.

Only the overall parameters explaining exposure levels were characterized: potential emission rates, based on ammonia potential surface emissivity and material surface area, and air exchange rates in the premises. These

TABLE 2: Numbers of airborne ammonia samples collected in each anaerobic digestion composting plant.

Campaign	A	B	C	D1	D2	E	F
Stationary sampling	29	35	41	42	21	18	30
Personal sampling	15	23	17	52	15	15	14
Loader sampling	3	6	11	3	3	7	9
Total measurements	47	64	69	97	39	40	53

investigations were generalized to all sites, to make their comparison easier.

2.3.1 Potential emission rate

The potential emission rate is a parameter defined to reflect the intensity of pollution sources in premises. It is calculated by multiplying the emission surface of a material present on the premises and the potential emissivity of this material. However, efficient capture and containment systems were taken into account, to subtract emission surfaces from the balance sheet. In the plants studied, the true efficiency of these systems was assessed using a smoke generator. However, in the simplified approach applied here, a binary factor was applied to efficiencies ($\eta=0\%$ or $\eta=100\%$). Material at various stages of transformation could be present in the same room. In this case, the potential emission rate was defined as:

$$E_j = \sum (1 - \eta_k) \cdot S_{k,i} \cdot e_i \quad (1)$$

with

E_j the potential emission rate in hall j ($\mu\text{g.s}^{-1}$),

η_k the capture efficiency of any device applied to source k (%),

$S_{k,i}$ the surface of material corresponding to source k at stage i (m^2),

e_i the potential surface emissivity of the material at stage i ($\mu\text{g.s}^{-1}.\text{m}^{-2}$).

Method used to estimate potential surface emissivity

Several devices are available to measure in situ surface emissivity (Eklund, 1992; Mosquera et al., 2010; Wheeler et al., 2010; Wolkoff et al., 1995). The approach applied here is described in the standard for the measurement of the emissivity of building materials (British Standards Institution, 2006) and consists of covering the emitting surface with a bell. A stream of clean air continuously sweeping through this bell becomes charged with pollutants which can then be analyzed. The air flow rate (Q_{spl}) and output concentration (C_s) can then be used to estimate the emission rate for the pollutant by applying $E=C_s \cdot Q_{\text{spl}}$. For anaerobic digestion composting sites, a variant of this method was implemented, whereby sampled material was placed in a container to analyze emissions. This choice was governed by the following needs/constraints:

- To estimate emissions derived from liquid effluents, particularly for liquids circulating in gutters;
- To access restricted areas (risks linked to the movement of loaders, risk of collapse of the piled stored materials). In these cases, the samples were collected by site employees;
- To obtain average values. To this end, samples from distinct areas of a storage facility were mixed.

The measuring device implemented was designed using elements used for exposure measurements: Gilian GilAir Plus sampling pumps, BW GasAlertMicro5 real-time detector (INRS, 2009), and a plastic bucket with a lid (15-L dosing bucket available in DIY shops). A pump was used to draw air from the container ($Q_{\text{spl}}=1 \text{ L/min}$), and one or more additional pumps diluted the sample (up to 15-fold)

to reduce the moisture content and ensure non-saturating ammonia concentrations reached the analyzer. To avoid using inlet air pre-charged with pollutants, the measurement device was installed outdoors.

A bibliographic study showed that the air velocity at the surface of a material affects ammonia emissions. In his literature review, Ni (Ni, 1999) listed 13 empirical models to evaluate the extent of this effect. Most of these models apply a factor corresponding to surface air velocities in $V^{0.8}$. Thus, compared to an emissive material stored in an area sheltered from air currents (0.2 m.s^{-1}), the same material stored in a ventilated area (0.8 m.s^{-1}) would have 3-fold higher emissivity. In most measurement methods, the emissivity of materials is assessed using devices swept by low air velocities. Consequently, the values obtained are underestimated compared to materials stored in the open. A corrective factor could be applied based on known velocities. However, due to the risk of piles collapsing, the air velocities at the surface of stored materials could not be assessed. Consequently, the emission rates estimated here are default values, which explains the use of the term "potential" applied to all notions of emissivity or emission rate in this article.

2.3.2 Air exchange rates

General ventilation is usually an essential lever for reducing exposure. The background concentration results from dilution of emission rates by air renewal rates ("perfectly mixed room" modelling (Sandberg, 1981)). However, ventilation can induce air movement and, as indicated above, in the case of ammonia, can lead to increased emission rates, which would cancel out the benefits of air renewal. Despite this uncertain context, the general ventilation in plants was assessed by measuring the air exchange rate (R). This theoretical value indicates the number of times the air volume contained in a room is replaced every hour. For these experiments, R was measured by gas tracing, by seeding the air in the room with a gas not present in the environment (SF_6) and monitoring its decrease in concentration over time (Laussmann & Helm, 2004).

3. RESULTS AND DISCUSSION

3.1 Exposure determinants

3.1.1 Air exchange rates

The air exchange rates measured by tracer gas on sites A to F are presented in Table 3. Overall, the rates measured were within the order of magnitude of those expected for industrial sites (typically between 2 and 4 changes per hour), with only two cases showing significantly different R values. In one of these cases, the very high value was the result of operating with open doors (separation area, site C, $R=13.0$). This situation reflected the configuration of the premises during the exposure measurement campaign, which was undertaken during a heat wave. The other case was linked to a ventilation failure due to lack of preventive maintenance (refinement area, site D, $R=0.3$). In this situation, the air exchange measured corresponded to that provided by natural ventilation.

TABLE 3: Air exchange rates measured (ACH *).

Site	A	B	C	D2	E	F
Separation area	4.9	3.4	13.0	2.2	1.8	1.8
Digestate-structuring area	7.1		4.9	2.1		4.9
Composting tunnels area		3.2	2.3	3.3	1.9	2.4
Refinement area			3.0	0.3	1.4	2.0

* ACH: Air Changes Per Hour

3.1.2 Potential ammonia emissivity of various materials

The process of anaerobic digestion composting of waste involves several phases that are generally reproduced across all sites (see supplementary Figure S2, online edition). Over the course of the process, the material undergoes various transformations and as a result extensive variations in ammonia emission can be measured (Figure 1).

Before the digestion phase (separation phase), ammonia emissions were generally low. Similarly, low emissivity was recorded on site C for inputs that had undergone pre-fermentation. After the digestion phase, emissivity was at its maximum (maturation). On all sites, the peak of emissivity was recorded after digestion. The levels reached differed between sites (with a 5-fold difference between maximum and minimum). Emissions then decreased gradually (sieving, storage). Before composting, a structuring agent is often added to the material (generally green waste that does not emit ammonia), which could explain the drop in emissivity. At the end of the composting phase, the material is sieved to retain the finest fraction as the finished product. The emissivity of the sieved product was generally low, except in the case of site F, where emissivity was observed to increase. Note that on this site, composting cycles are

brief (approximately two weeks). Sieve rejects and stored material were also monitored, showing emissivity levels comparable to those of the sieved product. The ammonia emissivity of the compost produced appeared to correlate with the emissivity of the digestate; the lower emissivity of the digestates on sites B and E leads to the production of materials emitting little or no ammonia. In contrast, on sites where the digestate was highly emissive (especially C and F), the materials at the end of the process continued to emit significant amounts of ammonia.

Concerning ammonia emissivity of materials from literature, comparison is not obvious because values obtained are rarely expressed by squared meter of matter. Comparable results from anaerobic digestion sector were not available but some values were measured from composting windrows between 120 and 460 $\mu\text{g/s/m}^2$ (ADEME et al., 2011). Other available measurements concerned cow dung or pig manure and ranged from 50 to 350 $\mu\text{g/s/m}^2$ (Borka et al., 2000; Dinuccio et al., 2008; Hoff et al., 2006). The emissions rates estimated in anaerobic digestion plants are of the same order of magnitude although the nature of the treated material is different and it cannot be ensured that the same stages of the process are considered.

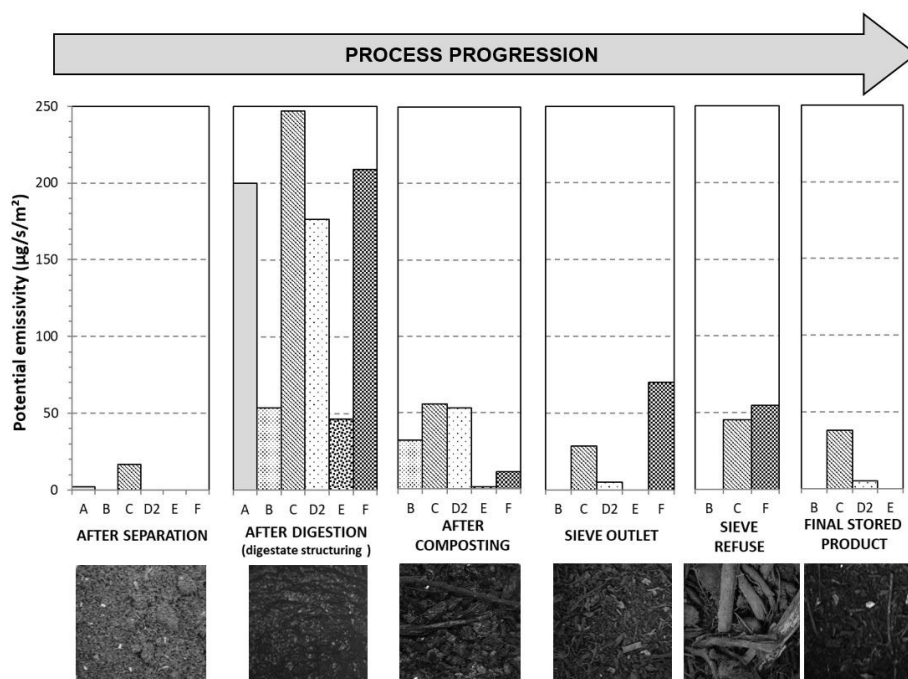


FIGURE 1: Potential emissivity from the material measured over the course of anaerobic digestion composting (a missing letter means that the measurement could not be performed). The pictures show the material at the different steps in the process.

3.1.3 Potential emission rates in each zone

Figure 2 shows potential emission rates for the different work areas on sites A to F, normalized per unit area for each of the zones where the activity takes place. The calculation of the emissivity in the composting tunnel area was made difficult by the fact that the emissive sources varied over time as tunnels were filled and emptied. To simplify estimations of potential emission rates, a static situation was considered based on the average activity in this zone during the measurement campaign. The material present in this zone may be digestate-structuring mixture, composted material directly removed from the tunnel, and/or material maturing in silos if the maturation phase takes place in the same zone.

3.2 Results from ambient air sampling

Ambient ammonia concentrations detected in the various areas are presented in Figure 3, with boxplots representing combined results for the six sites. An alternative presentation of this data is provided in Figure 4. This view highlights differences between the sites, and reveals links to trends identified from the potential emission rates Figure 2. To objectively analyze these two graphs (Figure 2 and the part of Figure 4 bounded by the dotted line), it should be kept in mind that the modelling of potential emissions is simplified, and that – due to the mobility of material in the storage facilities and loader traffic – the ambient samplers could not be positioned in the desired locations to optimize measurement of the average ammonia concentrations in the premises.

In line with the very low emissivity of the material in the reception and separation areas, (Figure 1 and Figure 2), the results for ambient samples showed that the ammonia content measured before anaerobic digestion was low, not exceeding a few ppm_v. The presence of ammonia in these areas is generally due to the early decomposition of biowaste (site C), but could also be explained by the

transfer of pollutants between premises, as in the case of site D.

Values recorded in the digestate-structuring area, just after anaerobic digestion, were often very high due to the handling of ammonium-rich digestates at this stage (Figure 1) and any material left in the open air will be a source of significant emissions. For sites lacking this type of area, these emissions generally occurred in the composting area. For two sites (C and D), the average values measured were very high (almost 40 ppm_v for D). In contrast, for two other sites (A and F) where digestate transfer and treatment are contained, and emissions are captured, the potential emission rate is very low and ambient concentrations were correspondingly low (average of 3 ppm_v). On these sites, emissions will only occur during maintenance operations or production incidents. Conversely, on site D, poor design of the capture systems, lack of rigour with covering of the material transfer line systems, and leakage from the material processing tanks resulted in a high potential emission rate and consequently a high concentration despite substantial general ventilation (Table 3).

In the composting tunnels area, average ammonia levels measured varied depending on the sites. In the case of three measurement campaigns (C, D2 and F), very high values (up to 45 ppm_v) were recorded, with increased concentrations during intensive tunnel emptying. For the three other campaigns (B, D1, and E), the values measured were systematically lower than the French OEL-8h value of 10 ppm_v (INRS, 2021, 2022a, 2022b), even though the same intensive tunnel emptying task was being performed on two of the low-level sites (B and E). On site E, the potential emission rate in this area was almost nil because only small quantities of low-emissivity material were present. On the other sites, the values per square meter were of the same order of magnitude, but the differences between sites were more pronounced for ambient concentrations. In some sites, maintenance operations requiring opening

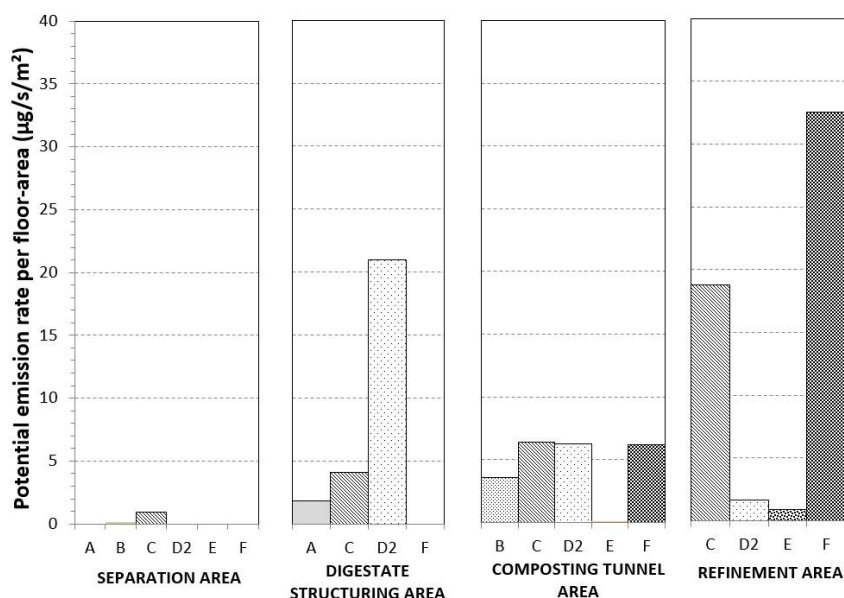


FIGURE 2: Potential emission rates in the different treatment zones, normalized with respect to floor-area.

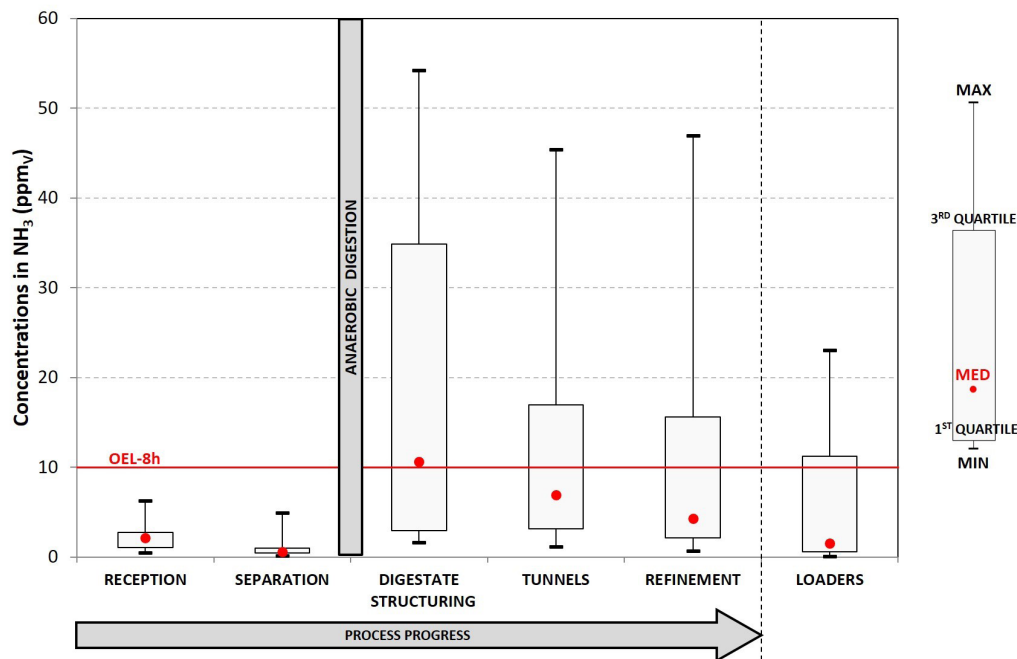


FIGURE 3: Ambient ammonia concentrations measured in various areas in six anaerobic digestion composting plants (boxplot).

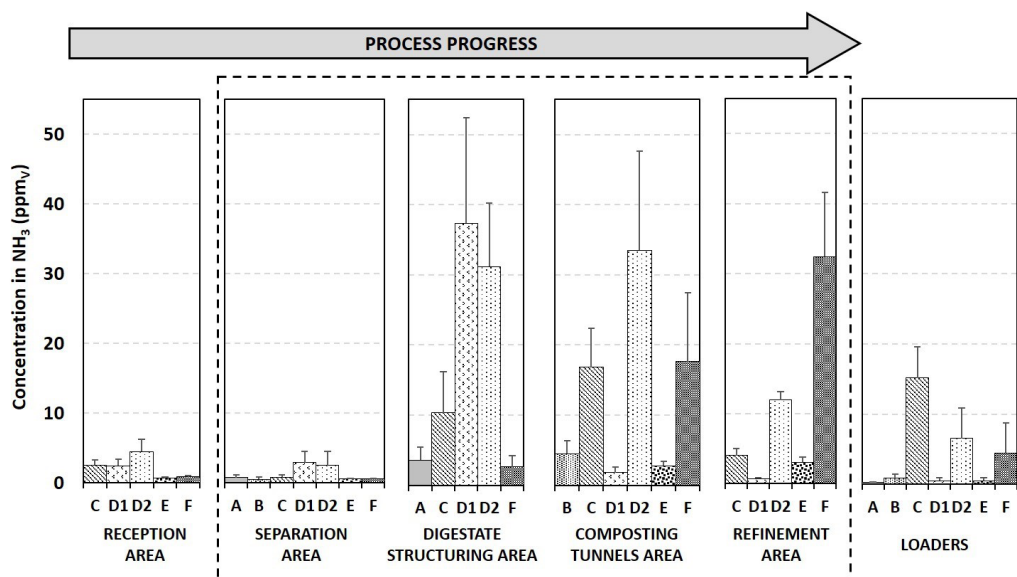


FIGURE 4: Ambient ammonia concentrations measured in various areas in six anaerobic digestion composting plants (bar chart view).

of composting tunnels explain the levels of ammonia detected in ambient samples.

In the refinement area, mainly sieved compost and sieve rejects are present. In this area in sites C and F, very high potential emission rates were measured. These rates were around 15 to 20 times higher than those measured for the other sites. The high emissivity of the material in this zone combined with the storage of large quantities of material could explain the high ambient concentrations measured – of the same order of magnitude as in the tunnels area – especially in the case of site F. Concentrations were low (i.e., less than 5 ppm_v) for three measurement campaigns (C, D1, and E). For two other campaigns, slight-

ly higher values were recorded (between 10 and 15 ppm_v for campaign D2, and for campaign F, six values exceeded the French OEL-8h value).

On site F, the general ventilation system used a considerable airflow rate, which, nevertheless, did not allow satisfactory concentrations to be achieved. It would be difficult to increase this flow rate because of the vast size of the premises. On site C, the ambient concentrations measured were low relative to the corresponding potential emission rates. This difference could be explained by the fact that the zone was open to the exterior and the ambient sampling point was placed close to a wall. The low emissivity of the material could explain the low potential

emission rates measured on sites D (campaign D2) and E. Nevertheless, during campaign D2, a malfunction in the ventilation system (Table 3) led to a non-negligible concentration level.

In the case of loaders, results varied depending on the site, and particularly on the area where loaders were used. Four campaigns revealed very low ammonia concentrations in the loaders (A, B, D1, and E), with low ambient ammonia concentrations (up to 8 ppm_v) measured in the areas where they were used. In contrast, for three other measurement campaigns (C, D2, and F), the areas where loaders were operated had high ammonia levels, and several values exceeding the French OEL-8h value were measured in cabs, especially on site C. These values were often due to problems with the cab filtration systems or to inappropriate practices by operators.

Concerning ammonia ambient measurements from literature, Pöther et al. found that the majority of the shift average values for ammonia ranged from 3 to 13 mg.m⁻³ (i.e. between 4 and 18 ppm_v approximately) but a maximum value was obtained in the delivery category, corresponding to digestate-structuring area (above 20 mg.m⁻³, i.e. 28 ppm_v approximately) (Pöther et al., 2021). Bakhiyi et al. found very low values in different areas of several plants, ranging from 1 to 4 ppm_v (Bakhiyi et al., 2018).

3.3 Results from personal air samples

Personal air sampling was also carried out (Figure 5), the operations leading to the highest exposure levels results correlated with the results of ambient air samples. In the reception, separation and digestate-structuring areas, workers regularly intervene to perform verifications, for ongoing maintenance, or to deal with malfunctions. The composting tunnels area is frequented by loader drivers emptying and filling the tunnels, by maintenance operators when an intervention is necessary, and by shift managers as they do their rounds of the whole site.

Personal sampler values measured for operators working in reception and separation areas were very low (less than 1 ppm_v).

For maintenance operators, concentrations were variable, depending on the tasks performed and the areas frequented (see Figure 5). Two workers from distinct sites were particularly exposed: one was working in a dehydration area which was significantly polluted with ammonia, the other was working in a composting tunnels area. Several values exceeded the French OEL-8h threshold, in particular for these two operators.

Loader drivers were generally well protected by cab filtration when systems were adequately verified and maintained. Nevertheless, this was not always the case, and operators occasionally applied inappropriate practices, resulting in potentially significant exposure to ammonia and other pollutants. On one site, the levels measured in loaders were high, with three values exceeding the French OEL-8h, and one concentration of 9 ppm_v. For a loader driver from another site, a concentration close to 11 ppm_v was recorded, but this driver was fitted with RPE when leaving his loader to intervene in the polluted area.

Production operators work in all areas throughout sites. This role was present in two plants, in both cases individual exposure values were less than or equal to 5 ppm_v.

Polyvalent operators and shift managers generally work site-wide. Like for maintenance operators, their overall exposure during their shift depended on the level of pollution in the areas frequented, and on the time spent in these areas. On one site, significant exposure to ammonia was recorded during the two campaigns conducted. Although only one value exceeding the French OEL-8h was obtained, in many cases concentrations exceeded 5 ppm_v.

Based on these measurements and observations, personalized recommendations were made to each plant, but general recommendations were also highlighted. These include the following:

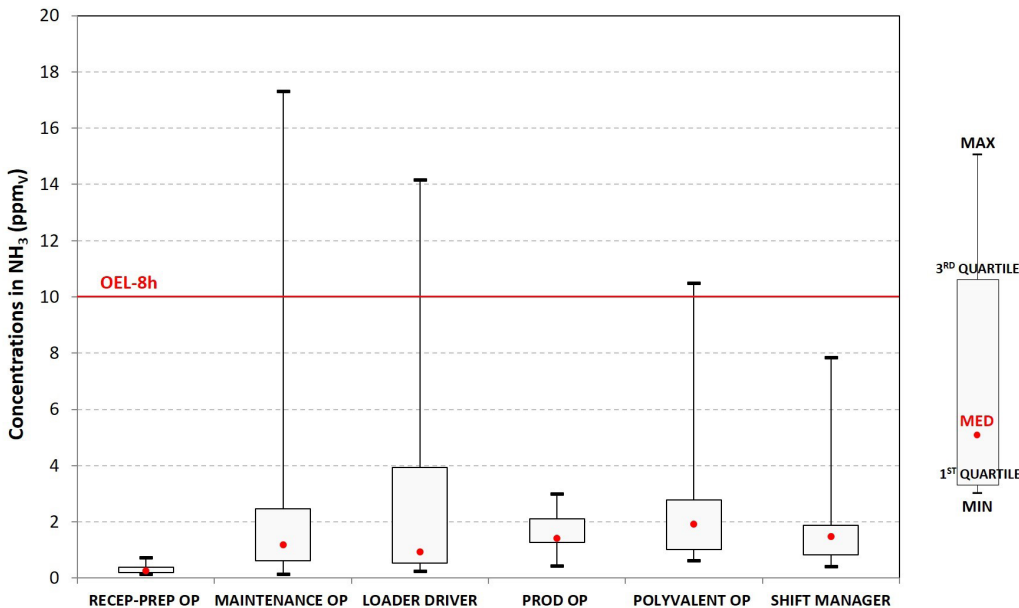


FIGURE 5: Individual ammonia concentrations measured for operators working on six anaerobic digestion composting plants.

- Well-designed and -maintained ventilation systems (sufficient renewal rates, localized capture systems, covered conveyor-belts, among others);
- Ease of cleaning and maintenance (prioritizing level-floor zones, well-organized cleaning systems, among others);
- Sufficiently long treatment cycles (namely for composting) and limited material stocks on the premises;
- Separated activities to avoid exposure due to co-activity;
- Monitored prevention measures and systems (maintenance of equipment: real-time detectors, respiratory protective equipment [RPE] and loader filter systems, ...);
- Employee education to raise awareness of waste management risks and improve practices.

4. CONCLUSIONS

The present work sought to assess occupational exposure to ammonia in anaerobic digestion composting plants. Measurements were performed to sample ambient air in work areas, in workers' breathing zone and in loader cabs. Estimation of explanatory variables consolidated the interpretation of exposure levels.

The results showed significant variability between sites depending on organization, particularly with regard to quantities of waste treated, residence times of materials in the different work areas, pre-treatments applied, anaerobic digestion process implemented, and any post-treatment processes, among other aspects. At the end of each measurement campaign, the results were used to suggest tailored technical and organizational improvements, which were presented to operators and management staff.

Among the explanatory variables identified as useful, the emissivity of the materials was one of the most significant; this parameter is easy to measure using equipment that is already available to occupational hygienists. For this study, measurements were taken using a real-time detector, but conventional filter-based sampling could also be used.

The explanatory variables make it possible to better target the areas with the highest emissions, and thus to prioritize the implementation of novel preventive solutions aiming to reduce emissions at source (process improvement, management of stored quantities, etc.), and/or to improve existing preventive systems (containment, capture at source, general ventilation). In particular, the method identified the most emissive materials, for which it was recommended to limit stocks and to set up local exhaust ventilation for the parts of the process concerned. This knowledge of source flows provides a more objective picture of emissions than purely visual observations (e.g., materials emitting a flow of water vapour wrongly considered to be charged with ammonia).

The approach used in this study, combining air sampling and material emissivity measurements alongside an assessment of ventilation, could be used for an extensive panel of chemical or biological pollutants, and is applicable across all waste management sectors. Its implementation would contribute to significantly improving the information

provided to workers, making them aware of their potential exposure, and could serve as a guide for occupational hygienists in their selection of preventive measures to reduce exposure levels.

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HONG KONG'S CIRCULAR ECONOMY ON PLASTICS: ARE THE ECONOMICS FAILING?

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ABSTRACT

In the Special Administrative Region Hong Kong, municipal solid waste has grown substantially over the past two decades with plastic waste accounting for an increasingly large proportion. In response, the Hong Kong government trialled several schemes that centred on economic mechanisms and the circular economic principles of recycling and reduction to manage plastic bags and plastic beverage containers. As the results of these measures have been met with strong criticism, the present paper assesses the extent to which therein applied economic mechanisms realised a circular management of these streams. To explore this question, the paper relies on quantitative and qualitative field research combined with an analysis of a specifically designed public opinion survey. The findings indicate that price setting is a key element towards circularity for waste plastics in Hong Kong, yet it is not the sole determinant. Using an evolutionary institutional framework the paper shows that besides economics the incorporation of consumers' and recycling stakeholders' interests are key for effective regulations towards a circular economy on plastics in the city.

1. INTRODUCTION

The Hong Kong Special Administrative Region, a global financial and trading centre, is characterized by high population density, high living costs and high levels of consumption. Given the scarcity in land and natural resources, Hong Kong not only critically depends on imports for nearly all products of everyday use. The city moreover has little space to accommodate for goods discarded at their end-of-life. With residents' affluence at an all-time high, municipal solid waste (MSW) generation has soared to 1.51 kg/cap/day in 2022 (EPD, 2023). These amounts are largely disposed in landfills (69 wgt%), about a quarter is exported for recycling (26 wgt%) and merely a small fraction (5 wgt%) is recycled domestically (ibidem). Within MSW, a continuously growing and in treatment challenging stream are post-consumer waste plastics. Of it, the city produces about 2,369 tons/day, rendering plastics into the second largest fraction (21.3%) of overall MSW (ibidem).

Given that existing landfills have operated near saturation levels for a considerable amount of time and free space for new landfills is scarce, more sustainable approaches are urgently needed. Additionally, global environmental dynamics that feature strong local impacts such as climate change and growing resource shortages further warrant adequate system adaptations in Hong Kong. One such concept that aligns with sustainable development and climate

change mitigation through resource conservation and their efficient use is the circular economy (CE). While literature has identified over 221 definitions (Kirchherr et al., 2023), at its core the CE is an evolving blueprint for systemic transition. It stipulates the replacement of globally predominating linear models (extract-produce-consume-dispose) with those that loop resources within socioeconomic systems (i.e. extract-produce-consume-recycle). In this sense, the concept calls for conserving resources to the best possible extent and simultaneously increasing efficiencies in resource use. So despite its frequent association with recycling, the CE bears much wider strategies for application encapsulated in the 10-R hierarchy of CE principles. These encompass the practices of refuse, rethink, reduce, reuse, repair, refurbish, remanufacture, repurpose, recycle and recover (Potting et al., 2017).

In Hong Kong, where high emission levels, climate change and the potential of rising sea-levels constitute very real threats to self-preservation, the government has produced no shortage in strategy papers aimed at enhancing the city's sustainable performance (EPD, 2005; HKGov, 2017). Yet, in the domain of WM, measures yielding substantial results have been falling behind intentions. While the reduction of MSW generation at the source to alleviate



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landfills has been a key tenet, practical measures remain fragmented and ineffective (Wan et al., 2018). This shortcoming is most visible when comparing actual MSW generation with the government's waste reduction plans issued since 1998 (Figure 1). For post-consumer plastics as well as for all other recyclable fractions most evidence points to unsustainable management. For example, estimations from 2020 point out that 80% of plastic beverage containers end up in local landfills (DWW, 2020). Further aggravating the situation are the waste import restrictions (the National Sword policy) implemented in Mainland China in 2017, which limit the import of most recyclables (BEC, 2021) and therefore cut-off Hong Kong from its major recyclable waste export market. Simultaneously, affluent lifestyles manifesting in conspicuous consumption (Lau, 2019) as well as the impact of COVID-19 triggering a packaging frenzy in the city have further increased the need to devise local, effective solutions.

In this context, the Hong Kong government has issued measures to address the plastic challenge. These include raising the levy on single-use plastic shopping and retail bags in 2022, a partial ban of single-use plastic table ware and food and beverage containers in 2023, the planning of an extended producer responsibility (EPR) scheme for plastic beverage containers for 2025 and a household waste charging scheme for 2024. The temporal sequence of these measures indicates an acknowledgment of the problem's urgency among governmental stakeholders. From a researcher's perspective, this raises the question to which extent such institutional (rule-based) actions can be effective in yielding CE related results.

To investigate this issue within a reasonable scope, the present contribution focusses on two post-consumer plastic waste streams, i.e. single-use waste plastic beverage containers (WPBCs) and single-use plastic bags (SUPBs). To benchmark the inquiry against the market-oriented, laissez-faire approach generally adopted by the Hong Kong government in WM (Lee, 2020; LEGCOPEA, 2023), the study aims to assess the economic effectiveness of official CE measures. The respective research question therefore asks to which extent are economic mechanisms effective to realize CE practices for WPBCs and SUPBs in Hong Kong? Specifically, the paper explores if Hong Kong's CE

on plastics (1) is predominately about price-setting and (2) if the official approach manages to 'get prices right'. The implicit hypothesis pursued in this paper is that the government's economic incentives are insufficient to foster a CE management of SUPBs and WPBCs.

Over the following paragraphs, the manuscript will first introduce the materials and methods used for the analysis. The subsequent section will then present the economic mechanisms implemented by the government for a circular management of SUPBs and WPBCs. After that the responses of non-governmental stakeholders to the official economic incentives will be analysed so as to assess whether official economic stimuli correspond to the stakeholders' own preferences. The concluding part will merge discussion and conclusion to elaborate on the role of economic factors in Hong Kong's CE on post-consumer plastics.

2. MATERIALS AND METHODS

Regarding materials and data, the paper builds its analysis on three types of data sources. In first instance, it draws on field research by the author on recycling structures for WPBCs conducted during late 2021. Related activities of material collection comprised interviews with collecting and recycling stakeholders, in parts belonging to the informal recycling sector, as well as observational quantitative data recording. The aim was to quantitatively ascertain transacted WPBC quantities along the value chain as well as an assessment of the practices and patterns adopted by collectors and recyclers when managing WPBCs. A detailed outline of obtained data is provided in Steuer and Chen (2023b). Second, the manuscript also presents so far unpublished survey results on waste plastic management and attitudes among Hong Kong residents. For these two surveys, conducted in May 2023 and September 2023, a total of 820 and 1,004 respondents were interviewed via the Hong Kong Public Opinion Research Institute (HKPORI) and Qualtrics. The questionnaires were disseminated online and comprised two types of questions: (1) multiple choice questions to capture waste single-use plastic generation quantities; (2) five-point normed questions along a Likert-scale to quantify behavioural patterns, routines and preferences of consumers on their engagement with

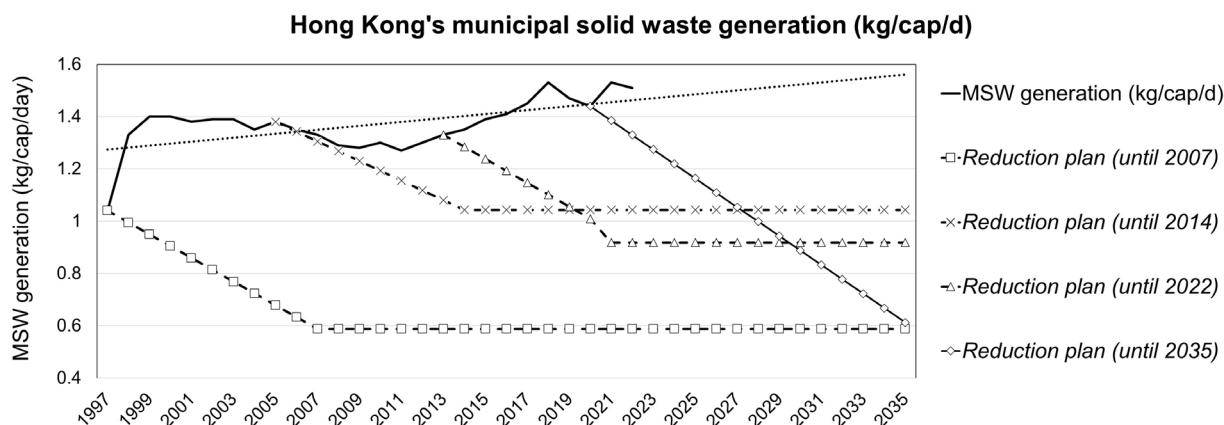


FIGURE 1: Official waste reduction plans vis-à-vis actual MSW generation in Hong Kong (the author based on EPD, 2001-2023).

single-use plastic waste products. In third instance, desk research was used to screen official documents and policy papers, corporate (sustainability) reports, public surveys and working papers from non-governmental organizations on Hong Kong's CE for waste plastics.

In terms of analytical methodology, the study uses an evolutionary institutional framework previously developed by the author for the assessment of CE performance in mainland China and Hong Kong (Steuer, 2021; Steuer, 2022). Its main purpose is to capture the process of institutional change and whether this dynamic results in effective outcomes. For the current study, the subject of analysis are the economic mechanisms, in particular the setting of prices for CE instruments, i.e. the levy system for SUPBs and the redemption value for an EPR on WPBCs in Hong Kong. As for whether the mechanisms are effective, two conditions must hold: First, the mechanism (a rule or institution) must achieve a quantitatively verifiable improvement, e.g. a reduction in SUPBs sold or consumed. Second, the mechanism must be 'interest-inclusive', i.e., it must account for the interests/ preferences of all involved stakeholders. For example, an officially implemented EPR on WPBCs must provide for deposit or redemption values that are in line with expectations of beverage retailers, waste generators, collectors, recyclers etc. In case of falling short in being interest-inclusive, the implication is that stakeholders do not participate in the system (generators do not return, collectors do not recover etc.) and likely develop their own solutions to handle WPBCs (such as discarding these in general waste bins, rejection to recover etc.).

The term of institutions in this context refers to man-made solutions to a problem, a challenge or a novel context. In related literature, institutions are often described as systems of rules, that structure and guide behaviour of and interaction between stakeholders (Groenewegen et al.,

2010). The evolutionary factor implies a system change over time, such as a gradual transition from a linear to a circular economic system. Here change can be envisaged as an unfolding interplay between novel challenges, problems or contexts and corresponding institutional solutions from different stakeholder groups (Mantzavinos, 2000). These solutions can be formal (regulations, laws, schemes etc.) or informal (routines, habits, norms) in nature. As new challenges affect diverse stakeholders at the same time, the analytical model illustrates an interplay between these agents. Each affected group does respond to the problem in their own specific manner: They craft institutions to solve the challenge at hand and once put into practice stakeholders may learn about how their as well as other stakeholders' institutions on the given challenge perform in reality. Learning is dependent on capturing information from feedback loops, such as through data collection, individual accounting efforts, observation of waste recovery patterns etc. The so acquired learning insights shape stakeholder experiences and value ideals and moreover decide if and how they adjust or refine their own institutions: Here shortcomings in achieving effective rules – by excluding other parties or because of quantitative target shortcomings – may lead to rule modifications or entire rule overhauls. A flow diagram illustrates a generalized evolutionary process of institutional change (Figure 2).

In the current study, the aim will be to identify whether CE related rules and practices on SUPBs and WPBCs have changed. And, if such changes can be traced back to feedback loops, information exchange or other interactions between governmental and non-governmental stakeholders. Hereby, a key, but not the exclusive focus will be changes in official pricing and other economic incentive structures that the government devises with the aim of attracting participation of ever more stakeholder groups.

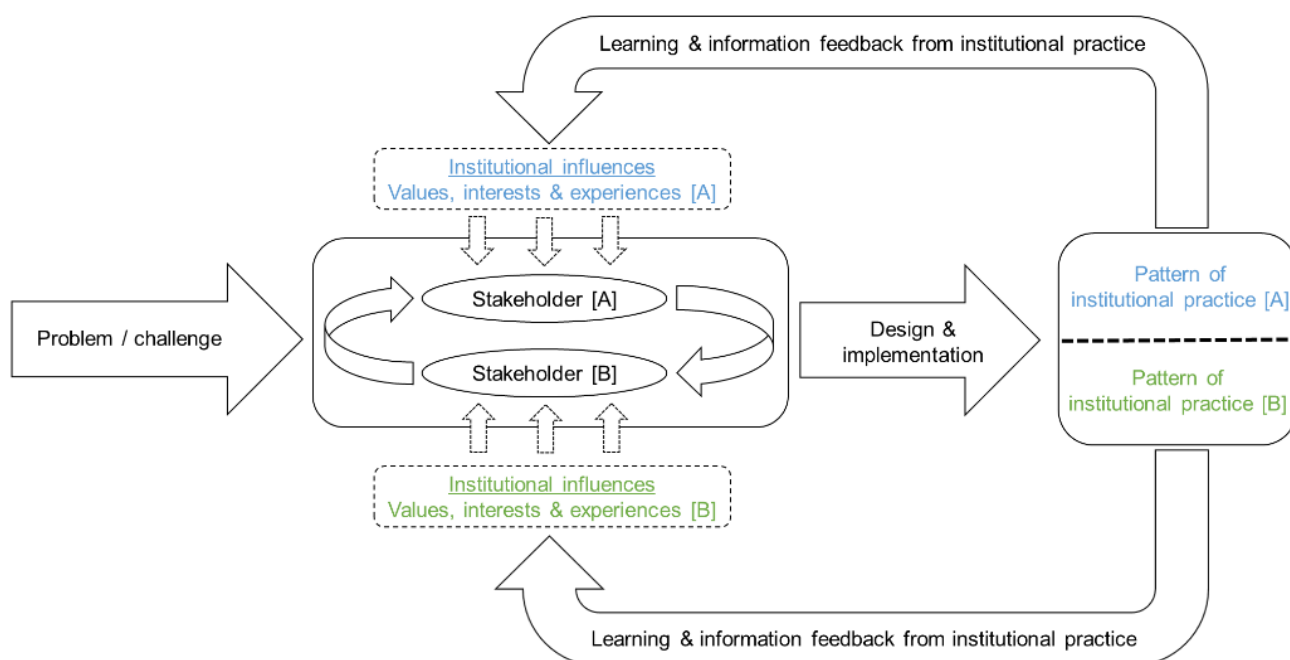


FIGURE 2: The evolutionary process of institutional change in a socio-economic system (based on Steuer, 2021).

3. RESULTS

3.1 The Economic Mechanisms in Hong Kong's Circular Economy on Single-Use Plastic Bags and Beverage Containers

3.1.1 Regulatory framework

The city's government regulates aspects of environmental protection and sustainability via single statutes (Tong, 2021). In contrast to other jurisdictions, Hong Kong has not formulated a CE law and CE relevant legislation is by and large fragmented and formulated as ordinances. The fundament for WM and CE measures is the 1980 Waste Disposal Ordinance, which has been extended by Product Eco-responsibility Ordinances for glass beverage containers, WEEE and plastic shopping bags. The piecemeal nature of this approach bears shortcomings in terms of measures being fragmented and with exception of the Product Eco-responsibility Ordinances often bearing overly voluntary and tacit characteristics. That is for example the case of source separation measures for MSW from households, which have failed to realise residents' participation (Wan et al., 2018). In regard to plastic waste, a direct comparison with the EU's institutional framework indicates that Hong Kong gave the issue too little attention. Issued regulations, such as the very recent waste charging scheme that was expected to take effect in 2024, address MSW recyclables in their entirety rather than setting up specific stipulations for individual waste streams. In terms of qualitative differences on promoted CE principles for managing post-consumer plastics, an earlier analysis by the author demonstrated that the EU's regulations predominately stressed 'refuse' and 'recycling' approaches. Hong Kong's regulatory stance on the other hand focusses on 'reduction', yet without notable achievements in curbing generation (Steuer and Chen, 2023a).

3.1.2 The levy for single-use plastic bags

The levy on SUPBs emerged as a result of the government's waste reduction strategy with the aim to promote a 'plastic-free' culture in the early 2000s when overall MSW amounts had increased notably (Deloitte, 2018). In 2009, the levy was introduced under the Product Eco-responsibility Ordinance and is next to glass beverage containers and WEEE one of Hong Kong's current mandatory (Extended) Producer Responsibility Schemes (PRS) (Tong, 2021). Launched in 2009 as a pilot, it initially covered a selected number of retailers, e.g. supermarkets, convenience stores, personal health stores etc. and levied 0.5 HKD (0.06 EUR) on every SUPB (EPD, n.d.). The scheme was extended to all retailers in the city in 2015, exemptions were abandoned, and the charging level was further raised to 1 HKD (0.12 EUR) in 2022 with the government promoting the reuse concept of 'bring your own bag' (ibidem).

Along with this development the price adjustment as a consequence of generated SUPBs deserves attention. The initial measure in 2009, indicated a 5% decrease in bags generated from the pilot-covered outlets over the following year (EPD, 2011). The coverage extension in 2015 achieved an even more notable decrease of 25%, i.e. a reduction of 5.24 billion to 3.93 billion SUPBs recorded at landfills (EPD,

2016). Yet, over the following years a rebound in growth rates of discarded SUPBs was assessed through landfill sampling, which indicate an annual median increase of 4.7% over 2016-2022 (EPD, 2017-2023). The resulting levy increase to 1 HKD by late 2022 is a direct response to this rebound (BEC, 2021). As of 2023, the scheme appears to have yielded some success according to Hong Kong's Environmental Protection Department (EPD): Over inspections across the entire city, it was found that retailers overall adhered to the levy scheme, had substantially reduced the variety of SUPB types for sale and helped distributing information on the practice of the levy (ACE, 2023). While most recent MSW statistics from 2023 have not yet been published, section 3.2.1 will explore to which extent its raise resonated with consumer preferences.

3.1.3 The Hong Kong EPR for waste plastic beverage containers

For WPBCs a PRS is currently under design and expected to be launched in 2025. Several pilots aiming to apply EPR specific mechanisms to WPBC recovery have been tested in the past. At their core the most controversial factor pertains to cost and price mechanisms. As of now, the common agreement among governmental and non-governmental stakeholders is that an economic incentive has to be applied to encourage the return of containers by consumers (Tong, 2021). In contrast to already existing PRS on glass bottles and WEEE, the scheme for plastic beverage containers will use a market-led approach aiming to outsource responsibility to commercial stakeholders (LEGCOPEA, 2023). The government will neither collect a levy from producers, importers and retailers nor coordinate downstream recovery services. Rather official stakeholders will set statutory recovery targets that beverage suppliers will have to meet. Moreover, coordinating duties including the implementation of a producer responsibility organization will have to be organised by beverage suppliers, while retailers with a retail floor area of 200 m² or above have to serve as designated return points and provide takeback and rebate services (EPD, 2021b). Takeback services cover plastic beverage containers and cartons containing between 100ml and 2 litres, and for these items scheme operators are obliged to provide rebates. As of 2023, a rebate amount of 0.1 HKD (0.011 EUR) is the current official benchmark (LEGCOPEA, 2023). As discussed in the sections below, this rebate level is far behind international EPR standards for WPBCs and emerged as result of lobbying from contesting interest groups.

3.2 Assessing the (Economic) Preferences of Non-Governmental Stakeholders

3.2.1 Scheme operators: Beverage manufacturers, retailers and recyclers

The variety of corporate stakeholders potentially affected by Hong Kong's PRS on WPBCs has recently merged into one dominant representative group to voice its interest. Drink Without Waste (DWW), which was established in 2017, comprises producers, retailers and recycling operatives as well as NGOs. Quite in line with the government's

intent, it supports the PRS and aims to recover 70-90% of beverage containers upon the scheme's taking-effect. Beyond recycling, the initiative further intends to realise CE goals for WPBCs pertaining to reduction (via installing and operating beverage dispensers), and redesign (via sustainable packaging design standards) (DWW, 2020).

There is ample documentation of some individual measures by corporate entities within the group striving to fulfil these objectives. This includes weight reduction, reduction of virgin PET and full substitution via recycle by local beverage companies, single-use plastic bottle removal by hotel chains and the setting up of water dispensers by property management (Deloitte, 2018). At its centre, the initiative's efforts, are geared towards developing a recycling network for WPBCs. The key cornerstones are (1) to create a dense and easily-accessible network of return points; (2) adopt a value-on-return scheme (VRS) instead of a Deposit Return Scheme (DRS), which DWW considers to minimize costs and facilitates the money-for-bottle transaction process; (3) to keep the VRS at a low level, which was initially set at 0.05 HKD (0.005 EUR); this in turn (4) was considered sufficient to incentivise "not only consumers but also... domestic helpers, building and street cleaners, and other informal collectors" to engage as primary collecting stakeholders (DWW, 2020). Both latter aspects underscore the prominent economic interest of the group's solution, which will be further analysed in the discussion section.

In regard to recycling, the perspective of collecting and preprocessing stakeholders deserves to be outlined separately. In general, recyclers in Hong Kong face high rental as well as relatively high labour costs, the latter of which being in parts due to weak source-separation of MSW recyclables by Hong Kong households (Steuer and Chen, 2023a). Further complicating matters is that Hong Kong does hardly host any manufacturing companies, which could make use of recycle as production feedstock (BEC, 2021). This implies that recycled materials have to be sold on international markets, where Hong Kong's cost-competitiveness is substantially undercut by comparable recycle produce from China. Yet, to evaluate the feasibility for WPBC recycling systems, DWW has between 2020-2021 conducted a one-year pilot assessing the collection and

preprocessing capacities of existing private-informal WM structures in Hong Kong (Steuer et al., 2022). In terms of incentives the pilot provided 0.1 HKD (0.06 EUR) for the entire operating costs, including subsidies for collectors (0.05 HKD/bottle) and pre-processors (0.035 HKD/bottle). At that time, the author was tasked with assessing the pilot performance, which was essentially done via quantitative documentation in the field and interviews with collecting and pre-processing (sorting, cleaning, baling) stakeholders. Quantitatively, the results indicated a bottle recovery capacity of 3% (based on sale volumes) by the surveyed pre-processors during the pilot. Would the pilot have been extended to all active pre-processors in Hong Kong the recovery rate could have been raised to 23%. Yet, once subsidies were stopped, recovery rates dropped substantially (Steuer and Chen, 2023b). The qualitative results from the interviews shed light on the relatively low recovery rate: Collecting as well as recycling stakeholders indicated their dislike for recovering WPBCs, due to reasons of hygiene and low reimbursed per-unit-value. Furthermore, transport, sorting, cleaning and baling were considered as cumbersome compared to the predominately handled recyclable streams of paper, cardboard and metals. A longitudinal analysis of preferences for and against CE practices for WPBC management further underscored the unchanged willingness of collectors and recyclers to engage in WPBC recovery. Over the past few years, both stakeholder groups have expressed a clear rejection towards recycling this waste stream (Figure 3).

3.2.2 Consumers

A major driver behind the launch of the SUPB levy outlined in section 3.1.2e were environmental NGOs lobbying government towards using the argument of consumer opinion (Lau, 2019). Yet, in comparison to other measures such as municipal waste charging, recycling infrastructure or an encompassing PRS, public preferences for the relatively simple SUPB levy merely ranked 6th out of seven CE concepts (Wan et al., 2018). In regard to consumer behaviour, the initial levy of 2015 had a limited effect as SUPB generation rebounded again soon after dipping for a single year (BEC, 2021). One potential factor explaining



FIGURE 3: Binary Interest Node Schematic (BINS) mapping stakeholder interests for and against CE management of WPBCs (reproduced from Steuer and Chen, 2023b).

this trend can be discerned from consumer sentiment on what this stakeholder group perceives to be the minimum charging level needed to discourage consumption. In 2022, a governmental-led consumer survey found that 74.5% or 6,651 out of 8,804 surveyed Hong Kong residents would consider at least 1 HKD adequate to reduce consumption. The same study conducted a telephone-survey among 1,003 residents on what charging level would constitute the most effective deterrent. Presented with a range from a minimum of 1 HKD to over 5 HKD, interviewees' responses resulted in a weighted mean of 2.2 HKD (0.24 EUR) (CSD, 2022). Against this background the government's 0.5 HKD level from 2015 appears to be too low for curbing SUPB consumption.

Regarding WPBCs, relative early pilots on return schemes in 2005 and 2007 provided insights on the role of (monetary) incentives for Hong Kong residents' participation in the CE. Both schemes were funded and organised by the EPD in collaboration with Swire Coca-Cola, Vitasoy and Watsons Water as main partners and conducted within sites of selected residential quarters for a year each. There residents could redeem WPBCs (sized between 480-800 ml) for demonetized 'reward points'. Points were allocated by specifically set-up collection points on a per-container basis and had to be uploaded to a specific redemption card. Having collected a certain number of points, consumers could exchange these for souvenirs or beverages. Setting aside the scheme's peculiar logic of returning containers to be rewarded with more containers for consumption, the pilots showed little innovation and overall failed to encourage residents' participation. Based on the general attitude reflected in residents' feedback, it turned out that these rather cherished convenience (access and opening hours of the WPBC return points) over other factors (Chan, 2011).

Over a decade later that perspective still appears dominant. A public opinion survey by the renowned HKPORI among 976 respondents, revealed that economic rewards for engaging in the return of WPBCs only ranked a 4.4/10, while aspects of hygiene, return convenience and number of return locations achieved an 8.3/10, 6.7/10 and 6.9/10, respectively (HKPORI, 2022). For return convenience and locations, public preferences indicated an over 60% sup-

port for collection stations within residential areas, i.e. close to one's home, while bus stations and retail ranked second with 57% and 54%, respectively (HKPORI, 2021). The fact of economic mechanisms as relatively weak drivers is corroborated by the author's own survey, which indicates that around 55% of consumers would not feel (more) motivated by economic incentives to segregate WPBCs and other single-use plastics (Figure 4).

The HKPORI has in this context conducted a very insightful survey, which aimed to explore the threshold value of a PRS inducing consumers to return their WPBCs. The findings show that notwithstanding whether DRS or VRS, a value of 1 HKD (0.12 EUR) would induce 62% and 78% of 927 respondents to return their emptied containers (Figure 5) (HKPORI, 2021). What this high benchmark means for the overall institutional dynamics of Hong Kong's CE on plastics will be discussed in the subsequent concluding section.

4. DISCUSSION AND CONCLUSIONS

4.1 Single-use plastic bags

From the perspective of the analytical model, the official implementation of the levy for SUPBs is essentially the result of an institutional responsiveness dynamic between governmental and consumer stakeholders: the positive information and learning feedback from consumers in support of the levy in 2011 provided a strong motivator for Hong Kong's government to expand the scheme and raising the levy to 0.5 HKD per bag (EPD, n.d.). Apart from consumer support, the most recent adaption to 1 HKD in 2022 can be traced back to the information feedback from increased waste plastic bag generation. Consumer sentiment captured via a 2022 consumer survey yet further suggested that to render the scheme even more effective a levy of 2 HKD (0.23 EUR) would be viable as such a price level still enjoys broad popular support (LEGCO, 2022). This underscores the argument of the paper regarding prices being a key mechanism to drive the CE practice of reduction for SUPBs in Hong Kong. Additionally, the case of SUPBs indicates that institutional adjustment accounting for involved stakeholder interests leads towards regulatory effectiveness. While statistical data for SUPB waste gen-

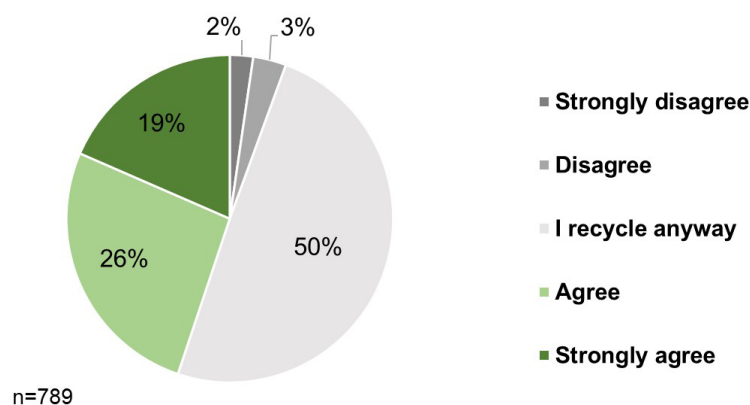


FIGURE 4: Interviewees on whether economic incentives (tax reduction, reimbursement, etc.) motivate individual separation of single-use plastic waste (public opinion survey conducted by the author).

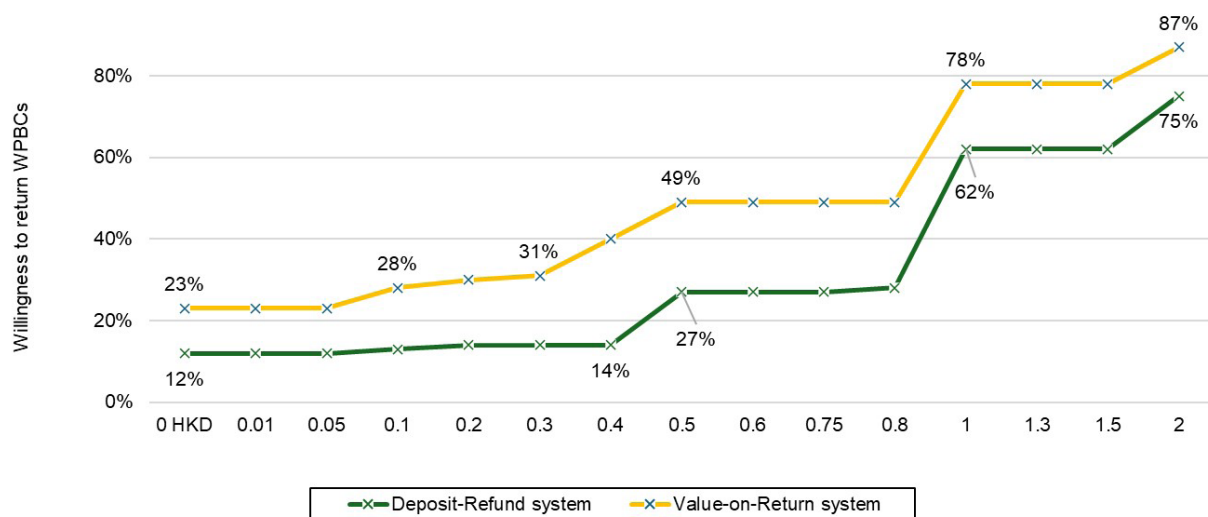


FIGURE 5: Willingness to return WPBCs for financial refunds (reproduced from HKPORI, 2021).

eration for 2023 is not yet available, the comparison with other jurisdictions indicates that the official levy value is set well with international benchmarks. For the purpose of comparison, the levy of each jurisdiction was divided by the Big Mac Index, a means to adjust country differences in consumer purchasing power between different countries. The resulting ratio enables a ranking of each region/ country, while the reduction rate provides an overall indication of the levies' impact (Table 1). So, while Hong Kong ranks well compared to other countries, its low reduction impact must be seen in light of the comparatively short implementation period.

4.2 Waste plastic beverage containers

For the stream of WPBCs, the path of institutional evolution is substantially more complex as preferences of scheme operators and consumers are each impacting CE performance and price setting. The substantially more voiceful stakeholder group were the scheme operators, represented by DWW, which used its notable public relations activities to influence the ongoing PRS design. For this group, the cost element plays a central driver. One factor emphasized by the beverage producers in 2018 was that virgin PET containers cost around 0.014 HKD/unit (Deloitte, 2018). Any redemption value imposed on containers via a PRS would therefore have a notable impact on retail prices and was seen as a potential disincentivizing factor for consumption. Hence the initiative's strategy

consisted of suggesting a redemption value of 0.05. Another argument of DWW for this stance was the potential for system fraud, which was assumed to occur should the redemption value be set too high (DWW, 2020). An online query on that challenge in the context of WPBCs however indicates that fraudulent practices can be easily prevented via digitalization, electronic financial transfers and single unit counting (Tucker, 2022). These measures mirror what the government has outlined in terms of requirements for its PRS (EPD, 2021a and 2021b), which in turn renders this danger relatively minimal. Yet, to DWW's credit, regarding scheme operations, some of its proposed objectives have gone beyond the government's demand, such as the inclusion of all single-use plastic beverage containers in the scheme, the emphasis on recycling network development and inclusion of informal stakeholders (DWW, 2020).

Another economic mechanism promoted by DWW is to replace the initially envisaged DRS with a VRS (DWW, 2021). While this approach has been criticized by green groups as failing to make consumers accountable for their consumption (personal communication with Mr. L., May 2021), there is yet another angle to it. DWW has notably highlighted their intention to include low-income groups, e.g. household helpers, informal collectors and cleaners of residential buildings into their design for a PRS (DWW, 2020). The DWW's argument therefore was that a VRS could induce these stakeholders to recover otherwise discarded containers for a small monetary return. These considerations how-

TABLE 1: Using the Big Mac Index to compare economic viability of SUPB levy benchmarks (the author based on ACE, 2023 and https://scaan.net/plastic_global/).

Country/ city	Implementation (year)	Levy effectiveness period (years)	Levy (EUR)	Big Mac Index (BMI) (EUR, 2023)	Levy/ BMI *100	Reduction
Ireland	2002	5	0.20	4.36	4.59	94%
Hong Kong	2022	0.33	0.11	2.45	4.48	60%
Portugal	2015	2	0.10	4.36	2.29	74%
UK	2015	8	0.06	4.06	1.48	98%
South Africa	2003	9	0.02	2.14	0.93	44%

ever neglect two critical aspects: During the author's field research, it was indeed found that informal collectors and cleaners delivered the majority of WPBCs over the DWW pilot in 2021 amounting to 78-91%wgt. of overall recovered quantities. Yet, once the subsidy ended, collected WPBCs reduced by about 1/3 primarily due to informal collectors ending their engagement (Steuer and Chen, 2023b). As this pattern underscores the critical factor of pricing and value setting for a future PRS in Hong Kong, another aspect of concern is system sustainability. During interviews on the ground, the author and colleagues found that informal collectors and cleaners were mostly about 60 years of age and older. In turn solely relying on a low redemption value to incentivize collection among low-income groups will not be sustainable over the long term and at best constitute a temporal externalisation of WPBC collection.

Given the notable efforts of DWW, the government has picked up several suggestions of the group, for example by leaving the question of DRS vis-à-vis VRS open to scheme operators as well as supporting the DWW as key authority regarding the consultancy for the further PRS design. Yet on the matter of value setting, relevant government documents show a tendency among lawmakers to push for a potential raise of the reimbursement value from 0.1 to 0.2 HKD (0.023 EUR) (LEGCOPEA, 2023). This would imply that the information feedback from the corporate interest group did not fully capture governmental stakeholders' position for decision-making and institutional design.

For consumers, the factor of economic value setting does play a substantially less important role. As indicated in the previous sections, this group clearly supports a PRS for WPBCs. Yet, as a recent findings have shown, to motivate a majority of Hong Kong residents into participation the rebate value has to be set at around 1 HKD (0.12

EUR) (see Figure 5). That level is somewhat corroborated by comparisons with international EPRs for WPBCs. Here again resorting to the Big Mac Index and how it compares to the suggested deposit/ redemption value, the officially suggested 0.1 HKD and 0.2 HKD values appear comparatively low. Rather, a redemption value of around 0.5 HKD would put Hong Kong closer to international best practice systems: As shown in Table 2 below, the Deposit/BMI ratio and the return rate show a strong positive correlation ($r_s=0.854$) for countries with an EPR for WPBCs. Hence, raising bottle deposits in line with the BMI's purchasing power parity benchmark appears to align well with higher recovery rates. Assessed from this angle, Hong Kong's decision-makers' current proposal of 0.1 and 0.2 HKD would thus appear to fall short of getting the prices right, which for the case of WPBCs provides support for the research hypothesis. By implication, the container value should at least be raised to 0.5 HKD (0.054 EUR) to come closer to the international Deposit/BMI median of 3.

4.3 Concluding remarks: The role of financial mechanisms vis-à-vis other institutional solutions

So are prices really the most decisive factor for Hong Kong's CE on plastics? A review of consumer sentiment would on first sight underscore this assessment. Among this group there is a strong support for a PRS on WPBCs and if raised to 1 HKD (0.12 EUR) or even 2 HKD (0.24 EUR) would induce up to 78% and 87% of consumers to return their bottles (Figure 5). Yet, besides purely economic incentives, the government should also incorporate lessons from previous behavioural evidence of this stakeholder group: Prior pilots on the stream failed due to neglect of WM infrastructure providing convenience and accessibility. Moreover, the author's longitudinal analysis of CE preferences among consumers shows that there is interest in WPBC re-

TABLE 2: Correlating international deposit values along the Big Mac Index with WPBC return rates (the author's calculations based on return rates provided by Reloop, 2020).

Country	Deposit Value (EUR)	Return rate (PET) 2016	Return rate (PET) 2019	Big Mac Index (BMI) (in 2020, EUR)	Deposit/BMI*100
Croatia	0.06	87%	89%	2.93	2
Estonia	0.10	87%	86.9%	4.12	2.4
Lithuania	0.10	92%	92%	4.12	2.4
Sweden	0.11	84.9%	84.1%	5.04	2.2
Norway	0.16	88.2%	89.4%	5.16	3.1
Denmark	0.20	90%	94%	4.03	5
Finland	0.20	92%	90%	4.12	4.9
Germany	0.25	98%	97%	4.21	5.9
Netherlands	0.25	95%	95%	4.12	6.1
Median	0.14	89%	90%	4.12	3
Hong Kong (2022) (Pilot: 0.1 HKD)	0.011	n.a.	n.a.	2.22	0.5
Hong Kong (2023) (Official deliberation: 0.2 HKD)	0.023	n.a.	n.a.	2.22	1
Hong Kong (int. standardised benchmark: 0.5 HKD)	0.054	n.a.	n.a.	2.22	2.4

cycling systems, but in a similar manner various reduction approaches to beverage containers as a business concept enjoy support (Figure 3). Indirectly, this particularity points to a rather reasonable assessment that not everything is about economic incentives in the circular economy: Surely, prices play their role in triggering or disincentivising certain behaviours. Yet, potentially more important are questions of overall institutional design. This pertains to developing rule systems and business models that fit into the cultural context of a region where the CE should take hold. More specifically, the very idea of designing a usable solution – a repairable product, a compostable packaging material, a sharable device – in a circular manner that aligns with preferences in the target society are key.

So while SUPB levies are likely to rise and a PRS for WPBCs will most probably be instituted in the near future, currently operated systems to spur recycling are in need of an overhaul. Unfortunate in this regard is the government monitored, privately operated MSW recyclable take-back station network GREEN@COMMUNITY, which has so far failed to incorporate consumer feedback. The network density is low, impairing return delivery convenience; redemption is demonetized by providing GREEN\$, a type of digital reward point, that can be exchanged for food products and other daily necessities; informal collectors and residential cleaners, who deliver the majority of WPBCs, yet prefer to supply the cash-rewarding traditional pre-processor network. As the latter received a median of 600 kg/month, it dwarfed the GREEN@COMMUNITY stations' reclaim performance of 244 kg/month (Steuer and Chen, 2023b). Against the backdrop that the formal institutional development would have to act on two fronts – adjusting prices and developing WM infrastructure – to account for public interest inclusiveness and thereby incentivize participation, the question is whether recycling is the best CE option for managing WPBCs. Previous work of the author and colleagues employing a projective LCA has shown that a combined refuse-reuse scenario with alternative packaging would be environmentally more beneficial than recycling WPBCs (Steuer and Chen, 2023b). Given the high anticipated costs for WM infrastructure development in Hong Kong, alternative CE approaches beyond recycling single-use beverage containers are worth considering to achieve a sustainable, circular system in the city.

What however constitutes a major set-back for Hong Kong's CE ambitions (beyond the single stream of waste plastics), is the repeal of Hong Kong's municipal waste charging scheme, a detrimental decision officially announced on 27 May 2024. The scheme intended to apply the polluter-pays principle by charging households for residual waste not segregated in recycling bins. Without doubt, its implementation would have required more efforts and imposed costs (around 0.11 HKD or 0.013 EUR per litre) on residents. Vice-versa, however, the measure could have effectively paved the way for more elaborate EPR schemes, such as for WPBCs, and reduction efforts by consumers. The simple logic that was missed out on by rescinding the scheme is that making polluters pay enforces stakeholders to rethink consumption patterns. By extension, adequate price-setting in waste charging that accounts for different

income groups, will contribute to resource conservation, which ultimately represents one of the key environmental objectives of the CE. The point to be made is that economic mechanisms can best deliver their impact in a CE, if a regulatory framework sets an effective playing field: Societies have to accept that anthropogenic dynamics affect the natural environment and offsetting these by marked-oriented internalisations will fall short of being effective. In Hong Kong, the shortcomings have been evident for the cases of SUPBs and WPBCs: Low associated 'costs' do not drive consumers or producers to align with CE principles such as refuse, rethink, reduce or recycle. Instead a mandatory framework is necessary to outline stakeholder responsibilities, set resource-efficiency targets and develop a long-term pathway towards sustainable resource use.

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The present study involved research on human subjects, which was approved by the Human Research Ethics Committee (HREP-2023-0394) of the Hong Kong University of Science and Technology.

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RECYCLING OF PHOTOVOLTAIC CELLS – A REVIEW

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ABSTRACT

The growth of solar energy has been remarkable between 2013 and 2022 on EU level, with a doubling of capacity from 80 GW to 197 GW. To recover valuable materials and mitigate environmental impact, it is imperative to have effective waste management practices in place. While recycling technologies are still in the experimental phase, the EU has put in place legal frameworks for manufacturer liability. There are continuous efforts being made to develop commercially viable technologies for the management of end-of-life PV module waste. An important part of the system of renewable sources is the establishment of a high-quality and efficient waste management system, which is contained within the concept of the circular economy. PV modules will at some point reach the end of their life cycle, which is estimated to be around 25 years, and become waste, therefore a lot of effort is currently being put into research and development of different waste treatment technologies of PV modules. They are still in the experimental phase, and few technologies become commercially available. Also, the management of waste generated at the end of the life cycle of photovoltaic technology is not yet fully regulated. However, at the level of the European Union, there is a legal framework for the liability of PV module manufacturers according to Directive on electric waste and electronic equipment. This paper is a detailed review of recycling technologies based on existing literature.

1. INTRODUCTION

1.1 Renewable energy and European Union policy

Renewable energy sources are energy sources that naturally replenish and do not deplete like fossil fuels. Renewable energy sources include hydroelectric power, wind energy, solar energy, marine energy, geothermal energy, and bioenergy. The total global capacity of renewable energy sources in 2013 was 1,567 TW, and in 2022 was 3,382 TW (IEA PVPS, 2023).

The ultimate goals of the growth share of renewable energy sources are reduction of harmful gas emissions, the impact on the environment and the prevention of resource depletion. Their use can help decrease the reliance on fossil fuels and increase energy sustainability, which will be beneficial for the environment and economy in the long run. According to (Isherwood, 2022) solar energy occupied 8,5% of the total capacity of all renewable sources in 2013, and more than 30% in 2022. Therefore, solar photovoltaic (PV) technology is widely recognised as key to achieving fully decarbonised energy generation, and as such is a vital tool in combatting climate change. Still, the production process of PV modules may involve some greenhouse gas emissions, but they do not produce any greenhouse gas emissions during their use.

Also, a new look at the energy strategy of the European Union (EU) is also a consequence of the Russian-Ukrainian war conflict that has been ongoing since February 2022. Therefore, the EC's REPowerEU plan was adopted in May 2022, where the EU proposes to end the energy dependence on Russian fossil fuels. Now it is the time for the European Union to turn to a rapid increase in renewables to establish energy independence from Russia. Hence, some member countries increased the targets for renewable electricity, introduced higher auction volumes and improved remuneration for distributed PV, reduced permitting timelines for solar and wind plants, increased grid capacity for new renewable energy projects etc. (RES Croatia, 2022).

If capacities of different types of renewable energy sources are compared in the period 2013-2022 in the EU, it can be seen a significant growth primarily in solar energy, followed by wind energy, while the other resources have almost the same capacities (Figure 1). In a period of 9 years, solar energy capacities have more than doubled, from 80 GW in 2013 to 197 GW in 2022.

It is important to emphasize that a significant part of the system of renewable sources also includes the establishment of a quality and efficient waste management system. Therefore, it is necessary to establish effective and



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sustainable waste management mechanisms and move away from the traditional practice of landfilling waste without treatment, which is contained in the circular economy of the European Union. The circular economy advocates an industrial economy that is consciously renewable, aims to use renewable energy sources, reduces, and strives to eliminate the use of chemicals and waste generation, and creates new jobs and green technology. Considering that the technologies of renewable energy sources are relatively new and that the transition is still at an early stage, the management of waste generated at the end of the life cycle (EOL) of photovoltaic (PV) technology is not yet fully regulated (Heath et al., 2020). However, all PV modules eventually reach the end of their life. Ensuring that the materials contained within modules at the end of their life are not confined to landfills has been an initial policy focus in the EU and other regions. Hence, PV modules are sorted based on the power of their installation facility and the time they entered the market, thus permitting differentiated processing according to technical and economic criteria. This uses an approach known as extended producer responsibility (EPR), which now represents one of the most important meta principles of environmental policy (Sica et al., 2018). The biggest challenge is likely to come from PV systems that were manufactured before the extended producer responsibility policy and from systems manufactured by manufacturers that no longer exist.

The EU has pioneered waste electric and electronic equipment (WEEE) regulations including PV waste specific collection, recovery, and recycling targets Directive on Waste Electrical and Electronic Equipment, according to end-of-life PV modules are hazardous waste belonging to the category of WEEE since 2012. The WEEE legislation has set up the minimum rules and targets for the dispos-

al of photovoltaic modules in EU member states. Revised Directive 2012/19/EU from 2019 suggests the minimum collection rate annually shall be 65% of the average weight of WEEE generated on the territory of that member state concerned, or alternatively 85% of WEEE generated on the territory of that member state. Minimum targets applicable to PV modules from 2018: 85% of weight shall be recovered, and 80% shall be prepared for re-use and recycled (European Union, 2012). However, every European country must transform this EU legislation into national law and can further refine or strengthen the rules behind the collection and recycling process. The main problem arising from this type of waste is the presence of environmentally toxic substances and the poor biodegradability of waste, which takes up significant space when deposited in landfills.

Currently, there is no pressure to ensure recycling capacity for photovoltaic panels since their lifespan is relatively long (25-30 years), they are still relatively new technology, and significant quantities of panels have yet to reach end-of-life stages which is anticipated by the early 2030s. . EoL processing technologies for photovoltaic panels are not yet green, but scientists and engineers around the world are working on improvements in terms of efficiency, environmental friendliness, and low-cost technology. As the quantity of waste photovoltaic panels increases, countries will have to establish their own disposal structures or hand over their photovoltaic waste to other countries, incurring high costs and without reclaiming the materials obtained through recycling. Currently, damaged photovoltaic modules covered by warranties mostly end up with the manufacturers who ensure proper disposal by handing them over to processors (Salim et al., 2019). All products should be designed and engineered for reuse after EoL. Thus, a closed loop implies a circular industrial system within a cir-

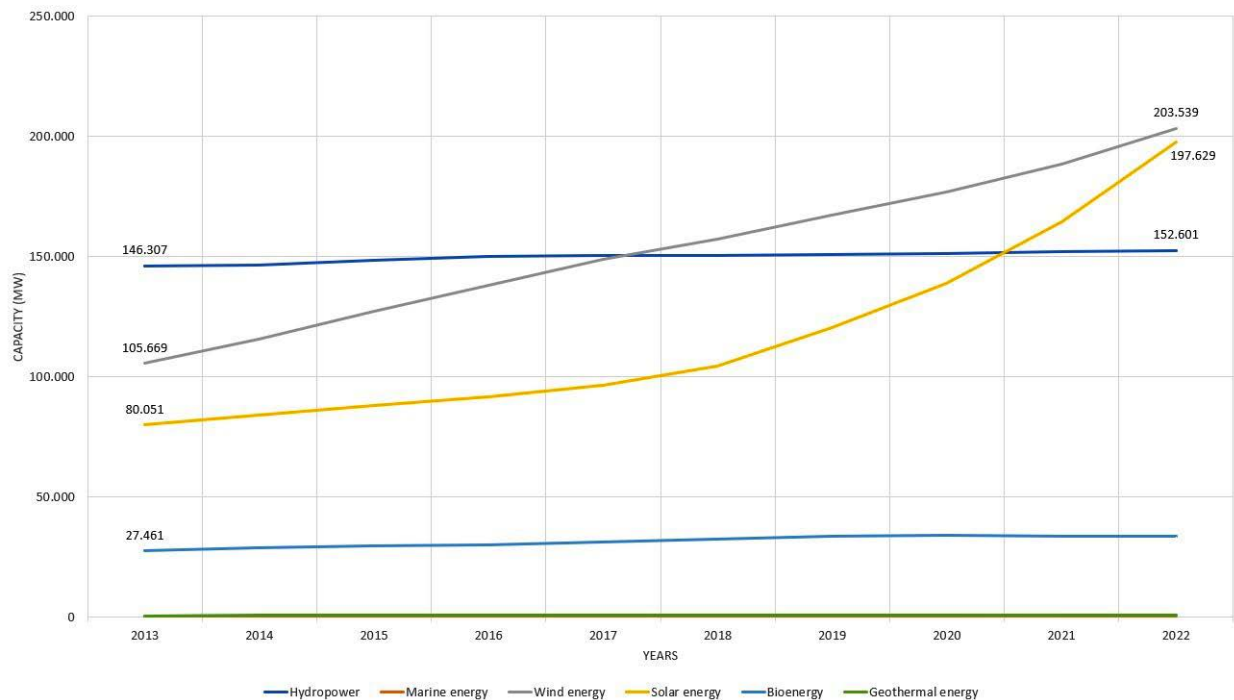


FIGURE 1: Total renewable energy capacity in the European Union 2013-2022. Data according to (IRENA, 2023).

cular economy, where the photovoltaic panel is designed in such a way that all the materials used for its production in the EoL can become a primary resource without creating waste and sacrificing quality, the whole process can be observed by up-cycling (Contreras-Lisperguer et al., 2017).

Researchers have placed focuses more on the PV waste in the countries where PV was initially installed or PV installation has kept rapid growth, such as Mexico, Italy, Spain, Australia, the OECD countries, USA, Belgium, India, and China (Song et al., 2023). Since the PV power capacity is increasing widely industrial-scale disposal or recycling is a vital economic and environmental aspect to consider (A. Omar et al., 2023).

Although waste management in accordance with the circular economy is covered, the work is focused on the review of PV module recycling. The introduction of the article provides an overview of the current installed capacity of renewable energy sources as well as solar energy. Trends in the installed capacity of individual renewable sources in the period 2013-2022 are presented. Then the shares of individual PV technologies on the market are presented. Later, a description of PV technologies is given with the specified materials used for individual components and a comparison of the technologies. The greatest emphasis is on the recycling process of PV technologies, so different recycling processes and the obtained results are listed.

1.2 Market and quantities of PV panels

PV module market can also be tracked through PV module capacity. The global picture of the growth of the capacity of PV modules is shown in Figure 2. For the period 2013-2022 an intensive growth of capacity is visible, especially for the area of Asia.

Although different types of PV cells are under development, only technologies that meet the criteria for service

life (> 15 years), efficiency (> 14%) and cost (< 0.3 USD/Wp) and that are not significantly constrained in resources can represent a major technological trend in solar energy application (Benda & Černá, 2020). This is also confirmed by the representation of each PV technology on current market. Therefore, the market share of silicon-based panels is projected to decrease to 44,8% until 2030, while third-generation PV panels are predicted to reach 44,1% in 2030 (IEA International Energy Agency; International Renewable Energy Agency, 2016).

Currently, c-Si technologies account for more than 95% of the overall cell production. Therefore, this review article focuses mostly on silicon PV module recycling techniques, although the main characteristics of other solar module technologies will be mentioned hereinafter. In 2015, technology of polycrystalline silicon PV module dominated the market with a share of 69,5%, and in 2019 monocrystalline silicon PV technology with a share of 65,5% (Figure 3). Amorphous silicon cells are dominant type of thin film solar cells and accounts about 13% of market share (Sander et al., 2007).

Regarding the estimation of PV waste, Figure 4 shows an estimation of cumulative PV waste up to 2050 for regular and early-loss scenarios on global level. The early-loss scenario gathers following assumptions, according to (IEA International Energy Agency; International Renewable Energy Agency, 2016):

- 0,5% of PV panels (by installed PV capacity in MW) is assumed to reach end-of-life because of damage during transport and installation phases.
- 0,5% of PV panels will become waste within two years due to bad installation.
- 2% will become waste after 10 years.
- 4% will become waste after 15 years due to technical failures.

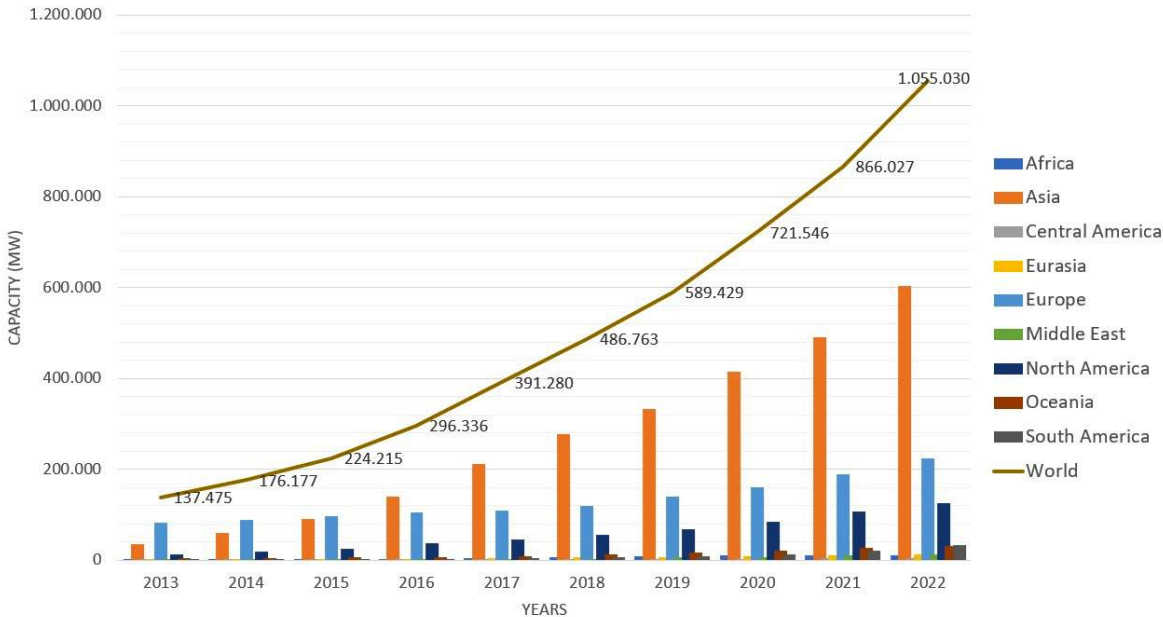


FIGURE 2: PV technology installed capacity globally 2013-2022. Data according to (IRENA, 2023).

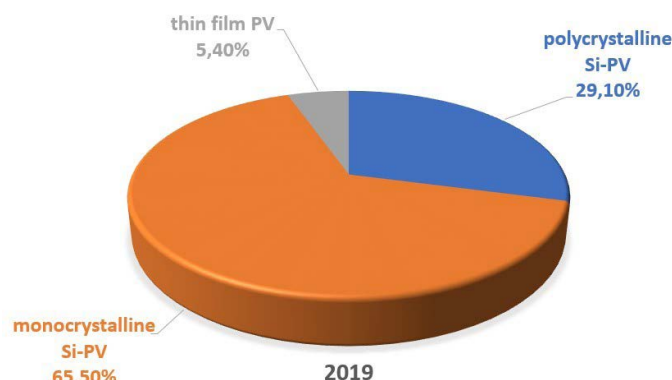


FIGURE 3: PV technology market share in 2019. Data according to (Benda & Černá, 2020).

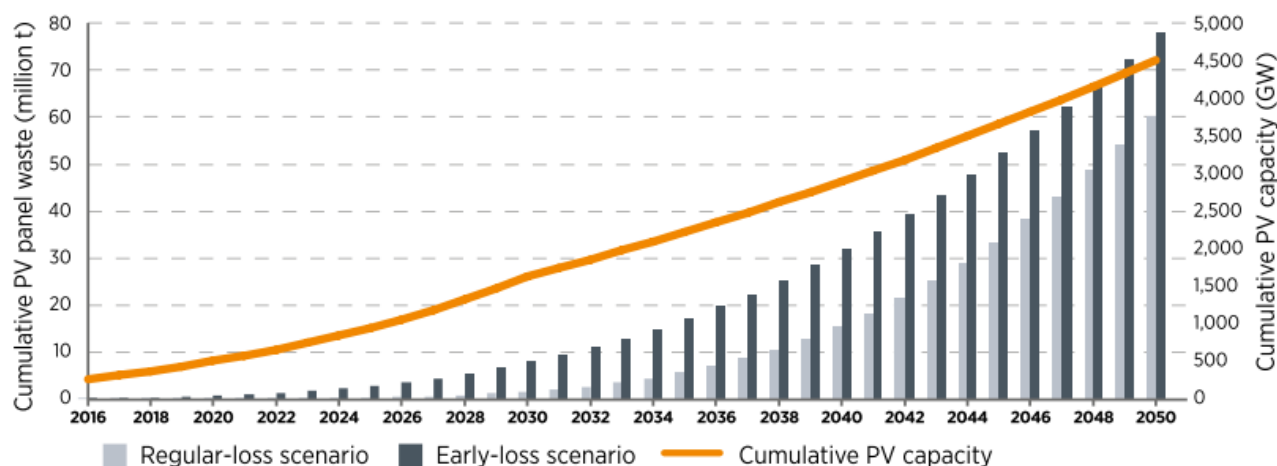


FIGURE 4: Estimated cumulative global waste (million t) of EoL PV panels. (IRENA & IEA-PVPS, 2016).

The early-loss scenario projection estimates much higher total PV waste streams, with 250.000 t alone by the end of 2016. This estimate would rise to 8 million t in 2030 and total 78 million t in 2050 on global level. The early-loss scenario assumes a higher percentage of early PV panel failure than the regular-loss scenario. In the regular-loss scenario, PV panel waste amounts to 43.500 t by the end of 2016 with an increase predicted to 1,7 million t in 2030. An even more drastic increase to approximately 60 million t could be expected by 2050 (IEA International Energy Agency; International Renewable Energy Agency, 2016).

The capacity of CPV modules in the world in 2022 was <1% of the total capacity of solar energy sources, which was 6,6 GW. A similar share of CVP module capacity was in the European Union in 2022 and amounts a little more than 1%, which was 2,3 GW (IEA PVPS, 2023).

2. TECHNOLOGY OF PHOTOVOLTAIC PANELS

2.1 Classification

Solar radiation is converted into direct current electricity by photovoltaic cell often called solar cells. When a material absorbs photons (the unit of electromagnetic radiation) with energy above a certain threshold, the photovol-

taic effect caused electrons to move within semiconductor material. Solar cell technology is based on this basic physical mechanism called photovoltaic effect (Voudoukis, 2018).

Interconnected PV modules make-up a solar panel or module, which can then be further connected into solar arrays or fields (Figure 5). It is important to note that solar panels perform best when exposed to direct sunlight, but they can still generate electricity even in cloudy weather, although in smaller quantities (Y. Xu et al., 2018). Typically, PV modules are rated from 290 W to 600 W, depending on the technology and the size (International Energy Agency, 2022).

The most used classification of solar cells is the one where they are divided according to the cell production technology. Therefore, there are three generation of photovoltaic cells presented in Table 1.

2.2 1st generation of solar panels

Wafer based crystalline silicon cells are classified as first generation of solar cells. There are two main types: monocrystalline silicon and polycrystalline silicon solar cells. Figure 6 shows wafer based crystalline silicon cell design.

Silicon is environmentally safe and the second most abundant chemical element in the Earth's crust. However, it

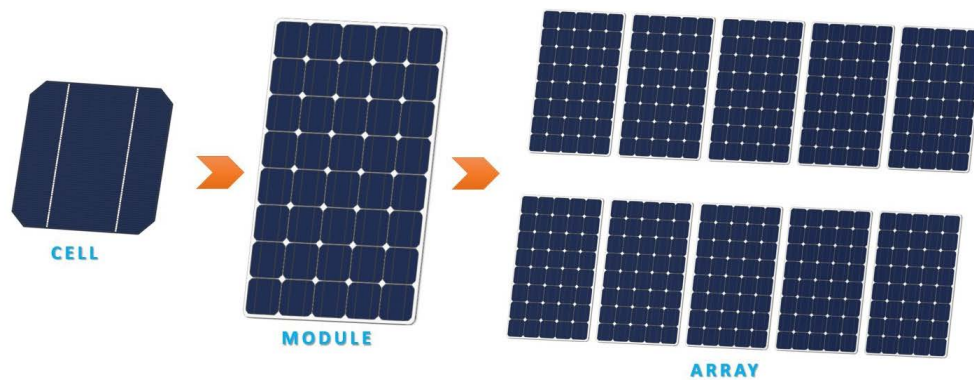


FIGURE 5: PV cell - module - array.

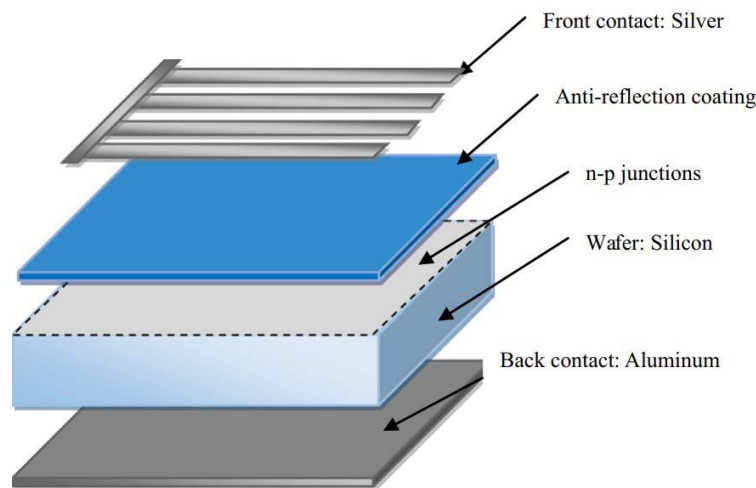


FIGURE 6: Wafer based crystalline silicon cells design (Tao & Yu, 2015).

is mainly found in the form of silicon dioxide and is therefore not suitable for direct use in photovoltaic modules. A significant portion of the costs associated with silicon goes to its extraction and processing to obtain a clean and uniform structure (Kumar et al., 2020).

TABLE 1: Most used classification of solar cells.

1st generation	Wafer-based crystalline silicon cells	Monocrystalline silicon (mono c-Si) solar cells Polycrystalline silicon (poly c-Si) solar cells
2nd generation	Single junction thin film cells	Amorphous silicon (a-Si) and Micromorph silicon (μ -Si) thin film solar cells Cadmium Telluride (CdTe) thin film solar cells Copper-indium-gallium-diselenide (CIGS) and Copper-indium-diselenide (CIS) solar cells
3rd generation	Devices with potential efficiency above the Shockley-Queisser limit (tandem cells) and emerging technologies using new materials	Dye sensitized (DSSC) solar cells or light absorbing dye* Concentrated solar cells Perovskite solar cells Quantum dot solar cells

* Also called organic cells

Only a few manufacturers control the whole value chain from quartz to solar cells. After production of 100% pure silicon rods, boron and phosphorus are added using a diffusion machine. Electrical contacts are then placed to connect each solar cell to another as well as to the junction box. The thinner the electrical contacts, the more sunlight the cell can absorb. After that, tin-coated copper strips are placed between the cells. Because pure silicon is still too shiny and can reflect up to one-third of sunlight, an anti-reflective coating is applied to the cell. Anti-reflective coatings are made of an oxide such as tantalum pentoxide (Ta_2O_5), titanium dioxide (TiO_2), silicon monoxide (SiO), silicon dioxide (SiO_2), silicon nitride (Si_3N_4), aluminium oxide (Al_2O_3), ITO (indium-tin-oxide), tin oxide (SnO_2). After these steps, the solar cells are usually sealed in ethylene vinyl acetate copolymer (EVA), placed in a frame that is sealed with silicone adhesive or polybutylene sealants and covered with a mylar back and a glass or plastic cover on the front. Other materials such as thermoplastic polyurethane (TPU), polyvinyl butyral (PVB) and silicone can also be used as encapsulating materials, the function of which is to bond the various layers and components together. The back layer or back sheet is usually made of Tedlar (trade name of PVF – polyvinyl fluoride) based or PET (polyethylene terephthalate) based. Chemicals and solvents like hydrochloric

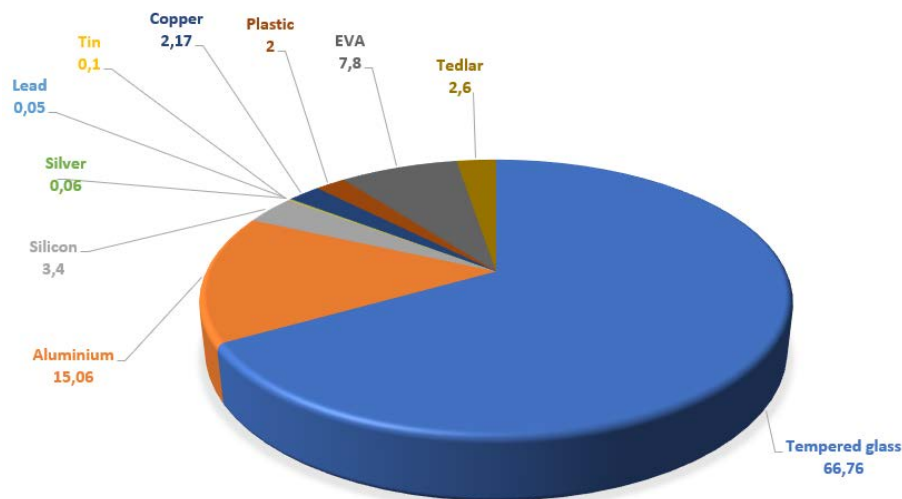


FIGURE 7: Average mass composition according to materials for c-Si module.

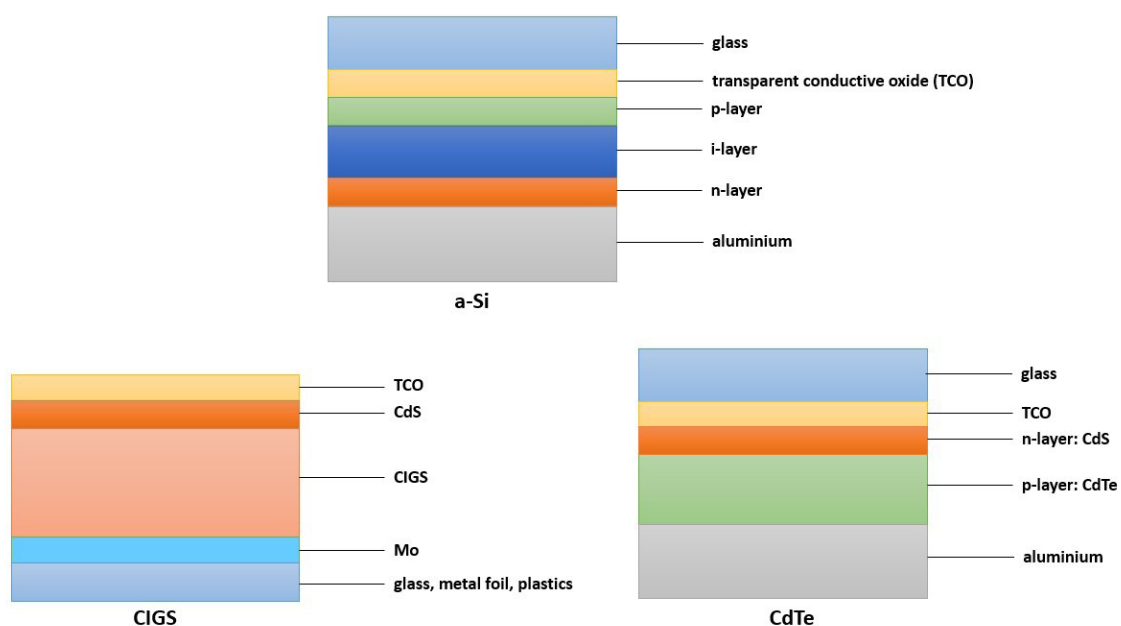


FIGURE 8: Thin-film cells (a-Si, CIGS, CdTe) design.

acid, nitric acid, hydrogen fluoride, acetone and ethanol are used for cleaning wafers and removing impurities during the fabrication process (MSK Instruments, 2024).

A typical crystalline PV panel with aluminium frame and 60 cells has a capacity of 270 Watt peak (Wp) and weights 18,6 kg (IEA International Energy Agency; International Renewable Energy Agency, 2016). When the average of the specified ranges is calculated, a pie chart with a grand total of 100% is obtained (Figure 7).

2.3 2nd generation of solar panels

Thin-film PVs are classified as the second generation of solar systems. Commonly used thin film solar cells are amorphous silicon (a-Si), micromorph silicon (μ -Si), cadmium telluride (CdTe) and copper-indium-gallium-diselenide (CIGS) and copper-indium-diselenide (CIS) thin film solar

cells. Some of the varieties of a-Si modules are amorphous silicon carbide (a-SiC), amorphous silicon germanium (a-SiGe) and amorphous silicon-nitride (a-SiN) (IEA International Energy Agency; International Renewable Energy Agency, 2016).

A thin-film module consists of several solar cells that are wired together, as shown in Figure 8. The module consists of thin semiconducting layer (CdTe, CIGS or a-Si) obtained using chemical vapor deposition techniques and a substrate - glass, polymer, or metal. Laser-scribing is used to delineate individual cells and make electrical connections between cells. Materials also used for thin-film PV cells are gallium arsenide (GaAs), copper and titanium dioxide (TiO₂). The typical structure of transparent conducting oxide (TCO) layer consists either of tin oxide (SnO₂) and indium tin oxide (ITO), or of zinc oxide (ZnO) (Savvilitidou

et al., 2017). The thickness of thin film cells varies from 80 nm – 0,5 μm (Dhilipan et al., 2022).

For a standard CdTe panel, 110 Wp can be assumed on average for 12 kg. A CIGS panel usually holds a capacity of 160 Wp and 20 kg (IEA International Energy Agency; International Renewable Energy Agency, 2016).

Although there is no toxic material in the final product of a-Si modules, the manufacturing processes do involve toxic and pyrophoric gasses such as germane, phosphorus, trimethylborane, silane, hydrogen and so on (X. Deng & Schiff, n.d.).

2.4 3rd generation of solar panels

This is a generation that is still in the development and laboratory testing phase. Devices with potential efficiency above the Shockley-Queisser (S-Q) limit and emerging technologies using new materials are classified as third generation of solar cells. The S-Q model describes the efficiency limit for a single-junction solar cell under one-sun illumination. Efficiency beyond the S-Q limit can potentially be achieved for a single-junction solar cell by using the process of multiple exciton generation (e.g. in quantum dots), up or down conversion of incident light, or limiting radiative emission (raising the cell voltage (Polman et al., 2016). Theoretically, tandem modules composed of several single-junction cells can be used for efficiency above the Q-S limit. Some third generation PV modules are listed below.

Concentrating PV (CPV) module is a complex system that uses mirrors and optics to collect high levels of sunlight to concentrate incoming radiation using collectors and mirrors. They can be classified into three types according to the semiconductor material and the type of tracker used: high concentrator PV (HCPV) modules, low concentrator PV (LCPV) modules and compound parabolic concentrators (CPC). HCPV modules are made of multijunction cells with semiconductor alloys which contains elements from groups III and IV in the periodic table and are built with double-axis sun trackers. LCPV modules are composed of c-Si or other semiconductor materials and are built with double or single-axis sun trackers. Unlike the previous two types of CPV, CPCs are designed without sun trackers and are concave in shape so that the ratio of reflection of incident light to the absorber is very high.

The cost of HCPV and LCPV modules is very high, which makes them inaccessible for conventional PV application. Hence, they are mostly used for space application (Kumar et al., 2020).

Organic cells use dye or organic semiconductors as the active light-harvesting layer. These are photochemical cells that are based on chemical processes to produce electricity from light, not using standard semiconductors. Photoelectrochemical cells include dye sensitized (Grätzel) cells and polymer (plastic) cells. This technology has been researched a lot over the past few years and is currently the fastest growing solar technology. Despite the low production costs, stable products are not yet available on the market due to corrosion problems due to exposure to water (Polman et al., 2016).

Quantum dot solar cells are single-junction designs that use colloidal quantum dots (CQDs) with lead sulphide

(PbS) that have a bandgap that can be tuned into the far infrared, frequencies that are typically difficult to achieve with traditional solar cells. Half of the solar energy reaching the Earth is in the infrared, most in the near infrared region. Moreover, CQDs offer easy synthesis and preparation. While suspended in a colloidal liquid form they can be easily handled throughout production, with a chimney as the most complex equipment required. CQDs are typically synthesized in small batches but can be mass-produced (Polman et al., 2016).

Perovskite solar cells are derived from the ABX₃ crystal structure of absorber materials, which is called a perovskite structure, where A and B are cations and X is an anion. In 2021, the perovskite solar cell achieved an efficiency of 28% in the silicon-based tandem and 23,26% in the CIGS-based tandem (International Energy Agency, 2022). Although perovskite solar cells have become very efficient in a very short time, there are still many challenges before they become a competitive commercial technology. For example, they degrade faster than silicon solar panels, especially in a harsh environment, but their methods of production are low-cost.

2.5 Comparison of PV technologies

Photovoltaic panels directly convert solar energy into electricity without using mechanical moving parts that are prone to wear and tear. They do not contain liquids or gases that can unexpectedly escape from the production facility, and they do not pollute the environment (including thermal pollution). They do not require additional fuel or auxiliary energy sources to function and have a fast response time, reaching full power almost instantly. Due to their modular nature, they have a wide range of applications, from small standalone units with just a few watts of power to autonomous power sources of several kilowatts, and even centralized power plants in the megawatt range. They can be located wherever electrical energy is needed. However, they do not produce electricity continuously, so grid storage systems such as batteries or grid connections are required (Hossein Karami Lakeh, 2024).

Thin-film modules have lower power and durability than the first-generation c-Si solar panels (Kumar et al., 2020). However, a-Si modules use less raw material than crystalline silicon modules. Amorphous-Si PV modules are light, flexible and radiation resistant, making them desirable for energy and space applications. Still, they are more appropriate in micro-power applications such as a solar-powered calculator and high-temperature environments hence they are less sensitive to temperature than c-Si solar cells (H. Kang, 2021). Previously, a-Si and μ -Si had a significant market share but failed to keep pace with the cost of crystalline silicon cells and the increasing efficiency of other thin film technologies (IEA International Energy Agency; International Renewable Energy Agency, 2016).

In terms of module efficiency, crystalline silicon solar panels have the highest efficiency, which is confirmed by their wide use. Silicon has a band gap energy of 1.1 eV, meaning it can absorb photons with an energy of up to 1100 nm. Hence, most used PV cells utilize silicon var-

iations altered by doping process to make them suitable semiconductors (Ghazy et al., 2021).

PERC (Passivated Emitter Rear Contact) solar cells are designed with an extra layer on the bottom of the solar cell which adds an additional 5% efficiency. This additional layer is called a dielectric passivation layer which has become the dominant commercial technology. An additional dielectric passivation layer reduces electron recombination, which affects the increase in electric current production. It also increases the solar cell's ability to capture light due to the layer's reflective properties, so photons have more opportunities to generate electricity (backside passivation). And finally, PERC reflects wavelengths above 1180 nm. In the conventional Si-cell, such wavelengths are easily absorbed by backside metallization layer and converted into heat. Longer wavelengths are especially present during the morning and evening or during cloudy days (Ghazy et al., 2021).

Monocrystalline panels have higher energy efficiency and have gained the largest market share in the last few years, 85% of c-Si (International Energy Agency, 2022). The difference in efficiency depends on the TCO performance, semiconductor material quality, deposition uniformity, encapsulation loss, busbar shadow loss and electrical loss.

Regarding module efficiency, (Luo et al., 2017) reviewed potential induced degradation (PID) of c-Si and thin-film PV modules. The progression of PIDs in c-Si modules depends on the applied voltage, humidity, and temperature. Four types of test methods are available to assess cell and module level PID sensitivity. At the module level, chamber PID and AI PID tests are often used, while at the cell level, PID sensitivity can be tested by corona discharge PID set-up or bias voltage application on a module-like layer stack.

According to the prediction shown in Figure 9, by 2030 there will be a decrease in the mass share of materials in the PV module. This affects the reduction of production costs, resource conservation, reduced use of rare and hazardous substances, increased module efficiency and lower

module prices. However, the thinner the wafer, the more fragile it is and therefore breaks more easily during the recycling process, so if you want to preserve an intact wafer, the recycling process becomes difficult.

In the manufacturing of silicon PV modules, the production costs of monocrystalline and polycrystalline silicon PV cells are reduced. Therefore, material consumption for silicon cells has decreased significantly from about 16 grams/Wp in 2006 to less than 4 grams/Wp in 2016 (IEA International Energy Agency; International Renewable Energy Agency, 2016).

PV module manufacturers have introduced a wide variety of advanced module technologies to the market with higher electrical power output, more weather resistance and lower cost. So, there are frameless modules that effectively eliminate module current leakage, glass-glass modules replace the back sheet with heat-strengthened glass to ensure high performance in harsh environments, bifacial modules benefit from additional power generated from the rear, advanced module bonding reduces copper ribbon and lead solder of tandem modules for higher efficiency. Therefore, bifacial photovoltaic technology is considered a promising alternative, as it can generate more energy than conventional monofacial PV technology by absorbing sunlight from both sides. Compared to monofacial PV, bifacial PV module has an additional layer of anti-reflective coating and back contacts instead of a back surface on the back of the PV module. Moreover, to increase efficiency, there is module technology in tandem design (Gu et al., 2020). However, the cost of this technology prevents their widespread commercial use, (Nanayakkara et al., 2017) found that the cost of both tandem module structures (CdTe/CIS and CGIS/CIS) on an equal-area basis is approximately 30 USD/m² higher than single-junction alternatives, although their efficiency is 4% higher.

Except for the third generation of cells, c-Si panels have the lowest mass in relation to the energy produced (0,102

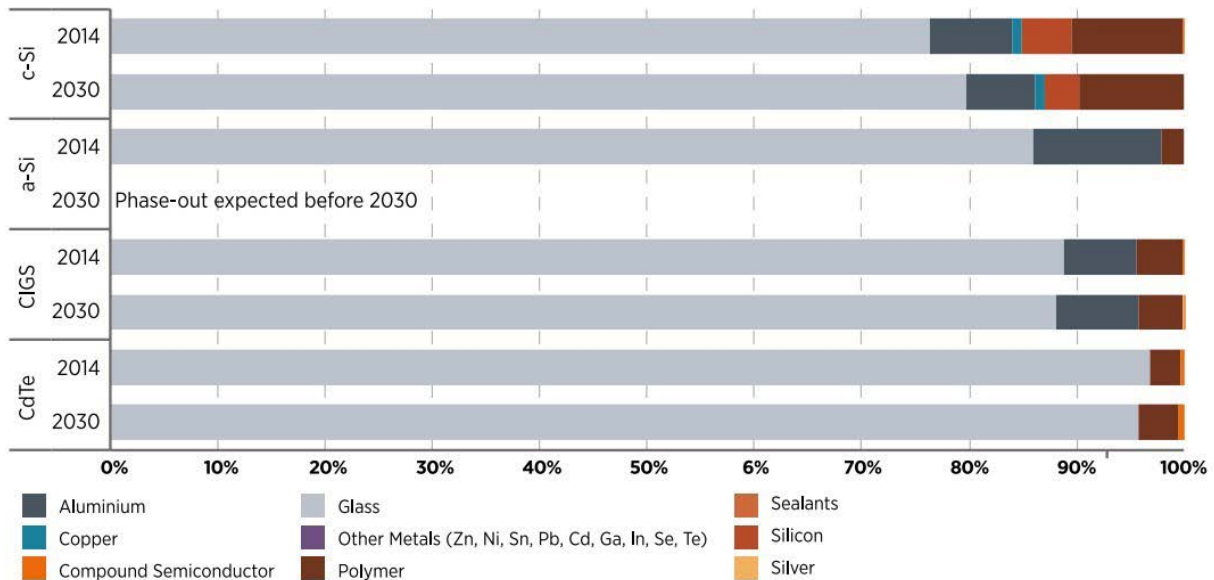


FIGURE 9: Total renewable energy capacity in the European Union 2013-2022. Data according to (IRENA, 2023).

kg/Wp), which is another reason why c-Si panels dominate the market (Divya et al., 2023).

3. EOL PV MANAGEMENT & CIRCULAR ECONOMY

As already emphasized earlier, the European Union provides a legislative framework for the extended responsibility of PV module manufactures at the European level through the Directive on waste electrical and electronic equipment (European Union, 2012). End-of-life management works on the principle of extended producer responsibility, which means that producers must be physically and financially responsible for the environmental consequences of their products until the end of their life cycle. This approach encourages the creation of more environmentally friendly products with a reduced environmental impact. Moreover, the EPR can be used to establish funds to support the financing of efficient collection, treatment, recycling, and disposal systems. Although producers bear the cost of waste management, these additional costs can, unfortunately, still be passed on to consumers, leading to higher product prices.

However, the concept of a circular economy involves the reducing waste and closing the loops within industrial ecosystems. This is achieved by transforming goods at the end of their life into valuable resources (A. Omar et al., 2023). An appropriate EoL management strategy for solar photovoltaic systems is crucial, not only to prevent and/or mitigate future environmental issues but also to reduce the demand for rare materials (Salim et al., 2019). Waste management of PV modules at the end of their life offers options related to each of the 3Rs of sustainable waste management: Reduce, Reuse, Recycle. The solar panel industry is indeed developing rapidly, and the installed capacities are increasing every day, which makes it almost impossible to completely prevent or reduce the generation of waste at the end of their life cycle. Therefore, the focus should be on proper waste management in other segments of the waste hierarchy: reuse, recycling, energy recovery and, the last resort, disposal. The disposal option creates additional costs, and it does not recover the intrinsic values of the materials present in the PV modules (Lunardi et al., 2018). Thus, (Tao & Yu, 2015) provided an overview of commercially available PV modules and their waste management pathways that include waste recycling, remanufacturing, and reuse. The current design of PV modules makes it difficult not only to repair and maintain modules, reuse, and remanufacture components, but also limits the possibilities of recovering secondary materials (Fiandra et al., 2023). (Masoumian & Kopacek, 2015) pointed out that reuse refers to the further use of a worn or damaged PV module or its parts. The reasonableness of such reuse is strongly related to the expected lifetime of the module or part of a module. Some of the materials used to manufacture PV modules will either be disposed of or incinerated as they can neither be recycled nor reused (Contreras-Lispeguier et al., 2017). However, it has been estimated that about two-thirds of these PV modules can be repaired or refurbished, and only about 10% of PV modules are recy-

cled worldwide (Lunardi et al., 2018). Therefore, approximately 50% of PV waste can be diverted from the recycling stream. The ratio is likely to be even higher as functional photovoltaic modules that are retired also enter the waste stream. A report by (IEA International Energy Agency; International Renewable Energy Agency, 2016) estimates that up to 80% of the PV waste stream will consist of products with premature failures in next 10 years, such as manufacturing defects or damages caused during transportation.

During the production of PV modules, certain hazardous and toxic substances such as Cd and Pb are used in small quantities. These substances will be present even at the end of their lifetime, so this should be considered to avoid their harmful effects on people and the environment (Rathore & Panwar, 2022). Rare materials such as ruthenium, gallium, indium, and tellurium are essential components of some type of photovoltaic panels. Some materials contained in the PV modules are classified as hazardous. The main concern regards the presence of cadmium in thin film module, and of lead and chromium in crystalline modules (Tammaro et al., 2015). If special requirements for handling and disposal are not implemented, these hazardous substances could potentially be released into the environment at the end of the panels' life. Therefore, this transitional period should be utilized for financing and developing efficient and environmentally friendly recycling technologies to maximize the recycling of materials contained in solar modules in the future (Salim et al., 2019).

Although standards exist today to produce PV modules, there are only limited insights and hardly almost no standards on the characterization, reliability testing, certification or labelling for second-life PV modules (Tsanakas et al., 2020).

4. REVIEW OF PV PANEL RECYCLING METHODS

After increasing the installed capacity of PV modules, more and more attention is being paid to the end of life of PV modules, especially in the EU. As stated earlier, the minimum targets applicable to PV modules from 2018 are that 85% of the weight must be recovered and 80% prepared for reuse and recycled. This requires improvements of the recycling process because downcycling is no longer efficient enough. Despite the great potential interest of researchers in PV recycling, primary data on PV recycling is still very scarce (Ardente et al., 2019).

In this section, a detailed review of recycling technology is made according to the literature regarding mathematical modelling, software simulation, experimental tests, and practical application. There are different methods of evaluating the recycling process for different types of PV modules through energy, economic and environment/social analyses that can be confirmed by laboratory tests, pilot projects or commercial technology. (Lunardi et al., 2018) present a review of the recycling process of PV modules using examples of pilot and commercially used recycling procedures. A large part of these analyses remains only in the research and development phase, and a smaller part of the recycling process becomes commercial. There are

several different review articles. Some of them deal with a systematic quantitative literature review on the EoL of PV panels (Mahmoudi et al., 2019), some review the recycling status and different recycling methods of solar panels (Divya et al., 2023; Isherwood, 2022), the environmental impact (De Sousa et al., 2023; Divya et al., 2023b; Rathore & Panwar, 2022), or recycling policy (Chowdhury et al., 2020). (McDonald & Pearce, 2010) emphasized that the economic motivation for recycling most PV modules is unfavourable without appropriate policy. (Seo et al., 2021) concluded that the recyclability of c-Si PV panels is guaranteed by recycling technologies that can reduce the environmental burden. However, to promote the recycling industry, it is necessary to ensure a sufficient level of value-added opportunity.

With the development of the third generation of photovoltaic modules, whose goal is to increase the efficiency of the modules, tandem or multi-junction p-n modules appear, which require a more complex recycling process since they consist of several single-junction solar modules. The recyclability of PV module components can be determined by the materials from which these components are made (Figure 10). Recycling procedures for c-Si PV panels differ from those applied to thin-film PV panels due to their different module designs. There are also precious and rare metals such as silver (Ag), gallium (Ga), indium (In), germanium (Ge), cadmium (Cd) and tellurium (Te). As for thin-film panels, they use less material than c-Si panels, but there are concerns about the availability and toxicity of materials such as tellurium (Te), indium (In), and cadmium (Cd) (Savvilitidou et al., 2017). Tellurium has a harmful effect on the liver, while cadmium can cause lung cancer, and affects bones and kidneys. Indium compounds damage the heart, kidney, and liver, and may be teratogenic. Conventional ma-

terials in PV panels are aluminium (Al), copper (Cu) and glass, as well as high-purity energy-intensive material such as the silicon (Si). The great economic importance of silicon for Europe, together with the high supply risk, justified its inclusion in the list of Critical Raw Materials (CRMs) of the EU. The main global producers of silicon are China (66%), the USA (8%), Norway (6%) and France (4%), and the main sourcing countries are Norway (30%), France (20%), China (11%), Germany and Spain (both 6%) (Critical Raw Materials Resilience: Charting a Path towards Greater Security and Sustainability, 2020). Glass and aluminium production are also energy-intensive processes. Therefore, avoiding the disposal of these materials in landfills would contribute to significant environmental gains (Molano et al., 2022).

Figure 10 shows the typical structure of a Si-based PV module. Recycling of photovoltaic panels is a process that involves collecting, disassembling, and processing the various components of the panels for reuse. This presents a significant challenge, especially with the laminate encapsulant (typically EVA) used in PV modules, due to the high temperature processes (requiring significant energy consumption), the use of solvents, and treatment of fumes during encapsulate removal (Pagnanelli et al., 2016). Different materials for encapsulants such as PVB and EVA, or some unspecified materials can affect the behaviour of the module during the recycling process (S. Kang et al., 2012). Another important fact is that not all materials can be recycled, for example the EVA layer in the module has hardly been recycled so far. . Despite being a thermoplastic, the amount of vinyl acetate can cause the polymer to cross-link, which in turn makes its recycling unfeasible. Removing polymers using high temperatures will normally generate atmospheric emission that might be toxic according to the method and the polymers. The use of organic solvents

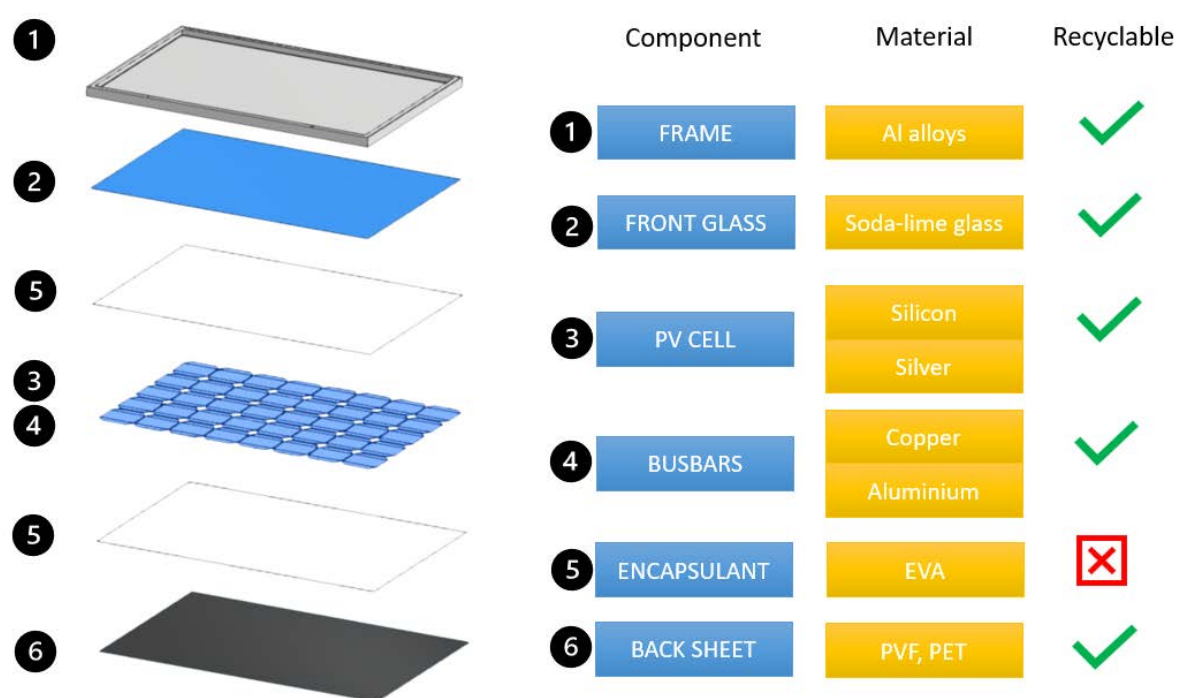


FIGURE 10: Recyclability of c-Si PV module's components.

can assist in the component separation phase with objective to separate or remove the encapsulant. This is most complex step of recycling PV modules. In industrial application, the use of organic solvents is hardly feasible because it requires a closed process (P. Dias & Veit, 2018). In contrast, recycling the aluminium frame, glass and plastic back is not technologically demanding. (Zhang et al., 2013) showed after testing that thermal separation, from an economic and ecological point of view, is a more favourable alternative compared to chemical processes that require the use of expensive and toxic reagents.

Various physical, mechanical, chemical, and thermal methods can be applied to recycle PV modules, depending on the composition of the module back sheet, but these methods often do not allow full recovery of all PV panel components (Chitra et al., 2020; Sica et al., 2018). Moreover, the procedures that can be used during processing and recycling of PV modules can be divided according to Table 2 with the corresponding references from the literature for a certain type of PV technology.

4.1 Experimental study and real system operation

4.1.1 PV Cycle

PV Cycle was the first to establish a downstream PV module recycling process and waste logistics across the EU (Tsanakas et al., 2020). This recycling process of c-Si PV modules achieved a recycling rate of 96% in 2016 (Lunardi et al., 2018). The process starts with the separation of the aluminium frame and junction box of PV module. This is followed by mechanical processes (shredding and sorting by laser or vibration) that are used to extract the remaining module materials. Recovered materials are glass, aluminium, and silicon.

4.1.2 FRELP

In the early stage of research and development/pilot, Sasil, in Italy, within the framework of the EU LIFE project FRELP (Full Recovery End of Life Photovoltaic process), investigated a new PV recycling process line. The project did not achieve any of the set goals. However, partial results have been achieved with the design of a prototype for sep-

TABLE 2: List of procedures used during processing and recycling of PV modules.

Recycling procedure	Subprocess	Reference for c-Si modules	Reference for thin-film modules
Physical/mechanical	Cryogenic treatment	(Zhang et al., 2013)	
	Shredding/milling	(Azeumo et al., 2019; Fiandra et al., 2019; Granata et al., 2014)	(Berger et al., 2010)
	Cutting		
	Disassembling (physical disintegration)	(Azeumo et al., 2019; Belançon et al., 2022; Fiandra et al., 2019; Granata et al., 2014; Huang et al., 2017)	(Granata et al., 2014; Savvilitidou et al., 2017)
	Dry and wet mechanical processes: blasting, scribing, attrition, sieving, floatation	(Azeumo et al., 2019)	(Berger et al., 2010)
	Density separation	(Azeumo et al., 2019)	
	Electrostatic separation		
	Magnetic separation		
	Separation according different physical or chemical properties	(Lim et al., 2022)	
Chemical	Acid/base treatment	(Doi et al., 2001)	
	Solvent treatment	(X. Xu et al., 2021)	(Savvilitidou et al., 2017)
	Combination of organic solvents and heat treatment by an assisted ultrasonic method to improve the dissolution rate of the EVA	(Azeumo et al., 2019; Kim & Lee, 2012)	
	Chemical etching	(Cui et al., 2022; Huang et al., 2017; S. Kang et al., 2012; Klugmann-Radziemska & Ostrowski, 2010; Lim et al., 2022; X. Xu et al., 2021)	
	Precipitation		
	Hydrometallurgy: leaching	(Cerchier et al., 2021)	
Thermal	Pyrolysis		
	Burning	(Cui et al., 2022; Huang et al., 2017; S. Kang et al., 2012; Lim et al., 2022; Tammaro et al., 2015; Wang et al., 2012; X. Xu et al., 2021)	(Granata et al., 2014)
	Pyrometallurgy		
Electrometallurgical	Electrowinning	(Cui et al., 2022; Huang et al., 2017)	
	Electrolysis		

arating the EVA polymer back sheet from the glass and a robotic system for separating and restoring the aluminium frame and connectors from PV panels (LIFE Programme, 2016). The FREL process is a combination of physical procedures, acid leaching and electrolysis where c-Si solar panels are economically justified in terms of waste disposal. The process is designed to increase the mass of the recovered material while reducing waste and considers the stages of transport and discharge, disassembly, cable treatment, glass separation, glass refining, cutting, sieving, acid leaching, filtration, electrolysis, neutralization, filtering, and incineration as well as waste disposal in sanitary landfills of hazardous waste and disposal of urban solid waste. A pilot project was implemented that processes around 1.300 modules per day (Cui et al., 2022), i.e. with a processing capacity of 1 t/h to 8000 t/year of waste c-Si PV panels (Latunussa et al., 2016). Expectations from that pilot project were high, which is confirmed by numerous LCA analyses. (Ardenete et al., 2019) estimated a recycling rate for the FREL process of 83%. Recovery rate for the FREL process: insulated cables (100%), silver (94%) at 99% purity, copper (97%) at 99,993% purity, aluminium scrap (99,4%), silicon (97%) at metallurgical grade purity, glass cullet (98%) at 98% purity (Cui et al., 2022; Latunussa et al., 2016). The pilot project of FREL process was interrupted in spring 2016 because of the small volume of waste panels currently collected. Secondly, mass incineration of large amounts of PV scraps with fluorinated plastics could become problematic for the plant which would risk exceeding legal limits for HF emissions. Finally, collection of the bottom ashes can be technically difficult and characterised by potential losses.

Some companies are working to formalize the recycling, repair/refurbishment and reuse segment in PV value chains and propose customized technical standards and regularity framework for EoL of PV modules.

4.1.3 ReSiELP

ReSiELP was a European project that aimed at recovery of valuable materials (aluminum, glass, copper, silicon, and silver) from EoL silicon PV modules. The recovery of silicon was carried out through thermo - mechanical and hydrometallurgical purification. Pilot plant was designed and built to upscale recycling technology to TRL 7, with a 1500 panels/year capacity by the group of metallurgy of Padova University (Cerchier et al., 2021). PV glass reuse for building materials production, sustainable mortar and concrete formulations were studied and seemed comparable with commercial products (Cerchier et al., 2022).

4.1.4 Veolia

In 2018, Veolia has opened their European PV module mechanical recycling plant in Southern France (in contract with PV CYCLE France). With a 95% recovery ratio, their recycling process isolates all PV component materials, from the glass to the aluminium frame, junction box and connection cables. The plant capacity is from 1.800 t in 2018 and will grow to 4.000 t. (Veolia Group, 2017). Recovered materials are then directed to various industries. The aluminium frame material is directed towards aluminium-based

refinery; polymeric materials are recycled to fuel used in cement industry; silicon cells to precious metal sectors; cables and connectors are crushed and sold in the form of copper shots; whereas glass, recovered as clean cullet, is used in the glass manufacturing industry. The glass recovered from such PV recycling process gets eventually contaminated with iron (Fe) due to the involved shredding process, and therefore cannot be considered as a low-Fe (i.e. high value) glass product (Tsanakas et al., 2020).

4.1.5 Solar World

Solar World is another pioneer in PV recycling c-Si panels based on the thermal treatment method, with which EVA is eliminated by burning at 600°C, followed by manual separation of metal, silicon cells and glass. The silicon cells are then etched again, and at the end of such a process the clean wafers can be reused, while the glass and metal are sent for recycling (Tao & Yu, 2015b). Solar World's recovery ratios typically exceeded 84% of the module weight, namely, 90% of the glass and 95% of the semiconductor materials. Solar World and its recycling subsidiary are no longer in business (Lunardi et al., 2018).

4.1.6 Loser Chemie

On the other hand, Loser Chemie's recycling process for c-Si PV starts with crushing and mechanical separation. Afterwards, chemical treatment is performed, followed by solid-liquid separation. As a last step of the recycling process, glass is chemically treated, whereas aluminium-based metallization is recovered. (Tsanakas et al., 2020) In 2019, they won an award for their Aqualux project, which uses water-jet technology to break down silicon-based PV modules into their individual parts. In doing so, the individual layers of the complex composite are separated using a special nozzle technique. Therefore, it is possible to detach the Tedlar and at the same time to obtain an EVA silicon cell mixture in which the former silicon cells with their contacts are present. Connecting lines containing copper are very clean and can be directly recycled. To separate the EVA from the silicon cell, they use an aluminium chloride solution to dissolve the aluminium back contact, destroying the bond between the silicon cell and the plastic. The silver is not lost by this process and can be recovered and poured into silver ingots by the hydrometallurgical process (PES Solar, n.d.).

4.1.7 Japan

In Japan, NPC Group has developed a recycling process for c-Si PV modules that can recover the glass without any destructive process. Recycling involves automatic separation of aluminium frames after PV module processing using a hot knife/cutter blade. Glass, solar cells matrix and EVA sheet can be directed to further recycling (Tsanakas et al., 2020).

4.1.8 China

In China, on the other hand, a PV recycling pilot plant recycles c-Si PV waste using cryogenic abrasive machining at -196°C and electrostatic separation. The mixture after abrasive machining, which contains EVA particles, Tedlar

particles and a mixture of silicon, silver and aluminium powder is separated by electrostatic separation. Mechanical processing does not involve a chemical process but cannot recycle silicon for reprocessing into new wafers due to its low purity (Zhang et al., 2013).

In the thin film PV segment, First Solar and ANTEC Solar are at the recycling technology forefront. They operate an advanced downstream program encompassing collection, transportation, and recycling of their CdTe modules (Tsanakas et al., 2020).

4.1.9 First Solar

Approximately 90% by mass of each collected First Solar PV module is recycled into new products (Tao & Yu, 2015). The EoL PV modules are shredded, and hammer milled into pieces small enough to break lamination bonds. Then, semiconductor films are removed by the leaching process with addition of acid and hydrogen peroxide in a slowly rotating, stainless steel drum. The remaining amount in the drum is then passed down to a classifier which separated the solid (including glass and laminate materials) from the metal-rich liquid. A vibrating screen separates the glass from the larger pieces to laminate material. The glass is further cleaned to remove any residual semiconductor material and then packaged for recycling. Meanwhile, the metal-rich liquid is pumped to the precipitation unit. The metal compounds are precipitated in three stages at increasing pH and then concentrated in a thickening tank. The resulting unrefined metal-rich material is packaged for processing by a third party to produce tellurium and cadmium products. This commercial process can recover 90% of the glass for use in new products and 95% of the semiconductor materials for use in new solar modules. First Solar is still operating and according to its web page recycle CdTe semiconductors (First Solar, 2024)

4.1.10 ANTEC Solar

ANTEC Solar has established a pilot plant based on the recycling process of thin-film photovoltaic modules CdTe/CdS. In this process, the modules are first physically dismantled into fragments. Subsequently, the module fragments are exposed to an oxygen-containing atmosphere at a temperature of at least 300°C, causing the pyrolysis of EVA. After that, the module fragments are exposed to a gas atmosphere containing chlorine at a temperature higher than 400°C, resulting in the etching process, where CdCl_2 and TeCl_4 are generated. The produced CdCl_2 and TeCl_4 are then condensed and deposited through cooling (Tao & Yu, 2015).

4.1.11 EVERP

Currently there is ongoing EVERP project funded by European Commission (HORIZON-CL5-2022-D3-03). The aim of project is to demonstrate two innovative technologies to delaminate the different layers of the PV panel. Through combination with recycling processes it will be possible to recover glass with less than 1% impurities, encapsulant and backsheet polymers with a purity over 99%, and silver with a purity of 99% (EVERP, 2024).

4.1.12 PHOTORAMA

Project PHOTORAMA was funded by European Commission (Horizon 2020) and lead by Italian research agency Enea and Italy-based heterojunction PV manufacturer Enel Green Power. Its objectives is a pilot line with established all needed interrelated successive steps: auto disassembly, smart separation and innovative recovery of metals from solar cells (Si, Ag, In, Ga). Project is based on new delamination technique based on a wire diamond technology, developed and patented by the CEA in order to separate a module's glass, backsheet, and cells in a single process, in such a way that the single parts of the module can then be treated individually for the recycling phase (Lovato et al., 2021; PHOTORAMA, 2024).

4.1.13 Experimental studies

(Azeumo et al., 2019) in their work processed polycrystalline-Si PV modules using a chemical and physical processes. The physical process involved disassembling the aluminium alloy frame and then cutting the modules into homogeneous samples to perform small-scale laboratory operations. Material separation in the PV module is achieved using a circuit consisting of operations like crushing/milling, sieving, and heavy medium separation. The plastics and most of EVA was recovered. After that, the sink (mainly composed of glass and metal) and the float (polymers and EVA residues) are sent to the recovery of the dense medium separator which is mainly pumped back to the dense medium separator after the first drainage screen, and partially regenerated after water spraying and magnetic separation. At the end of process, the sink is then milled and sieved to obtain the glass and metal fraction. The chemical process used solvents to separate the layers of glass, polycrystalline silicon, and polymer. The separation of EVA and the sheet of polymer was carried out using 99,8% pure toluene at different temperatures and with or without the use of ultrasound and thermal pretreatment. They concluded that the concentration of toluene for dissolving of EVA layer has a more important role during separation than the power of ultrasound. The type of solvent used to remove the EVA layer can mostly define the efficiency of the delamination (Savvilitidou et al., 2017).

After removing the Al frame, the recovery of the glass sheets of the c-Si PV panel was assisted by the thermal mechanical process (Belançon et al., 2022). Although the glass bottle has approximately the same composition as the cover glass in PV modules, the latter has higher purity and transparency. Therefore, it would be a waste to mix such a high-quality material with the glass of coloured glass bottles.

(P. R. Dias et al., 2022) also provided experimental part of study where they shredded c-Si module into a smaller than 2 mm parts, quartered homogenous mix. Ten such samples were prepared for the electrostatic separation of materials into electrostatic conductive and non-conductive fractions. For evaluation of metal distribution in each fraction, the outputs were digested in nitric acid (65% concentration), to leach silver and copper and then hydrochloric acid (38% concentration), to leach aluminium at room tem-

perature. The average mass distribution after the electrostatic separation had 3,34 wt% contained in the conductor fraction, while the remaining 96,66 wt% in the non-conductor fraction.

(de Oliveira et al., 2020) performed experiments in silver recovery and showed that is possible to recover up to 99.98% of the silver from photovoltaic cells by combining processes involving acid leaching by nitric acid followed by electroprecipitation process.

(Klugmann-Radziemska & Ostrowski, 2010) performed experimental recycling of c-Si PV cells (monocrystalline and polycrystalline). The process consists of chemically removing the Ag coating and chemically etching the rest of module with organic solutions with different concentrations and temperatures during the process to obtain a pure silicon wafer. Despite the efforts made to formulate the composition of a universal etching solution, the results showed that the solution must be modified according to the type of PV cells being recycled.

Chemical etching recipes must be adapted to different cell technologies. Numerous mixtures were tested, mainly based on the following compounds: hydrofluoric acid (HF), hexafluorisilicic acid (H_2SiF_6), nitric acid (HNO_3), acetic acid (CH_3COOH), hydrogen peroxide (H_2O_2) and distilled water (H_2O).

(Savvilitidou et al., 2017) in their experimental study disassembled Al frame, junction boxes and cables from CIS and tandem a-Si:H/ $\mu\text{c-Si:H}$ PV modules. Individual parts (panel, junction-box, and cables) were shredded and pulverized to study the toxicity of each part as well as the toxicity of the entire module for disposal. Most of the toxic elements in the junction box samples have high measured values. The absence of Cd in CIS PV panels, which is commonly used as CdS in the window layer of solar cells, is probably due to the replacement of Cd with Zn, which is less toxic. They also compared the measured concentration of toxic materials with Total Threshold Limit Concentration (TTLCL). This method determines the toxicity of the sample, if the samples exceed the threshold value they are classified as hazardous. For the CIS, all samples are hazardous for Se, one edge sample is characterized as hazardous for Cu, Hg and Zn, while the overall CIS sample and a-Si:H/ $\mu\text{c-Si:H}$ junction box sample are toxic for Cu. These results raise concerns about the potential risks of inappropriate module disposal and environmental risk may be underestimated or overestimated through a laboratory simulated scenario. Still, TCLP test proves that no sample leachate exceeds the threshold limits. They experimented with different treatments to successfully remove the EVA layer. They concluded that removing the EVA layer from the CIS panel is more challenging as it takes 4 days using lactic acid.

(Granata et al., 2014) in their experimental research used polycrystalline silicon PV module, a-Si and CdTe PV module. They concluded that, to obtain the highest mass recovery, for all three types of PV modules, the process scheme should be: crushing by two blade rotors crusher, crushing with a hammer, thermal treatment at 650°C of fractions larger than 1 mm and sieving with a sieve of 0,08 mm. With these procedures, about 85% of the total weight of the panel can be recovered as glass from frac-

tions $d > 0,08$ mm, while fractions $d < 0,08$ mm can be processed for further extraction of silicon, cadmium, tellurium, and zinc.

(Huang et al., 2017) proposed a three-step wafer-Si PV module recycling process with two new technologies in their experiment. The first technology was sequential electrowinning, which allows the extraction of multiple metals one by one from Si modules, Ag, Pb, Cu and Sn. The recovery rates of Ag and Cu were 74% and 83%, respectively, but they claim it can be increased to 95%. The purity of the recovered metals is over 99%. Another technology presented was sheet resistance monitoring which maximizes the amount of solar-grade Si recovered from the module.

(Tammaro et al., 2015) in their experimental tests heated three different c-Si panels in a furnace at 600°C and kept them at that temperature for 30 minutes. At the end of the thermal treatment, the PV cell and the glass layer are separated. Moreover, relevant amounts of hazardous metals, such as chromium and lead, have been released and may cause environmental concerns if the fumes are not properly treated.

(Cui et al., 2022) achieved the separation of silver, lead, tin, and copper in the laboratory (leaching, electrolysis, and etching from bare silicon solar cells). After removing the c-Si PV module frame and cables, organic compounds are burned in a furnace, leaving behind glass, silicon wafers and other residues. The recovered silicon is then leached with nitric acid to form soluble metal salts. The leach solution is treated with a two-stage electrowinning process to recover silver, copper, and lead oxide, while the remaining wafers are etched with hydrofluoric acid and sodium hydroxide sequentially to remove contact metallization and passivation layers to recover silicon.

(X. Xu et al., 2021) presented in their work a novel process that integrates solvothermal swelling and thermal decomposition for non-destructive Si-wafer recovery. First, the EVA swelling method is applied by exchanging organic vapours in a solvothermal reactor. This represents a significant saving of time and amount of solvent compared to previous processes in which the module was immersed in a solvent. In addition, the introduction of ethanol is effective for removing the back layer to accelerate EVA swelling and avoid emissions of fluorine-containing gases. The key to successful recovery from a non-destructive Si-wafer is to achieve rapid EVA release by placing the gas in a horizontal direction during thermal decomposition through pre-formed gas release channels by pre-swelling the EVA. Therefore, the proposed process provides to be a feasible method for widespread non-destructive recovery of Si-wafers.

Research by (Camargo et al., 2021) explained that the metal content of crystalline silicon PV modules can vary because of factors such as the PV panel brand, country of origin, year of manufacture, among others. The thermal degradation followed by manual segregation increased the silver concentration concerning the ground PV laminate due to the removal of polymers, the glass fraction and copper ribbons.

After reviewing the experimental recycling processes of different PV modules, the conclusion is that the delam-

ination process (removing of the encapsulant) most often takes place either mechanically (shredding/milling) or thermally (burn off EVA layer) or chemically (dissolving in organic or inorganic solvents/acids). In terms of material separation, the physical method of separating glass and aluminium is the most common approach, followed by dry and wet mechanical processes such as vacuum blasting, scribing, attrition, sieving and flotation. Of the electrometallurgical processes in the part of c-Si PV module material extraction, sequential electrowinning is used, which successfully recovers Ag, Cu and Pb. Moreover, regarding the recycling of thin-film PV modules, a combination of mechanical/physical and chemical processes is most often used.

The goal of recycling photovoltaic modules is to obtain valuable materials such as Ag, Al, Pb, Cu and Si. Chemical etching in different solvents and at different temperatures is most often used to remove various coatings or residues after material separation and recover valuable materials.

4.2 Analysis of PV modules – software simulations

There are many studies based on the EoL solar panel Life Cycle Assessment (LCA) methodology with the aim of obtaining an assessment of the efficiency of waste management system, the impact of different waste management options on the environment and human health, the efficiency of the system in relation to the recovery rate, energy savings and so on.

There is an initiative of the International Energy Agency under the Photovoltaic Power Systems Programme called Task 12 and several reports have been published based on LCA analysis of solar panels, and some reports are dedicated to EoL management of solar panels. Some of the objectives of Task 12 are the accurate quantification of the environmental profile of PV compared to other energy technologies which can be performed using LCA describing energy, material, and emission flows in all phases of the PV life cycle modules. The next objective of Task 12, EoL management options for PV modules, can be investigated using recycling analyses and other waste management options.

(Belançon et al., 2022) validated the simple manual removal of glass sheet as a proof of concept for the future reuse of glass sheets from c-Si PVs. They suggest developing some kind of mechanical separation machine to separate the glass from the rest of the PV module. Estimated energy savings could be around 10 TWh due to avoided demand for new cover glass production and carbon saving equal to half the weight of the glass. Limitation to this procedure is that was performed on unbroken glass.

(Daniela-Abigail et al., 2022) in their work provide environmental, economic, and social dimensions of PV waste for the Yucatan region of Mexico through life cycle environmental analysis and life cycle cost analysis. The analysis shows a reduction of 83% in the category of human toxicity (non-cancer effects) and freshwater ecotoxicity, while the depletion of mineral, fossil, and renewable resources show a 66% lower use of the FREL P recycling system compared to the system without recycling case. The categories of human toxicity (cancer effects), photochemical ozone forma-

tion, terrestrial eutrophication and marine eutrophication show reductions between 32% and 42%. Since the categories of photochemical ozone formation and terrestrial eutrophication are influenced by salvage mostly aluminium, and to a lesser extent glass, silicon, and silver. Regarding the categories of climate change, particulate matter and land use, the impact was reduced by 28%, 22% and 20%, respectively. The categories of water resource dependence and freshwater eutrophication show a change of 1%. (P. R. Dias et al., 2022) in their study proposed an economical and environmentally friendly processing method using LCA and LCC. This process consists of removing the module frame, laminate shredding and concentrating the material using electrostatic separation. The proposed technology is electrostatic separation to obtain an estimate of material separation efficiency. The latter gives two fractions: a valuable mixture (2-3% of total weight of module) of silver, copper, aluminium and silicon, and a mixture mainly of glass, silicon, and polymers. An economic evaluation shows that this process is more cost-effective than FREL P under some conditions such as: smaller amounts of waste (< 4 kt/y), if there is no market for recovered glass, and when the end-processing industry is located afar. They were also conducted an experimental part of recycling, described later in the experimental part of review.

There is a gap in the scientific literature when it comes to assessing the environmental impacts resulting from different methods of recycling and reusing PV module materials. (Contreras Lisperguer et al., 2020) conducted an LCA analysis using data for the FREL P process due to the lack of data for other types of processes. The LCA analysis was carried out for the scenarios of open material cycle and closed material cycle with the aim of assessing the impact on the environment.

In their study (Lim et al., 2022) analysed an optimal recycling process with high resource recovery efficiency first at the laboratory level, which consists of module delamination, acid etching and sequential electrodeposition. The obtained results are further applied in the LCA analysis where this recycling system is compared with waste disposal. The chemical they use are widely available and cost-effective, namely HNO_3 , H_2SO_4 , H_2O_2 and NaOH . High recoveries of 86%, 95% and 97% were achieved for silver, lead, and aluminium, respectively, representing a combined resource recovery of 86%.

(Contreras Lisperguer et al., 2020) used LCA methodology to assess the environmental impact of EoL c-Si PV panels in a closed-loop-material-cycle (CLMC) scenario and compared it with a CLMC scenario of CdTe panels. In this study, it was estimated that the transport of decommissioned panels has the greatest impact on the environment, and in the following categories - fossil fuel depletion, metal depletion, climate change, ozone depletion and particulate matter formation. Wafer recovery should be the optimal method of PV panel recycling at the EoL, and thermochemical processes must be avoided during the delamination process. The use of organic solvents can be considered as an alternative way to reduce emission of toxic gases, the formation of fly ash and sludge, and to avoid damage to the solar cell material in the delamination phase. For PV tech-

nology to be fully sustainable, the use of incineration, pyrolysis and toxic acids that generate harmful environmental impacts should be avoided.

The recoverable percentage of each material used to produce the modules is particularly high for glass and Al (97% and 100%, respectively), smaller but still significant for Cu and Te (around 80%), and less significant for rare metals such as In and Ga (75% and 99%, respectively), which only represent about 1% of the total panel weight (Sica et al., 2018).

In their study (R. Deng et al., 2019) presented a techno-economic review of three main recycling approaches: glass recycling, mechanical recycling and thermal recycling compared to landfilling of PV panels. Upcycling (high-value recycling) aims to recycle not only glass and Al frame, but also more valuable components, such as Ag, Si, Al, and Cu. This process is divided into module delamination (can be mechanical, thermal, or chemical), leaching/etching (to separate silicon and metal), metal extraction (electrolysis, metal replacement, chemical precipitation).

(Ardente et al., 2019) compared the FREL process and the recycling process currently used in Europe (baseline process) by applying LCA analysis. The overall recycling rate achieved by baseline process (according to visits to two recycling plants in Italy and Spain) is around 24%, well below the current minimum target of 80% reuse and recycling, as set by the WEEE Directive. The process consists of dismantling the panel frame and cables, which are subsequently sorted for recycling, then the glass is separated with a hammer or grinding, or it is directly shredded with other WEEE waste. The estimated basic process can separate up to 10% of glass, while PV cells and plastic are disposed of in landfill with shredding residues. In this study, the environmental impact of burning three types of different background layers (two fluorine plastics – TPT, KPK and fluorine-free plastic PPE) was evaluated. Compared to incineration of TPT back film, the FREL process applied to fluorine-free PV waste is characterized by lower impact on acidification, human toxicity, ozone depletion and cumulative energy demand (CED). Incineration of PV waste without fluorine has a slightly greater impact on climate change due to higher carbon content of PPE. Therefore, incineration of waste with KPK always has the greatest effects, due to the higher content of fluorine in the background layer. Hence, (Ardente et al., 2019) pointed out that the thermal treatment of PV modules must be carefully evaluated as they may be responsible for emissions of pollutants such as hydrogen fluoride that is potentially released by the combustion of halogenated plastics in the back sheet of PV panel. Unlike (Ardente et al., 2019), (Aryan et al., 2018) compared EoL management options (incineration and pyrolysis) for two types of back sheet materials that are typical for c-Si PV modules. They are PET based back sheet (fluorine free) and PVDF based (fluorinated). The results of the analysis, along with the assumptions made during the modelling, show that the fluorine-free back sheet shows a better environmental performance compared to the fluorinated back sheet in both incineration and pyrolysis scenarios.

(Latunussa et al., 2016) analysed by LCA a FREL process for recycling c-Si PV panels based on a series of phys-

ical (mechanical and thermal) treatments followed by acid leaching and electrolysis. The total impact of the process on climate change is about 370 kg CO_{2eq}. This is due to the incineration of PV sandwiches (34%), transport (29%) and metal recovery treatments (24%).

4.3 Mathematical models

(Molano et al., 2022) proposed a reverse logistics planning model for the collection of EoL PV panels, which is expected to support the integration of existing recycling technologies (FREL) and collection schemes using a holistic approach to ensure feasibility and reduce impact on environment. They emphasized that transport is one of the main activities that can put an additional burden on the environment. However, from the economic perspective the feasibility of reverse logistics highly depends on the price of commodities and recovered materials.

(Paiano, 2015) estimated waste production for the periods (2012-2038) and (2039-2050) with the aim of managing this waste to improve proper waste management of hazardous substances, as well as the importance of recovery and recycling of valuable resources. Given that the calculated amount of cadmium contained in EoL CdTe thin films is 877 t in 2050, its potential leaching due to nitric acid or rain is between 254-351 t. The amount of lead contained in the EoL of c-Si and CIGS modules in 2050 is 3381 t, so the relative value is between 439,5-3043 t.

(Choi & Fthenakis, 2014) developed a framework of mathematical modelling to evaluate economic feasibility of the macro- and micro-level planning of recycling process of c-Si PVs at the example of Germany. The yield of recovered cells is highly dependent on the type, design and condition of the modules being processed. Design-dependent factors that affect results of the thermal process are type of laminate, the crystal type and the dimensions of the embedded cells, the material and dimensions of the connections and solders.

4.4 Economic aspects of recycling technologies

Although the expected lifetime of PV modules is 25-30 years, based on several studies (Daniela-Abigail et al., 2022; Foles et al., 2020; M. A. Omar & Mahmoud, 2018) it can be concluded that the payback period of PV modules (without battery storage) for residential applications is between 5-12 years. The difference between the payback period of various locations is mainly because of the available solar radiation resource, ambient temperature, and the economic variables, especially inflation and the discount rate (Daniela-Abigail et al., 2022). Compared to other electronic products, silicon PV modules have a very low cost per kilogram, which makes them unattractive for recycling. Accordingly, the product price of new PV modules is sensitive to additional recycling costs (R. Deng et al., 2019). The recovered quantities of aluminium and glass cullets are much higher than the quantities of copper and silicon by weight. Therefore, Al and glass prices are relatively more sensitive to the recycling plant profits (Choi & Fthenakis, 2014). Also, whole glass has a higher economic value than broken glass (P. Dias & Veit, 2018).

(Molano et al., 2022) collected market prices of recovered materials in Australian dollars for 2022, here Australian dollars are converted to US dollars at the current exchange rate (for 2022) – data showed in the Table 3.

Additional interest in PV recycling is also shown by the recent high number of patents developed for the removal of encapsulant from the laminated structure of PV, which is one of most difficult and important goals for photovoltaic recycling technologies. A list of patents related to PV module recycling can be found in the literature (Padoan et al., 2019). To become economically viable (R. Deng et al., 2019) concluded that both mechanical and thermal processing of recycling should aim to reduce costs by at least 30%, reaching USD 400-500 per ton and USD 700-800 per ton, respectively, with improved material recovery. New technologies for PV modules recycling should aim to reduce the cost and complexity of recycling processing and improve the mechanical recovery of the intact silicon wafer. Furthermore, life cycle analysis for the Yucatan region indicated that the cost of electricity could be reduced by 2% to reach 0,146 USD/kWh. The only condition is that it is accompanied by a recycling incentive of 15% of the initial cost (Daniela-Abigail et al., 2022).

5. CONCLUSIONS

Photovoltaic panels have become a key technology in the production of clean-green energy, but as their use expands, it is crucial to consider the long-term environmental impacts. Proper management of photovoltaic panels at the end-of-life is of great importance. These modules contain materials that are recyclable and if properly recycled can be used to manufacture new panels or other products. However, no commercial recycling process can currently recover all valuable materials from photovoltaic panels, and there is no consensus on the best way to achieve that goal.

It often costs more to break down the panel and recover the raw materials than the actual value of those materials. Hence, if more materials could be recovered with higher purity, they would have a higher market value and make recycling more economically attractive. However, extraction of high-purity materials requires multiple processing steps, which increases recycling costs. Solving this dilemma could be key to the development of a sustainable photovoltaic panel recycling industry. What is important now is to extend the lifetime of photovoltaic panels as much as possible, to design modules with a long-term vision of processing module waste to enable the recovery of valuable materials to the circular economy loop. It is also im-

portant to use environment friendly materials, as well as to increase the efficiency of the module.

The development of new PV technologies alone does not guarantee their market acceptance, therefore new technologies must have adequate economic, environmental, social, and political support. Just as it took 25 years for PERC technology (Blakers, 2019) to move from development to widespread commercial use, it might take that long to commercialize new PV or recycling technology.

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TABLE 3: Prices (USD/kg) of recovered materials in 2022 (Molano et al., 2022).

Output – recovered material	Price (USD/kg)
Aluminium scrap	1,181
Glass scrap	0,0754
Copper scrap	5,27
Silicon metal (metallurgical grade)	1,428
Silver	786,19

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RELIABILITY OF GUELPH PERMEAMETER FOR DETERMINING THE HYDRAULIC CONDUCTIVITY OF MINERAL LINERS

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ABSTRACT

The aim of this study is to evaluate whether the Guelph permeameter can be effectively used as a quick and reliable instrument to measure the hydraulic conductivity of compacted mineral liners. The investigation was carried out on two distinct areas of the final capping of a municipal solid waste landfill in Italy. On the first test site, hydraulic conductivity tests were performed using the Guelph permeameter, assessing reliability and repeatability of test results. On the second one, many undisturbed soil samples were taken and tested in the laboratory using a modified oedometric apparatus as a falling-head permeameter. Further hydraulic conductivity tests with Guelph permeameter were also performed in the holes made by soil sampling. The campaign was integrated with in situ density tests using the calibrated sand method, to check the uniformity of the compaction degree of liners. A final comparison between hydraulic conductivity obtained in situ and in the laboratory was proposed and critically analysed. The experiments performed on the two test sites with the Guelph permeameter and their statistical processing demonstrate that the instrument is not only economical and easy to use, but also reliable if used with appropriate precautions.

1. INTRODUCTION

The main environmental problems of a municipal solid waste (MSW) landfill are associated with the control of leachate and biogas emissions into the surrounding environment. Mineral liners are compacted at the bottom and top of landfills to achieve suitable hydraulic conductivities. National regulations generally establish the minimum thickness of liners and their maximum hydraulic conductivity k , according to the nature of waste stocked inside the landfill (European Council, 1999, 2018). No further details are generally provided on the experimental procedure to determine the coefficient k , which can be obtained, directly or indirectly, using either laboratory or field tests. The choice of a specific procedure depends on many variables, such as nature, water content and degree of saturation of soil, fluid permeant, required accuracy of instruments, available time and economical budget.

Many studies have shown that hydraulic conductivity can vary considerably depending on the applied experimental procedure (Fallico et al., 2006). Moreover, field measurements often provided values of k smaller than those obtained through laboratory procedures. This could be due to the presence of air trapped during wetting, which caus-

es partial saturation of the soil, hence a lower permeability (Fodor et al., 2011; Jačka et al., 2014; Rezaei et al., 2016).

The Guelph, auger-hole, and falling-head permeameter methods in clay to sandy loam soils were compared by Gallichand (Gallichand et al., 1990). The results showed that the geometric mean of K from the Guelph was less than that obtained by the falling-head method. It was also found that Guelph permeameter measurements underestimated the K_s value compared to the laboratory falling-head method, with differences even exceeding an order of magnitude (Feki et al., 2020).

Similar results were also found in studies carried out in the agricultural sector. Soto & Kiang (Soto & Kiang, 2018) found that the hydraulic conductivities resulting from tests with a double ring infiltrometer are, on average, 2.2 times greater than those obtained with the Guelph permeameter. However, agricultural soils are characterized by hydraulic conductivities 2-3 orders of magnitude greater than those investigated in this paper.

In this paper hydraulic conductivity obtained with laboratory falling-head tests and with Guelph permeameter on two test sites are compared.

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2. PERMEABILITY TESTS

2.1 Field permeability test with Guelph permeameter

The Guelph permeameter allows the on-site determination of the hydraulic conductivity of soils. It is easy to transport, place and use, cheaper than other permeability tests, and can provide reliable data if properly used. Furthermore, it can measure the three-dimensional flow of the permeating liquid, resulting from the horizontal and vertical components. The Guelph (2800K1 model) permeameter was used in this study. To interpret the experimental results, the following equations were used (Elrick et al., 1989; Elrick & Reynolds, 1992; Reynolds et al., 1983, 1985; Reynolds, 1986; Reynolds & Elrick, 1985, 1987; Zhang et al., 1998). The lowering rate R of the reservoir water level is considered equal to:

$$R_i = \frac{H_i - H_{i-1}}{t_i - t_{i-1}} \left[\frac{\text{m}}{\text{s}} \right] \quad (1)$$

where (Figure 1):

H are the initial and final hydraulic heads in a specific time interval (m);

t is the cumulative reading time (s);

i is the number of measurements,

and its mean ($\bar{R}$) is:

$$\bar{R} = \frac{1}{I} \cdot \sum_{i=1}^I R_i \left[\frac{\text{m}}{\text{s}} \right] \quad (2)$$

where I is the total number of measurements.

The first R values are generally neglected as water flow has not yet stabilized. Flow rate (Q) can be expressed by the following equation:

$$Q = A \cdot \bar{R} \left[\frac{\text{m}^3}{\text{s}} \right] \quad (3)$$

where A is the cross-sectional area of the tank, equal to 216 mm², when only the internal tank is used, and to 3522 mm², when both tanks are used. Finally, hydraulic conductivity (K_{fs}) is calculated as follows:

$$K_{fs} = \frac{C \cdot Q}{2 \cdot \pi \cdot H^2 + \pi \cdot a^2 \cdot C + 2 \cdot \pi \cdot \frac{H}{\alpha^*}} \quad (4)$$

where:

C : dimensionless shape factor,

Q : water flow rate (m³/s),

H : water height in well (m),

a : well radius (m),

α^* : macroscopic capillary length (m⁻¹).

α^* and C are calculated using Table 1.

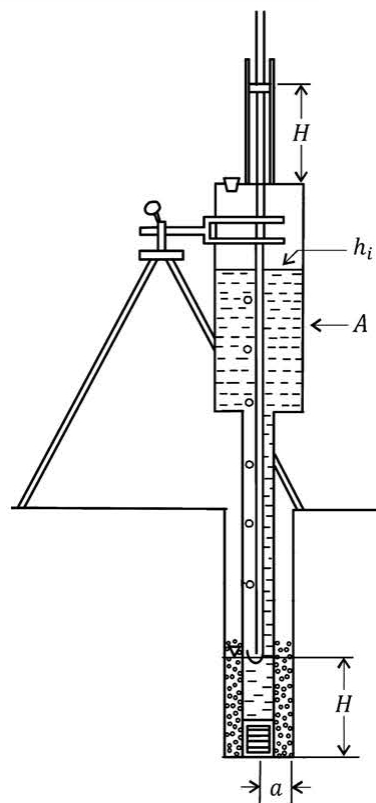


FIGURE 1: Working section of the Guelph permeameter.

The Guelph permeameter is essentially a Mariotte bottle and allows you to maintain a constant head within a small cylindrical well dug into the ground with special tools. Water seeps into the soil through the wet bottom and sidewalls and is replaced by new water flowing from the Guelph permeameter to maintain the level in the sump. By measuring the water level in the Guelph permeameter tank in two successive instants, measurement of the water infiltrated in a specific time can be obtained. From these measurements it is possible to obtain an estimate of the hydraulic conductivity at field saturation (K_{fs}), which is a fundamental parameter for modelling the water seepage through soils (Bagarello & Iovino, 2010).

Particular attention should be paid to the following aspects:

- Field saturated hydraulic conductivity (K_{fs}) - Compacted mineral liners always have a degree of saturation less

TABLE 1: Soil categories with respective α^* and C (Reynolds et al., 2008).

Soil category	α^* (m ⁻¹)	C
Compacted, structureless, clayey, or silty materials such as landfill caps and liners, lacustrine, or marine sediments, etc.	1	$\left(\frac{H/a}{2.081 + 0.121 (H/a)} \right)^{0.672}$
Porous materials that are both fine textured and massive. Unstructured clayey and silty soils, as well as fine structureless sandy materials may be included	4	$\left(\frac{H/a}{1.992 + 0.091 (H/a)} \right)^{0.683}$
Most structured and medium textured materials. Structured clayey and loamy soils, as well as unstructured medium sands may be included. This category is generally the most appropriate for agricultural soils	12	$\left(\frac{H/a}{2.074 + 0.093 (H/a)} \right)^{0.754}$
Coarse and gravelly sands. Highly structured soils with large and numerous cracks and macropores may also be included	36	

than 100% and this strongly influences the infiltration process (Reynolds et al., 1983). For this reason, K_{fs} is used in equation (4) in substitution of fully saturated hydraulic conductivity (K_s), used only in laboratory tests.

- Macroscopic capillary length (α^*) - This parameter, tabulated and dependent only on soil nature, is more reliable in partially saturated soils. However, K_{fs} does not strongly depend on α^* , particularly for high H/a ratio (Elrick et al., 1989).
- Shape factor (C) - This factor which is also tabulated, derives from numerical analysis and depends mainly on the H/a ratio and soil structure (α^*). Estimation of C factor becomes less reliable if the soil is close to saturation. However, for most applications, the tabulated values may be considered reliable (Reynolds & Elrick, 1987).

To obtain valid experimental results, some practical precautions must be taken as follows:

- Point measurements - The test aims to evaluate the punctual hydraulic conductivity of a compacted mineral liner. Cracks or preferential paths may be present in sampled soil, where hydraulic conductivity is much higher. Therefore, a statistically significant number of measurements must be made to correctly assess hydraulic conductivity.
- Soil homogeneity - All formulas assume a homogeneous, isotropic and uniformly wetted soil. If the test is carried out on heterogeneous soils, the estimation of α^* is difficult and hydraulic conductivity could give unlikely results (Reynolds, 1986). Therefore, when analysing a stratified soil with different permeabilities, it is recommended to carry out many tests at different depths corresponding to the soil stratification (Xiang et al., 1997).
- Achievement of steady state condition - In very permeable soils the steady state condition is achieved quite quickly (order of minutes), while a longer period (order of hours) could be required in poorly permeable soils. Therefore, the test duration should be adapted to the nature of the investigated soil.
- Gravity influence - Gravity can affect the experimental results. However, the amount of flow that can be attributed to gravity decreases rapidly as the H/a ratio increases; for $H/a \geq 5$ the gravitational flow can be neglected, having an influence less than <5% (Reynolds et al., 1983).
- Hole smearing - During the excavation a thin smeared layer can be generated along the lateral surface of the hole, especially with fine-grained soil having higher water content. An underestimation of permeability, due to the pore closure around the test hole, could be obtained (Reynolds, 1986). To limit smearing, a sharp auger should be used, making a gentle downward pressure, and the hole surface can be scrubbed with a brush or small sprocket (Reynolds, 1986). The infiltration rate values in smeared wells could be less than half of those not smeared (Koppi & Geering, 1986).
- Well collapse - Test wells could collapse during the excavation or during the test in granular soils. This prob-

lem can be overcome by using a pointed casing (Reynolds & Lewis, 2012).

- Temperature changes - Heating of water inside the tank, due to direct solar radiation, can prevent air from rising due to its thermal expansion. Furthermore, hydraulic conductivity is inversely proportional to water viscosity, which is inversely proportional to its temperature. Therefore, K_{fs} will increase with the water temperature: an increase of 15°C can lead to an increase in measured permeability up to 45% (Reynolds & Topp, 2008).

2.2 Falling head permeability test in a modified oedometric cell

Falling-head permeability test in an oedometric cell is suitable for low permeability soils, since the test duration is short and even small flow rates can be easily measured using a graduated burette. The test follows the ISO 17892-11 procedure and is carried out using an oedometric apparatus, modified for permeability tests (ISO 17892-11:2019, 2019). Each sample is previously saturated and consolidated with a set of increasing vertical pressures. The hydraulic conductivity of a saturated soil sample (K_s) can be calculated as follows:

$$K_s = \frac{a \cdot L}{A \cdot (t_2 - t_1)} \cdot \ln \frac{h_1}{h_2} \quad \left[\frac{m}{s} \right] \quad (5)$$

where:

A: burette area (m²);

L: sample height (m);

A: cross-sectional area of sample (m²)

h_1 and h_2 : initial and final water level (m);

t_1 and t_2 : initial and final testing time (s).

2.3 Planning of the tests

The on-site experimentation was carried out in a municipal solid waste (MSW) landfill, located at Grumolo delle Abbadesse in the north-eastern part of Italy. The landfill area covers approximately 160,000 m² and has a total capacity of 1,125,000 tons of MSW. Compacted mineral barrier and rainwater drainage of the landfill cover were made using silty clay and silty sand respectively, coming from local excavations. Furthermore, the biogas drainage layer was made with selected inert waste, able to perform the required function of a gravelly layer. The liquid limit (w_L) and the plasticity index (PI) of the silty clay used in the compacted barrier were between 44-46% and 19-22%, respectively.

Two different areas were selected on the temporary cover of the landfill (Figure 2) to test the Guelph permeameter. Clayey soil was spread over the test area in layers 0.40 m thick, and compacted with four passes of a dumper loaded with 45 tons of sand, obtaining a degree of compaction around 95 % of standard Proctor dry density (corresponding to a compaction energy of 593 kJ/m³). Although the dumper is not a vehicle specifically designed for compaction, it was demonstrated that it is suitable for this purpose (Moraci et al., 2018).

2.3.1 First test field

The first test field aimed to evaluate reliability and repeatability of the experimental results. A rectangular area

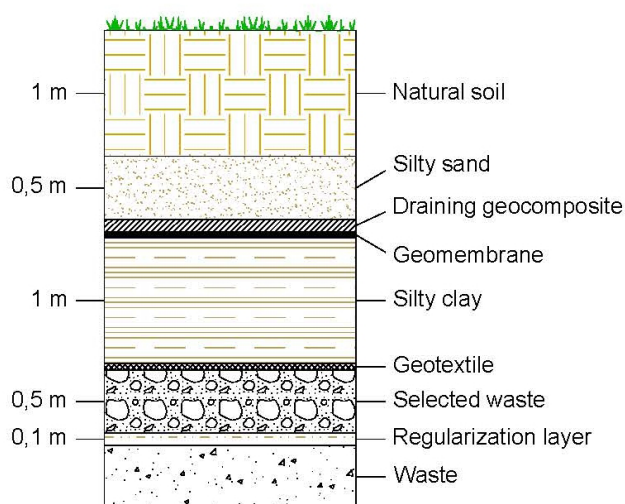


FIGURE 2: Section of the final cover in the Grumolo delle Abbadesse landfill.

of 9.0 m x 4.0 m of mineral liner was selected (Figure 3). Clayey soils were spread and compacted obtaining a final layer 2 m thick. Before test execution, the shallowest layer, 0.3 m thick, was removed. Twenty-four hydraulic conductivity tests were performed using the Guelph permeameter. In most tests, an interval t of 2 minutes between readings was enough to achieve a constant lowering rate. However, in case of clay lenses, the interval was increased to 5 minutes to achieve a higher difference between consecutive readings. In most tests, both tanks (total area $A = 3522 \text{ mm}^2$) of the permeameter were used. When clay lenses were met, only the internal tank was used ($A = 216 \text{ mm}^2$) to get adequate resolution when taking a reading. Three

different test pit depths were chosen: 0.2 m (A), 0.3 m (B) and 0.5 m (C). It was thus possible to obtain results that were more representative of the entire layer, characterizing possible soil inhomogeneity. A tripod was used to minimize the permeameter settlement, sometimes digging to position the legs correctly. The well was dug with the augers supplied in the kit, so its radius was always equal to 30 mm.

A hydraulic head in the well $H = 0.15 \text{ m}$ was chosen, suitable for moderately permeable soils, from silty sands to silty clays (Elrick et al., 1989). The macroscopic capillary length $\alpha^* = 1 \text{ m}^{-1}$ and the shape factor C were defined using Table 1. In general, the volume tested roughly corresponds to the size of the wetted zone around the test well. In moderately permeable soil, this corresponds to a spheroid extending from 0.1 m to 0.2 m outward from the submerged surface of the test hole (Elrick et al., 1989). Therefore, a spacing of 1 m among the holes was chosen (Figure 3). A hydraulic head of 0.15 m was assumed (Figure 4).

2.3.2 Second test field

The second test field was designed to evaluate the difference between the results of the two types of permeability tests, after having characterized the investigated soil. A rectangular area of 9.5m x 4m was prepared using the same soil of the previous testing area (Figure 5). In this case soil was selected before laying, to isolate the most homogeneous fraction possible. The final thickness was greater than 1 m, also considering the effects of compression and swelling. Again, the first 0.3 m of soil was removed just before starting each test. For the laboratory analyses, twelve cylindrical samples of 80 mm in diameter and 300 mm in length were taken. For comparing in situ and in laboratory permeability, twelve hydraulic conductiv-

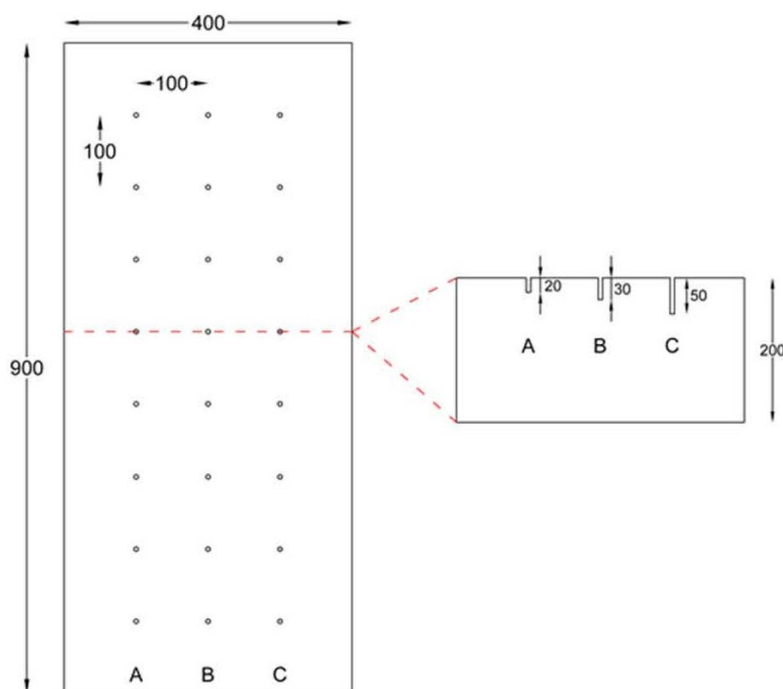


FIGURE 3: Plan and section of the first test site (measurements expressed in cm).

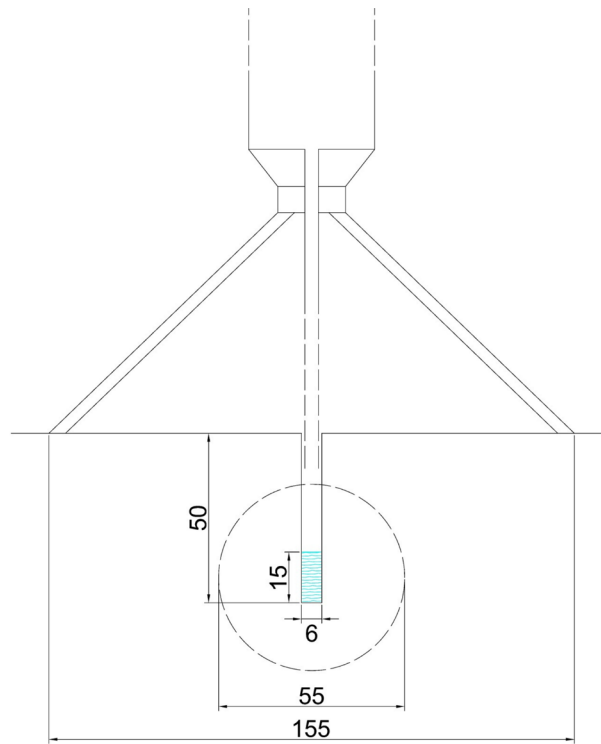


FIGURE 4: Wet zone and tripod foot around the test hole (measurements expressed in cm).

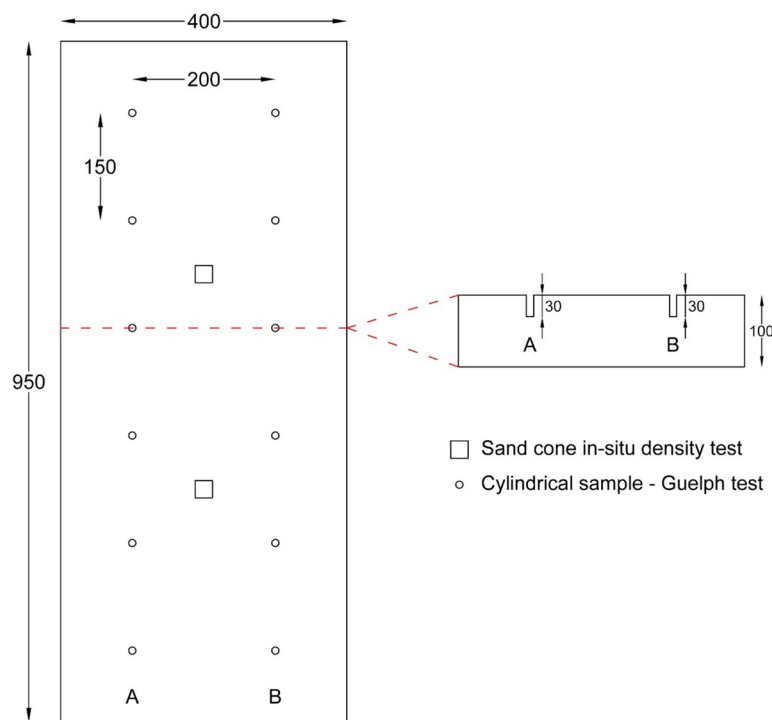


FIGURE 5: Plan and section of the second test site (measurements expressed in cm).

ity tests with Guelph permeameter were performed in the holes where sampling was made. In almost all tests, only the internal reservoir ($A = 216 \text{ mm}^2$) was used, with a time interval between readings of five minutes. Only in the case of sand lenses was it necessary to use both tanks ($A = 3522 \text{ mm}^2$) and a time interval of two minutes, to obtain a

clear difference between consecutive measurements. Well sizes were determined by the envelope for the laboratory sampling (80 mm in diameter and 300 mm in height), then the radius a of the well equal to 40 mm. To maintain the ratio $H/a \geq 5$ and to avoid the influence of gravity, a hydraulic head $H = 0.23 \text{ m}$ was chosen. As for the first test field, the

macroscopic capillary length $\alpha^* = 1 \text{ m}^{-1}$ and the form factor C were defined by means of Table 1.

Tests were preceded by two in situ density tests with a sand volumenometer. Laboratory analyses included, for each sample, the soil classification by particle size distribution and sedimentation: twelve sieves with descending diameters from 100 to 0.063 mm were used and then the remaining soil was suspended in an ASTM 152H hydrometer with distilled water and sodium hexametaphosphate ($\text{Na}_6\text{P}_6\text{O}_{18}$) at a controlled temperature of 20°C. Then twelve falling head hydraulic conductivity tests were performed in a modified oedometric cell: from the lowest part of each cylindrical sample a disk of 4000 mm² in area and 20 mm in height was obtained. Each sample was consolidated at 25 kPa and several falling head hydraulic conductivity test cycles were carried out until the system stabilized. The area a of the burette was equal to 19.6 mm². The test was performed at 20°C. The initial water level in the burette was $h_1 = 1 \text{ m}$. To minimize the inevitable heterogeneity of the investigated soil, the area of both test fields was reduced to the minimum possible size, considering the radius of influence of the wet ground around the holes and the minimum space required for the deployment of the tripod.

2.4 Test results

The processing is based on the following statistical indicators:

$$\text{arithmetic mean} \quad \bar{x} = \frac{1}{n} \cdot \sum_{i=1}^n x_i \quad (6)$$

$$\text{median} \quad \text{Med}(x) = \frac{x_{(\frac{n}{2})} + x_{(\frac{n}{2}+1)}}{2} \quad (7)$$

$$\text{standard deviation} \quad \sigma = \sqrt{\frac{1}{n-1} \cdot \sum_{i=1}^n (x_i - \bar{x})^2} \quad (8)$$

$$\text{variance} \quad \sigma^2 = \frac{1}{n-1} \cdot \sum_{i=1}^n (x_i - \bar{x})^2 \quad (9)$$

$$\text{coefficient of variation } C_v \quad C_v = \frac{\sigma}{\bar{x}} \quad (10)$$

where

n is number of data.

In addition, the Bland-Altman plot was used to statistically assess the agreement between Guelph permeameter results and laboratory data (Braždžionytė & Macas, 2006; Ghosh et al., 2019). In the Bland-Altman plot, the mean of two groups of data is plotted on the x axis and the difference between them on the y axis. The mean of the difference between the two methods is defined as bias. The 95 % upper and lower limits of agreement are equal to ± 1.96 times the standard deviation of the difference.

2.4.1 Results of hydraulic conductivity tests with Guelph permeameter on the first test field

On the first test field soil was quite heterogeneous, with frequent sand and clay lenses (Figure 6). For this reason, the measured hydraulic conductivity varied in the area, also in samples located at the same depth (Figure 7). The maximum permeability ($k = 1 \cdot 10^{-8} \text{ m/s}$), required by many national regulations, was exceeded in most cases. Two of the twenty-four tests were invalidated by the sinking of the permeameter in the well. The average permeability on

TABLE 2: Statistical parameter of Guelph tests on the first test site.

	$k_{fs} \text{ (m/s)}$
$\bar{x} \text{ (m/s)}$	3.8E-07
median (m/s)	2.2E-07
$\sigma \text{ (m/s)}$	6.4E-07
$\sigma^2 \text{ (m/s)}^2$	4.0E-13
C_v	1.68

site k_{fs} was $3.8 \cdot 10^{-7} \text{ m/s}$ and the median was $2.2 \cdot 10^{-7} \text{ m/s}$, compatible with a loamy soil, while standard deviation and coefficient of variation showed high values (Table 2).

In conclusion, despite the high soil heterogeneity, the preliminary study was successfully completed, since the results are reliable and reproducible, with consistency across the data.

2.4.2 Results of hydraulic conductivity tests with Guelph permeameter on the second test field

The grain size distribution curves of soils are reported in Figure 8. They are quite homogeneous and can be classi-

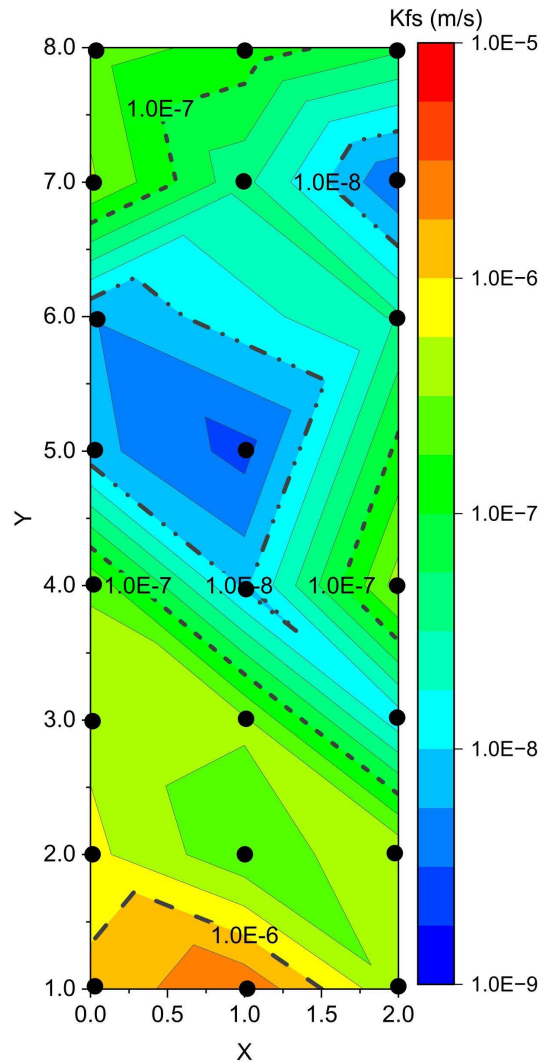


FIGURE 6: Distribution of the hydraulic conductivity determined with Guelph tests on the first test site.

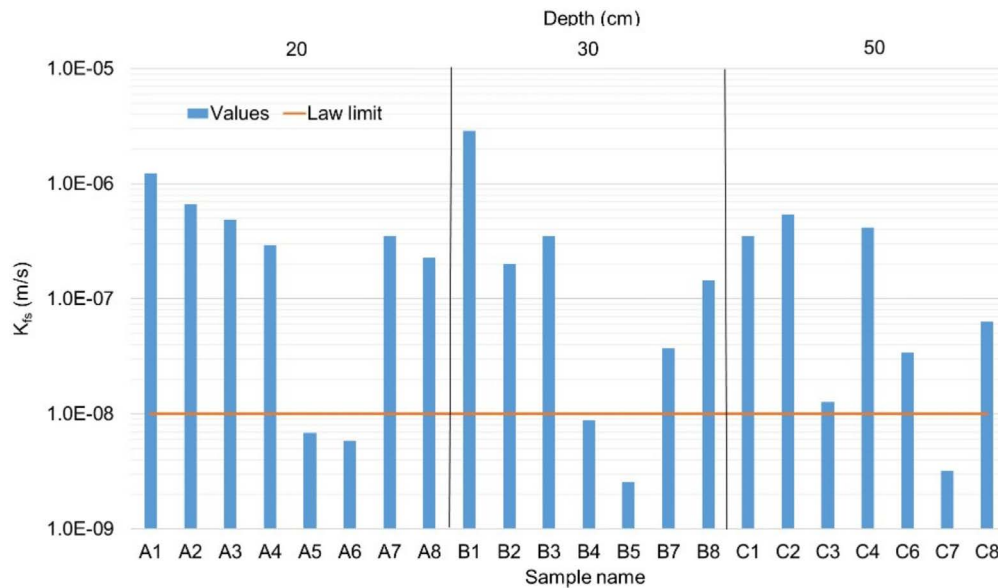


FIGURE 7: Hydraulic conductivity determined with the Guelph tests on the first test site.

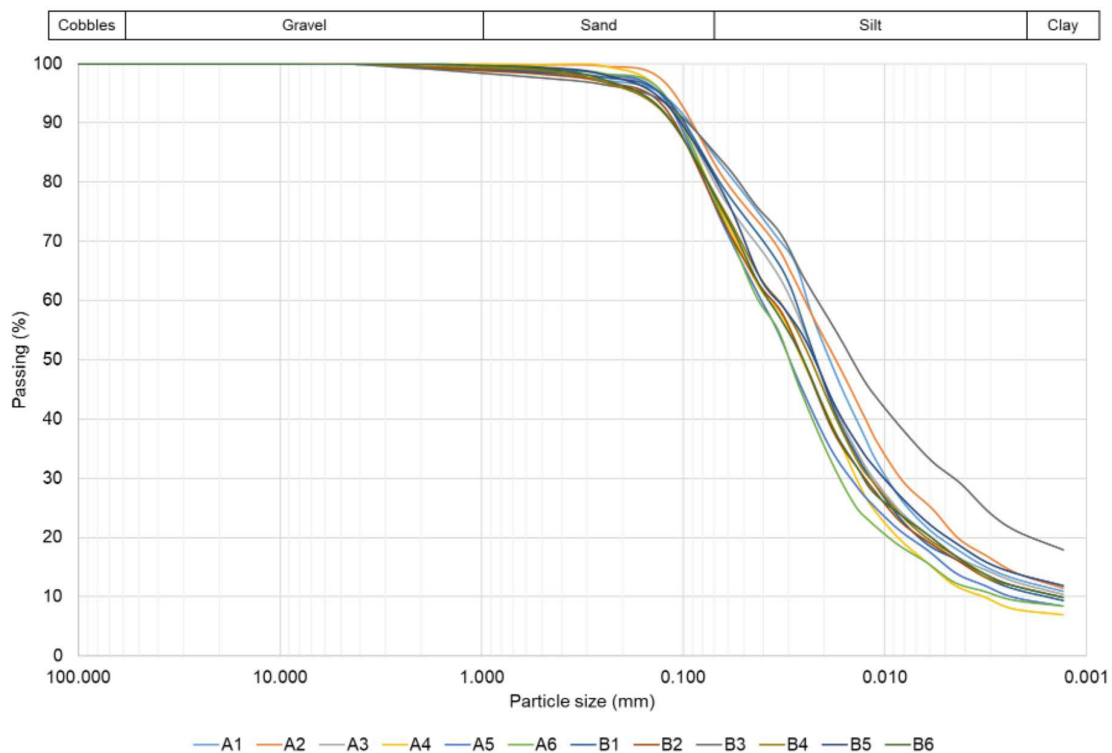


FIGURE 8: Grain size distribution of the investigated soil samples.

fied as silt (65.2%) with a small percentage of sand (22.7%) and clay (11.8%) (Table 3). Values of density obtained on site were very similar to those determined in the laboratory.

On the second test field, the results of the permeability tests both in the laboratory and on site were less dispersed (Figure 9 and Figure 10). The maximum coefficient of permeability, required by most European regulations, was respected in most samples. Two of the twelve tests were neglected due to instrument errors. The mean and the median

on-site hydraulic conductivity were lower than those on the previous test site (1.1×10^{-8} m/s and 5.3×10^{-9} m/s) by one order of magnitude. The corresponding laboratory samples reported a slightly higher mean value (1.2×10^{-8} m/s and 9.1×10^{-9}) (Table 4).

Figure 11 shows the hydraulic conductivity values vs the clay percentage: the higher the clay content, the lower the permeability. No noticeable trend in permeability was detected with depth (Figure 12).

TABLE 3: Properties of the different soil samples.

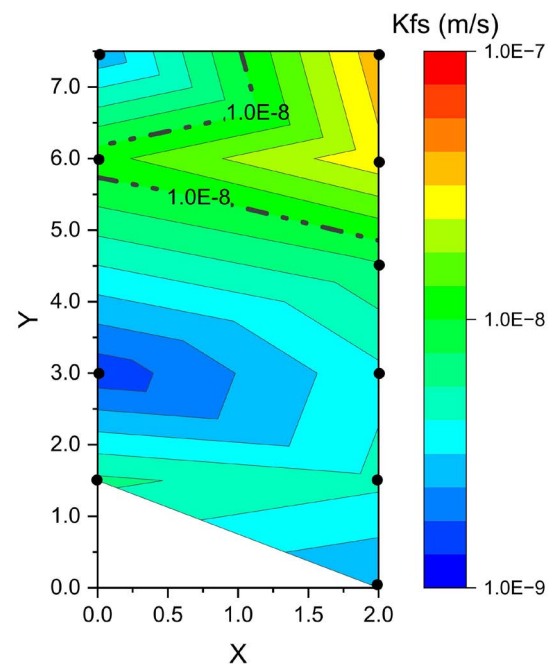
	laboratory	onsite
w (%)	19,4	19,0
ρ (Mg/m ³)	1,96	1,94
ρ_d (Mg/m ³)	1,64	1,63
gravel (%)	0,3	
sand (%)	22,7	
silt (%)	65,2	
clay (%)	11,8	
Soil description	Sandy clayey silt	

TABLE 4: Results of the permeability test on the second test site.

	Guelph	Laboratory	
	k_{fs}	k_s	k_{fs}/k_s
$\bar{x}$ (m/s)	1.1E-08	1.2E-08	2.06
median (m/s)	5.3E-9	4.4E-9	0.58
90 th percentile	3.0E-8	3.0E-9	7.10
σ (m/s)	1.3E-08	1.3E-08	
σ^2 (m/s) ²	1.6E-16	1.6E-16	
C_v	1.19	1.05	

As before, for both laboratory and in-situ permeability tests, the values of the standard deviation and of the coefficient of variation were high but very similar between the two types of tests (Table 4). As expected, the laboratory samples reported different hydraulic conductivity than the on-site tests. Indeed, the two types of tests adopt distinct tools and test conditions to determine the hydraulic conductivity. In particular:

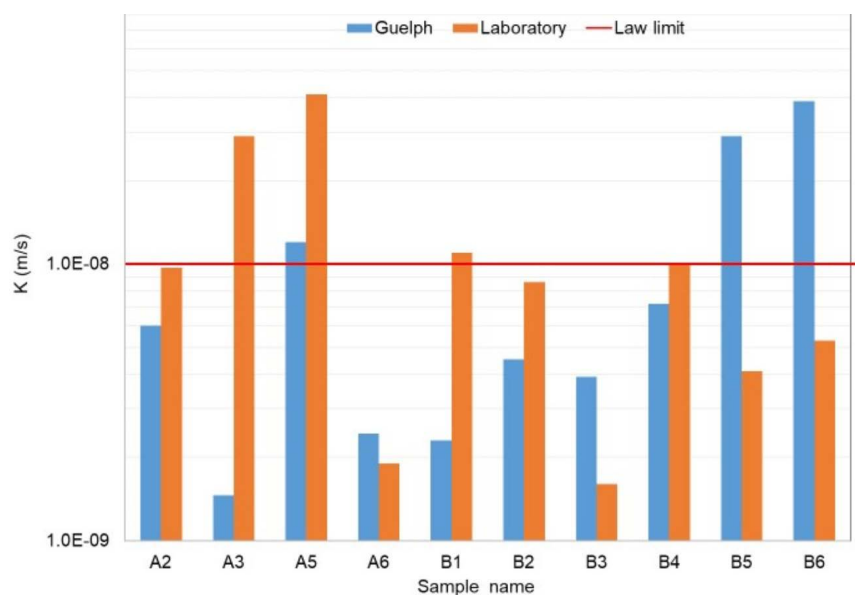
- Difference between sample sizes: the laboratory samples were discs of 4000 mm² in area and 20 mm in height, while the volume sampled on site was a cylinder

**FIGURE 9:** Distribution of the hydraulic conductivity determined with Guelph tests on the second test site.

of 5000 mm² in area and 230 mm in height.

- The laboratory tests were carried out at a controlled temperature of 20°C, while it was not possible to control the external temperature on site (ranging from 10°C to 23°C).
- The flow direction in the laboratory tests was vertical, while the on-site test measured the combination of both vertical and horizontal permeabilities.

Despite these disparities in test conditions, the difference between the results of the field and laboratory permeability tests was surprisingly minimal. The mean and medi-

**FIGURE 10:** Hydraulic conductivity measured on the second test site (ordered by name).

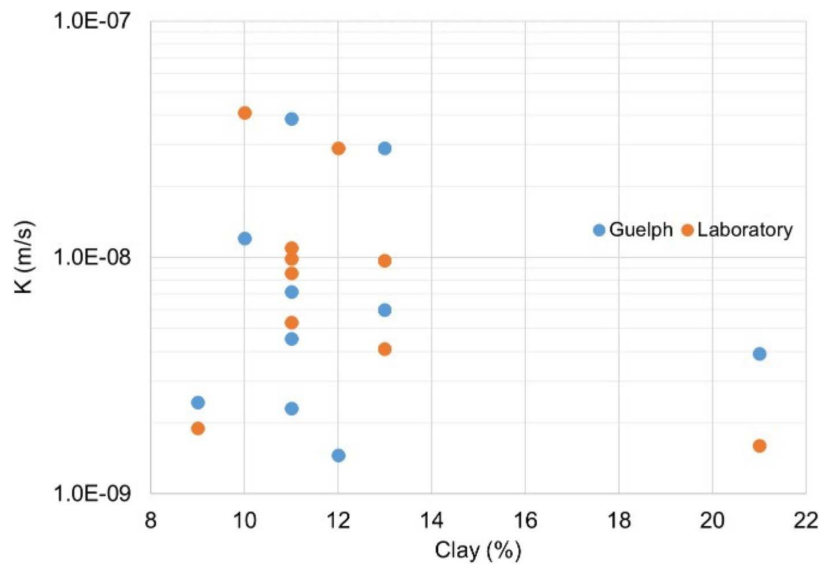


FIGURE 11: Hydraulic conductivity measured on the second test site (ordered by clay fraction).

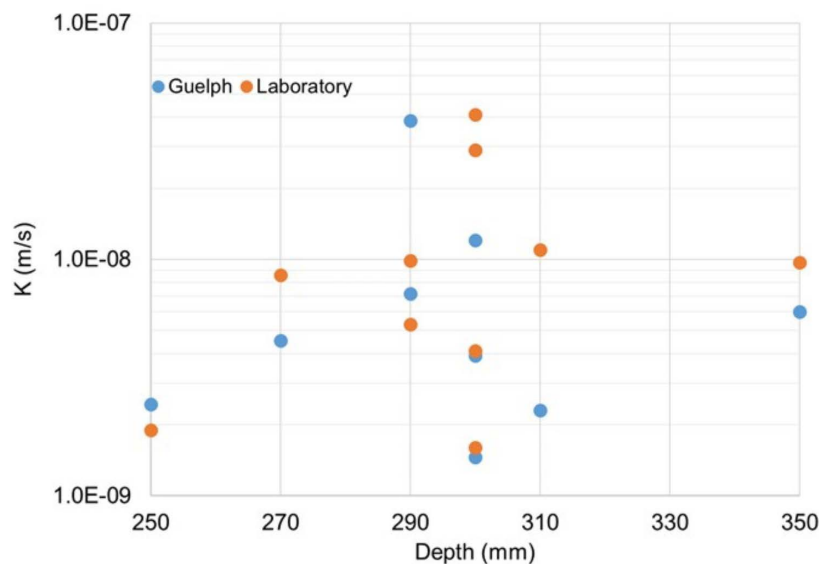


FIGURE 12: Hydraulic conductivity measured on the second test site (ordered by depth).

an k_{fs}/k_s ratios are 2.06 and 0.58, respectively, values that point to the existence of some outliers (Table 4).

A reference bold line, representing the equality between the hydraulic conductivity obtained in the laboratory and in situ, was inserted diagonally in the Figure 13. Other dashed lines, drawn in the same figure, represent the percentual deviation of the Guelph hydraulic conductivity from the above-mentioned reference line. The comparison between laboratory and Guelph hydraulic conductivity data shows a good agreement with some outliers.

Hydraulic conductivity data used in Bland-Altman plot were log-transformed since the hydraulic conductivity generally follows lognormal distribution (Loáiciga, 2015). Figure 14 shows a negative close to zero bias, the width of the 95% upper and lower limits and the absence of an evident trend between the difference and the mean.

Finally, in Figure 15 the k_{fs}/k_s ratio is shown as a function of the clay percentage and is mostly less than unity. In conclusion, considering the significant differences in procedures and conditions between laboratory and in situ permeability tests, the results are remarkably similar, and the Guelph permeameter provided very reliable results.

3. CONCLUSIONS

In the present study, the Guelph permeameter was tested on two testing areas, located on the provisional cover of the Grumolo delle Abbadesse MSW landfill. The mineral liner of the capping system was composed of clayey soils excavated in the landfill site. On the first testing area, 24 hydraulic conductivity tests were performed with a Guelph permeameter. On the second testing area 12 cylindrical samples were collected for laboratory analysis and hydraulic conductivity

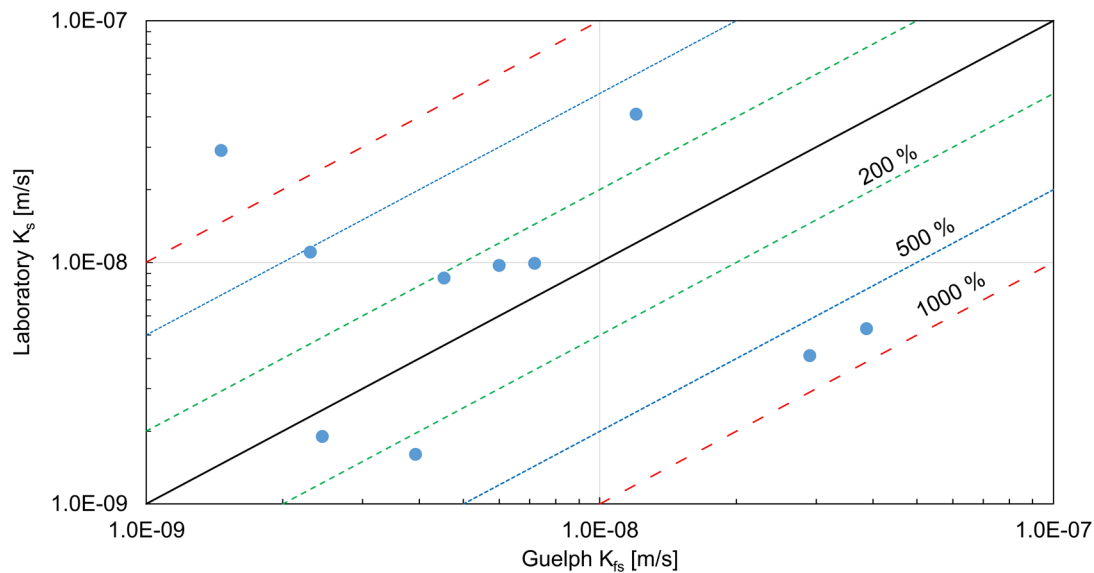


FIGURE 13: Hydraulic conductivity measured on the second test site and in laboratory.

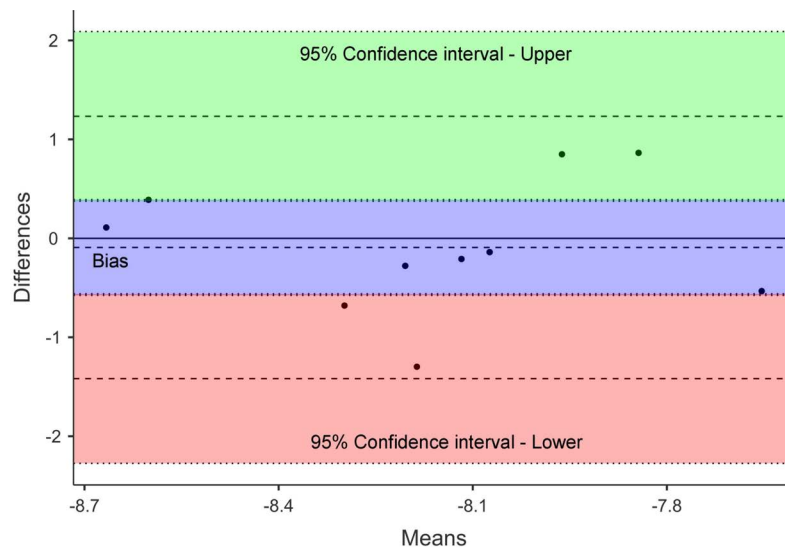


FIGURE 14: Bland-Altman plot between data measured on the second test site and in laboratory.

tests with the Guelph permeameter were performed in the holes made for the sample collection. Two in-situ density tests were also carried out. In the laboratory, sieve analyses and falling head permeability tests were performed.

The main purpose of the first test field was to evaluate reliability and reproducibility of the results obtained with the Guelph permeameter. The experimental results confirm this attitude; if the investigated soil is homogeneous enough, the hydraulic conductivity remains in a range comparable to that of laboratory tests. A greater dispersion of the values was found in the presence of sandy and clayey lenses, detected visually in some points of the investigated area (figure 6) which made the tested soil non-homogeneous.

The purpose of the second testing area was to evaluate the difference between hydraulic conductivities determined with the Guelph permeameter tests and those obtained from laboratory permeability tests.

On this second site, the soil was more homogeneous than on the first one and was classified as a sandy clayey silt. In situ density tests were found to be consistent with laboratory values. In this second test, given the greater accuracy in the selection of soils so that their homogeneity was granted, hydraulic conductivities were more than an order of magnitude lower than in the first test. Although the in situ and laboratory permeability tests differed in terms of test conditions and type of instrumentation, the difference in hydraulic conductivity values determined between corresponding samples was minimal.

In conclusion, the experiments performed with the Guelph permeameter and their statistical processing show its reliability, making it an instrument not only cheap and easy to use, but also reliable if used with the appropriate precautions. In particular, the Guelph permeameter becomes a valid alternative to laboratory permeability tests to

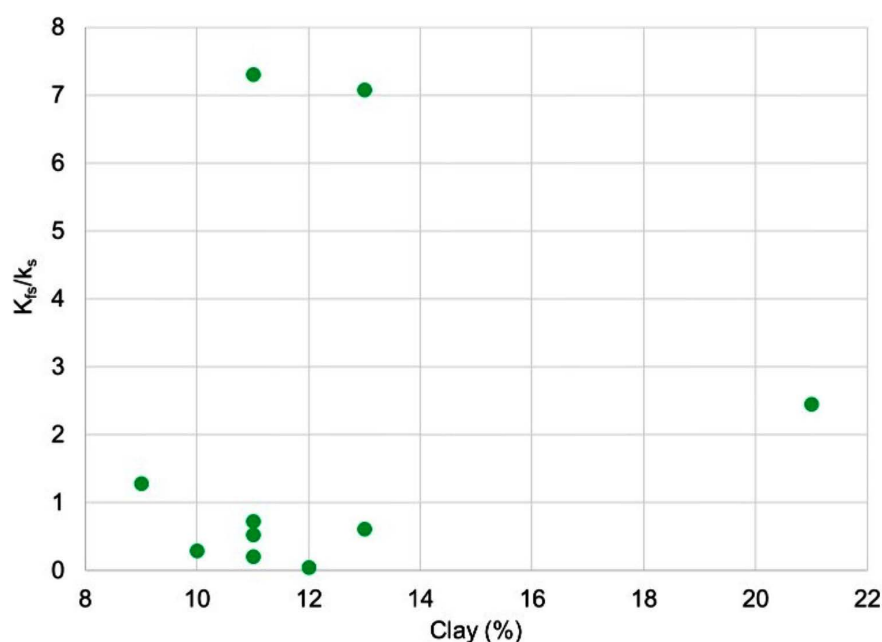


FIGURE 15: Ratio of on-site and laboratory hydraulic conductivities vs clay fraction.

define the permeability of compacted mineral layers quickly and precisely in landfills. The Guelph permeameter can be used with the aim of intervening during the construction of the barrier without slowing down the construction processes. Based on the values of the hydraulic conductivity detected on-site and according to the limits imposed by the regulation, the soil properties can be quickly corrected by adding or removing water or by soil mixing.

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ENVIRONMENTAL LAW ISSUES REGARDING PFAS POLLUTION IN WATER AROUND LANDFILLS IN SWEDEN

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ABSTRACT

In this paper, the regulatory framework for PFAS is addressed, using the traditional legal method to analyze the rules and structures of existing laws and regulations in Sweden. The aim is to analyze and problematize the legal situation regarding PFAS in water around landfills, with a focus on leachate. The starting point is the updated risk assessment of PFAS and the requirements for a non-toxic and healthy environment. The study shows strengths and weaknesses with the environmental legal framework for PFAS contamination at landfills in Sweden. Four central problems have been highlighted: (1) production: waste containing PFAS continues to be sent to landfills; (2) old landfills that are not in use but pose a high risk of spreading PFAS; (3) landfills in operation, which comply with current regulations but have large concentrations of PFAS in leachate (and surrounding water); and (4) PFAS propensity for long-range dispersal. The revised health-based guideline from the European Food Safety Authority underscores the heightened danger of PFAS, emphasizing the need to minimise the risk of long-term exposure. It takes time to implement changes to the legal system. While it is possible to regulate purification and permitted levels of PFAS in leachate through permit conditions, this opportunity is rarely used. A large responsibility therefore rests on the authorities responsible for the licensing of landfills to align the individual permits with up-to-date scientific knowledge on PFAS risks.

1. INTRODUCTION

The top step in the European waste hierarchy is preventing the generation of waste and the last step is disposal, including landfilling (Directive 2008/98/EC, Article 4). This means that landfilling should be considered as the last resort for managing waste, limited to what cannot be further valorised within the Circular Economy concept. Despite this, landfilling is still the main method of handling waste in large parts of the world; it is seen as a simple and cost-efficient way to dispose of waste. On the other hand, when the Circular Economy principle is effectively applied, hazardous substances contained in the original products may accumulate both in the recycled materials and in residues disposed of in landfills, including PFAS (Cossu et al., 2020; Pivato et al., 2024). For this reason, the design of the landfill, in line with the principle of environmental sustainability, plays a crucial role in reducing the risk of environmental pollution and its harmful effects on human health (Grossule and Stegmann, 2020). Although strict requirements are often placed on active landfills, the many older landfills pose a great risk of pollution to soil, water, and air (SGI, 2011).

In Sweden, 4.4 million tons of waste (excluding extractive waste) were landfilled in 2020, which is an increase of 300,000 tons compared to 2018 (SEPA, 2022:36). Since the beginning of the 1990s, Sweden has worked diligently to select and promote other treatment methods to reduce landfilling, including material recycling with easily accessible recycling centers, as well as clear guidance for municipalities to reduce the amount of waste. As a result of stricter legislation and a series of bans on disposal of certain types of waste introduced in the 21st century, a large part of the active landfills in Sweden have been closed, and most of them will be covered by 2030. The materials used for the final coverings are layered to, among other things, prevent rainwater from seeping through and getting contaminated (Swedish Waste Management, 2022:3; 2023). There are currently 370 landfills in Sweden that submit environmental reports¹, and about 6,700 landfills are registered in the EBH-portal² (Miljösamverkan, 2022:8). However, the closed landfills continue to affect their surroundings for a very long time (SGI, 2023).

Per- and polyfluoroalkyl substances (PFAS) are a collective name for a group of highly fluorinated, industrially produced, and environmentally hazardous substances that

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have been used in large-scale production since the early 1950s (KEMI, 2021:9). There are over 12,000 different PFAS structures that have been identified (EPA, 2021), and of these, roughly 4,700 have been inventoried so far (OECD, 2018:16). Since many PFAS are grease-, dirt-, and water-repellent, it makes them very useful in products like textiles, lubricants, and food packaging materials. Other PFAS also have surface-active properties that make them useful in, for example, cookware, cleaning agents, paints, ski wax, and cosmetics (KEMI, 2015:32-49). PFAS are very effective, and therefore only low concentrations are needed to achieve the desired function in the product (KEMI, 2015:66). Due to the strong bond between carbon and fluorine, PFAS are extremely persistent and do not fully degrade under natural conditions. This means that PFAS remains indefinitely in the environment in some form (KEMI, 2021:16).

Perfluoroalkyl sulfonic acids (PFSA) and perfluorocarboxylic acids (PFCA), such as perfluorooctane sulfonate (PFOS) and perfluorooctanoic acid (PFOA), respectively, are the PFAS that have mainly been studied in the last 20 years (KEMI, 2021:13). Thus, for the majority of PFAS and their effects on health and the environment, there is a lack of knowledge. There are, however, strong reasons to consider all PFAS as harmful to health (Bonato et. al., 2020; Lohmann et. al., 2020; Fenton et. al., 2021). Studies show that PFAS may affect reproduction, the immune system, birth weight, cholesterol levels in the blood, and cause cancer (Bonato et al., 2020; EFSA, 2020:3-9; ATSDR, 2021 Ch. 2; Fenton et al., 2021; Racz and Kempisty, 2021:3). Although available epidemiological studies show this correlation between exposure to PFAS and the mentioned health risks, it has not yet been possible to establish a cause-and-effect relationship for most of these health risks (ATSDR, 2021:26). Today, PFAS are found everywhere in the environment all over the world, and almost all people have PFAS in varying degrees in their bodies³ (KEMI, 2021:9).

Many PFAS have high water solubility, which, together with their persistence, gives them high mobility and a great potential for long-distance transport by water. Some PFAS (e.g. fluorotelomer alcohols (FTOHs), perfluorobutanoate (PFBA)) can be released into the air from materials, production processes, incineration, wastewater treatment, and landfills (Ahrens et al., 2011; Hamid and Li, 2016; MPCA, 2022; Björklund et al., 2023); and spread long distances through the air, which constitutes an important spread path to soil and surface water globally. This means that PFAS, via atmospheric deposition, can also be added to areas that otherwise have no influence from civilization or direct sources, like arctic environments. PFAS can enter groundwater by precipitation, irrigation, or flooding when they are present in the soil. In groundwater, the contaminants, due to their relatively high tendency to dissolve in water, can spread over long distances. Emissions to surface water can occur via runoff from contaminated areas, contaminated groundwater, emissions from, for example, extinguishing operations, treatment plants, landfills, or atmospheric deposition (KEMI, 2021:34-36; Racz and Kempisty, 2021:7-9).

PFAS has accumulated in both active and closed landfills due to decades of consumer use and subsequent dis-

posal of products, industrial waste, and sludge containing PFAS (Bolan et al., 2021:120). Knowledge about the composition, concentrations, and waste streams of PFAS in Sweden is often inadequate because only a few percent of the Swedish landfills have been investigated regarding PFAS. As a result, PFAS are spread to surrounding water and soil, where they pose a threat to humans and the environment for a long time to come (Miljösamverkan, 2022:3 and 8).

The aim of this paper is to analyze and problematize the regulations regarding PFAS in water near landfills, with a focus on leachate, in light of an updated risk assessment of PFAS and the requirements for a non-toxic and healthy environment. The study will highlight the strengths and weaknesses of the environmental legal framework for PFAS pollution at landfills in Sweden.

After the introduction, data on PFAS from Environmental Cooperation Sweden (Swe: Miljösamverkan Sverige)⁴ is presented to give an overview of PFAS findings at landfills in Sweden. Thereafter, the revision of the health-based guideline value from the European Food Safety Authority is briefly summarized to show a changed view of the risk assessment of PFAS. This is followed by an account of the regulation of PFAS on the international and European Union (EU) level, with a particular focus on waste (landfill) and water legislation. Lastly, the results are discussed and conclusions are presented.

2. MATERIALS AND METHODS

The analysis is based on a qualitative legal review of the regulations for PFAS. A traditional legal method has been applied, which in short means searching for a solution to a legal problem by analyzing and interpreting current law based on the generally recognized sources of law (Kleineman, 2018:21; Sandgren, 2009:118). The starting point for the analysis has been a review of the international regulation of PFAS. As an EU member state, Sweden is obligated to fulfil the requirements of secondary EU law (i.e., directives, regulations, and decisions), which means that certain areas of Swedish legislation are strongly influenced by EU law and, in some cases, directly transferable. This applies in particular to the waste and water legislation, as these matters are to a large degree harmonized at the EU level, thus limiting the possibilities for strictly national legislative solutions. Consequently, the central focus of the analysis has been relevant EU regulation, which has been followed by national legislation. To gain a greater understanding of the problem and the threat that PFAS in landfills poses to human health and the environment, materials from various authorities and organizations have also been studied.

3. PFAS UPDATE

3.1 Environmental Cooperation Sweden

With an emphasis on PFAS in leachate, Environmental Cooperation Sweden established a case manager support program for PFAS in landfills in January 2022. Measurement data on PFAS in leachate from landfills, as well as in surrounding groundwater and surface water, was collected

from 165 Swedish landfills in different phases. Leachate data was obtained from 117 landfills. The number of PFAS analyzed varied between 1 and 34. The investigated landfills had not installed treatment technologies to effectively remove PFAS. The general methods that existed for removing pollution were, for example, sedimentation ponds and wetlands. PFAS were present in measurable levels in all leachate samples, even after existing treatment methods. Since PFAS hardly breaks down, existing treatment methods are not sufficient to effectively remove all PFAS (Miljösamverkan, 2022:9 and 12).

Furthermore, the results showed that surface water and groundwater adjacent to the landfills contained PFAS, often in concentrations that exceeded the Water Authority's assessment criteria (environmental quality standards for water) for PFAS-11 and PFOS. Groundwater and surface water bodies upstream of the landfills also contained PFAS. This may be because the spread of PFAS from landfills is greater than estimated and/or that there are uncertainties regarding the direction of the water flow. A more likely reason is however that there are other sources of PFAS within the catchment area. High concentrations of PFAS found in groundwater and surface water were often also found in high concentrations in leachate (Miljösamverkan, 2022:13). Figure 1 shows some highlights of the project findings.

PFAS with the highest concentrations in leachate were PFBA, PFBS, PFPeA, PFHxA, and PFOS, with a maximum concentration that varied between 22800 and 74600 ng/L and a mean value that varied between 427 and 1242 ng/L. The most common PFAS in leachate were PFHxA, PFOA, PFOS, PFPeA, and PFHpA, with a detection frequency between 96 and 99%, and PFHxS, PFBS, and PFBA had a detection frequency between 89 and 94%. At least one PFAS was present in 83–84% of all examined surface and groundwater. With a detection frequency of 53–71%, eight PFAS—PFOS, PFOA, PFHxA, PFBA, PFHxS, PFHpA, PFPeA, and PFBS—were discovered in more than half of the surface and groundwater samples analyzed (Miljösamverkan, 2022:10-13).

3.2 European Food Safety Authority guideline value

The European Food Safety Authority (EFSA) has previously carried out a risk assessment of PFOS and PFOA. In 2008, a so-called tolerable daily intake (TDI) was determined, based on animal studies, of 150 ng/kg of body weight and day for PFOS and 1500 ng/kg of body weight and day for PFOA. The TDI corresponds to the amount of these substances that a person can ingest every day throughout their life without risking any impact on health (EFSA, 2008:2-4).

According to EFSA's 2020 intake estimates, the weekly exposure of the European population to per-weekly PFAS exposure ranges from 3 to 22 ng/kg of body weight. Fish, fruit, and eggs were the foods that contributed the most to exposure to PFAS. In areas with PFAS-contaminated water, exposure to PFAS via drinking water can be a significant source (EFSA, 2020:39 and 147).

EFSA has updated its risk assessment of PFAS based on epidemiological studies and their effect on the immune system. In the latest evaluation from 2020, a joint tolerable weekly intake (TWI) is specified for four different PFAS: PFOA, PFNA, PFHxS, and PFOS. The assumption is that these PFAS have similar properties and are equally toxic. The evaluation states an overall TWI of 4.4 ng/kg of body weight per week. The TWI value is the exposure level one should maintain throughout one's lifespan (EFSA, 2020:7-8).

4. RESULTS

4.1 Global regulation on PFAS

The first reports on the risk and harm of PFAS came in the early 2000s (Giesy and Kannan, 2001; Hansen et. al., 2001). Since then, techniques have been created and are still being developed to, among other things, detect lower concentrations of PFAS in water. This has created a greater awareness of the problem with PFAS.

At the global level, the relevant instrument is the 2001 Stockholm Convention on Persistent Organic Pollutants ('Stockholm Convention'). The convention aims to protect human health and the environment from the harmful ef-

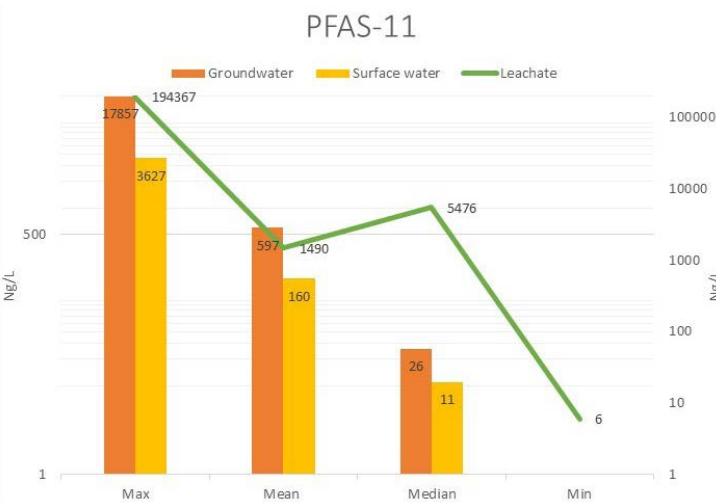


FIGURE 1: PFAS-11 found in leachate, groundwater, and surface water (Miljösamverkan, 2022:10 and 13).

fects of persistent organic pollutants (POPs) by prohibiting, phasing out, or limiting the use and production of listed substances, but also to limit or eliminate, where possible, emissions of unintentional byproducts. In addition to production and use, the convention also covers waste management and environmental monitoring (UN, 2001; UNEP, 2020). PFOS, its salts, and related substances were added to the annex of the convention in 2009 (UNEP, 2009). In 2019, the annex was updated based on new assessments on PFOS, adding PFOA, its salts, and related substances to the list (UNEP, 2019a; UNEP, 2019b). The latest PFAS addition to the convention was made in 2022 with perfluorohexanesulfonic acid (PFHxS), its salts, and related substances (UNEP, 2022). Compliance with the convention would imply a global phase-out of these three PFAS and their substances. Considering the risk profile and the risk management evaluation of PFAS, we can expect more PFAS to be added to the list in the future. Already, there is a proposal to include long-chain (C9–C20) perfluorocarboxylic acids (PFCAs), their salts, and related compounds in the Stockholm Convention Annex (UNEP, 2021).

4.2 EU regulation on PFAS

The first EU legal act relating to one PFAS substance was adopted in 2006 with regard to restrictions on the marketing and use of certain dangerous substances and preparations. Even if PFOA were suspected to have a similar risk profile, it was PFOS that were first regulated to be restricted based on the assessment that PFOS are persistent, bioaccumulative, and toxic to mammalian species (Directive 2006/122/EC, Recitals 1 and 7).

The Stockholm Convention's requirements for limiting PFOS, PFOA, PFHxS, and substances that can be broken down into these substances have been introduced into EU legislation through the Regulation on Persistent Organic Pollutants ('the POPs Regulation'). The POPs Regulation aims to protect human health and the environment by eliminating, phasing out, or limiting the production and use of POPs, as defined in the Stockholm Convention or the Protocol to the 1979 Convention on Long-Range Transboundary Air Pollution on Persistent Organic Pollutants (CLRTAP) (Regulation (EU) 2019/1021, Article 1 and Annexes; Regulation (EU) 2022/2400). Since PFOS, PFOA, and PFHxS are included in the POPs Regulation, they are prohibited to manufacture, place on the market, and use within the EU, except for certain special exceptions, including use for laboratory-scale research. The objective is to control waste that contains or is polluted by POPs and to reduce, if at all possible, POP emissions. The aim and objectives of the regulation have been formulated with the precautionary principle in mind, which means that a lack of scientific agreement regarding the risk of damage caused by the POPs must not mean that precautionary measures are not taken (Regulation (EU) 2019/1021, Articles 1, 3, and 4; Regulation (EU) 2022/2400).

Producers or holders of waste must, as far as possible, avoid contamination of the waste with the substances listed in Annex IV of the regulation. To ensure that the POP content is destroyed or irreversibly transformed, contaminated waste must, in most cases, be disposed of or re-

covered quickly (Regulation (EU) 2019/1021, Article 7). It is the responsibility of each Member State to ensure that the production, collection, and transport of contaminated waste, as well as its storage and treatment, is carried out under conditions that protect human health and the environment (Regulation (EU) 2019/1021, Article 7(6); Directive 2008/98/EC, Article 17).

The presence of PFAS in waste is regulated by limit values (Regulation (EU) 2019/1021, Annex IV). Since November 2022, limit values for PFOA and PFHxS have been added to the annex following the addition of these substances to the Stockholm Convention (Regulation (EU) 2022/2400).

The POPs Regulation Annex IV shows a Low POP Content Level (LPCL), and Annex V shows a Maximum POP Content Level (MPCL). When waste contaminated with POPs is to be destroyed or irreversibly transformed, the LPCL is specified in concentrations, in mg/kg. For PFOS and its derivatives, the limit value is 50 mg/kg; for PFOA and its salts, 1 mg/kg; for PFOA-related compounds, 40 mg/kg; for PFHxS and its salts, 1 mg/kg; and for PFHxS-related compounds, 40 mg/kg (Regulation (EU) 2019/1021, Annex IV; Regulation (EU) 2022/2400). If the waste has a level between LPCL and MPCL, it can be stored permanently in a bedrock formation, salt mine, or landfill for hazardous waste, provided that the waste has been stabilized and a decision has been made by the appropriate national authority (Regulation (EU) 2019/1021, Annex V). In Sweden, the appropriate national authority is the Swedish Environmental Protection Agency (SEPA).

Exemptions from the regulations pertaining to the destruction or irreversible transformation of PFOA and PFHxS in accordance with the higher levels in Annex V may be requested if there is a lack of accessible technology and the application is environmentally appropriate. According to SEPA's interpretation, it is not possible to grant such a dispensation for waste contaminated with PFOS since the upper and lower limits for PFOS are the same. All waste containing more than 50 mg PFOS/kg dry matter must be destroyed or irreversibly transformed. This means that the waste must be handled in such a way that PFOS is not at risk of spreading further, e.g., through incineration. In this case, recycling or disposal methods are not allowed, as this may lead to further contamination of the waste. However, waste containing less than 50 mg PFOS/kg dry matter needs to be disposed of or go through recovery operations in a manner that is compatible with current EU regulations. In general, if the waste contains concentrations below the LPCL of the respective POP substance, it does not need to be destroyed or irreversibly transformed. The waste must instead be recycled or disposed of in a way that is compatible with the appropriate EU legislation and gives the best results for the environment (Regulation (EU) 2019/1021, Annexes IV-V; SEPA, 2023; SEPA, 2024).

4.3 Landfills and PFAS regulation

The main Swedish regulation on waste landfills is the Ordinance (2001:512) on landfilling. This ordinance was adopted to incorporate the requirements of the EU Directive 1999/31/EC on the landfill of waste in Swedish law. The latter aims to prevent, or where possible reduce, the

negative effects that landfills have on land, air, surface water, groundwater, and human health (Article 1). To support the EU's transition to a circular economy, the directive was amended in 2018 (Directive (EU) 2018/850, Recital 2). One goal of the directive is to limit the proportion of landfilled municipal waste to 10% by 2035 at the latest (Directive 1999/31/EG; Directive (EU) 2018/850, Articles 1 and 5(5)). A couple of key points in the directive are that Member States must implement national strategies to try to reduce the amount of biodegradable waste going to landfills; only waste that has been treated may be deposited; and landfill owners must apply for a permit, which, among other things, must include information on methods to prevent and reduce pollution (Directive 1999/31/EG, Articles 5(1), 6(a) and 7; Directive (EU) 2018/850).

After a landfill is closed, it continues to affect its surroundings for a very long time. To limit the leakage of leachate, EU requirements for the final landfill cover are, in Sweden, transposed into the Ordinance (2001:512) on landfilling. As an example, the final cover for landfills for hazardous waste must be designed so that the amount of leachate passing through the cover does not exceed 5 L/m² per year. It is the operator's responsibility to ensure that the limit is not exceeded. Landfills in operation must be protected against surface and groundwater entering the landfill (Swedish Ordinance 2001:512, Para. 23 and 31).

There are three categories of landfills. The division is based on the type of waste that is landfilled: hazardous waste, non-hazardous waste, and inert waste (Directive 1999/31/EG, Article 4; Swedish Ordinance 2001:512, Para. 7). A correct classification of the waste is important. To assess whether a waste has hazardous properties, Annex III to Directive 2008/98/EC on waste ('the Waste Framework Directive') is applicable (Swedish Waste Ordinance 2020:614, Ch. 2 Para. 2). When a waste contains a substance that is classified with at least one of the hazard class codes found in Annex III of the Waste Framework Directive, it shall be classified as hazardous waste. The substance PFOS is reproductively toxic (H 10), and the limit for when waste with PFOS content must be classified as hazardous is therefore set at 0.3% (3000 mg/kg). The limit for when a waste is classified as hazardous in terms of PFOS, PFOA, and PFHxS can thus be significantly higher than what follows from the requirements for destruction or irreversible conversion according to the POPs regulation (SEPA, 2023).

SEPA's regulations on disposal, criteria, and procedures for receiving waste at waste disposal facilities (NFS 2004:10)⁵ contain acceptance criteria and limit values for waste that is deposited. The regulation currently has no acceptance criteria for any PFAS substance. However, before waste is deposited, the operator must have acquired as much knowledge as possible of the waste's composition, leachability, and other properties, as well as the effects of the waste in general and in the long term (Swedish Ordinance 2001:512, Para. 16).

Waste that contains pollutants that lack acceptance criteria must nevertheless be examined to assess whether it meets the landfill class in which it has been classified (NFS 2004:10, Para. 4). The limit values for PFOS, PFOA,

and PFHxS, according to the POPs regulation, must therefore be taken into account when receiving waste at a landfill. There are no general guidelines on what requirements should be set for any substance in leachate that, after local treatment, is released to the recipient or is led to a sewage treatment plant. However, the landfill owner must have knowledge of the PFAS levels found in leachate leaving the landfill. Special precautions such as stabilization, covering of the masses, deposition in separate cells, etc. may be necessary. These measures must be assessed and tested in each individual case, in keeping with the precautionary principle. Conditions are set for each landfill during the permit assessment in accordance with the Swedish Environmental Code (SEC) (SEPA, 2023).

With the support of Ch. 9 SEC and the Swedish environmental assessment ordinance (2013:251) regarding environmentally hazardous activities and health protection, a permit for landfill operations may be granted. The application for a permit is assessed based on the general rules of consideration, according to which requirements for knowledge, caution and resource management can be set, and results in concrete conditions for landfill operations (Ch. 2 SEC). Common conditions include that leachate from the facilities must be collected and stored in ponds, and the impact of the emissions on the recipient must be controlled. Landfilling can also constitute water operations that, in connection with the design of landfills and the handling of leachate, may be subject to a permit or notification requirement following Ch. 11, s. 9 and 9a SEC. The leachate itself is considered waste (Swedish Ordinance 2020:614, Annex II).

There are currently only a few landfills that have conditions in their permits for the release of PFAS. According to Environmental Cooperation Sweden, there are only conditions or probationary conditions regarding PFOS for three landfills in Sweden. In these cases, the PFOS content in outgoing water must not exceed 80–100 ng/L. The project team currently assesses that around 20 ng/L of PFAS in outgoing leachate can be seen as a benchmark for when the supervisory authority should require an investigation of appropriate measures. However, an assessment should be made in each individual case. Lower levels may be appropriate for a particularly sensitive recipient, and higher levels may be acceptable under certain conditions (Miljösamverkan, 2022:26 and 35).

The situation is, however, largely characterized by the absence of clear guidelines for the authorities. On top of that, in the few cases where limit values have been set, there is disagreement regarding what the appropriate level is. An example is the judgement from 2021 of the Swedish Environmental Court of Appeal in case M 2164-19. The case concerned conditions for a landfill, where the County Administrative Board in Jönköping County demanded a limit value for PFOS of 20 ng/L to ensure that there is no deterioration of the status of the water body with regard to PFOS and the environmental quality standards (EQS) based on calculations of dilution in the recipient. In their judgment, the court issued provisional regulations stating that the PFOS content as an annual average may not exceed 80 ng/L in outgoing treated water.

In the event that no limit values are set, the supervisory authority always has the opportunity to issue injunctions and prohibitions that are needed in an individual case, with the support of Ch. 26 and Ch. 2 SEC. The supervisory authority could, for example, issue an injunction for control of a waste facility and its leachate.

For waste treatment facilities, there are requirements according to the EU's Best Available Technique (BAT) conclusions for waste treatment (Decision (EU) 2018/1147). These provisions are mainly implemented through the Swedish Industrial Emissions Ordinance (2013:250). According to the BAT conclusions, PFOS and PFOA in water discharges from waste treatment facilities must be monitored once every six months (Decision (EU) 2018/1147, Annex). The BAT conclusions apply in parallel with the requirements following the permit assessment. A specific BAT requirement is also enshrined in the SEC, which means that requirements regarding BAT can be set based on both the SEC (Ch. 2 Para. 3) and the BAT conclusions.

A number of Swedish sewage treatment plants receive leachate from landfills. The measures required for these plants are based on the requirements set according to the voluntary certification system Revaq⁶. Normally, leachate from landfills should not be connected to Revaq-certified sewage treatment plants; instead, the leachate must be disconnected, and reconnection is not permitted after the certificate is granted. To be allowed to maintain a connection to a certified sewage treatment plant, the disconnection must lead to the raw water protection being damaged or a recipient for the leachate being missing, at the same time as the leachate must be purified to an acceptable level according to the Revaq manual for assessment of leachate. So far, the manual only contains levels and tolerable allowances for PFOS⁷ (Allmyr and Sternbeck, 2017:4, 8, and 12; Miljösamverkan, 2022:34).

4.4 Water and PFAS regulation

The EU Water Framework Directive (WFD) was adopted in 2000 and aims to protect and improve all water within the EU (Directive 2000/60/EC). The directive is supplemented by three subsidiary directives: The Priority Substances Directive (Directive 2013/39/EU), which specifies limit values and regulates 45 priority substances to be investigated in water management; the Groundwater Directive (Directive 2006/118/EC), containing criteria for assessing good chemical groundwater status and for detecting and reversing upward trends in the occurrence of contaminants; and the Drinking Water Directive (Directive 2020/2184). In Sweden, the WFD was introduced into Swedish legislation in 2004 through Ch. 5 SEC, the Swedish Ordinance (2004:660) on management of the quality of the water environment, and the Swedish Ordinance (2017:868) with county board instructions.

Based on the WFD, the Water Authorities⁸ in Sweden have established EQS for the priority substances that are specified in the Priority Substances Directive, which only lists one PFAS, PFOS. The Swedish Agency for Marine and Water Management has, among other things, the task of protecting, restoring, and ensuring that fresh water and sea resources are used sustainably. In their regulation (HVMFS

2019:25), an EQS for PFOS in surface water is set to 0.65 ng/L as an annual average value (Annex 6), while for surface water that is a source of drinking water, there is an EQS for PFAS-11 at 90 ng/L (Annex 5(4)). These limit values are legally binding for the water authority's risk, status, and action assessments. Leachate is mainly formed by rainwater that infiltrates the landfill, but in old landfills, ground and surface water seep into the landfill and thus form leachate. If the soil beneath the landfill is unsaturated with moisture, such as in an inflow area, the leachate can infiltrate through the soil, and in this way, some contaminants are retained in the soil while others are transported further and risk contaminating the groundwater (SEPA, 2008:5).

When it comes to PFAS in groundwater, in 2016, the Water Authorities produced EQS for the total concentration of PFAS-11. At a concentration above 90 ng/L, the groundwater body was considered to have an unsatisfactory status. Already at 18 ng/L, i.e., at a concentration sum of 20% of the EQS, measures were to be taken to reverse the trend (SGU-FS 2013:2; Swedish Water Authorities, 2016). In 2022, the European Commission presented a proposal for a revision of the WFD, the Priority Substances Directive, and the Groundwater Directive with the aim of improving the protection of EU citizens and the environment by, among other things, enabling a faster process of inadequately managing chemical threats (European Commission, 2022). From March 2023, a new general threshold value for PFAS in groundwater applies within the Swedish water administration, which is based on the commission's proposal. In the Geological Survey of Sweden's regulation on mapping, risk assessment, and classification of groundwater status, it is established that the new general threshold value pertains to the weighted sum of PFAS-24 and is set at 4.4 ng PFOA equivalents/L. The threshold value means a total sum of all PFAS content after these have been individually converted to PFOA equivalents based on the substances' toxicity factors (SGU-FS 2023:1, Annex 3). This threshold value has no legal effect but will exist as support for the Water Authorities in their work. The guidelines decided by the Water Authorities for the sum of PFAS-11 are legally valid until 2027 at the latest when the Water Authorities next decide on threshold values (KEMI, 2023a:2).

At the end of 2020, a new directive (2020/2184) on the quality of water intended for human consumption was adopted. The goal is to protect human health from the harmful effects of all kinds of contamination of drinking water (Recital 2). Limit values for PFAS-20 were introduced in the directive (Annex III, Part B). Member States must have taken the necessary measures by January 12, 2026, at the latest to ensure that the drinking water is compatible with the new, legally binding limit values (Article 25). It is estimated that around 20 million people in Europe may be exposed to drinking water with higher PFAS levels than the new recommended limit values determined within the EU (KEMI, 2023b).

In the new drinking water directive, a parameter value of 100 ng/L is set for PFAS-20 (Directive 2020/2184, Annex I, Part B). This legislation entered into force in Sweden on January 1, 2023, and applies to municipal drinking water and drinking water from public and commercial business-

es (LIVSFS 2022:12, Para. 2). The Swedish Food Agency's previous, non-legally binding recommendations for drinking water referred to PFAS-11, with an action limit of 90 ng/L and an upper guideline value for when you should avoid drinking the water of 900 ng/L (Swedish Food Agency, 2021:10). Based on EFSA's statement from 2020, Sweden has now chosen to set a national guideline value of 4 ng/L for PFAS-4 and 100 ng/L for PFAS-21, applicable from January 1, 2026 (LIVSFS 2022:12, Annex 1). As a comparison, it can be mentioned that the Danish Environmental Protection Agency has chosen to lower the limit value to 2 ng/L for PFAS-4 (Danish Miljøministeriet, 2021).

In 2013, it was discovered that the drinking water for residents in Kallinge, in southern Sweden, and the surrounding area and the northwestern parts of Ronneby municipality were contaminated with high levels of PFAS. The contamination had spread from a nearby fire drill area to the ground, the groundwater, and on to the drinking water. The levels of PFAS measured in the drinking water were among the highest measured in the world, at 10,380 ng/L. A group of 165 people came together to sue the municipal water company for the health risks of living with high levels of PFAS in the blood. Ten years later, in December 2023, the Swedish Supreme Court announced that the municipal water company was responsible for personal injuries in the form of elevated levels of PFAS in the blood, which means physical deterioration of the body, and must therefore compensate the plaintiffs (Case T 486-23).

5. DISCUSSION

In this paper, we have highlighted the legal situation regarding PFAS in landfills. We have focused on the regulation of waste and water in Sweden. This includes an analysis of the functionality of the legislation when it comes to the management of PFAS in landfills in relation to the overarching goal of protecting human health and the environment.

When it comes to the specific legal regulations around PFAS and landfills, the study points to some central problems: (1) production, we continue to send waste containing PFAS to landfills; (2) old landfills, which are not in use but pose a high risk of spreading PFAS; (3) landfills in operation, which comply with current regulations but have large concentrations of PFAS in their leachate (and surrounding water); and (4) PFAS propensity for long-range dispersion. What we have concluded is the following:

The precautionary principle is a well-established principle that permeates both international, EU, and national law. The principle briefly means that precautionary measures must be taken when an activity poses a risk to human health or the environment. It goes along with the idea that it is "better to be safe than sorry" and entails that although scientific evaluations cannot assess the risk with certainty, precautionary measures should be taken to avoid a potentially hazardous effect of, for example, a product or process. Provided that measures taken are proportionate to the desired level of protection, it is not discriminatory and is consistent with previous measures in similar situations (Pettersson & Johansson, 2022). We do not have knowledge about all PFAS, but we do know that PFAS are per-

sistent in the environment, bioaccumulative and that some may be toxic to humans and animals. Although PFAS may not cause acute problems, we know that long-term exposure implies a high risk of several serious health problems in humans. Despite this, so far, only three PFAS are banned within the EU. Moreover, there is no comprehensive legislation that covers all PFAS; instead, individual PFAS are regulated at different legislative levels. Often, when an individual PFAS is regulated, that substance is replaced with another unregulated PFAS. There are simply too many substances, and the process of deciding whether to add a substance to a restriction or ban list takes too long. In that way, the legislation has no effect when it comes to reducing the risk.

There are currently two EU proposals submitted regarding PFAS. One of the proposals concerns banning PFAS in fire-fighting foam (ECHA, 2022), and the other is a restriction proposal that intends to treat PFAS as a group and manage PFAS uniformly within the EU (ECHA, 2023). Fire-fighting foam containing PFAS has caused serious environmental contamination, both of soil and drinking water. The proposal to ban PFAS in fire-fighting foam would mean that no further PFAS are spread in that way. What remains is the problem of contamination that already exists and that needs to be managed. If we do not act, it is estimated that around 4.4 million tons of PFAS will end up in the environment in the next 30 years (ECHA, 2023). The proposal to treat PFAS as a group would prevent the problem of one substance replacing another as soon as one is restricted. The positive effects of PFAS restrictions are already noticeable, and the concentrations of PFOS seem to have decreased on the Swedish west coast. Studies from the national monitoring program for environmental toxins in biota showed a decrease in PFOS levels in herring liver at the sampling station Fladen during the period 2004-2016 (Bignert et al., 2017:286). However, even if PFAS are banned or restricted from now on, the problem of contaminated waste will remain for a long time. The waste that we have already generated will not disappear. We must thus both reduce the PFAS emissions that occur today and find ways to handle PFAS pollution from existing sources, such as landfills.

This brings us to the problem of environmental risks associated with old landfills that are not in operation. It is estimated that there are several thousand older landfills in Sweden that were closed before the current regulation came into force. These have been closed according to earlier, and much lower, requirements for landfilling or before requirements for protective measures even existed. As a result, the environmental protection surrounding these landfills is generally very low. As an example, there is no or only limited monitoring of leachate (SGI, 2014:13). The responsibility to inventory closed landfills and assess the risk to human health and the environment lies with the municipalities (NFS 2020:6, Para. 17). Although this work must be included in each municipality's waste plan, the pace of the inventory varies between municipalities because it can be costly, it requires commitment, and it is not clear how the risk assessment should be made or how the result should be implemented. Another issue concerns who can be held legally responsible for the remediation of a contaminated

area. Even if the starting point is the Polluter Pays Principle (PPP), it is in practice often complicated to, for example, determine who is the polluter, or to find the polluter. This will however not be discussed further here.

The previously set EQS for PFOS and PFAS-11 are exceeded in some water bodies today. This entails the risk that PFAS contaminates our ground and drinking water and that the problem may increase over time. This brings us to the problem of active landfills that comply with current regulations but that, despite this, still have high levels of PFAS in their leachate, which spreads to the surrounding waters. The regulation for PFAS in waste only regulates three PFAS so far, and the threshold for waste to be considered hazardous is high. As there are no national guidelines for treatment and permit levels for PFAS in leachate, it is difficult for the authorities to regulate PFAS in the operator's permit, although it has become more common for authorities to request measurements of PFAS in leachate and recipients. It is important to set requirements for supervision and control of PFAS, and the regulation needs to be both clearer and more uniform. These shortcomings point to the necessity of information regarding PFAS, including education, innovation, and voluntary measures.

The problems with PFAS contamination have gained increasing attention lately. In addition to the fact that the media has been diligent in reporting and informing about what PFAS is and how it spreads, there are regularly published news stories such as "Swedish crayfish can be stopped—too high levels of Pfas"⁹, which made headlines in Sweden during the crayfish season in August 2022. Another interesting headline from March 2023 was "New research: 43 types of PFAS in common toilet paper"¹⁰. There is also a great focus and interest in research, both in conducting studies of PFAS effects and in the development and evaluation of various treatment techniques, not least within the waste industry. In addition to legal measures, voluntary measures and actions will be crucial to achieve sustainable development for future generations and to meet both national and EU environmental goals such as a toxic-free environment, responsible management of chemicals and waste, and good health and well-being.

Environmental Cooperation Sweden has reported that in Sweden there is currently only one full-scale waste management facility that treats PFAS in leachate, but that several new ones are in the works. At the Telge Recycling plant in Södertälje, Sweden, they use foam fractionation, and the degree of leachate purification is at least 90% for 6:2 FTS, PFDA, PFHpA, PFHxS, PFNA, PFOA, and PFOS. However, for PFBA, which is a short-chain substance that occurred in the highest concentrations but was also one of the most common in the project findings, the degree of purification was 0% (Miljösamverkan, 2022:28). This shows that different purification techniques are effective for different PFAS, and to achieve an acceptable result, a combination of several techniques may be required, and this needs to be further studied. This is crucial to subsequently be able to set conditions on treatment in the landfill's permit.

The last problem we intend to highlight is the fact that PFAS tend to travel long distances, both via air and water. PFAS is thus an international problem, and concerted ef-

forts to deal with the problems that these substances cause are necessary. A study from 2022 shows that the levels of PFAS in rainwater often exceed both the US Environmental Protection Agency Lifetime Drinking Water Health Advisory, the Danish limit value for drinking water, and the EQS for EU surface water (Cousins et. al., 2022). Since leachate is mainly caused by rainwater infiltrating landfills, the fact that rainwater contains high concentrations of PFAS is a serious problem. This circles us back to the problem (1) of production and the importance of regulating the use of PFAS already at the production stage to try to reduce the load, and the risk it poses, with the concentrations that are now in the environment. Chemicals are needed in our society, but to achieve environmental goals, chemical policy must be reformed, and innovative chemistry must be encouraged.

6. CONCLUSIONS

The revision of the health-based guideline value from EFSA (EFSA, 2020) shows that PFAS are more dangerous than previously estimated and that it is crucial that the risk of long-term exposure to PFAS is minimized. Most of the Swedish assessment bases for limit and guideline values build on EFSA's tolerable daily intake for PFOS from 2008 (EFSA, 2008), and these need to be updated. Today, three PFAS in waste are regulated through the EU Regulation on Persistent Organic Pollutants (2019/1021), and to assess whether a waste has hazardous properties, Annex III to Directive 2008/98/EC on waste is applicable. The problems with PFAS are receiving more and more attention, not least in the waste industry, with the development and evaluation of effective purification techniques.

Since it takes time to implement updates within the legal system, a large responsibility is placed on the authorities that assess and issue permits for water operations or environmentally hazardous activities, including landfills. While it is possible to regulate purification and permitted levels of PFAS in leachate through conditions in permits for water operations and environmentally hazardous activities in accordance with the Swedish Environmental Code, this opportunity is rarely used. Here, a change needs to take place; acceptable levels of risk based on scientific knowledge need to be established, and the assessment of individual permits needs to be aligned with these requirements.

From a global perspective, international treaties and agreements provide a framework for countries to work together to address the environmental and health risks associated with PFAS. However, there is a need for more research on the global impact of PFAS contamination and the effectiveness of international agreements, such as the Stockholm Convention, in addressing these issues. It will require more collaboration and learning from each other as we move forward.

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¹ All environmentally hazardous activities that require a permit in accordance with Ch. 9 of the Swedish Environmental Code must submit an environmental report every year to the Swedish County Administrative Board. Environmental reporting aims to monitor and follow up on businesses adherence to the conditions specified in their applicable permit, as well as enable them to enhance their environmental work (Swedish Environmental Code, Ch. 26 Para. 20).

² Swedish database with a manager's guide on contaminated areas. Everything from modern landfills in operation to smaller, closed landfills is registered. EBH-portal is operated by the Swedish County Administrative Boards. <https://www.ebhportalen.se/>.

³ As an example, in 2013, among the world's highest levels of PFAS were measured in people in Kallinge, Sweden. The highest level was 1800 ng/ml, the lowest level was 91 ng/ml, and the average level was about 600 ng/ml. A "normal" person can have about 10 ng/ml of PFAS in the body (Case T 486-23).

⁴ A collaboration between county administrative boards, the Swedish Environmental Protection Agency, the Swedish Board of Agriculture, and the Swedish Agency for Marine and Water Management. The goal of the coalition is to increase consensus between the different county administrative boards and thereby create a more uniform exercise of authority through supervisory guidance, supervision, and assessments in the area of Swedish environmental law.

⁵ SEPA's regulation is designed in accordance with the Landfill Directive (Council Directive 1999/31/EC of 26 April 1999 on the Landfill of Waste) and the Council Decision of 19 December 2002 on procedures for the reception of waste at landfills in accordance with Article 16 and Annex II to Directive 1999/31/EC. The directive is a minimum directive, and the established acceptance criteria are not exhaustive.

⁶ Revaq is a certification system that aims to improve the quality of plant nutrients from sludge by reducing the flow of hazardous substances to sewage treatment plants.

⁷ In Revaq's manual, the reporting limit for PFOS to be investigated and evaluated in leachate is set at 5 ng/L. The assessment system is based on how leachate affects the sludge quality and its risk of negative effects on health and the environment. In sludge, PFOS is set at a low risk level of 0.12 mg/kg dry substance (Allmyr and Sternbeck, 2017:8 and 12).

⁸ Sweden is divided into five different water districts, based on the borders of the major sea basins and catchment areas, and in each district, the government has appointed one of the County Administrative Boards to act as water district authority. <https://www.vattenmyndigheterna.se/om-vattenmyndigheterna.html> (accessed 2023-06-02).

⁹ Swe: Svenska kräftor kan stoppas – för höga halter Pfas. Title on Swedish TV4 on September 22, 2022. <https://www.tv4.se/artikel/67sfKlvFP6Uy-Y6rOUqMxSj/svenska-kraeftor-stoppas-foer-hoega-halter-miljoegift> (accessed 2023-06-22).

¹⁰ Swe: Ny forskning: 43 sorter PFAS i vanligt toalettpapper. Title in Today's PS Life Science (Swe: Dagens PS Life Science) on March 17, 2023. <https://www.dagensps.se/life-science/ny-forskning-43-sorter-pfas-i-vanligt-toalettpapper/> (accessed 2023-06-22).

THE TACIT END-OF-WASTE STATUS IS de facto USED IN THE EU AND CRUCIAL FOR THE CIRCULAR ECONOMY

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ABSTRACT

For a well-functioning circular economy, recovered materials with a specific purpose, that have a market, and meet the legal requirements for both products and the environment should have a product status. However, at EU level, EoW has only been established for certain metals and cullet glass, and implementation at national level has slowed down. One potential solution is tacit EoW, supported in our opinion by articles 6(1) and 6(4) of the Waste Framework Directive (WFD), which can be understood as placing responsibility on economy operators with possible oversight by authorities. Such an interpretation is not currently accepted by the authorities in most European countries, where a formal "case by case" authorization is at least required. This position paper examines the status of various waste streams in the EU and illustrates that tacit EoW is already widely used (1 billion of tonnes?) in regulation and waste management practices. This includes: Reuse of material as a product after preparation (level 2 of the waste hierarchy); Recycling of materials (level 3 of the waste hierarchy) to produce new products; Recycling in environmental applications, such as fertilisers, soil conditioners, and growing substrates, as well as road base-ment and backfilling operations; Energy production from various biomass and wood waste categories. Granting tacit product status consistent with level 3 of the waste hierarchy to other materials processed by certified industrial facilities would benefit the circular economy. Ten materials ranked by the EU Joint Research Centre could immediately benefit of tacit EoW.

1. THE IMPLEMENTATION OF THE END-OF-WASTE PROCEDURE

The advantages of EoW status are greater use of recycled materials in products (with certified quality) and therefore less new consumption (energy, raw materials, oil, water, etc.) and fewer emissions (waste, CO₂, used water ...) while increasing EU's strategic autonomy.

Not only the recycling industry asks for that status but also the producers. To reflect the whole value chain perspective, in a document *"No time to waste: EU-wide end-of-waste criteria for recycled rubber from tyres"* (EuRIC and ETRMA 2022), it is stated that *"Today, rubber circularity from end-of-life tyres (ELT) is at stake, and the tyre value chain (from producers to recyclers) is ready for taking this step at EU level right now."* (Figure 1).

The Waste Framework Directive of 2008 revised in 2018 (EU 2018) defines criteria for end-of-waste status: existence of commonly used specific applications, existence of a market or a demand, fulfilment of technical requirements for the specific applications (meeting existing legislation and standards applicable to the products the

waste streams substitute), and the absence of any overall adverse environmental or human health impacts in the intended use.

According to many authors, the process of regulatory EoW has halted or at least slowed down since 2016. A review is presented in (Hennebert et al., 2024a), and the reasons are: *"... excessive slowness, excessive precautionary principle, a part of arbitrary commission "expert" judgment who's some can be opposed to, delegation to local authorities that have not always the expertise, single waste by single waste approach with individual (small business) demanders that have not always the resources for best quality dossiers"*.

Twenty-seven years ago, before the Waste Framework Directive of 2008, the need for inclusion of resource in the definition of waste was already stressed (Bontoux and Leone, 1997; Pongrácz and Pohjola, 1997). The experts of EU Joint Research Centre (JRC) proposed a distinction between waste and non-waste (Bontoux and Leone, 1997). The JRC provides independent, evidence-based knowledge and science, supporting EU policies to positively impact society (EC, 2024). The JRC did many propositions since

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Key benefits of EU-wide EoW for recycled rubber from tyres



FIGURE 1: Key benefits of EU-wide end-of-waste for recycled rubber from tyres (European Recycling Industries' Confederation - EuRIC and European Tyre & Rubber Manufacturers Association - ETRMA 2022).

1997. In a report on end-of-waste criteria for waste paper: *"The Directive introduces the possibility that certain waste streams having undergone a recovery operation and fulfilling certain criteria – so-called End-of-waste criteria – can cease to be waste."* (Villanueva and Eder, 2011). In a report on end-of-waste criteria for biodegradable waste subjected to biological treatment (compost & digestate): *"The purpose of end-of-waste criteria is to avoid confusion about the waste definition and to clarify when certain waste that has undergone recovery ceases to be waste. Recycling should be supported by creating legal certainty and an equal level playing field and by removing unnecessary administrative burdens."* (Saveyn and Eder, 2013). These suggestions have been included as unsaid end-of-waste in the Fertiliser Regulation 1009/2019/EU (EU 2019). In a report on end-of-waste criteria for waste plastic for conversion: *"The purpose of end-of-waste criteria is to avoid confusion about the waste definition and to clarify when certain waste that has undergone recovery ceases to be waste. Recycling should be supported by creating legal certainty and an equal level playing field compared to virgin material production, and by removing unnecessary administrative burdens. The end-of-waste criteria are defined as to provide a high level of environmental protection and an environmental and economic benefit to the recycling chain of the material under study."* (Villanueva and Eder, 2014). In a report of JRC of 2022 ranking priority of different waste streams for EoW status, *"This study supports the implementation of both the Circular Economy Action Plan and the Waste Framework Directive by identifying a list of priority waste or by-product streams (scoping) and deriving the most suitable candidate streams for which to develop further EU-wide end-of-waste or by-product criteria."* (Orveillon et al., 2022).

On the other hand, a more liberal approach, rooted in responsibility, self-regulation, and a risk-based strategy, encourages reuse and recycling (Hennebert et al., 2024a). Striking the right balance between a centralized precautionary stance (*"The administration rejects due to*

potential hazards to people and the environment...") and a decentralized approach that promotes industry-led, risk-conscious solutions for reuse or specific applications in secondary materials is a pressing question. In a report for the European Commission (EC, 2020), the German Federal Environment Agency and ARCADIS Belgium proposed that the economic operators take the responsibility of the end-of-waste status, without prior approval by the authorities, and with ex-post control if necessary: *"Simplifying certain aspects of the decision-making process by referring to the 'ex-post' assessment of the EoW decisions taken by the economic operators."* (p. 131).

Different solutions to circumvent/solve the halting of EoW were proposed: *"non-waste", "waste use potential", "certified material", "non-waste product", "secondary raw material", "premat"* (synthesis in Hennebert et al. 2024a). In any case, creating a third category between waste and product is legally and practically unfeasible. The best option seems to move from waste to product status by a *"tacit"* end-of-waste procedure.

A more demanding solution can be considered: *"We argue that in a longer-term perspective the aim should be to erase the border between waste and chemicals regulation and create a single regime for the regulation of materials and their flow"* (Alaranta and Turunen, 2021). However, simultaneously advocate additional transitional precautionary measures, without any specific arguments of effective circulation of toxic substances. Matter benefiting of EoW procedure are products (articles or preparation and substances). The product regulation is continuously updated and more stringent. In name of potential hazard and risk (probability or frequency of exposure and damage of a target by hazardous substances in the intended use), the authors are in effect jeopardising their *"longer term perspective"*.

Recycled materials fulfil specifications (under the responsibility of the producer) to be sold. These materials actually become products when used by the buyer.

Examples are old papers, metal scraps, garden and hedges materials. This procedure happens without ex-ante control or administrative authorization. It is in fact a tacit EoW procedure.

The objective of this position paper is to show that tacit EoW is in fact already very widespread for hundreds of millions of tonnes of materials in the EU. To demonstrate this, it is necessary to broaden the scope and include combustibles, fertilizers, road foundations materials, contaminated soils, technosols, even fumes and treated water. The whole image then becomes obvious and the horrible notion of waste destined to be landfilled (far from consumers) weakens. The main point of this paper is that hundreds of millions of tonnes of materials are managed by specific regulations equivalent to EoW or by unsaid tacit EoW. It should be understood and recognised as such and applied to other resources streams.

The interpretation that articles 6(1) and 6(4) of the Waste Framework Directive (WFD) can be understood as placing responsibility on economy operators with possible oversight by authorities is not currently accepted by the authorities in most European countries, where a formal “case by case” authorization is at least required. For the proposed interpretation to become generally accepted, a modification of the WFD or clear Guidelines from the Commission should be necessary.

2. WHAT SHOULD BE THE TACIT EOW PROCEDURE?

The law “for the green industry” in France in 2023 (FR 2023) gives to the products of production facilities made from waste an end-of-waste status (with the same conditions as in the Waste Framework Directive) without prior approval by the administration, that can ex-post control their actions, including on requirement of NGOs or industries.

The proposition of the German Federal Environment Agency and ARCADIS Belgium (EC, 2020) and the French law are in line with the Article 6(1) of the WFD (EU, 2018) which states: “*Member States shall take appropriate measures to ensure that waste which has undergone a recycling or other recovery operation is considered to have ceased to be waste if it complies with the following conditions...*” and in line with the Article 6(4) which states: “*Where criteria have not been set at either Union or national level under paragraph 2 or 3, respectively, a Member State may decide on a case-by-case basis, or take appropriate measures to verify, that certain waste has ceased to be waste on the basis of the conditions laid down in paragraph 1...*”: “verify that certain waste has ceased to be waste” means that the regional producers or a single producer can claim the end-of-waste status and the authority checks whether the conditions are met. In our opinion, ex-post decision making would fall under the category of appropriate measures of articles 6(1) and 6(4). This is not the present legal situation as explained above.

The European Recycling Industries’ Confederation (EuRIC) pledge for a tacit end-of-waste status of recycled matter at the exit of certified industrial facilities (EuRIC 2019, 2020, 2022, 2023, Hennebert et al., 2024a).

This question is open to discussion since 2017 during the consultation on Chemical, Product, Waste interface (EC, 2017). The European Consumer Organisation BEUC pledge for an active monitoring and control: “*We call on the Commission, in cooperation with Member States, to ensure that only safe recycled materials of known quality is placed on the market. ... For this reason, robust end-of-waste criteria that strike the right balance between encouraging recycling and avoiding re-injecting hazardous substances into the economy are required*” (BEUC, 2017).

The tacit EoW should apply at EU level: “*The difference in national EoW criteria imposes unequal costs on [end-of-life tyres] ELT recyclers, depending on the testing requirements of the national criteria. Some countries have more stringent requirements for chemical testing of batches whereas others do not. The lack of alignment and absence of EoW criteria in some Member States also leads to issues with exports from the EU: those Members States with ELT EoW criteria are able to export ELT as a product without the need to comply with waste shipment regulations and hence have a competitive advantage.*” (EuRIC and ETRMA, 2022).

2.1 The case of hazardous waste

Removing waste status results in fewer administrative constraints, particularly for hazardous waste, which must be closely controlled. Hazardous means for waste that the limits of one or more of the 15 hazard properties HP 1 to HP 15 and the POP concentration are exceeded (summary in Hennebert, 2019). With a calculation method close to the one of products, 50% of the hazardous waste are hazardous by ecotoxicity HP 14 ‘Ecotoxic’ and 20% are toxic for humans HP 6 ‘Acute toxic’ (Hennebert et al., 2016).

Non-hazardous waste is neither explosive nor oxidizing nor combustible (test of flame propagation speed) nor toxic for humans nor ecotoxic nor POP. It therefore does not present any risk whatever the use. Whether non-hazardous materials leaving recycling facilities circulate under waste status or under product status is ultimately only of administrative importance.

Hazardous waste must be used only for defined purposes so that the risk(s) for humans and the environment are acceptable. These hazardous materials leaving the recycling facilities will circulate under publicly tightly controlled waste status (including the waste tracking slip). In the event of end-of-waste status, these materials will circulate under the more self-controlled status of product. It is therefore logical and necessary that the EoW declarations or authorisations for hazardous waste are more subject to control by the administration, as provided for by the law “for green industry” in France in 2023 (FR 2023).

3. TACIT EOW FOR DIFFERENT CASES OF REUSE AND RECYCLING

This section presents different cases of reuse and recycling, and the tacit status of associated waste. The different cases are presented at Figure 2.

In the WFD, “*material recovery*” means any recovery operation, other than energy recovery and the reprocessing

End-of-waste cases: Situation and Propositions

Tacit EoW?

✓ = should have

Reuse as products (after preparation, reparation, cleaning, decontamination...)	Done
• Compliance with product specifications (including migration rate)	
Recycling up to product	Done
• Compliance with product specifications (including migration rate)	
Recycling as material for products not in contact with the environment	✓
• Compliance with product specifications (including migration rate); EU level: metals,, glass or National level	
Recycling as industrial product in contact with the environment	✓
• Construction material, granular solids: compliance with Regulation 305/2011 including release of substances (CEN TC 351, recent methods). Also used: compliance with acceptance of inert waste landfill	
Recycling in the environment as	
• Fertilisers, soil improver, growing medium, biowaste-derived compost, digestate: compliance with Regulation 1009/2019/EU with nutrient content and environmental standards (metals, microbiology)	Done
• Backfilling: non-hazardous waste (non ecotoxic – Waste Framework Directive 2008/98/EC revised 2018), and in practice compliance with acceptance of inert waste landfill	Done
• Treated NH contaminated soils (Immobilisation of elements and destruction of organics): as « Backfilling » ?	✓
• Created NH soils or substitute of vegetated topsoil (technosols): proposed as « Backfilling » + biological assessment	✓
• Waste waters after treatment (out of scope of the EU Waste Framework Directive)	Done
• Fumes and gaseous effluents emitted into the atmosphere after treatment (idem)	Done

FIGURE 2: Different cases of reuse and recycling, and the tacit status of associated waste (NH = non-hazardous).

into materials that are to be used as fuels or other means to generate energy. It includes, inter alia, preparing for re-use, recycling and backfilling” and ““backfilling” means any recovery operation where suitable non-hazardous waste is used for purposes of reclamation in excavated areas or for engineering purposes in landscaping. Waste used for backfilling must substitute non-waste materials, be suitable for the aforementioned purposes, and be limited to the amount strictly necessary to achieve those purposes”. So, material recovery includes preparation for re-use, recycling, backfilling and other operations excepted fuel or energy production.

3.1 Case of re-use as product after preparation

For objects or materials that are reused (eventually after cleaning – e.g. textile or after industrial decontamination – e.g. active coal), the compliance with the specifications of the corresponding product is fulfilled. It is in fact a tacit end-of-waste procedure.

3.2 Case of recycling up to products

This case is the recycling (separate collecting, sorting, cleaning, eventual first transformation) of material from waste as raw material for new product. It is similar to 3.1.

3.3 Case of recycling as material for products not in contact with the environment

For “waste” (resources) that are recycled into industrial or consumer product not in contact with the environment, the compliance with the specifications of the corresponding product will be achieved and is legally sufficient. Due to legislation on products and protection of environment and human health, the products normally

have no adverse effects on health or the environment under their recommended use. These wastes can be hazardous in the sense of the waste hazard classification. The intended use will limit the exposure and hence the risk. For example, for plastics and textile fibres, some additives make the plastic hazardous when they are used at their functional concentrations. From 233 additives, it has been found that 63 additives at their functional concentration (= 27% of 233) make plastic hazardous (Hennebert, 2023). Substitutes are most of the time available. The principle of essential use (use only when no alternative is available) should be promoted. Specific methods and tests for each material and each use have been developed. For instance, the migration rate is defined in product regulations.

For end-of-waste criteria, compliance with the specifications of the corresponding product is legally sufficient. The products normally have no adverse effects on health or the environment under their recommended use. It could be not necessary to develop specific methods and tests. For products not in contact with the environment, specifications elaborated by the producers are available and the recycler “only” has to comply. Consequently, tacit EoW status should be granted for materials sold at the output of certified recycling facilities (EuRIC, 2023; Hennebert et al., 2024a).

3.4 Case of recycling as product in contact with the environment

A risk assessment on end-of-waste criteria is justified in the event of use in contact with the environment. There are already, for example, specifications for building materials in Regulation 305/2011 (EU 2011), developed in CEN TC 351 (Construction products: Assessment of release of

dangerous substances). The concentration limits in solid and leachate used for acceptance into inert waste landfill, without any impermeable barrier (Decision 2003/33/EC, EC 2003) are widely used for different materials in contact with the environment. A leachate means in this context the initially deionised water that has passed through a column of material (percolation test) or that has been in contact with the material (batch leaching test), that is filtered, and analysed.

How were obtained the leachate concentrations used for acceptance into inert waste landfill (EC 2003)? The method includes an integrated model of emission of elements and some compounds by waste, the vertical transport of these contaminants through the unsaturated soil layer by vertically draining water and their interaction with that soil. This is followed by mixing and horizontal transport and adsorption/desorption within the saturated zone by horizontally moving ground water (Hjelmar et al., 2001; Hjelmar, 2012; Hennebert, 2022). The principle is that from the release of contaminants in leachate from a waste (by column percolation tests), the expected concentration at the point of compliance can be assessed. Using the target concentration of elements and substances in the groundwater, the corresponding maximum leaching concentration of elements and substances from waste can then be calculated by “reverse modelling”. The target concentrations in the groundwater are typically the ones for water that can be made drinkable or the ones of drinking water of the World Health Organization or of the EU. It is a risk approach: the maximum leachable concentration derived from reactive transport reverse modelling from waste, based on published parameters for modelling. That method was used for the case of mercury in non-sulphidic mine tailings and was successfully proposed to the COP4 of the Minamata Convention ‘Make mercury history’ (Hennebert, 2022).

Demonstrating that a recycled material in contact with the environment does not harm the environment (a condition for EoW status) is in many cases much more demanding than simple control that the specifications of a product are met. It’s a logical scientific question: demonstrating no action(s) requires more data than demonstrating an action. If no action is taken, all cases of cause and effect must be documented. For a specific action, fewer cases need to be studied. Long-term studies (years) are sometimes necessary and fully justified to assess that the use is not harmful.

3.5 Case of material recovery in the environment as beneficial substances for soils, backfilling, treated contaminated soils and sediments, technosols

The biowaste-derived compost, and methane digestate benefit of a tacit EoW, after an exhaustive study by the JRC, if they comply with Regulation 1009/2019/EU (EU 2019) with nutrient content and environmental standards (metals, microbiology), together with the (primary) fertilisers, soil improver and growing medium. The backfilling (Waste Framework Directive 2008/98/EC revised 2018) as recovery operation of non-hazardous waste is allowed since 2018 and in practice is a case of EoW status. Most of

the time, compliance is verified with acceptance into inert waste landfill (EC 2003).

The treated non-hazardous contaminated soils and sediments (immobilisation of (eco)toxic elements and destruction of (eco)toxic organics), and the man-made non-hazardous soils or substitute of vegetated topsoil (technosols) could be treated as backfilling.

A clear EU framework for the evaluation of the biological functionality of these soils and sediments is lacking. How to assess the biological impact of the intended use in natural environment? And what is the acceptable concentration of elements in the leachate or the effect in a laboratory biotest? Some authors of biotests de facto look for a zero effect whatever the dose. When wastes are used in soil as an amendment, conditioner or fertilizer, it must be recognized that the relevant concentrations in the leachate or effects in biotests at agronomic rates are not the zero concentration or effect, but rather the concentrations or an effect comparable to those of agronomic rates of amendments or fertilizers used in agriculture for a long time. For example, clean wood ash at high levels will be phytotoxic in itself due to alkalinity, but at the recommended dose of ± 1 litre/m² (0.25 kg/m²) it is a sustainable source of alkalinity, potassium and phosphorus.

Another point is that it is important to promote the recycling of N, P, K, Ca, Mg, S and trace nutrients Fe, B, Mn, Zn, Cu, Mo, Ni, etc. from waste, taking into account the entire life cycle of industrial waste production. For N, P, K, the offsets are oil or natural gas consumption, extraction of phosphate rock and attack with sulfuric acid, with some African ores too rich in cadmium, mining kainite and sylvinit and purification with enormous piles of NaCl salinizing for instance the Rhine valley in Alsace (FR) on the one hand, and the harms of non-cycling on the other hand: either incineration/combustion (energy for drying, air pollution control residues and ashes to be landfilled, unlimited legacy of landfills for the next generations), or direct landfill (methane emission, space consumption, illimited legacy). Life cycle analyses are the best method to consider the global advantages and disadvantages.

The fate of wastewater and gaseous effluents, which do not have waste status in the WFD, but which are released into the environment after treatment and in compliance with concentration and flow limits, is a sort of tacit status of EoW.

3.6 Summary of EoW cases

The different cases are sketched in Figure 2.

4. TACIT EOW STATUS FOR NON-HAZARDOUS MATERIAL IN CONTACT WITH THE ENVIRONMENT: THE CASE OF MSWI BOTTOM ASHES FOR ROAD FOUNDATIONS IN FRANCE SINCE 1994, AND 5 OTHER MINERAL NON-HAZARDOUS MATERIALS IN FRANCE

The French Circular of 1994 relating to the elimination of incineration bottom ash of municipal solid waste incinerator (MSWI) from urban waste defined the

conditions to valorise the bottom ashes in road, parking and construction basements. It can be assessed with the EU 2018 criteria for EoW status (Hennebert, 2024b). The Circular anticipated the four conditions for EoW:

- a) Specific purpose
 - i. The Circular defines the uses as road, parking or construction basement
- b) Existence of a market or demand
 - i. Evidence at the time of the Circular. In 2020 in France, 2.4 Mt/year were maturated and valorised (from 13 Mt/y of municipal solid waste generating 2.7 Mt/y bottom ashes) (ADEME, 2022)
- c) Fulfilling of technical requirement
 - i. Three categories of BA are defined in the Circular: recoverable, maturable and storable, depending on regular analyses, by a tier or by the operator, with leaching concentration limits for classification. Recommendations are formulated for initial characterisation and for periodic sampling. The report on the initial characterisation must be communicated to the Inspection of classified installations. The environmental assessment was later completed with geochemical modelling (Bellenfant and Guyonnet, 2009) and percolation tests and lysimeters studies (CEREMA, 2015).
 - ii. Maturation platforms are installations classified for environmental protection. The BA must be recoverable after a maximum of 12 months, otherwise it is landfilled in landfills for non-hazardous waste. Traceability of the different batches in the platform is mandatory.
 - iii. A quality protocol is recommended. If it exists, it should be transmitted to the Inspection of classified installations.
- d) Absence on adverse effects in the intended use
 - i. Intended use with a protective layer, in controlled conditions, not in the perimeter of production of drinking water, at least 30 m away from any watercourse and at an altitude 1 m higher than the highest known level of waters in the area.
 - ii. The analytical methods (leaching in 3 times 10 litre/kg, the French leaching test at that time) and the concentration limits were defined in the Circular to minimise emissions and transfer to groundwater.

As the Circular requests that the producer samples and analyses by himself or by tier independent laboratory: *"It is not the responsibility of the inspection of classified installations to certify the quality of the bottom ash produced by an installation. It is up to the operator to be able to demonstrate to users of the bottom ash it produces that the recovery conditions set by this text and the authorisation order which governs the operation of its installation are respected"*, it was the first legal document establishing de facto a tacit end-of-waste procedure.

That tacit EoW was reinforced in 2011 by a Guidance document of a public technical institute aligning with the Decision 2003/33/EC on landfill acceptance criteria for inert waste (CEREMA, 2011), with a tiered approach of European batch leaching test, percolation test (CEREMA,

2011), completed in 2015 for the level 3 of environmental characterisation with lysimetric tests and experimental plots (CEREMA, 2015) and updated in 2023 (CEREMA, 2023).

That method was applied in dedicated guidance on steel slags (CEREMA, 2012a), bottom ash from non-hazardous waste incineration (CEREMA, 2012b, an update of the 1994 Circular), on deconstruction materials from building and public works (CEREMA, 2016), on foundry sands (CEREMA, 2019a) and ashes from pulverized coal-fired power plants (CEREMA, 2019b). That institute also managed research and development works on the valorisation of marine and land sediments in public works and concrete.

For total mineral non-hazardous waste, in 2020 in France, 130 Mt were recycled (backfilling non included), as road basements or granulates for non-structural concrete. These tacit end-of-waste procedures have contributed to that result. That fact illustrates that EU-wide tacit end-of-waste status should be granted for the materials supplied by certified industrial recycling facilities.

5. TACIT EOW AND SPECIFIC COMBUSTION REGIME FOR COMBUSTIBLES

The use of waste for thermal energy is particular since it involves products and waste: agricultural and forestry products are also used for this purpose, and the energy recovery of waste in principle only concerns waste that are not recyclable as material (level 4 of waste hierarchy).

The combustibles (solid, liquid, gaseous) include (see Table 1):

- products,
- waste with effectively tacit (unspoken) EoW,
- waste without EoW (in France) but with a special combustion regime (Solid Recovered Fuel also named Re-fuse Derived Fuel),
- non-hazardous and municipal waste excluded from EoW status, and
- hazardous waste excluded from EoW status.

The categories of biomass that are waste (EU, 2015), namely vegetable waste of agriculture and forestry, of food processing industry, of paper production, of cork, and wood waste without halogenated organic substances and heavy metals, benefit of tacit EoW status.

Non-recyclable refuse-derived combustibles (that are waste) have a specific category of facilities in France, distinct from that of waste incinerators, which is in fact a kind of escape of waste status by the valorisation operation.

Other gaseous, liquid or solid waste intended for use as combustible may have EoW status in France. The assessment of their environmental and health impact, depending on the composition of the waste, the combustion equipment (combustion performance, air pollution control) and the composition of the smoke must be done according to guidance (Collet and Partaix, 2021). The concentration limits in flue gases mainly come from (EU, 2015). This highlight that in real life there is a continuum between waste and products and that tacit EoW exists because it is an essential link of this continuum.

TABLE 1: The waste used as combustible have already in some cases tacit end-of-waste status: matter, regulatory regime for combustion or incineration and tacit end-of-waste status in France (NH = non-hazardous, H = hazardous).

Matter	Characteristics	Product or waste ?	NH or H?	Combustion or Incineration?	Possibility of EoW?
Biomass	Vegetable products of agriculture or forestry	Product	NH	Combustion	Already Products
	Vegetable waste of agriculture and forestry, of food processing industry, of paper production, of cork	Waste	NH	Combustion	Yes, Tacit
	Wood waste (without halogenated organic substances and heavy metals)	Waste	NH	Combustion	Yes, Tacit
Refuse-derived combustible ("solid recovered fuel")	Prepared mixture of non-recyclable plastics, paper, cardboard, wood and organic matter.	Waste	NH	Combustion with specific regime	No but specific Combustion regime
Gaseous, liquid or solid waste to be used as combustible	Organic waste, non-hazardous or hazardous (in particular treated wood waste for coating or with halogenated organic substances and heavy metals, wood waste from construction and demolition waste)	Waste	NH or H	Combustion if EoW Incineration	Yes, Explicit
Municipal waste	Including commercial waste, non-hazardous	Waste	NH	Incineration	No
Industrial waste	Hazardous	Waste	H	Incineration	No

The production of electricity or heat or liquid fuel (pyrolysis) or gas (gasification) from construction and demolition (C&D) waste wood and furniture waste wood will increase because there is not enough other primary wood for the EU flexible energy transition. Will these installations fall in the incineration regime (in case of waste status for wood) or in the combustion regime (with tacit or explicit EoW status for wood)? Investment and social acceptance are easier with product status.

6. THE TACIT EOW SHOULD BE GRANTED FOR ANY PREPARATION FOR REUSE AND RECYCLING OF MATERIALS IN CERTIFIED INDUSTRIAL LOOPS

6.1 EoW and the waste hierarchy

The product status by tacit EoW procedure is a logical consequence of recovery (Figure 3).

The material supplied by certified industrial recycling facilities should have EU-wide tacit end-of-waste status, because compliance is achieved. For the last level of the waste hierarchy, the term "sustainable landfill" should only be used for impermeable and non-self-reactive monostock for future recovery and EoW.

6.2 Waste streams that could immediately benefit of tacit end-of-waste status

The Joint Research Centre of the EU has ranked plastics, textiles, rubber (from end-of-life tyres), mineral fraction of construction and demolition wastes, paper and cardboard for priority to develop end-of-waste criteria at EU level (Orveillon et al., 2022). In total 12 wastes in 5 groups were top-ranked from about 70 candidates in 11 groups (Figure 4). Using the categorisation presented here (section 3), for these 12 wastes:

- 10 wastes will not be used in contact with the environment: 5 plastics, 3 textiles, paper and cardboard, and mineral wool,

- 2 wastes will be used in contact with the environment: rubber (for outdoor uses) and aggregates (for construction).

For the 10 wastes not used in contact with the environment, the technical or commercial specifications mandatory for commerce are already available, and the registration of material with sameness in REACH is probably available. Tacit EoW could be immediate.

For the 2 wastes used in contact with the environment, an in-depth dossier to verify the impact on the environment is compulsory. For rubber, the dossier is available (EuRIC and ERTMA, 2021). For aggregates, many propositions with tests and limit values have been available for 15 years (JRC, 2009; Delgado et al., 2009; Saveyn et al., 2014; Tecnalia, 2021). The potentially moving elements from the waste to the environment are the leachable ones and the leachate concentration limits developed for inert waste landfill acceptance (see 3.4), eventually with some adaptations, are available and should be used since they are included in the EU landfill acceptance decision (EC, 2003).

7. CONCLUSIONS

The end-of-waste status is a consequence of the material recovery operations. The legal classification of discarded materials should be determined by their place in the waste management hierarchy. The wastes that are reused or recycled in certified industrial facilities should benefit of a product status by a tacit EoW procedure. This paper demonstrates that it already is the case for many wastes. The definition of waste should be narrowed down (Johannson, 2023) to exclude the matters that are resources.

A wide range of waste is already ranked by the Joint Research Centre of the EU for potential EoW (Orveillon et al., 2022), and particularly 10 ranked in the first five groups, not used in contact with the environment. The technical or commercial specifications mandatory for commerce of these 10 wastes are already available, and the registration of material with sameness in REACH is probably available.

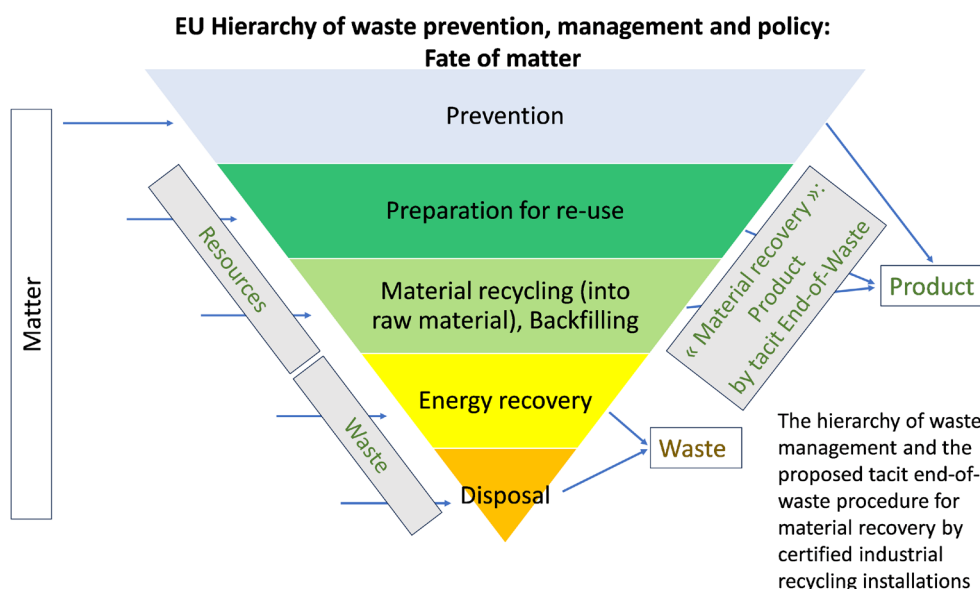


FIGURE 3: The relation between the waste hierarchy and the tacit end-of-waste procedure.

Figure A: List of priority streams grouped per category, and ranked based on their overall potential, according to the scoring system and data used for the present study

<u>Waste/by-product categories and priority streams:</u>		Overall potential
1 - Plastics:		
Polyethylene terephthalate (from plastic waste)		63
Low- and high-density polyethylene (from plastic waste)		60
Mixed plastics (from plastic waste)		57
Polystyrene and expanded polystyrene (from plastic waste)		57
Polypropylene (from plastic waste)		55
2 - Textiles:		
Separately collected clothes and other textiles prepared for re-use		60
Cellulosic fibres (from textile waste)		56
Mixed fibres (from textile waste)		55
3 - Rubber:		
Rubber (from end-of-life tyres)		59
4 - Mineral fractions of construction and demolition waste:		
Aggregates (from demolition waste)		57
Mineral wool (from demolition waste)		55
5 - Paper and cardboard:		
Paper and cardboard waste		55

FIGURE 4: Priority streams of waste for EU end-of-waste criteria (Orveillon et al. 2022).

Tacit EoW could be immediate. This objective should be achieved in a harmonized way across the EU. For each material and each use, the four EoW conditions are met in EuRIC affiliated companies. An end-of-waste dossier should be made public, and the material should be registered in REACH if necessary. The tacit status of EoW at European level in accordance with Article 6(1) and 6(4) of the Waste Framework Directive should then be requested to Member States by the European Commission. Formally, this could be a request to the European Commission to ask all Member States that these materials produced in certified facilities and intended for a given use cease to be waste, in

accordance with Article 6(1) of the WFD (EU 2018) which states: “Member States shall take appropriate measures to ensure that waste which has been the subject of a recycling operation or another recovery operation is considered to have ceased to be waste if they meet the following conditions...”.

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HOUSEHOLD HAZARDOUS WASTE: GAUGING KNOWLEDGE LEVEL AND ITS IMPLICATION FOR DOMESTIC WASTE HANDLING AND DISPOSAL PRACTICES

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ABSTRACT

Household hazardous waste (HHW) refers to waste generated at the household level that can pose a danger to public and environmental health if not managed properly. The amount of waste, including HHW, has been increasing due to urbanisation, rising income levels, and changes in lifestyles. Unfortunately, most households are not aware of the potential risks associated with this waste, despite the serious health and environmental hazards it can pose. This creates a conundrum which this study seeks to unpack. A mixed-method approach addressed this gap, including a policy dialogue and a survey of 1,245 Greater Accra Metropolitan Area respondents. From the findings, most respondents disposed of aerosol cans, disposable sharps, unused or expired medication/drugs, fluorescent tubes and bulbs, and electronic waste as part of their household waste. Significantly, it was concerning that 68% of participants were unaware of the potential health hazards associated with improper disposal of HHW. Based on empirical evidence, the awareness levels were affected by gender, education level, and occupation. To address the issue of HHW and reduce its harmful impact on the environment and public health, it is essential to implement policies that encourage waste segregation, establish product stewardship programs, and conduct targeted education campaigns. These actions will help raise awareness about the significance of proper HHW disposal. Additionally, it is vital to have a strong political will to effectively enforce legislation that supports the shift towards more sustainable waste management practices, thus promoting sustainable development.

1. INTRODUCTION

Waste is a by-product of human activities. In the past, people conveniently and unobtrusively disposed of it in large open land spaces, as the population density was low. However, with rapid urbanisation, population increase, developing living standards, and lifestyle changes, the volume and content of waste have significantly changed, respectively (Kaza et al., 2018). It has been estimated that around 2 billion tonnes of solid waste are generated globally every year, and this number is expected to rise up to 3.4 billion by 2050, given the current trend. Additionally, there has been a significant increase in the demand for and usage of packaged products and electronics in the last few decades (Kaza et al., 2018). These dynamics in municipal solid waste (MSW) composition have been typified by an emerging waste stream, household hazardous waste, which is of critical public and ecological health concern.

Household Hazardous Waste (HHW) has been a matter of concern in developed countries since the 1980s, but there is no universally accepted definition or list of waste products considered as HHW. In fact, even within a single country like the United States, the list of HHW may vary from region to region (Wallace et al., 2018). Nevertheless, it is generally agreed that HHW has some common characteristics, including being toxic, corrosive, flammable, and reactive. HHW is generated from domestic consumption and includes unused, expired, or leftover consumer products like batteries, motor oil, brake fluid, etc. It also includes home improvement products like paint, caulk, and air fresheners, health and beauty products like hair dye, nail polish, and bath salts, as well as household cleaning products like stain removers, disinfectants, bleaches, shoe polish, and furniture polish (Wessel, 2004). Other examples of HHW are pesticides, insecticides, weedicides, fabric dyes, fluorescent tubes, light bulbs, and ink cartridges (Al-Tamimi et



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al., 2018). Other examples of HHW are pesticides, insecticides, weedicides, fabric dyes, fluorescent tubes, light bulbs, and ink cartridges (Delgado et al., 2007; Edokpayi et al., 2017; Inglezakis & Moustakas, 2015; Kontogianni et al., 2014).

Due to the increasing amount of waste and the emergence of hazardous waste streams including HHW, the waste management industry has become more crucial. It is necessary to properly manage and treat MSW coming from households, public and private areas, buildings, and commercial establishments to prevent significant environmental and public health hazards (Edokpayi et al., 2017). Despite the inherent substantial threats, landfilling remains the most preferred, used, and cheapest household waste management. Indeed, almost 70% of MSW is disposed of internationally in landfills (Delgado et al., 2007; Edokpayi et al., 2017; Inglezakis & Moustakas, 2015). Also, unlike the advanced nations, many emerging economies grapple with inadequate infrastructure and capacity to effectively handle the increasing MSW quantities and composition dynamics. For example, only 44% of sub-Saharan Africa's population can access proper MSW management services (Oteng-Ababio, 2023).

Although HHW constitutes a relatively small fraction (up to 3%) of MSW (UN-HABITAT, 2010), prior studies have indicated that its mismanagement poses severe environmental and public health risks. At home, expired pharmaceuticals, disposable sharps, home improvement products, etc., may cause poisoning, burning and injuries to both humans and domestic animals (Ansong et al., 2016; Gutberlet & Uddin, 2017; Roggof & Biderman, 2015). Beyond the point of generation, waste workers face HHW-related hazards, including fires and explosions caused by chemical reactions in waste compaction trucks, transfer stations, shredders, or landfills (Gutberlet & Uddin, 2017; Roggof & Biderman, 2015). HHW also poses soil, water, and air contamination risks when improperly disposed of. Therefore, it is essential to correctly identify and assess all potential hazards to protect against possible environmental impact (Kontogianni et al., 2014).

Effective HHW management is a function of public awareness, efficient collection, availability of the necessary infrastructure, and the existence and enforcement of the relevant laws. Table 1 presents various approaches implemented in different regions worldwide, which may be tailored to the context of other countries to proper HHW management. In the USA, some states have specific HHW laws. However, this is not the general case in Europe. Rather, the European Hazardous Waste Directive (no. 91/689/EDC) directs member countries to manage HHW along with other industrial hazardous waste (Inglezakis & Moustakas, 2015; Mudiyansele & Herat, 2022). South Africa's Waste Act of 2008 stands out as an illustration of a comprehensive legislation that offered the necessary legal structure guiding the development of the national strategy for HHW management. Adequate public awareness of what constitute HHW, relevant laws, and collection programmes promote HHW avoidance, reduction, and source separation (Fuller et al., 2018). Investment in collection programmes and treatment infrastructure is also crucial. Mobile or fixed

collection approaches, or a combination of both have been adopted. In regions that have made significant strides in HHW management, like the USA and Europe, product stewardship schemes, such as Extended Producer Responsibility (EPR), have been used to finance HHW management programmes (Cassel, 2018).

Unlike hazardous waste streams from industrial sources, HHW is not strictly regulated under any law in Ghana, including the National Environmental Sanitation Policy (NESP). This is largely because the amount of HHW generated in Ghana and the significance of their disposal practices are poorly understood. Generally, the amount of HHW generated is believed to be small, so disposal risks are negligible. However, there is a lack of reliable and clear information on their generation rates and disposal practices. With changes in legislation, for example, in 2016, when the Environmental Protection Agency (EPA) achieved the passage of the Hazardous and Electronic Waste Control and Management Act (Act 917) (Republic of Ghana, 2016) and the Legislative Instrument on Hazardous and Electronic Waste Control and Management Regulations (LI 2250), it has become important to acknowledge the hazardous elements contained in MSW and shift towards regulated, specialised and safe storage and handling processes throughout the entire value chain. As concerns about chemicals in household products increase, the potential environmental consequences from their disposal are also becoming more apparent, thus, giving a *raison d'être* for the current study.

Furthermore, existing studies on HHW in the global South highlight the need for data (Edokpayi et al., 2017) and more awareness regarding HHW management (Edokpayi et al., 2017; Mmereki et al., 2017; Ubachukwu et al., 2014). For instance, Ubachukwu et al. (2014) found low public awareness of HHW dangers in Nigeria's Enugu metropolis. Edokpayi et al. (2017) reported limited HHW awareness among tertiary students in Nigeria, Kenya, and South Africa, while Decharat et al. (2020) observed low to moderate HHW awareness in Thailand's Phatthalung province. The issues with HHW management, like the quantities, qualities, and the public's awareness levels, only worsen the already existing setbacks such as policy and legislation, limited financial resources, inadequate infrastructure and logistics, and insufficient institutional capacity (Edokpayi et al., 2017).

In Ghana, academic research and policy attention have primarily focused on specific HHW streams, such as electronic (see Daum et al., 2017; Oteng-Ababio, 2010; Grant and Oteng-Ababio, 2012; Owusu-Sekyere et al., 2022; Owusu-Twum et al., 2022) and household medical waste (Sasu et al., 2012; Udofia et al., 2017). Despite the fact that households generate a diverse range of household hazardous waste (HHW) items, no study has been carried out to investigate HHW in a broader sense. This knowledge gap is the reason behind the current work. The study aims to address concerns related to managing HHW generated in typical urban scenarios, using Metropolitan Accra as a laboratory. The primary objective of this study was to identify and quantify the predominant hazardous pollutants in MSW and develop specific profiles. To achieve this, we conducted a mixed-method study to assess common HHW in

TABLE 1: Key approaches to effective HHW management.

Intervention	Country	Advantages	Limitations	Reference
Legislation and enforcement				
1. Comprehensive Waste Management Act	Austria South Africa	Covers the management of all existing waste streams and those that may emerge in the future.	The process of formulating and enacting the comprehensive Act may take long or may be stifled by low political interest.	
2. HHW Specific legislation	USA	HHW is clearly defined or items that constitute HHW are explicitly indicated.	List or definition may vary geographically and may also be inexhaustive as new products emerge.	Wallace et al. (2018); Inglezakis and Moustakas (2015)
Collection approaches				
1. Collection events	USA, Austria, Germany, Sweden, Denmark	Special days dedicated or events for community-based HHW collection.	Limited number of collection times per year.	Fuller et al. (2018)
2. HHW collection facilities	USA, France, Hungary, Italy, UK, Spain	- Collection throughout the year. - May be co-located with MSW facilities, municipal parks, or fire stations.	- May require significant infrastructural investment. - Long distances to drop-off points may discourage public patronage of services.	Nightingale and Lewry (2018); Mudiyansele and Herat (2022)
3. Kerbside and door-to-door (DTD) collection		- Higher service patronage due to convenience to the public. - DTD offers equitable service access, especially for the disabled and homebound customers.	- Relatively higher per-unit cost. - Unreliable pick-up services will disincentivise the public from participating.	Quinn and Brunning (2018)
4. Retail collection	USA, Japan, Germany, Switzerland	HHW are retrieved from consumers through the same channel the products were purchased.	Both retailers and consumers may require financial incentives to participate.	Mudiyansele and Herat (2022)
Financing mechanisms				
1. Extended Producer Responsibility	USA, Europe	Effective EPR programmes shape product design and also finance separate HHW collection and treatment.	Requires strong coordination and collaboration between government agencies and industry stakeholders.	Fuller et al. (2018); Cassel (2018)
2. Taxation		Excise tax on household hazardous products to minimise consumption.		
Social marketing				
Public education and sensitisation	USA, Europe	- Foster change in consumer behaviour and motivate source segregation and participation in HHW collections. - May be combined with collection programs.	May not yield the intended outcome if the relevant laws are not enforced and collection programmes are not well implemented.	Fuller et al. (2018)

and around the study area and investigate domestic management practices. Firstly, a policy dialogue was used to understand the current state of HHW management from key stakeholders' viewpoint in Ghana. Subsequently, a concise review of the relevant policies and legislation was undertaken. Following this, we conducted a survey to assess the public's understanding and perspective on the matter. The findings were then triangulated and discussed. The findings of this study could be useful in deciding appropriate waste disposal methods and modifying existing ones. We will explore the research methodology and residents' understanding of HHW. Ultimately, our aim is to provide recommendations for a smoother transition towards sustainable environmental management.

2. MATERIALS AND METHODS

2.1 Context of the study area

Accra, Ghana's capital, has expanded spatially since its inception in the late 1870s. The city is located within the Greater Accra Metropolitan Area (GAMA), among the fast-growing urban conurbations in Sub-Saharan Africa (UN-HABITAT, 2009). The region comprises 29 self-governing districts (GSS, 2021) and covers 1,079 km² (see Figure 1). While immigration to the city area certainly contributes to its growth and expansion, residential mobility from more central locations to the urban fringes and natural population growth are equally important. According to recent estimates from the Ghana Statistical Service, GAMA contained

a population of approximately 4.6 million (2016) (Akubia & Bruns, 2019).

The GAMA region, with a population of almost 4.9 million, has become a functional area. It has strong territorial connections with surrounding communities and is a significant socio-political and economic hub, with territorial solid waste linkages with adjoining areas. The region is divided into three residential classes based on income, namely low, medium, and high-income zones reflecting varying levels of neighbourhood planning, densification, and infrastructural development (Songsore et al., 2009; Songsore & McGranahan, 1993). These areas have unique waste management systems that can be classified as formal or informal. The formal sector typically uses advanced vehicles, while informal players commonly rely on motorized tricycles, which can easily navigate through communities with poor road access. However, studies indicate that even in well-planned neighbourhoods, the services provided by waste management companies may be perceived as unreliable, leading to the growth of the informal sector in recent years. (Deku, 2017; Sarfo-Mensah et al., 2019). GAMA is estimated to produce 1.3 million tonnes of MSW annually, with a per capita production rate of 0.74 kg (Miezah et al., 2015) and a current waste collection rate of about 70% (Oteng-Ababio, 2023).

2.2 Data collection and analysis

To gain a better understanding of HHW awareness and management, we took an exploratory, sequential, mixed-method approach described by Creswell and Clark (2017). This method employs qualitative research to explore a topic and utilises the results to develop an instrument for subsequent quantitative research. As shown in Figure 2,

our first step was to hold a policy dialogue with various stakeholders. Nineteen participants, including five officials from waste management (collection, processing, and disposal) companies, three representatives from the informal waste sector (including the president of the main informal waste collectors association and two e-waste workers), three waste sector academics (from the University of Ghana and Kwame Nkrumah University of Science and Technology), two environmental legal professionals, one waste management consultant, and three public officials in the industry (two waste management directors (WMDs) and a representative from the Ministry of Sanitation and Water Resources). WMDs are responsible for city (metropolitan) waste management. After a brief introduction of HHW by the authors, a cross-section of the participants presented on pre-selected topics, including contemporary concerns regarding MSW streams, the existing legislative frameworks governing HHW globally and locally, current (local) governments' arrangements to manage HHW, and how the private and informal sectors were handling such waste stream along the waste management value chain. After the presentations and deliberations, the participants were divided into groups for a breakout session to discuss the topic further. They summarised the current state of knowledge and challenges associated with HHW management and proffered some policy recommendations. The policy dialogue highlighted the lack of knowledge on HHW, generating intense public discussions and commentary on the subject.

Based on the workshop findings, a structured questionnaire was designed and administered through a public survey to gather additional information on the management practices of HHW in the region. The questionnaire includ-

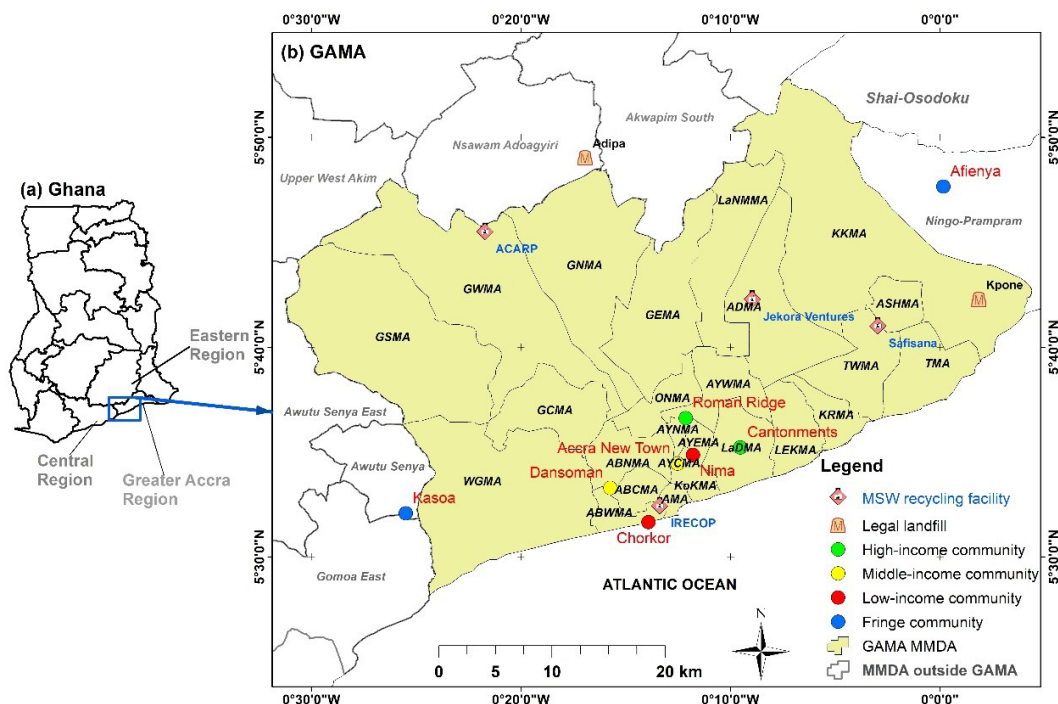


FIGURE 1: Map of the study area.

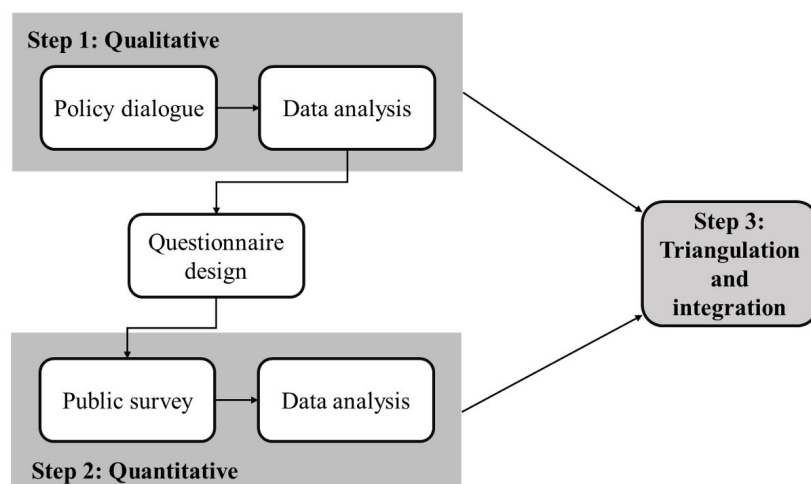


FIGURE 2: Process flow of sequential mixed-method approach used study.

ed questions related to the socio-demographic details of the respondents, the HHW streams that they manage, and their knowledge about the handling, disposal, and potential hazards of HHW. Proxy questions were also used to assess the level of awareness and understanding of the topic among the respondents. The study was conducted in eight communities, namely Chorkor, Nima, Dansoman, Accra New Town, Cantonments, Roman Ridge, Kasoa, and Afienya. These communities were selected purposively to reflect the geographical and socioeconomic diversity of the region and to account for some spillover dormitory communities, as shown in Figure 1. Entry into the communities, except for Cantonments, was facilitated by an Assemblymember of the area, who also provided guidance in identifying the community boundaries.

The required sample size was estimated at 1,230 using the Yamane's formula (Equation 1), based on the current population of the GAMA (GSS, 2021), Kasoa and Afienya, at a 2.85% margin error. The sample sizes for each community were determined using the proportional quota technique based on the 2010 population and housing census (PHC), which had community-level data available at the time of the study. The sample sizes for Cantonments and Afienya were slightly adjusted to reflect field conditions, including empty houses and significant community expansion compared to 2010, respectively. Respondents (household heads or their representatives, 18 years or older) were selected using a multi-stage sampling technique similar to the one described by Frimpong (2019). The study involved a total of 1,245 participants who were administered questionnaires in their local dialect by trained field officers.

$$n = N / (1 + N * e^2) \quad (1)$$

where n is the sample size, N is the population size, and e is the margin error.

We analysed the transcribed audio recording of the workshop and notes taken thematically using ATLAS.ti. This was also complemented by a mini review of the HHW-related laws in Ghana. Moreover, we used descriptive and inferential statistics in IBM SPSS (version 24) to examine the survey data. We evaluated the knowledge levels of the

respondents by using a scoring system where "0" represented an incorrect response, "1" was assigned to "I don't know", and "2" indicated a correct answer. We used the median scores of the respondents for the questions measuring HHW handling, disposal, and environmental impact as the aggregated scores. Median scores of less than one, equal to one, and more than one were interpreted as "low," "neutral," and "basic" knowledge levels, respectively. The term "basic" was used instead of "high" to describe the level of knowledge required for correct responses since the study questionnaire only included typical examples of HHW.

To determine if respondents' socioeconomic characteristics affected their likelihood of hearing about HHW, we used logistic regression. We inputted all variables into the equation using variable codes displayed in Table A.1. Beforehand, we assessed multicollinearity among predictor variables using variance inflation factor (VIF) and tolerance values, as outlined in Field (2013).

Finally, we discussed the study results by integrating and triangulating information gathered from both the qualitative and quantitative sources.

3. RESULTS

3.1 Respondents' sociodemographic profile

Table 2 displays that most participants (58.2%) were females, which is consistent with the recent census data indicating a higher proportion of females (about 51%) in the study area (GSS, 2021). Also, many females were at home during the survey period, particularly in low and middle-income communities. Additionally, about 51.8% of the respondents were young adults aged between 21 and 40 years, 34% were middle-aged adults aged between 41 and 60 years, and 11.4% were senior citizens above 60 years.

Out of all the respondents, only 9.6% had no formal education. Basic and secondary education was completed by 28.3% and 31.8% of the participants, respectively. Around 20% were government or private employees, while 31.8% were traders. The majority of the participants (51.4%) were married, followed by single (31.3%), widowed (6.6%),

TABLE 2: Sociodemographic characteristics of the respondents.

Sociodemographic characteristic		Frequency	%	Sociodemographic characteristic		Frequency	%
Gender	Male	725	58.2	Age	Up to 20	35	2.8
	Female	520	41.8		21 - 40	645	51.8
					41 - 60	423	34.0
					More than 60	142	11.4
Highest education level	None	119	9.6	Average monthly income (GHC)	0 - 500	463	37.2
	Primary	115	9.2		501 - 1000	358	28.8
	Basic	352	28.3		1001 - 1500	174	14.0
	Secondary	396	31.8		1501 - 2000	109	8.8
	Tertiary	263	21.1		Above 2000	141	11.3
Major occupation	Unemployed	183	14.7	Type of dwelling unit	Separate	290	23.3
	Government worker	90	7.2		Semi-detached	122	9.8
	Private company employee	159	12.8		Apartment/flat	135	10.8
	Artisan/craftsman	163	13.1		Compound house	657	52.8
	Trader	396	31.8		Improvised home	29	2.3
	Student	73	5.9		Other	12	1.0
	Other	181	14.5				

co-habiting (6.3%), and divorced (4.4%). Two-thirds of the participants earned a monthly income of GHC 1,000 (USD 105) or less, while only 11.3% earned more than GHC 2,000 (USD 210). The most prevalent dwelling unit was a compound house (52.8%), followed by separate houses (23.3%), apartments and flats (10.8%), and semi-detached houses (9.8%). A small percentage of individuals lived in improvised homes such as kiosks or containers.

3.2 Household solid waste management practices

Based on Table 3, the majority of the participants (54.7%) used authorised waste bins to store their waste, while others utilized improvised containers like old sacks and plastic bags. Figure 3 displays some of the various receptacles employed for waste storage. The type of storage container used varied depending on the economic status of the community. High-income households primarily used authorised waste bins (96.0%), while middle-income settlements used waste bins for about 50.2% of waste storage, with improvised containers and sacks accounting for 18.9% and 11.7%, respectively. Less than half of the respondents (37.9%) in low-income communities had authorized waste bins. It's ironic that the prevalence of authorized waste bins was slightly higher at 47.0% in fringe communities. This isn't surprising since most of the emerging middle-income families settle in the peri-urban neighbourhoods where land is relatively affordable.

From the data analysis, it was found that only 36.2% of the respondents practised waste separation at home. The most commonly separated materials were plastics, cardboard, and metals. Out of this percentage, 61.3% sold parts of their waste to itinerant buyers, which was more common in low and middle-income neighbourhoods. This indicates that monetary incentives may significantly drive household

waste segregation in the study area. Additionally, a significant proportion (23.1%) of people disposed of their waste through open burning. This method was more prevalent in dormitory communities on the outskirts (49.4%), which are currently underserved with official waste collection services in most cases.

The study found that 46.8% of participants reported that informal waste collection was the official arrangement in their neighbourhoods. These waste collectors typically use smaller vehicles like motorised tricycles to access areas with poor road accessibility. Also, 45% of respondents were served by waste management companies that use compaction trucks, while 7% had communal container systems. The remaining 4.8% had no official waste collection arrangement.

The study also provided insights into the availability of waste collection services in different neighbourhoods. In high-income areas, most households (86.7%) were served by waste management companies providing door-to-door services. In contrast, approximately 50 to 60% of households depend on informal operators for door-to-door collection in low-income, middle-income, and fringe communities. Communal waste disposal systems were primarily found in low-income neighbourhoods, serving about 16.7% of households. In communities located along the coast, residents often resorted to dumping their waste at the sea-side. In contrast, others would dispose of their waste in drains and bushes or burn it in open areas where there was no official waste collection arrangement.

3.3 Gauging respondents' awareness and knowledge levels

This section focuses on the inventory of household hazardous waste (HHW) as reported by our respondents.

TABLE 3: Household waste management practices.

MSWM practice		Frequency	% Low-income	% in community type			
				Middle-income	High-income	Fringe	
Primary storage	Waste bin	681	54.7	37.9	50.2	96.0	47.0
	Plastic bag	131	10.5	27.1	18.9	1.8	16.1
	Used sack	213	17.1	13.2	11.7	1.8	12.9
	Improvised container	216	17.4	21.8	18.9	0.4	22.9
	Others	4	0.3	0.0	0.2	0.0	1.2
	Total	1245	100.0	100.0	100.0	100.0	100.0
Official waste collection arrangement	Waste management company	516	41.4	26.8	36.6	86.7	28.1
	Private informal ('aboboyaa')	582	46.7	51.1	54.8	9.8	59.8
	Communal container	87	7.0	16.7	6.2	0.9	1.6
	Other (none)	60	4.8	5.4	2.4	2.7	10.4
	Total	1245	100.0	100.0	100.0	100.0	100.0
Waste management practices	Burying	13	1.0	1.6	0.4	0.9	1.6
	Open burning	288	23.1	12.6	18.5	18.2	49.4
	Separation	451	36.2	37.5	44.5	29.8	25.3
	Sell to itinerant buyers	279	22.4	38.5	23.6	6.2	14.5
	Reuse	123	9.9	18.6	10.6	1.8	4.8

**FIGURE 3:** Types of receptacles for waste storage. (a) waste bin, (b) sacks, (c) improvised containers, and (d) plastic bags.

It explores their awareness and knowledge regarding HHW and the management practices they adopt to deal with it. The aim is to understand better the types of hazardous waste found in households, respondents' awareness and knowledge levels, and their practices for managing such waste. The study examines the prevalence of HHW in households, respondents' familiarity with this waste, and their management strategies.

According to Figure 4, 41.3% of the respondents reported generating unused or expired medications/drugs, while 40.5% generated used or broken light bulbs. Most participants who generated these HHW fractions did so over one month or more, as depicted in Figure 5. Additionally, 23.0% of the participants generated various types of battery waste, such as dry cells, vehicle batteries, and power banks. These wastes were produced after more than three months (44.1%), monthly (17.8%), or quarterly (31.8%). Fur-

thermore, 16.7% of the participants reported generating expired paints and solvents, while 70.1% produced such waste generally after over three months.

More than half of the respondents (56.6%) confirmed that they dispose of sharps like used razor blades, needles, and lancets in their homes. These disposable sharps were generated mostly every month (33.6%), but also frequently over a week (24.0%), quarterly (18.6%), or beyond three months (19.7%). Most respondents (86.7%) generated compressed cans, such as body spray, spray starch, air freshener, and insecticide spray, every month (56.4%). About one-third of the respondents (32.5%) produced unused or expired pest control chemicals or e-waste (32.8%). Two-thirds of those who generated e-waste (66.4%) indicated it took over three months to produce them. However, a considerable proportion of the respondents produced expired or unused pest control chemicals on a weekly

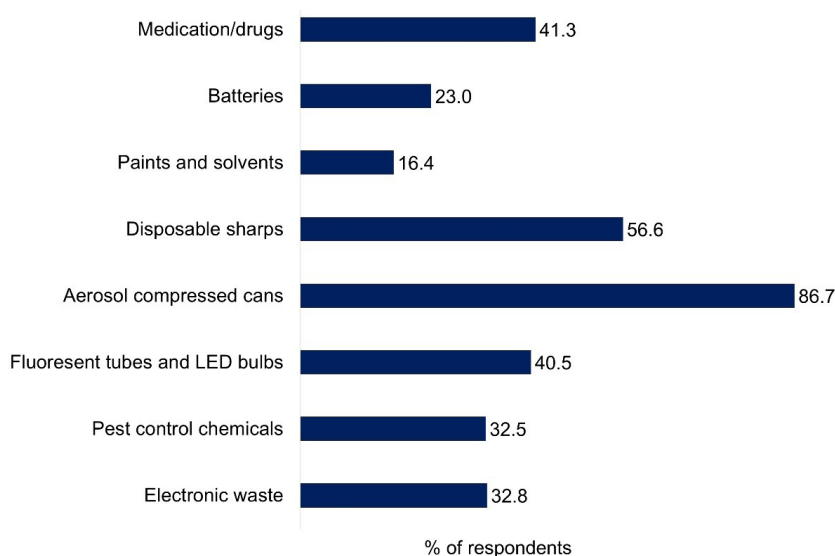


FIGURE 4: Percentage of respondents who generated various streams of household hazardous waste.

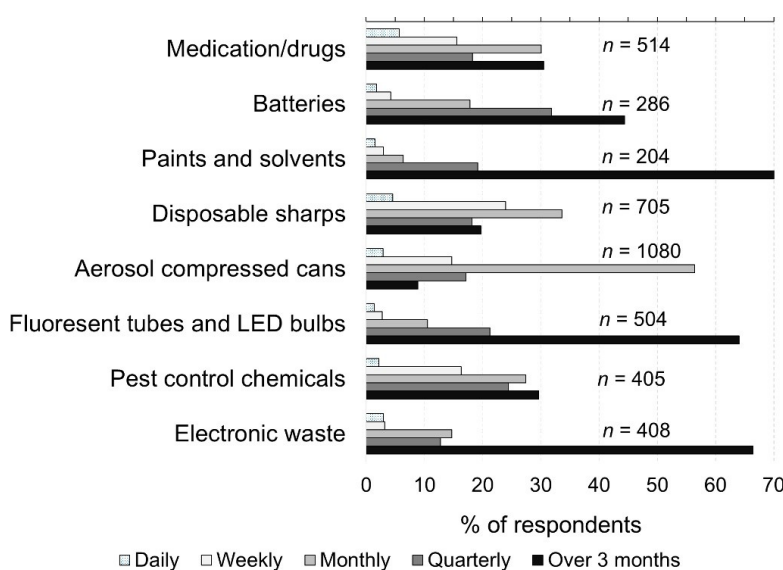


FIGURE 5: Frequency of generation of household hazardous waste streams.

(16.3%), monthly (27.4%), quarterly (24.4%), or over three months (29.6%) basis.

According to Figure 6, over 71% of the respondents disposed of hazardous household waste (HHW) by adding it to regular waste bins. This included medication, batteries, diapers, sanitary products, sharps, aerosol cans, bulbs, and pesticides. Paints and other hazardous solvents were also often added to waste bins, stored in backyards, or disposed of by other means. As for e-waste, 35.3% of the respondents added it to waste bins, 40.9% stored it in their backyards, and 23.0% disposed of it through other methods. Backyard storage was commonly used for battery products (15%), sharps (10.8%), and pesticides (13.1%).

Though the survey results appear interesting, they somehow parallel the qualitative data. For example, revelations from the policy dialogue confirmed the generation of these HHW fractions and their co-disposal with mainstream waste. To support this, an operator of an MSW treatment facility observed as follows:

"...The kind of waste coming to the processing plant is very abhorrent and negatively impacts our operations...I cannot mention the number of chemicals we receive in the form of liquids, powders, etc. These chemicals are negatively affecting the quality of the products that we are producing. Not only the quality, but the maturation period is also adversely affected." – Composting & Recycling Plant Operator 1

This observation highlights the challenges material recovery facilities face treating MSW contaminated with HHW, including extended compost production cycles and compromised compost quality.

The survey results indicate that there is a lack of awareness among the public regarding the proper handling of HHW. As per the findings of Figure 7, 67.6% of people had not even heard of HHW, and 75.7% were not aware of its treatment options. Even though more than half (56.6%) of the respondents believed that household waste should not be handled the same way, 52.6% were unsure or had low knowledge of the appropriate HHW handling methods. Upon further analysis of the data, it was discovered that individuals familiar with HHW, aware of its treatment methods, or read product labels were more likely to be from high-income neighbourhoods. This is likely due to the higher levels of education in these communities, which were later confirmed by the regression analysis findings.

According to the study, it was found that most participants (85.8%) had a basic understanding of the correct methods for disposing of Household Hazardous Waste (HHW) which is shown in Figure 8. Additionally, the majority (79.9%) were aware of the environmental harm that could result from improperly managed HHW. The level of knowledge varied depending on the type of HHW, with over 80% of the participants exhibiting a basic understanding of handling and disposing of aerosol cans, e-waste, and pest control chemicals. Approximately 75% of the participants had a similar level of knowledge regarding disposing of expired or unused medicines, while two-thirds were familiar with handling batteries, paints, and solvents. However, only about 50% had comparable knowledge regarding the disposal of disposable sharps. Furthermore, less than half of the participants (48.9%) admitted to reading product labels before discarding them.

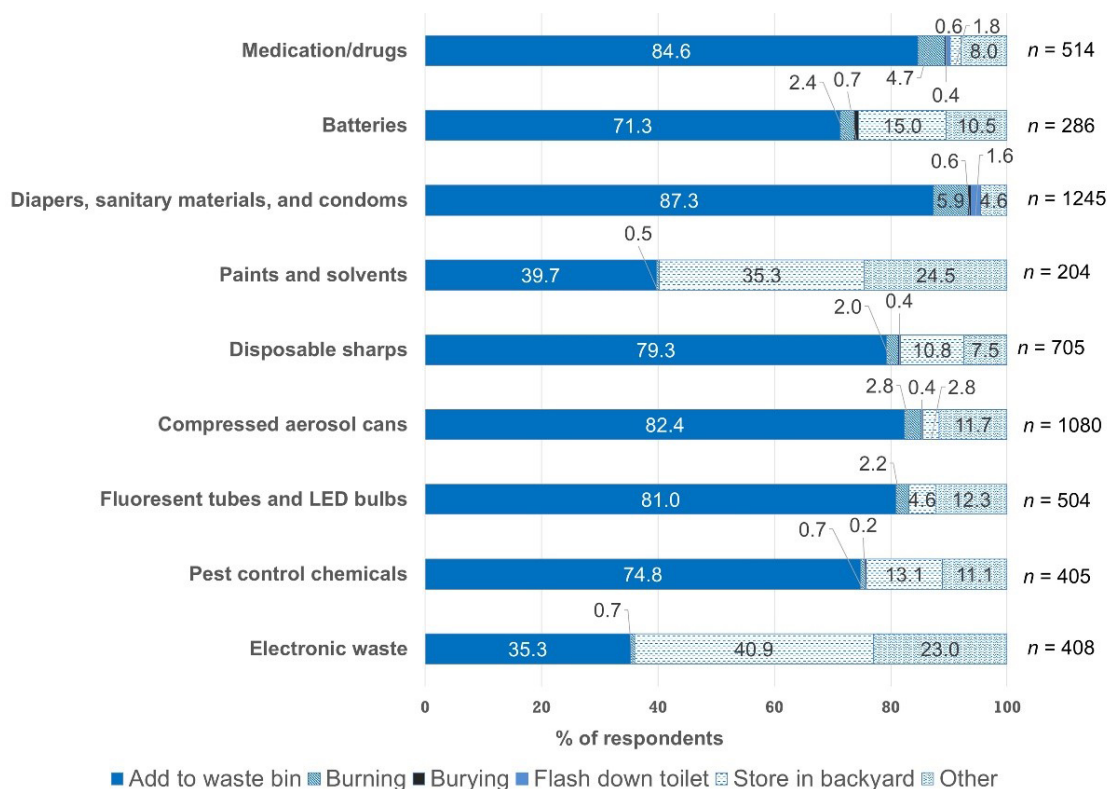


FIGURE 6: Distribution of household hazardous waste disposal methods.

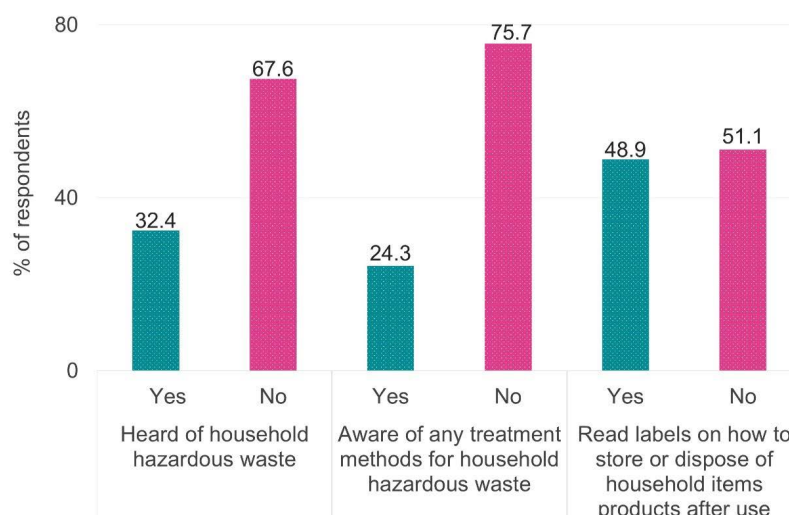


FIGURE 7: Household hazardous waste awareness rates the among respondents.

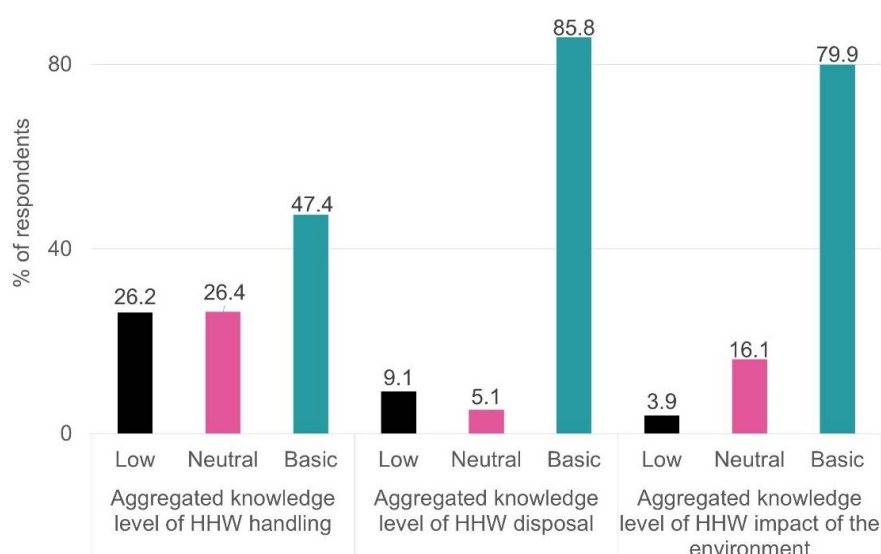


FIGURE 8: Respondents' knowledge levels regarding household hazardous waste handling, disposal, and environmental impact.

The study employed logistic regression analysis to investigate if awareness of HHW varied between different sociodemographic groups. The binary regression model accurately predicted whether a respondent had heard of HHW, with a model accuracy of 71.0%, a Chi-square (χ^2) value of 115.21 and a p-value of $<.0001$. The -2LogLikelihood value was 1452.6, R^2_{CS} was .09, while R^2_N was .12. Furthermore, the predictors had an acceptable level of multicollinearity, as the VIF was < 5 and the Tolerance was > 0.2 .

Based on the results presented in Table 4, the gender, education level, and major occupation of respondents were significant predictors of their awareness of HHW. However, age, income, and dwelling unit type did not significantly impact. The data indicates that being male increased the likelihood of hearing of HHW by 1.4 times (or 40%) compared to females (Exp (β) = 1.447, $p < 0.01$). Furthermore, higher levels of education increase the chances of hearing about HHW (Exp (β) = 1.217, $p < 0.01$). Among the different occupational groups, there was significant difference

in HHW awareness (Wald $\chi^2 = 16.25$, $df = 6$, $p < 0.01$). Specifically, government workers (Exp (β) = 2.741, $p \leq 0.001$) and students (Exp (β) = 1.886, $p < 0.05$) were 2.7 and 1.9 times, respectively, more likely to have heard of HHW than the unemployed.

During the policy dialogue, it was noted that there is a general lack of awareness and knowledge regarding HHW among both practitioners and the public. Although participants were familiar with some common HHW, such as car batteries, pesticides, and motor oil, they did not recognise other hazardous items, such as empty aerosol cans, fluorescent tubes, nail polishes, and creams. Here are some observations made during the workshop:

"It's surprising how little people know about household hazardous waste, even those who work in related fields. The main issue is that we don't always realize which home items can be hazardous. Many everyday household items pose a threat to our health and the environment, but we

TABLE 4: Relationship between sociodemographic profile and the likelihood of hearing of HHW.

Variable	β	Wald	df	Sig.	Exp (β)
Gender (ref: female)	0.369 **	7.286	1	0.007	1.447
Age	-0.011	0.014	1	0.906	0.989
Education	0.217 **	9.636	1	0.002	1.242
Average monthly income	0.104	2.995	1	0.084	1.109
Occupation (ref: unemployed)		18.096	6	0.006	
Artisan/craftsman	0.399	2.456	1	0.117	1.490
Government employee	1.008 ***	10.126	1	0.001	2.741
Private company employee	0.405	2.371	1	0.124	1.500
Trader	0.196	0.764	1	0.382	1.217
Student	0.634 *	4.134	1	0.042	1.886
Other	-0.092	0.127	1	0.721	0.912
Type of dwelling unit (ref: separate house)		3.142	5	0.678	
Apartment/flat	0.283	1.522	1	0.217	1.327
Compound house	-0.083	0.259	1	0.611	0.921
Improvised home	-0.248	0.252	1	0.616	0.781
Other	-20.453	0.000	1	0.999	0.000
Semi-detached	0.126	0.293	1	0.588	1.134
Constant	-1.934 ***	23.846	1	0.000	0.145

Note: ref means reference group; * $p \leq 0.05$; ** $p \leq 0.01$; *** $p \leq 0.001$; $N = 1245$; β = coefficient; Exp (β) = odds ratio

don't always recognize them as such." – Breakout Session Group 1

"Throughout today's discussions, a recurring theme has been the lack of knowledge or awareness around certain topics. As a group of well-educated, well-read and well-exposed individuals who grasp the complexities of various issues, it is both concerning and intriguing to note this trend. However, we also need to consider ordinary people who live in households with an average education level of junior high school or even lower. It is a cause for concern that they may not be aware of important topics. For instance, there seems to be a lack of awareness about the waste category we are currently discussing." – Academia Participant 1

During the workshop, it was revealed that informal workers handle and process 93-97% of Ghana's e-waste. The participants in the policy dialogue expressed concern that most workers in the e-waste industry have limited knowledge of the hazardous nature of the materials they handle. A consultant in the e-waste sector aptly described the consequences of leaving the handling of this waste stream to be managed by the informal players:

"Many workers in the informal sector do not wear personal protective equipment, such as gloves or boots, and are often exposed to unknown hazards. Although many of these workers are aware of the potential health risks associated with their work, their knowledge of these risks is often limited. Studies suggest that only 1-10% of workers have a comprehensive understanding of the risks they face, and even this level of knowledge may not be sufficient to change their behaviour." – MSW Management Consultant.

During the policy dialogue, three crucial factors were identified to address the issues surrounding the management of household hazardous waste (HHW) in Ghana. Several important points were raised, including raising awareness and intensifying public education on HHW. It was also emphasised that promoting and enforcing the segregation of household waste is necessary. The lack of specific laws regarding HHW makes it necessary to enact appropriate policies and legislation to govern its management and promote household waste separation. Additionally, it is critical to have enforcement mechanisms in place to ensure compliance. This would require adequately resourcing environmental health offices (in the districts and municipalities) and waste management departments (in the metropolises) to carry out their mandate. Furthermore, it was suggested that the country establish special courts to address environmental sanitation-related issues. Finally, it was necessary to establish special collection systems or arrangements with reliable services to sustain the public's interest in separating HHW. These insights were gathered from workshop participants:

"To summarise our position regarding household hazardous waste, although there are environmental laws in place in Ghana, no specific law or act currently addresses HHW from households. We believe that there should be a law created that clearly defines the term "household hazardous waste" and establishes penalties for those who violate it. This legislation would allow us to take legal action against offenders and protect our rights." – Legal Practitioner.

"...There is an urgent need to develop and enforce sanitation policies and legislation. Special courts should also be established to address sanitation-related issues." – Breakout Session Group 1

4. DISCUSSION

This study confirms that automotive products (e.g., vehicle batteries and motor oils, etc.), light bulbs, pesticides, home improvement products (e.g., paints, air fresheners, etc.), and household cleaners (e.g., disinfectants, bleaches, etc.) are familiar sources of HHW in urban sub-Saharan Africa (Edokpayi et al., 2017). The study highlights the everyday use of aerosol cans, such as insecticide sprays, body sprays, air fresheners, hair sprays, and others, as well as disposal sharps like razor blades, lancets, and injection needles, as well as expired and unused medications, fluorescent tubes, and e-waste among residents in the study area. This emphasises the need to pay attention to properly dispose of and manage these items. The prevalence of aerosol cans is attributed to the shift from traditional practices of using lime and lemon to control body odour to using containerised deodorants and the increased adoption of insecticide sprays in Ghana over the past few decades.

Additionally, the number of cases of non-communicable diseases, such as diabetes, heart disease, and cancer, has increased in the recent past (Ministry of Health, 2022). Managing diabetes at home may require frequent blood sugar level monitoring with glucometer kits, where the patient's finger is pricked with lancets for blood samples. In certain instances, patients self-administer insulin injections to regulate their glucose levels at home. These contribute to the widespread use of disposable sharps. The proportion of respondents who generated medication/drugs HHW in this study was 41.3%, less than the over 50% reported by Sasu et al. (2012).

4.1 HHW awareness and knowledge levels in GAMA

The results of this study show that around two-thirds of those surveyed were not familiar with HHW, and three-quarters did not know of any HHW treatment methods. Additionally, more than 50% of the respondents had insufficient knowledge of the proper handling of HHW or were uncertain about the correct procedures. These findings are similar to previous reports from Sub-Saharan Africa (Edokpayi et al., 2017; Ubachukwu et al., 2014) and Thailand (Decharat et al., 2020). These pieces of evidence underscore the necessity for heightened awareness and education in developing countries regarding HHW and its detrimental ecological and public health effects. Enhanced awareness can also drive public demand for government intervention, urging the development of policies and legal frameworks and the provision of logistical support for effective HHW management.

During a policy dialogue, it was discovered that even those working in the MSW management sector, who are expected to possess adequate knowledge about HHW, did not consider specific HHW fractions hazardous. Items such as vehicle batteries, pesticides, paints, thinners, and partially empty gas cylinders were quickly recognised as hazardous. However, other household hazardous items such as nail polishes, lipsticks, creams, and unused medication or drugs were not considered. It was also revealed that the management of HHW is entirely missing from Ghana's

waste management architecture and that no politician or sector minister has ever mentioned it. Although the country has hazardous waste legislation that targets industrial and commercial activities, there is no regulatory framework for HHW. This suggests that policymakers are unaware of the prevalence and risks of HHW. Therefore, raising more awareness among the public and key stakeholders about the risks associated with HHW is crucial. It should be noted that many countries in the sub-region, such as South Africa, Nigeria, and Kenya, do not have a legislative framework for Hazardous Household Waste (HHW). As a result, there is no specific definition of HHW for these countries (Mmereki et al., 2017). This lack of or variation in the HHW definition can affect the public's perception of what is hazardous and make it difficult to quantify and monitor HHW. This issue has been rightly pointed out as a cause of concern.

4.2 Implications of HHW knowledge level for HHW handling and disposal

As mentioned earlier, most of the participants were not familiar with HHW. More than half of the respondents needed guidance on how to handle HHW. Despite this, nearly 80% of the participants understood the proper disposal methods and the harmful effects of HHW on the environment. However, it is concerning that even with this knowledge, many respondents still disposed of HHW along with MSW. Our research aligns with previous studies (Sasu et al., 2012; Udofia et al., 2017), indicating that three out of four participants dispose of pharmaceutical waste with Municipal Solid Waste (MSW). Furthermore, the report highlights that over 80% of the study population disposes of sharp waste in household bins. However, our research contrasts with the situation in developed countries where safe receptacles are commercially available in pharmacies for individual use to store used needles safely. In Ghana, such safe containers are primarily found in medical facilities and are rarely available at the household level.

The study revealed that open burning of municipal solid waste was very common (23%) in the study area. However, we found that the deliberate burning of household hazardous waste was limited (0.5-4.7%). This result was not surprising since the study participants had basic knowledge about the harmful effects of HHW on the environment and public health. The study also found that in 2020, the Ghana Food and Drugs Authority (FDA) initiated the 'Take Back Unwanted Medicines from Consumers' (TBUM) Project to retrieve unused and expired medicines from households, starting from Accra (FDA, 2020). However, as rightly noted by Amoabeng et al. (2022), the lack of awareness of such programmes has limited their impact.

Regarding how to get rid of household hazardous products, there are usually instructions on the labels provided by the manufacturers. However, according to earlier studies Edokpayi et al. (2017), people in sub-Saharan Africa tend to ignore these instructions. Even those who read the labels do not follow the proper disposal instructions. Our study found that almost half of the respondents claimed to read the product labels before disposal, a claim which coincided with the number of our respondents who had

received at least a secondary education. This finding suggests that even those who claim to read the instructions still require help disposing of certain products due to a lack of separate disposal containers. Our observation was consistent, with over half of the respondents stating that certain household waste streams should be handled differently. These respondents also understood the consequences of improper hazardous waste management on public and environmental health. The irony is that despite this awareness among some respondents, many admitted to disposing of hazardous waste with regular municipal waste.

It appears that the handling and disposal practices of HHW in GAMA are influenced by several factors beyond the population's level of awareness and knowledge about the subject. Logistical, infrastructural, legal, and institutional gaps need attention. For example, the study found that 45% of respondents did not have approved waste bins and instead used sacks, crude containers, and other unapproved waste storage methods. This discourages the source separation of household waste. Additionally, previous studies (Deku, 2017; Sarfo-Mensah et al., 2019) have shown that waste collection services in the area are generally unreliable, raising concerns about whether the existing waste collection regime can support separate HHW collection. The unreliable waste collection services have created opportunities and economic niches for the growth of informal waste collectors, who serve a significant proportion (41.5%) of the residents in the area. Unfortunately, these informal waste collectors may not have enough knowledge and expertise in managing HHW, which poses a risk to themselves and the environment.

4.3 Policy implications and recommendations

The recent study highlights the urgent need to reassess the country's institutional and policy frameworks for managing MSW. Currently, there is no specific policy for the management of HHW. Therefore, it is crucial to have specific legislation that defines HHW and outlines a streamlined management process, from storage to disposal. Our study advocates adopting a comprehensive MSW legal framework similar to South Africa's Wastes Act and Austria's Waste Management Act. This framework should encompass handling various forms of waste, such as liquid, gas, solid, and particulate waste, recycling operations, and all parties' responsibilities throughout the waste management process. The policy must also highlight the obligations of governmental bodies, economic and financial mechanisms, EPR, and the movement of waste across borders as integral components.

Given that the country has numerous laws regarding environmental sanitation, yet their impact is limited, having specific legislation for HHW will not be enough to guarantee positive results. Previous studies have identified significant challenges in policy enforcement due to factors such as lack of political will, funding deficiencies, limited capacity and commitment of enforcement institutions, corruption, and poor sanitation culture. To overcome these obstacles, it is crucial to develop enforcement mechanisms. Environmental health officers must be appropriately recruited and adequately resourced to enforce the law at the community

level. Additionally, sanitation courts should be established to expedite offenders' prosecution and deter others from breaking the law.

We suggest implementing EPR programmes as part of the reform. The EPR system would hold producers accountable for managing their products at the end of their life cycle. Producers can finance HHW collection and disposal programmes through the EPR system to ensure proper disposal of hazardous household waste. The EPR financing scheme could also integrate the provision of appropriate storage receptacles for such programmes to encourage HHW separation at source. Currently, EPR programmes are limited to collecting plastic waste and e-waste; only a few such programmes exist in Ghana. These programmes are incentivised. In 2017, the Ghana Recycling Initiative by Private Enterprises (GRIPE) was established to promote recycling efforts among plastic producers worldwide. However, despite the good intentions, the initiative has not fulfilled its commitments and is often used as a marketing ploy to deflect blame for plastic pollution (Chellé & Dontoh, 2022).

The success of EPR in managing HHW depends on specific policies and regulations, as well as the active participation of producers in EPR initiatives. However, EPR schemes often struggle to succeed in developing countries due to poor waste management infrastructure, lack of transparency, and neglect of the informal sector (Talbot et al., 2022). Ghana faces the same challenges in integrating the informal sector into its waste management system. Therefore, policymakers should find ways to involve the informal sector to improve MSW management and build capacity. The government needs to expedite efforts to integrate the informal sector formally, as outlined in the 2020 National Waste Management Strategy (NWMS). The NWMS aims to address the integration issue by acknowledging, registering, and offering training to informal workers. However, progress in implementation has been slow. The NWMS could provide informal waste workers with necessary occupational health and safety training if successfully implemented.

Additionally, policymakers could combine EPR programs with economic instruments like packaging taxes and landfill fees to enhance their effectiveness (Vink, 2020). However, caution should be exercised as funds from such measures may only be allocated with a dedicated fund secretariat. This was seen with the 10% Environmental Excise Tax on plastic products in 2013 (Keesman, 2019). Therefore, the government should establish the necessary structures to ensure the timely release of funds accrued from financial instruments to sustain any EPR regime.

Effective waste management is crucial for Ghana, and this includes managing HHW. To achieve this, Ghana needs a comprehensive approach considering all forms of waste. The Integrated Sustainable Waste Management (ISWM) framework is a suitable basis for relevant policies because it incorporates the physical and governance structures necessary to manage all waste types (Letcher & Slack, 2019). The waste management hierarchy has proven to be a practical approach for managing HHW. However, the success of HHW management programmes depends on inclusivity - the involvement of the public, private and local government

institutions, national government, and producer responsibility organisations (Wilson et al., 2013). Therefore, Ghana's HHW management policies must prioritise educating and encouraging the public to reduce their consumption of HHW products. Strategies such as buying only the necessary quantities, opting for less hazardous alternatives, and fully utilising the products can help minimise HHW.

This study found that females, individuals with low formal education, and non-government workers were less likely to be aware of HHW. Therefore, policymakers must implement public awareness campaigns that are more targeted and inclusive to ensure that all sections of the public are informed about HHW. Additionally, research has shown that women are vital in managing household waste and are more likely to exhibit pro-environmental behaviours. (Lee et al., 2011; Odonkor & Sallar, 2021), which further emphasises their significant role in effective HHW management. Hence, failure to adequately raise awareness among women will hinder the success of HHW management efforts. In addition, it has been observed that Tuesdays are the best day for HHW education programmes in coastal communities, as most people do not go fishing on that day.

Efficient waste separation is crucial in reducing the depletion of natural resources, promoting recycling, and prolonging the lifespan of landfills by diverting waste. Households may practice waste separation for various reasons, such as financial incentives. Our research found that while 35-40% of respondents disposed of e-waste, paints, and solvents with regular waste, a similar percentage stored them in their backyards. Although paints and solvents can be stored for future use, e-waste is commonly hoarded for economic reasons. Our study also confirms that people from low- and middle-income communities are likelier to practice household waste separation and sell them for financial gain. However, recent studies suggest that personal ethical values also significantly shape people's intention to separate household waste (Oduro-Appiah et al., 2022). These internal normative pressures include separating waste as a show of respect for the environment and feeling guilty for not separating their waste to help protect the environment (Schwartz & Howard, 1981), which is deeply entrenched in the beliefs of the three major religions in the country. Therefore, campaign messages promoting HWS should reinforce moral obligations. Additionally, legislation and enforcement are necessary to encourage HHW segregation at the household level.

5. CONCLUSIONS

Our study aimed to assess participants' level of awareness and knowledge regarding Household Hazardous Waste (HHW). The results showed that HHW accounts for only a small percentage of Municipal Solid Waste (MSW) and is not widely known among the general public. However, some citizens have become more aware of HHW through local education and information initiatives. They can now identify hazardous materials in the waste generated in their daily lives. Despite its low volume, HHW should be collected separately and treated effectively. The current practice of disposing of HHW in conventional landfills can

lead to the release of potentially harmful materials into the environment, making it crucial to handle it with care.

Residents' knowledge level regarding hazardous waste disposal and its impact on the environment varied depending on the type of waste. This raises concerns about the health and safety risks associated with how some households in Ghana handle certain types of hazardous waste. Although many residents have basic knowledge about the adverse impact of hazardous waste on the environment and how to dispose of it properly, they still dispose of it with regular waste. Proper handling and disposal of hazardous household waste can be challenging due to various reasons such as lack of awareness, insufficient laws, poor enforcement, and logistical and infrastructural obstacles. Our study recommends educating the public about hazardous household waste and raising awareness through targeted campaigns to address this issue. It is crucial to enact hazardous household waste and household waste segregation regulations and appropriate enforcement regimes to combat this problem.

Effectively managing household hazardous waste involves addressing several essential aspects. Local authorities should promote and implement separate collection schemes designed for different groups of HHW items. These schemes should be reinforced where preliminary initiatives have already been taken. The government, producers, and retailers should collaborate to develop integrated take-back systems to ensure ease of use. It is crucial to support the existing legislative framework and use consistent terminology across different countries. This will enable reliable statistics to be collected. Reducing the amount of HHW generated should be a top priority. This can be accomplished by using less hazardous materials in manufacturing and making conscious daily decisions. These efforts will also positively impact the management of HHW, as prevention is key. It is important to note that there is a lack of accurate data regarding the amounts of HHW generated. Therefore, future research should examine the quantities of HHW fractions produced within the GAMA, starting with the most commonly encountered types.

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Extra contents

COLUMNS AND SPECIAL CONTENTS

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

Environmental Forensic

NEED FOR COMPREHENSIVE GUIDELINES FOR ENVIRONMENTAL FORENSIC INVESTIGATIONS

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1. INTRODUCTION

Identifying the polluter(s) and their relative contribution to the impacts of pollution is often the primary objective of an environmental forensic investigation. Such investigations need to be carried out in a manner meeting the requirements of the legal system that is in place designed to punish the polluters. The results need to be scientifically robust, but also presented in a way that is acceptable to the legal system, judge, and jury. These are important requirements because punishment to the polluters can depend on the quality of communication, as much as the severity of the impacts of pollution.

Just as in any specialized crime area, effective communication between the investigating experts and the judiciary is a challenge in environmental crimes (Custers, 2024). The inability of the judiciary to fully understand the technical nuances involved in the case and the inability of the technical experts to fully appreciate the importance of the legal requirements are the reasons behind this. An effective way to address this challenge is having peer reviewed, clear and objective guidelines and protocols, both for investigation and for the presentation of evidence. For the investigating experts, this will save a lot of effort as (s)he knows beforehand the expectations from the judiciary. It prevents the situation where a skilled opposing advocate finds it easy to get the experts' testimony struck from the records on procedural grounds, even when the expert is competent and his/her scientific methods reliable (Samad et al. 2015). As for the judiciary, it is now left with the easier job of verifying whether the guidelines are adhered to or not.

2. EXISTING GUIDELINES FOR ENVIRONMENTAL FORENSIC INVESTIGATION

The field of environmental forensics has grown significantly over the past couple of decades, and various organizations and institutions worldwide have come up with guidelines and protocols to standardize environmental forensic investigations. The most notable and explicit guideline is the one published by INTERPOL.

The "Pollution crime forensic manual" Volume I and II by INTERPOL provide guidelines on the sample collection procedure for some typical environmental crime scenarios and the subsequent documentation of the evidence (INTERPOL, 2014). A limitation of this manual when adopted for environmental forensic investigations is that it does not address the investigation in its entirety, but focuses on the sample collection and preservation procedures. INTERPOL has published specialized investigative manuals for illegal garbage discharge from vessels (INTERPOL, 2018) and illegal oil discharge from vessels (INTERPOL, 2007). The INTERPOL manuals (Figure 1) are obviously from the perspective of a law enforcing officer investigating a crime scene. But, for an environmental forensic expert, guidelines on the type of forensic tools suitable for a given situation, the most suitable sampling and analysis techniques for a given situation, availability of field kits, etc. are as important as the sample collection and preservation strategies.

Apart from the INTERPOL manual, there are guidelines prepared by many regulatory agencies that implicitly address some stages, mostly the sample collection part of an environmental forensic investigation. "The inspector's field sampling manual" by Environmental Canada is one such guideline 'that has been written to provide national standards and uniformity in environmental sampling for samplers including enforcement officers, other public officers and lay samplers' (Environmental Canada, 2005). This manual focuses on sampling for investigations of suspected violations, in addition to sampling for inspections, and emergency response. In fact, "The Inspector's Field Sampling Manual" was the basis for the 'Pollution crime forensic investigation manual' as acknowledged by INTERPOL.

The United States Environmental Protection Agency (US EPA) has Environmental Crime Task Force Teams stationed at various places to better support the enforcement of environmental crime and to deter crime before it happens. The Central Virginia Environmental Crimes Task Force has prepared a 'Resource guide for the investigation of environmental crimes' (The Central Virginia Environmen-



FIGURE 1: Manuals by INTERPOL for Environmental Crime Investigation.

tal Crimes Task Force, 2005). Although the purpose of this document is to provide practical information for the investigation of potential environmental crimes, its scope is also limited to evidence collection and preservation. There are manuals/handbooks prepared by many other US States on Environmental Crimes, like the 'New Jersey Environmental Crimes Handbook', 'Louisiana Environmental Crimes Handbook', etc., but the focus is on listing different types of environmental offenses and the legal provisions to tackle those offenses. The 'Royal Solomon Islands Police Force (RSIPF) Environmental Crime Manual', although elaborate, focuses on types of environmental offenses, legal provisions against those offenses, procedures in its trial and the punishments.

The environmental statutes of many countries have provisions regarding the procedure of collecting samples as part of environmental enforcement, for example Sections 10 and 11 of The Environment (Protection) Act, 1986 of India or Section 108 of the Environment Act, 1995 of the UK. Adherence to these provisions are important to convict the polluter, and hence important for an investigating expert to know.

The manuals, guidelines and statutory provisions discussed above do not go into detail on the vast variety of different techniques available to an environmental forensic expert. Although these manuals present very important in-

formation, the expectations from a comprehensive manual for environmental forensic investigation is that it should provide clear and in-depth details on all aspects of the investigation of an environmental crime. This would help an investigator to carry out the investigation effectively, even if (s)he is a novice to the field. More importantly, it will help the judiciary, which is otherwise not trained in the technical aspects of investigation, to verify if the procedures adopted in the investigation are scientifically sound and decide on the merits of the case.

3. SOME SUGGESTIONS FOR A COMPREHENSIVE ENVIRONMENTAL FORENSIC INVESTIGATION MANUAL

The realization that environmental crime is an important crime area has resulted in the formation of many specialized environmental crime enforcement agencies in countries across the world. There is also a suggestion to integrate environmental forensics to criminal investigation to tackle convergence of environmental crime with traditional crimes (Priya et al., 2023). Thus, there is an increasing demand for environmental forensic experts. The situation also calls for effective standards and guidelines for environmental forensic investigation. The American Society for Testing and Materials (ASTM) had set up a subcommittee,

E50.06, to frame comprehensive guidelines for environmental forensic investigation in 2005. But they could not take off due to non-technical issues (Petrisor, 2005).

The legal requirements regarding the investigation of an environmental crime may vary from place to place depending on the procedural and substantive laws in force relevant to the case at the place. However, the technical aspects like the sampling and analysis procedure to be adopted, methods to ensure integrity of samples, interpretation and presentation of the results of analyses, etc. are applicable to all jurisdictions. This allows formulation of a 'universal' technical manual for environmental forensic investigation.

There is a lot of scientific knowledge on environmental forensic investigation being generated in different parts of the world. The journal 'Environmental Forensics' focuses on publishing latest developments in this scientific area, the International Network of Environmental Forensics is a Royal Society of Chemistry group holding annual conferences to act as a forum for scientists, environmental consultants, regulators and lawyers to share information. There are also a range of excellent text books that provide detailed information on environmental forensics as a whole (Morrison and Murphy, 2015), environmental forensics methods (Mudge, 2008), as well as environmental forensics for specific groups of pollutants (Morrison and Murphy, 2006; Wang and Stout, 2007; O'Sullivan and Sandau, 2013). Over 3000 manuscripts have been published on the topic of "environmental forensics". So this is a well-researched area, yet it can still be a challenge to transfer this knowledge from the academic community into a structured legal setting. If this information, spread across different sources, can be brought to a common platform and updated periodically, it will be a great resource for the environmental forensic practitioners.

4. CONCLUDING REMARKS

Environmental forensics is important for the delivery of environmental justice. Although it is a relatively new area in many countries from a legal perspective, there has been significant scientific advancement in the area in the last two decades. Realizing the need for strict environmental

crime enforcement, many regulatory agencies have recognized the importance of environmental forensics. This has resulted in the development of many basic guidelines and manuals that can be useful for environmental forensic investigations. However, considering the amount of new knowledge being generated in the area, it is time the experts in the area come together under the aegis of an international regulatory agency like the INTERPOL, to develop a comprehensive manual for environmental forensics, that covers all aspects of an environmental crime investigation.

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Grassroots Innovations from Youth

MAPPING ZERO WASTE CHALLENGES WITH MIDDLE SCHOOLS STUDENTS IN BRAZIL AND CANADA

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1. INTRODUCTION

One of the main challenges in cities today is solid waste and its inadequate management, along with the consequences of this mismanagement for citizens. Additionally, urban areas face numerous other waste-related challenges that must be addressed. Urban sustainability hinges on effective solid waste management, particularly in light of the growing generation of plastics, population expansion, and increased consumption. Moreover, there are significant bottlenecks in waste governance, such as the absence of fiscal incentives and the lack of selective waste collection programs (Alfaia et al., 2017; Pacheco et al., 2012). These challenges have further increased by the surge in waste volumes since the pandemic (Forlani & Njie, 2022).

Climate change and accelerated urban growth can complicate the organization of our waste systems and exacerbate the problems associated with mismanagement. For example, inadequate waste collection services and facilities can lead to the clogging of urban drains during heavy rain episodes, causing flooding scenarios. The population is also more susceptible to diseases linked to waste exposure, such as Leptospirosis, cholera, and other infectious diseases. Lack of waste collection and littering cause environmental pollution, negatively impacting society, which is further aggravated by inequality (Owusu-Sekyere, Attakora-Amaniampong, & Aboagye, 2016). Selective waste collection, on the other hand, prevents the spread of diseases and ensures that waste is disposed of correctly, promoting a more circular economy and decreasing greenhouse gas emissions. Thus, improving solid waste management in our cities enhances the quality of life for all citizens, especially those in vulnerable situations.

Involving youth in the development of urban waste management policies is crucial for several reasons. Young people are future inhabitants and stewards of urban environments, making their engagement essential for sustainable solutions that directly affect them. Their fresh perspectives and innovative ideas inject vitality into policy discussions, driving initiatives that promote recycling, reduce waste generation, and advocate for responsible consumption (Marchezini & Trajber, 2016). Additionally,

empowering youth in decision-making processes fosters a sense of ownership and responsibility towards their communities, leading to more creative and effective strategies.

Ultimately, by including youth in policy design, cities can harness their energy, enthusiasm, and insight to create greener, cleaner, and more livable urban spaces for all (Kallio et al., 2013; Kallio et al., 2014; Kraftl, 2018; Walker, 2019; Driskell, 2017). Rather than merely observing their living conditions, children and youth can and want to contribute to changing their urban environment and actively engage in their community (Kania et al., 2020; Trajber et al., 2019). It's important to recognize the potential of young people's perspectives in driving changes in public policy (Walker, 2019).

Based on this context, our research explores ideas of zero waste with grassroots actors and youth in São Bento do Sapucaí (São Paulo), Gonçalves (Minas Gerais), Brazil, and Victoria, Canada. The study engages students aged between 12 and 14 years old, using participatory research methods, including community mapping, group discussions and workshops to capture their perceptions of waste issues in their environments. While there are some studies available on schools and food waste, there is a gap in the literature on the generation of household waste on school premises and their surroundings, recognizing the potential of students as innovators to address this issue. The aim of our study is to develop proposals for improving waste management in the study region. This article discusses initial results obtained from the interviews with teachers in Canada and Brazil and from the student engagement on the topic in Canada, highlighting the emerging innovative ideas.

2. ZERO WASTE AS A DRIVER FOR GRASS-ROOTS INNOVATION

The concept of Zero Waste serves as a catalyst for the design and management of products and processes and aims at systematically minimizing the volume and toxicity of waste materials, especially plastics (Zaman, 2022). It embodies a visionary approach encouraging individuals to alter their lifestyles and practices to prevent waste generation, while ensuring that all discarded materials are

designed to be utilized as resources for future purposes (Tamminen & Lobin, 2022). Zero Waste fundamentally challenges the notion that waste is inherently valueless and an inevitable byproduct of consumption.

Zero Waste is an aspirational goal and serves as a catalyst for enhancing waste management by addressing the escalating waste volumes and advocating for policy reforms. It also means dedicating resources to environmental education and training, raising awareness and fostering research to drive behavioral and consumption changes (Gutberlet, 2016). Many cities and educational institutions around the world are already embracing these principles; and the city of Victoria in Canada is a good example for the growing awareness about the need to change consumption and discard habits (City of Victoria, n. d.).

Youth participatory action research emerges as a potent strategy to amplify the voices of young people. Recent studies exploring geographical knowledge within urban environments have engaged children and youth, whose perspectives offer valuable insights into tackling complex challenges such as climate change, disaster risk reduction, and resource security, while also proposing effective solutions (Marchezini & Trajber, 2016). According to Trajber et al. (2019), the knowledge and perspectives of young people are often disregarded despite their disproportionate vulnerability to socio-environmental impacts. Working with social actors, that may have been previously outside the policy process, such as youth and grassroots organizations can fill in missing links in promoting waste reduction and resource diversion into the Circular Economy (Carenzo, Juarez & Becerra, 2022).

Therefore, fostering youth participation is crucial for achieving sustainable household waste management (Naldi, 2023, Perera, 2016, Pulubuhu & Alhaqqi, 2019). Meaningful engagement with youth requires a space to learn about these critical issues, and capacity building that supports youth as resilient innovators and champions.

3. PROJECT STEPS

The project envisages several steps, as outlined in Figure 1, to learn from youth and grassroots actors about waste challenges and zero waste perspectives.

At the outset of the research, we conducted a bibliographical and documentary review on household waste management issues related to the study areas. We identified and mapped zero waste initiatives in the study region, building a database. Once we identified the schools to work with, we presented the project to the school coordinators and teachers and planned the interviews with the teachers responsible for the grade 7 classroom. Next, we present our initial results from the engagement with the teacher and the students at Monterey Middle School in Victoria. The project received approval from the UVic Research Ethics Board (Protocol # 21-0406).

We conducted four 2-hour meetings and a final 1-hour meeting with the students from this classroom. The first meeting involved an introduction to the topic and group discussions to brainstorm on the challenges that come with waste and the perceptions of zero waste.

During the second meeting we applied the dynamic group discussion format of the “world café”, to develop an

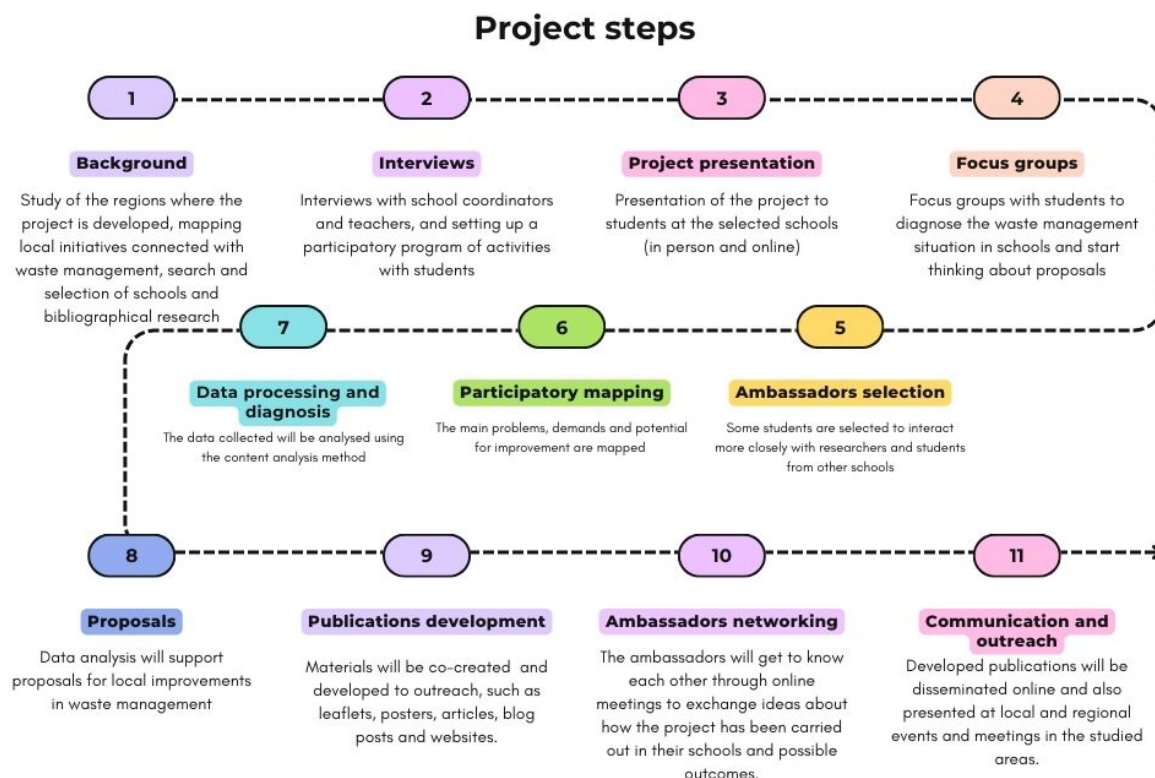


FIGURE 1: Flowchart of research steps (Source: the authors).

action plan with the students. They also identified students to act as ambassadors for the project, who would collaborate closely with the UVic team and future student ambassadors in Brazil. The third meeting exposed the students to a simulation on cooperative recycling in Brazil, supported by Virtual Reality (VR), to educate the students on participatory and grassroots initiatives in waste management. Additionally, a participatory mapping activity was conducted where students marked on a printed map of the school and its surroundings the main issues discussed in focus groups. During the fourth meeting we provided feedback on the mapping activity with the key findings. The students then collectively constructed a 'Dream Tree' (see Figure 2), where they wrote or drew their aspirations for a zero-waste future on a leaf which was glued to the tree. We also applied a questionnaire with 7 open ended questions and provided the students with 15 to 20 minutes to conclude this exercise. In our last meeting, the ambassadors presented their action plan to the classroom, and we finished with a presentation answering the question: where does our waste go? A similar research process will be applied over the next months in the schools identified in Brazil. In the following section we will discuss the findings from the interviews with teachers in both countries and from the interventions in the classroom in Victoria, Canada.

4. SOME INSIGHTS FROM TEACHERS AND STUDENTS IN MIDDLE SCHOOLS IN CANADA

Text The interviews with school coordinators and teachers in Brazil reveal limited knowledge about Zero Waste within schools. However, the interviewees expressed sig-

nificant concern about current waste management practices. It was noted that there is a need to replace the school's trash bins and ensure that they are placed in appropriate locations. Motivating students to reduce waste and use bins correctly appears to be a significant challenge, highlighting the necessity for initial education and awareness efforts. Another issue raised was the wastage of materials such as paper and notebooks. Furthermore, the thought seems to prevail that if rubbish is produced, it guarantees the jobs of the cleaners; thus, it is not perceived as an issue to change.

Another complex notion to work on is proper waste separation. When asked to segregate in too many categories, it seems to make the task to sort more difficult for the students. There have been lectures on selective waste collection, but as they are infrequent, the students end up forgetting the lessons. In other words, there is no continuity in promoting best practices. According to the coordinator of one of the schools, it is necessary to reinforce the theoretical part and find ways to turn theory into action.

At Monterey Middle School, interviews with two teachers revealed inconsistencies in waste management practices, a lack of information for both teachers and students regarding consumption/waste behaviour, and a general unfamiliarity with the concept of Zero Waste among school members. While students engage in discussions about social justice and environmental issues, including climate change, they struggle to connect their personal actions with broader environmental impacts. Specific concerns were raised about soft plastics from wrappers and snacks brought from home. The interviews suggest a need for



FIGURE 2: Collective construction of the 'Dream Tree' for zero waste (Source: the authors).

developing a comprehensive waste management plan tailored to the school's needs, which could serve as a model for other schools as well.

During the group discussions in the second workshop with the students, we asked the following questions:

i. How is waste currently taken care of in your school? What problems do you find? List the problems.

ii. How should the school deal with its waste (e.g., paper, plastics, food waste)? How could students, teachers and staff contribute to a better waste management?

iii. What could help schools in Victoria to become zero waste? What could you specifically contribute? What products could be created by your class to help change the situation?

The questions helped us gain insights into the primary waste management issues at Monterey school from the students' perspective, as well as generated ideas for solutions and awareness-raising initiatives (see Figure 3).

With the participatory mapping activity, we were able to locate the hotspots related to waste generation and accumulation in the neighbourhood. This information will help prioritize actions for a co-created waste management program for the school. Using these maps, students can pinpoint priority issues and show the real demands of the school.

The various other classroom interventions contributed to identify strategies for addressing the key waste-related issues and to generate suggestions for interventions both within and beyond the school. For example, students advocated for reducing paper and tape usage and suggested that teachers utilize more digital resources and print only when necessary. They also highlighted plastic tape as a significant contributor to the problem due to its non-biodegradable nature. Furthermore, students emphasized the importance of shifting perceptions around waste management. Currently, cleaning up garbage is often employed by the teachers as a punitive measure. The students propose a paradigm shift: promoting waste sorting through positive reinforcement rather than punishment.

Lastly, student ambassadors have committed to creating a poster and a presentation to disseminate information about Zero Waste in other classrooms. Their aim is to raise awareness among students about the issue and encourage sustainable practices.

5. FINAL CONSIDERATIONS AND NEXT STEPS

Based on first research findings, there is an urgent need to educate both students and staff on the importance of effective solid waste management within schools and their

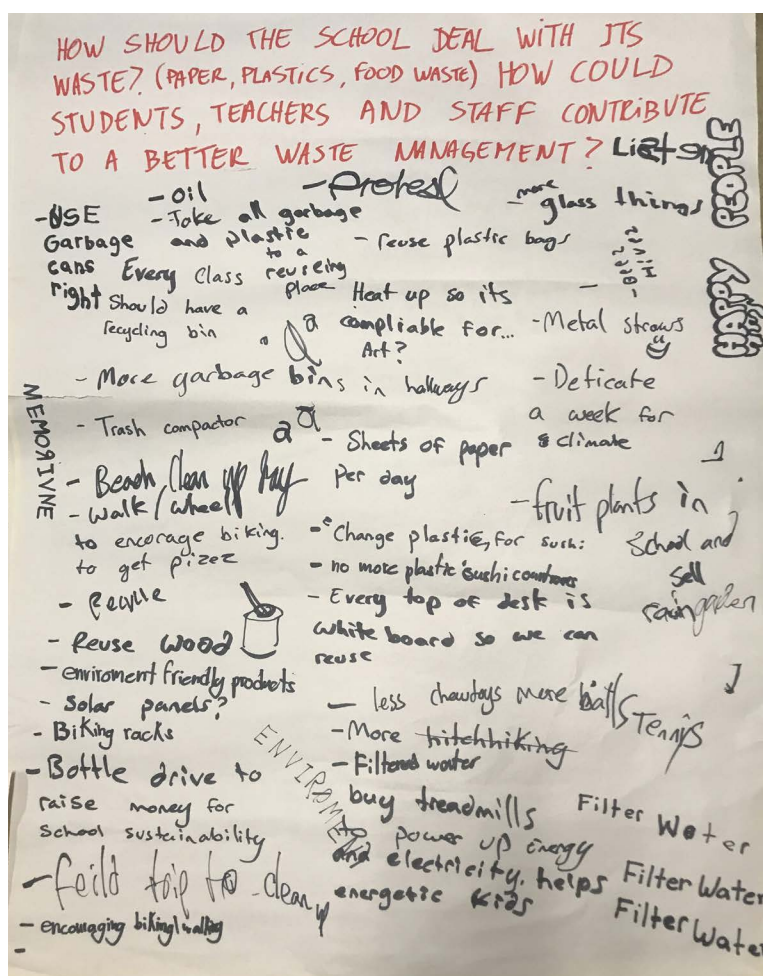


FIGURE 3: Group discussion sheet with students' perceptions (Source: the authors).

surrounding neighborhoods. This education should include teaching proper waste separation methods, assessing the types of waste generated, understanding the associated issues and implications within the school environment, and collaboratively developing an action plan for efficient and effective waste management.

The next steps in this project involve finalizing participatory activities in schools in both countries, completing the final products for each class (in the case of Monterey school the outreach material such as a presentation and a poster), and disseminating the project's results and products.

In Victoria, the plan is to distribute the final products to other schools, sharing acquired knowledge and promoting the necessary measures for transitioning to more efficient waste management in schools, raising awareness about waste challenges in urban settings. In Brazilian cities, the goal is to disseminate the produced materials to other schools, contributing to citizen education toward achieving Zero Waste goals and to also specifically target the semi-urban and rural areas adjacent to the towns under study.

Finally, we can already conclude that youth, as grassroots innovators, play a pivotal role in driving societal progress through their creativity, resourcefulness, and passion for change. Unlike established institutions, youth bring a fresh perspective unencumbered by traditional norms, allowing them to identify unconventional solutions to pressing challenges. By harnessing their skills and innate curiosity, youth innovators not only develop novel solutions but also inspire others to actively engage in problem-solving activities. Our research findings, so far, suggest that young people play an important role as grassroots innovators and providing them with the necessary support and giving them autonomy, they engage in promoting collective actions for change, particularly if these actions are supported by their teachers and can be followed up in the classroom.

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

by Rainer Stegmann

Last December I visited the beautiful picturesque Zhangjiajie natural park in Wulingyuan in Hunan Province, China. This UNESCO world heritage site consists of karst mountains with up to 200 m high sandstone pillars. After two days of hiking in this unique landscape (some call this area also Avatar mountains) my guide showed me a very special art museum: the Zhangjiajie Junsheng Sandstone Painting Institute. This museum was named after Li Junshen the founder of sand stone painting and houses only paintings created in this technique. "Sheng Painting is a new kind style which uses natural materials, like sand, stone, plants. It makes a breakthrough about drawing materials developed from minority 's art and absorbs the other style's advantages". <https://www.discoverzhangjiajie.com/junsheng-sandstone-painting-institute.html> In this institute there is also an education center where students learn this technique.

The artists use differently colored sand of different grain size and sprinkle it on stone plates and other materials. At the end the sand is fixed for preservation.

The results are amazing. I present 3 examples of very different character. One can only see the grains of sand when you get close to the painting. Amazing how he achieves all the nuances like in an oil painting.



Li Junsheng. Flowers.





Li Junsheng, Mountains.



Li Junsheng, Landscape.

These very beautiful sand paintings show the character of the objects in an impressive way. But what has this kind of art to do with waste? Sand and waste wood, plant residues that are also used as materials often become waste. Large constructions may produce a lot of soil without further use in that area. For non-useable sand and other kinds of uncontaminated soils and inert materials the landfill Category 0 was introduced in Europe with different requirements to be met. We experience more often that products suitable for further use like textiles, smart phones and sand become waste often due to non - available users and/or for economic reasons. To find ways for further use of such materials is one of the great challenges we are confronted with in waste management. Sand painting will not solve that problem but may point to it.



In the next issue of DETRITUS I will present a sculpture from N'Dorah, a well known artist born in Cameroun now living in Paris and Venice. She is exhibiting also at the Venice Biennale. She uses different materials including fibres from different origin.

CONTENTS

Editorial

CIRCULAR ECONOMY BETWEEN BUSINESS AND BIODIVERSITY

M.C. Lavagnolo, G. Felici and A. Lanzavecchia 1

Packaging

APPLICATION OF NANOCELLULOSE IN FOOD PACKAGING - A SWOT ANALYSIS

O. Kramer, J.K. Eikenbusch, F. Gheitasi and J. Wu 3

Biological Treatment

PROPORTIONAL EVALUATION OF WATER HYACINTH AS A RAW MATERIAL FOR BIOGAS, BIOMANURE AND SOIL CONDITIONER GENERATION VIA ANAEROBIC PROCESSES AT LAB AND PILOT SCALE

K. Janga, S. Begum and G. Rao Anupoju 12

OCCUPATIONAL EXPOSURE TO AMMONIA DURING INDUSTRIAL-SCALE ANAEROBIC DIGESTION COMPOSTING OF HOUSEHOLD WASTE

P. Battais, F. Bonthoux, S. Lechêne, J. Klingler, J. Grosjean, N. Monta and J. Kunz-Iffli 27

Recycling

HONG KONG'S CIRCULAR ECONOMY ON PLASTICS: ARE THE ECONOMICS FAILING?

B. Steuer 37

RECYCLING OF PHOTOVOLTAIC CELLS – A REVIEW

A.A. Vučinić, J. Bartulović, V. Zanki and I. Presečki 47

Biowaste to energy

MATHEMATICAL MODELLING OF QUANTIFYING BIOETHANOL FUEL PRODUCTION FROM PINEAPPLE WASTE

M.A. Imteaz, A.B.M. Sharif Hossain, A. Ahsan and P. Kaur 54

Landfilling

RELIABILITY OF GUELPH PERMEAMETER FOR DETERMINING THE HYDRAULIC CONDUCTIVITY OF MINERAL LINERS

Gi. Cortellazzo, F. Benedetti, S. Busana and M. Favaretti 66

ENVIRONMENTAL LAW ISSUES REGARDING PFAS POLLUTION IN WATER AROUND LANDFILLS IN SWEDEN

M. Johansson, M. Pettersson and J. Kumpiene 78

Policy and strategies

THE TACIT END-OF-WASTE STATUS IS de facto USED IN THE EU AND CRUCIAL FOR THE CIRCULAR ECONOMY

P. Hennebert 89

HOUSEHOLD HAZARDOUS WASTE: GAUGING KNOWLEDGE LEVEL AND ITS IMPLICATION FOR DOMESTIC WASTE HANDLING AND DISPOSAL PRACTICES

P.K. Dagadu, G. Sagoe and M. Oteng-Ababio 99

Columns

ENVIRONMENTAL FORENSIC

Need for comprehensive guidelines for Environmental Forensic Investigations I

GRASSROOTS INNOVATIONS FROM YOUTH / Mapping Zero Waste challenges with Middle Schools students in Brazil and Canada

IV

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

IX