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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 the Emerging Sources Citation Index (ESCI)**, **Clarivate Analytics, Web of Science**. 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

WHAT ABOUT RESIDUES FROM CIRCULAR ECONOMY AND ROLE OF LANDFILLING?

The current European policy for waste management is based on the concept of Circular Economy, intended as a global strategy consisting in a set of actions aimed at maintaining the value of the products over a longer period, saving non-renewable resources and minimising waste generation.

The emphasis placed subsequently on the recycling of waste has promoted throughout Europe a marked increase of separate collection, often perceived by politicians and citizens as the definitive solution for any waste disposal problem.

Consequently, the hierarchical waste management strategy (Prevention, Reuse, Recycling, Energy recovery, Landfilling) has frequently been approached in moral and demagogic terms rather than in technical terms, and the bad boys of waste management technologies, (i.e. incineration and landfilling) have been blacklisted as the forbearers of all evils, often being deemed superfluous.

In particular, landfilling has been banished as a hazardous system, obsolete and polluting, as strongly underlined on both a political and regulatory level.

This emphasis has resulted in the underestimation of a series of aspects that are currently emerging quite dramatically:

- not all wastes produced can be disposed of by means of separate collection and recycling;
- wastes cannot be recirculated endlessly;
- recycling activities in turn generate residues and wastes, even in the unlikely hypothesis of separate collection achieving a 100% rate;
- hazardous substances contained in the original products accumulate in the recycled materials posing risks for human health during their use and for the environment in case of uncontrolled disposal;
- the negative perception of landfills and incineration, largely stoked also by the communication strategies of the European Union, has fostered a non-acceptance amongst public opinion that today has turned heavily against the administrators attempting to establish the necessary facilities;
- although viewing landfills as an obsolete system to be abandoned, nothing has been done to date on a regulatory level to promote sustainability of the system and allow it to act as a virtuous sink to close the material loop and immobilize the abovementioned contaminants.

Recycling processes, in the same way as procedures used in the processing of natural resources, can be outlined in four distinct stages: extraction, selection, refinement and production (Figure 1).

Each of these stages produces residues that will subsequently need to undergo treatment in order to render innocuous or immobilise potential contaminants; lastly, a final sink will need to be identified in which to store the wastes safely and sustainably over an extended period of time, as is the case with any type of natural cycle .

On the basis of the flow chart provided in Figure 1, the total amount of waste/residue generated (R_{tot}) is thus obtained from the difference between the diverse waste flows (Wi) and the final products (Ph), as described by the following mass balance equation:

$$R_{tot} = \sum_{i,j,k,h} (R_i + R_{i,j} + R_{i,j,k} + R_{i,j,k,h}) = \sum_i W_i - \sum_h P_h$$

The amounts of residues, where they are generated and how they are treated and disposed of are clearly evident from the official statistical waste management data available for different countries.

Indeed, data routinely communicated relate to the quantities and percentages of wastes which, downstream of collection, are forwarded to the three main treatment options – recycling, thermal treatment and landfilling, whilst

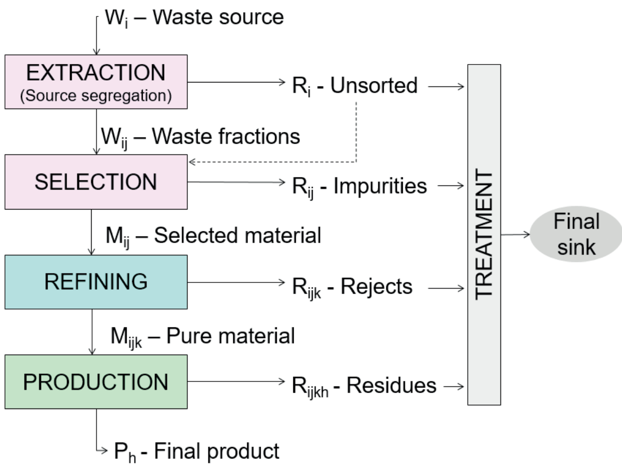

FIGURE 1: Recycling processes and related flows of materials (W: Waste, M: Valuable materials, R: Residues, P: Products) (i: different waste source, j: individual recovered fraction, k: pure materials from the individual recovered fraction, h: individual type of product).

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ignoring the disposal flows of treated waste. This however is misleading as it masks the effective use of both landfills and incineration. As an example, countries such as Germany, the Netherlands, Denmark and Japan would seem to have more or less eliminated the use of landfills, reducing this to a mere 2-3%, thereby endorsing the idea amongst the public in other countries that landfilling is superfluous.

Likewise, inclusion of the production of SRF (Solid Recovered Fuel) or RDF (Refuse Derived Fuel) in recycling conceals the fact that most of these materials ultimately end up in thermal treatment plants, while some are co-fired in other facilities (coal power plants, cement plants, etc.).

The entire international scenario would therefore be better represented by taking into account the actual disposal of wastes.

Indeed, considering how disposal represents the action or process of getting rid of something, the disposal of waste is actually achieved through the production of end products in recycling, together with gasification of material in thermal treatment and permanent depositing in landfilling.

Consequently, statistical data available worldwide should be updated to take into consideration residues originating from different treatment options (recycling, thermal treatment, landfilling) and the disposal routes according to mass balance flows schematised in Figure 2.

The following assumptions are made here in:

- only 60% of the recovered waste fractions become new products (as average considering the different materials);
- at least 20% of residues from the processing of recovered waste fractions are sent to thermal treatment;
- 20% of the residues from recycling processes are landfilled;
- the total amount of solid residues from thermal treatment ranges around 30%, including APC residues and fly ashes (30%), and bottom ashes (70%); assuming that half of bottom ashes are recycled, the disposal routes for thermal treatment residues could be calculated as 20% of total thermally treated waste to landfilling and 10% to recycling;
- no material is recovered from landfill, although using an Enhanced Landfill Mining concept this would be possible.

These assumptions are of course indicative and may vary significantly not only from one waste fraction to another but also from one country to another.

By implementing these assumptions in the statistical datasets describing waste management in different countries a situation diverse from that commonly represented is described. This is clearly evident from Figure 3, where the two different situations are represented using the triangular chart originally proposed by Cossu (2009).

Specifically, Figure 2A illustrates the percentage of use of the different treatment methods based on statistical data provided by The World Bank (2018) according to mass balance boundary A of Figure 2. Figure 2B provides actual disposal rates calculated according to the assumptions described in Figure 2 (mass balance boundary B).

Analysing the two graphs the following concluding remarks can be drawn:

- considerable amounts of residues are produced throughout the different recycling processes, including products which cannot be further recycled for technical or environmental reasons (accumulation of contaminants); these residues need a secure final sink in order to avoid diffuse pollution;
- landfilling plays a much higher crucial role in waste management strategy than generally recognised by authorities;
- thermal treatment represents an important option in countries with a high population density, where it is conveniently coupled with Circular Economy actions;
- "zero waste" (ZW) appears an even more unrealistic proposal, and can be considered solely when viewed as a conceptual trend;

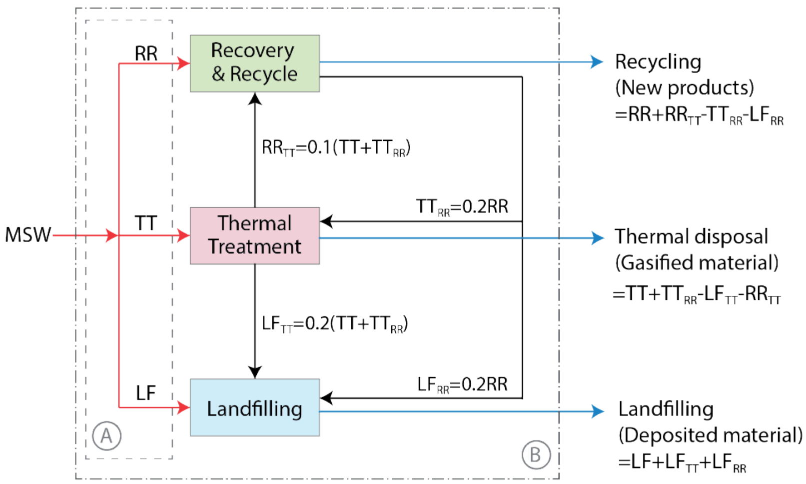

FIGURE 2: Mass balance flows of a Municipal Solid Waste (MSW) system, considering two different boundaries: A=waste treatment; B=waste disposal. (MSW: Amount of handled waste; RR, TT, LF: Amount of waste sent respectively to Recovery and Recycle, Thermal Treatment and Landfilling; TTRR, LFRR: Amount of residues (% by weight) from Recycling sent respectively to Thermal Treatment and Landfilling; RRTT, LFTT: Amount of residues (% by weight) from Thermal Treatment which are sent respectively to Recycling and Landfilling).

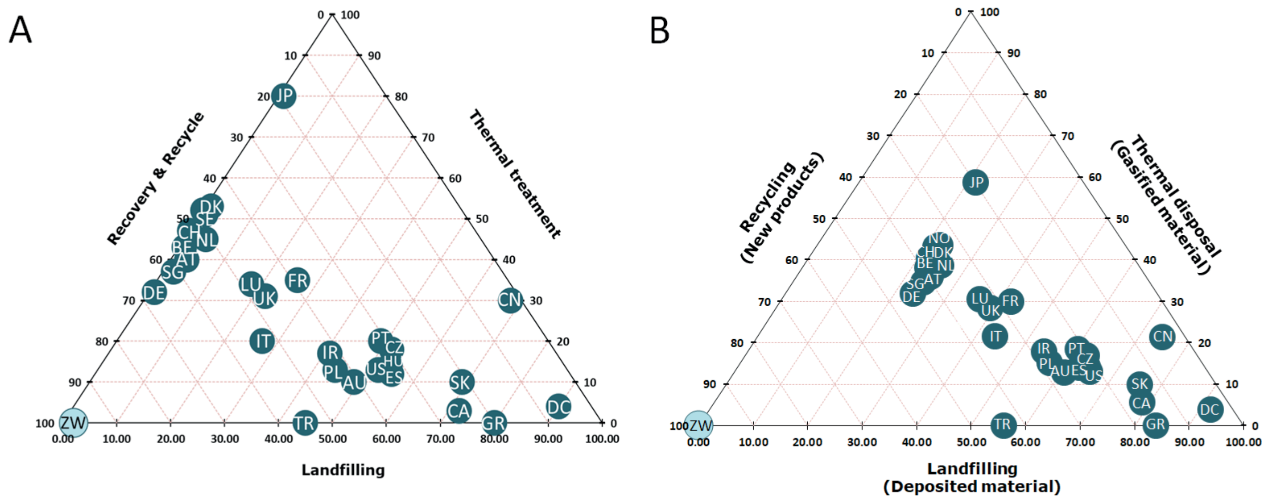

FIGURE 3: Graphical representation of the scenario of municipal solid waste management throughout the world arranged according to the three main options of treatment and disposal using the triangular chart proposed by Cossu (2009). Countries have been indicated using their e-mail country codes. DC = Developing Countries; ZW = Zero Waste. Chart A represents data derived from The World Bank (2018), Chart B describes the situation based on actual disposal rates, calculated according to assumptions described in Figure 2.

- landfilling terminology should be defined in order to differentiate the wide range of technological applications and quality of accepted waste (untreated waste, mechanical-biological pre-treatment, predominantly inorganic waste, etc);
- landfilling should be conceptually and technically remodelled in order to fulfil the fundamental strategic role of acting as a sink in Circular Economy strategies; this should be reflected in a new set of landfill regulations.

Some of the abovementioned aspects have been recognised by the European Union, which specifies in the Directive 2018/850/EC: "In order to ensure the reliability of data, it is important to lay down more precisely the rules according to which Member States should report municipal waste that has been landfilled". However, the results of this approach are not yet evident, also due to the failure of several countries to date to implement the Directive in national regulations.

In the same Directive, the EU maintains a negative approach towards landfilling ("A progressive reduction of landfilling is necessary to prevent detrimental impacts on human health and the environment") considering it a kind of dustbin in the waste management systems lacking the role of final sink to close the material loop, which would allow the diffusion of contaminants to be kept under control (Cossu, 2016).

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Letter to Editor

TREATING THE SYMPTOM? A MARXIST REFLECTION ON 'ZERO WASTE' AND SARDINIA 2019 SYMPOSIUM

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ABSTRACT

Written as a reflection on Sardinia 2019, the purpose of this letter is to draw attention to a perceived failure within waste management studies to adequately engage with the socio-economic and socio-political conditions that drive the prodution of waste. By way of a solution, it proposes a return to Marxist dialectics and modes of anlaysis in order to reframe contemporary debates on waste management practices to include more critical discussion and engagement with the root causes of waste, specifically capitalist production and class- addressing the illness rather than merely treating the symptoms.

1. INTRODUCTION

"Even an entire society, a nation, or all simultaneously existing societies taken together, are not owners of the earth, they are simply its possessors, its beneficiaries, and have to bequeath it in an improved state to succeeding generations, as boni patres familias."
-Karl Marx (1981), Capital, Volume III

Within waste management academic discourse, Marxist reflections on the interplay between waste and society are rare. Even from disciplines prone to ground their analyses in critical social theory, such as anthropology and geography, Marxist modes of analysis, based on class relations and social conflict, are infrequently utilised. The reasons for the unpopularity of Marx within the field are unclear. One explanation could be that researchers, working within an inherently multi- and inter-disciplinary space, aim to avoid excessive theorising in order make their work as accessible to as broad an audience as possible. Another reason could be the historically close partnerships that waste management scholars, and the STEM disciplines more broadly, have fostered with industry, and the financial and technological interdependence that has resulted. It can be hard to be critical of funders, and too often research agendas are tailored to meet the objectives of the capitalist class.

Material considerations aside, a final possible explanation, as articulated by Foster (1998), could be Marx's often contentious reputation on environmental issues, having been accused of subscribing to an overly 'productivist' view of history and labour (see Benton, 1989; Giddens, 1981) or being fundamentally anti-ecological (see Clark, 1989; Ferkiss, 1993). These criticisms have been controversial however, as a number of influential scholars, such as John Bellamy Foster and David Harvey, have maintained the opposite, insisting that a fundamental appreciation for the limits of growth and the perils of environmental degradation are inborn within Marx's basic formations of both communism and capitalism, and that an early conception of stewardship and sustainability is inherent within his vision for future societies, as the quote at the top section alludes (see Foster, 1998, 2010; Harvey, 1996). Regardless of the reason, Marx's lack of popularity within waste management studies is unfortunate, as, according to this author, it is a field that is in serious need of more critical and theoretically-grounded debate on the economic systems that continue to allow for the creation of unprecedented amounts of waste and the socio-political factors that serve as barriers to accessing waste management services.

This letter has been written in the wake of the 17th International Waste Management and Landfill Symposium, held in Sardinia, Italy between 30 September and 04 October, 2019. It is meant to serve as a reflection on the nearly 600 studies presented at Sardinia 2019, as well as a critical observation of intellectual trends within waste manage-

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ment studies. It is not meant as a criticism of any specific piece of work, rather it's in response to a perception by the author of a broader failure within the field to critically engage with the root causes of waste, namely capitalist production and accumulation, as well as a lack of nuanced analysis of class, one of the most significant socio-economic factors predicating access to waste management services, particularly within the Global South (Cornea et al., 2017; Getahun et al., 2012; Ndau & Tilley, 2018; Oyekale, 2015). Although Sardinia 2019 featured discussions on numerous innovations in waste management practices and technologies, it is my opinion that we, as a discipline, remain overly orientated towards simply managing one of the many societal consequences of capitalism, i.e. waste, without questioning or challenging its legitimacy or sustainability- essentially treating the symptom without addressing the illness.

Materialist criticisms of waste management studies aside, it is beyond the scope and scale of this piece to highlight all of the ways in which greater engagement with Marxist dialectics could add value to such a broad, international, and multi-disciplinary field. Rather, by way of example, I present a brief critique of the 'zero waste' paradigm; both as a set of principles guiding the redesign of resource life cycles within a circular economy, as well as a movement or lifestyle that shifts the emphasis of action to consumers, in order to highlight the depth of socio-economic and socio-political analysis currently lacking within the discipline. Although Marx's writings have been used as an analytical lens to critique the circular economy, notably by Valenzuela and Böhm (2017), critical Marxist critiques of the zero waste lifestyle movement have been absent within academic discourse, and this silence is particularly notable because critiques are taking place, often eloquently, on social media platforms. It is time for us, the academic community, to contextualise and elevate these discussions.

2. ' ZERO WASTE'

'Zero waste' as a concept has evolved beyond academic peroration to the level of normative buzzword that can be adapted to suit a specific objective or agenda (see Specter, 2019, for instance). As such, a variety of definitions exist for the term, depending on the primary focus area of the application, however, they can all be generalised to include an emphasis on waste prevention, re-use, and the comprehensive use of resources (Cole et al., 2014). On examination, two major manifestations[1] of the concept can be discerned within contemporary discourse. First, there is the notion of 'zero waste', as embraced within the confines of Sardinia 2019[2]: a set of principles for academics and waste management practitioners to guide the redesign of product life cycles so as to eliminate waste and maximise the recovery of resources, in line with the idea of a circular economy (Franco-Garcia et al., 2018). The second manifestation of 'zero waste', as a lifestyle or movement, is even more nebulous, but has resonated more profoundly within popular discourse. Touted in countless books, blogs, websites, and social media pages, the 'zero waste' lifestyle movement shifts responsibility for waste minimisation to consumers, advocating for more sustainable living through reduced consumption of single use products[3], re-use, and recycling (see Cohen, 2017; Johnson, 2013; Korst, 2012; Moss 2018). Less of a scientific methodology than a set of beliefs and best practices, the lifestyle movement broadens the narrative of 'zero waste' from the exclusive domain of industry and waste management practitioners towards a platform for collective action (see Figure 1).

Certainly, both aspects of the 'zero waste' paradigm have positive features- treating waste as a resource and instilling greater personal responsibility for waste minimisation are undoubtedly social goods. However, these narratives, as Valenzuela and Böhm (2017) describe, have also served to de-politicise the discourse around the unsustainability of capitalism, enabling ever increasing levels of consumption and waste, while legitimising unsustainable production and notions of limitless growth[4]. Moreover, this depoliticisation of capitalism reinforces class structures, a consequence that is explicit within the 'zero waste' lifestyle movement which shifts the burden of sustainability from capital to labour (see Harvey, 2014; Yates 2011).

In regards to the manifestation of 'zero waste' bound to the circular economy and widely discussed at Sardinia 2019, Valenzuela and Böhm (2017) provide a valuable example of how a Marxist reading or analytical approach can be utilised to broaden an investigation of waste or waste management practices to include thoughtful critique of the socio-economic and socio-political systems that underpin them. Their analysis fuses Marxist critique with psychoanalytic readings from French psychiatrist Jacques Lacan in order to interpret the materialist dynamics that place actors in the position to both produce and consume waste as a commodity, as well as the subjectivity that lies at the heart of waste as a socially constructed concept. The objective of their critique is the re-politicisation of waste, as a step towards interrupting the endless repitition of its creation, management, and attempts to eradicate it.

Valenzuela and Böhm's (2017) case study critically examines the circular strategies of Apple Inc., a globally iconic brand and one of the world's largest producers of e-waste. Over the past decade the company has espoused a significant commitment towards 'zero waste' principles, to the point where its dedication to circularilty has become central to its corporate ethos. According to Valenzuela and Böhm (2017, p. 46), both the design and features of Apple many products display an attempt towrds 'zero-waste' optimisation and timelessness, "everything is engineered to perfection, mimicking nature's wasteless cycles and systems." Apple branding and marketing leans into this perception, with products prominently featuring recyling logos, as well as graphics of leaves, trees and other representations of natures (beyond the obvious apple motif). Within Valenzuela and Böhm's (2017) Marxist-Lacanian frame of analysis, such a green and guilt-free imaginary is irresistible to the consumer searching for affirmation in a world inundated with commodities, waste, and recycling bins. They interpret this brand of circularity as a contradiction-engine in a Marxist-Lacanian sense; affirmation drives the subject to consume further, but the reality of the unpackaged product,

FIGURE 1: Zero waste meme derived from a quote by Anne-Marie Bonneau (Irwin, 2019).

designed to swiftly decay into obsolscence, cannot match the enjoyment derived from opting into the discourse on circularity, contributing to a disenchantment that can only be endured by further reinfocing of the trust in the guarantees of circularity espoused by the Apple brand and perpetuated by the endless purchase-consume-discard cycle (Valenzuela & Böhm, 2017). In their critique of growth capitalism and the circular economy, Valenzuela and Böhm (2017, p. 48) find the example of Apple powerful because it is illustrative of the way, as they state, "the image of a wasteless post-growth economy is never far away from the affective enjoyment that capital's 'green' rhetoric seeks to command in the experience of the sustainability-wary consumer." For the repolitisation of waste to emerge, the endless cycle of consumption must be interrupted. Thus, the re-politicisation of waste must entail the interruption of the self-affirming sustainaible image of consumption that is enabled through the purchase of 'green' commodities (Valenzuela & Böhm, 2017).

Valenzuela and Böhm's (2017) commentary directly confronts the contradictions inherent in the notions of sustainable capitalist production and consumption which lie at the heart of the 'zero waste'/circular economy paradigm. Their critique echoes the Marxist notion of the 'fetishation' of consumption, what Harvey (2014) and Yates (2011) have characterised as a yearning for the facades of the exchangable fruits of labour, such as luxury goods and name brands, which serve to mask the loss of one's own humanity that occurred during their production and subsequent exchange. To Valenzuela and Böhm (2017), sustainability can be seen as having been driven to the logical conclusion of this fetishation of consumption when it in turn is tranfsformed into fetishised content, typified by Apple's 'green' and guilt-free imaginary and its hordes of devoted fans who line the streets for each new product release, and becomes an essential part of the discourse and practice through which capitalism organises and legitimises itself. Their critique has been highlighted because it adds, what I feel, is important and critically needed theoretical and ontological depth to waste management discourses and provides a good example of how Marxist analysis remains relevant within our field.

In regards to the 'zero waste' lifestyle or movement, however, no such critique has emerged from academia, although criticism has been mooted within online blogs, editorials, and other journalistic platforms (see Sattlegger, 2019; Tan, 2019, for example). As previously described, the 'zero waste' lifestyle centres on a platform of waste minimisation, what best-selling author Bea Johnson (2013) has termed her five R's of waste avoidance: refuse, reduce, reuse, recycle, and rot. Within this paradigm, sustainability is an indvidualised responsibility, where each consumer is encouraged to make decisions deemed best for society. Sustainable consumption becomes both the illness and the cure. However, as Sattlegger (2019) notes, this individualisation of responsibility also serves to trivilise structural barriers (such as capitlist produciton and class) to a sustainable society.

When responsibility has been individualised within a capitalistic system it is left to the market to initiate or respond to changes in consumption patterns, and in regards to the 'zero waste' movement, the market has responded enthusiastically, offering a wide-range of products, such as metal straws, re-usable coffee cups, and glass jars, to name a few, to facilitate a consumer's transition to a 'zero waste' lifestyle (see Figure 2). This capitalistic response reveals the sustainable consumption contradiction that lies at the heart of the 'zero waste' movement. Moreover, as Tan (2019) points out, these products often have more significant environmental impacts in their production and disposal than the single-use products they were designed to replace. Thus, the movement has not served to disrupt consumption, rather to refocus it in other, more envrionmentally ambiguous directions. For consumers, a 'zero waste' lifestye and the myriad of 'sustainable' consumption practices it has spawned may lead to a clear conscience- Valenzuela and Böhm's (2017) fetishation of sustainability- but for capital, the sustainable consumption contradiction has instead functioned to bolster capitalistic production, opening up new markets for purposely-designed goods and providing a green-washed public image burnished by the veneer of sustainable production (Sattlegger, 2019).

Finally, the 'zero waste' movement cannot be analysed without interrogating the role that its complicated relationship with capitalistic production and consumption plays in discounting, and often aggrevating, class dynamics and inequality. The 'zero waste' movement individualises action, but as Sattlegger (2019) rightly notes, different distributions of income, wealth and knowledge create disparities between individual's freedom for action. For the poor, the difficulties of coping with the challenges that emerge within everyday life often leave little room for considersations to sustainable consumption. For instance, how much freedom does an individual in a rural area, with little time

FIGURE 2: A critique of the consumption inherent in the 'zero waste' lifestyle movement (Larson, 2019.

and few financial resources, have to make sustainable choices? They simply consume what is available from a limited range of market options. Thus, a 'zero waste' lifestyle has largely become the preserve of the priveleged: a small group of rich and highly educated consumers who have the time and resources to affect such a change in living. As Tan (2019), notes, even if these privileged individuals successfully become 'zero waste' (ignoring the problem of consumption), it does not discount their disproportianate environmental impacts in other areas, such as energy and CO_2 emissions, or negate the class disparity and social inequality that equipped them to make the change to begin with. These critiques raise a number of questions that the 'zero waste' movement has not been able to adequality answer, for instance: how much should the poor be concerned with waste management challenges caused by the rich, and can they be faulted for aspiring to similar patterns of consumption? This remains a fertile space for grounded theoretical discussion within within waste management studies. Moreover, these critiques point to the inherent intersectionality of these debates, challenging prospective researchers to consider the ways in which their points of analysis may mask potentially classist, racist, ableist, sexist, or other discriminating narratives.

3. MOVING FORWARD

In a Marxist waste management discourse, the underyling social structures that drive the creation of waste and structural access to waste management services must be examined. Marxist waste management studies should also attempt to change the basic structures of society. However, these conversations are not currently happening within our field, and most seem content to merely engage with waste as a point of reality, rather than engage with it as a consequence of our socio-economic and socio-political systems, that may or may not be fundamentally illegitimate[5].

Why are these discussions not happening in spaces such as Sardinia 2019, and why has Marx proved unpopular as an analytical lens within the discipline? I previously offered a number of possible explanations including the multi-disciplinarity of waste management studies, its historically close relationship with capital[6], and Marx's often criticised take on the environment. Regardless of the reason, it is important to note that outside of academia, on various social media platforms, such as Facebook, Twitter and Instagram, these conversations are occurring, and often with the nuanced dialectical understanding of class-structures and capitalistic power dynamics which academics and practitioners within waste management studies have too frequently ignored. For instance, Intersectionelle, a Canadian-based Facebook page launched in 2013, regularly features class-concious critiques of the 'zero waste' movement in addition to a wide range of left of centre content. Their response to the meme of the quote by Anne-Marie Bonneau cited previously properly centres the role of the capitalist class in the creation of our global waste crisis, and attempts to reframe the solution from a movement towards individual action to one for systemic change and class awareness- jokes on 'consumption' aside (see Figure 3). Green Memes for Communalist Dreams, another Facebook page that traffics heavily in memes, promotes a broader social ecology platform and often posts content critical of liberal environmentalism, green capitalism, and the sustainability movement. Finally, Turning Green, on Instagram, although promoting content catered to those pursuing a 'zero waste' lifestyle does so with an intersectionality and class-awareness atypical of similar pages. These are just a few of the many critical voices that have proliferated across social media that have spoken specifically to the 'zero waste' movement. Although I'm reluctant to assign too much weight to individual commentators, nonetheless, it is important to not discount the power of these platforms to shape public discourse, as well as to reflect broader undercurrents of discontent within society with our current global socio-economic systems (see Ballantyne, 2017). Finally, these critiques of 'zero waste' are more poignant for their absence within waste management academic circles, which should be best positioned to contextualise and inform public debate.

The purpose of this letter has not been to comment critically on individual pieces of work. Rather, it has been written to reflect on how Marxist frames of analysis might reframe contemporary debates on waste management practices to include more critical discussion and engagement with the root causes of waste- rather than merely treating the symptoms. Moreover, the work of Valenzuela and Böhm (2017), has been highlighted as one of the few examples of how Marx's writings have been utilised as an analytical lens within our field, and as a successful blueprint for waste management academics who feel that some level of Marxist critique may add depth to their analysis. Not every scholarly contribution within waste management studies needs to include nuanced class critique or make an original theoretical contribution, particularly those coming from the STEM disciplines, nonetheless, there is certainly room for broader multi-disciplinary awareness and concern for the systemic socio-economic and socio-political conditions that created, and continues to create, our waste problem. A thought to Marx may help.

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FIGURE 3: A class-conscious critique of zero waste from social media (Intersectionelle, 2019).

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[1] A third, but lesser discussed manifestation of the 'zero waste' movement has also been discerned in the opposition to waste to energy, i.e. incineration, in the European Union.
[2] Sardinia 2019, for instance, featured multiple panels and lectures on the circular economy, in addition to more than a dozen papers. It would be impossible to list them all. Please refer to the conference proceedings for a complete list.
[3] This distinction is important, as the movement remains profoundly materialist, with most 'zero waste' lifestyle platforms sporting a wide range of products available for sale in order to facilitate more sustainable living.
[4] Doeland (2019, 1), in a recent editorial for Detritus, also alludes to the danger of 'zero waste' narratives normalising and destigmatising unsustainable capitalist consumption, noting that "the resourcification of waste strips it of its power as a doomsayer urging us to curb our consumption".
[5] One notable exception, that often contains highly innovative and creative academic and research-based discussion of these issues is Discard Studies. Serving as an online hub for scholars, activists, environmentalists, and other communities engaged in waste, Discard Studies provides a platform for critical and informed discussion around the relationships between waste and society, contextualised within broader sociocultural-eocnomic analysis. Their monthly compilation 'The Dirt' seeks to assemble recent articles, job postings, and calls for participation relevant within the discipline.
[6] This dynamic alone could be a worthwhile object of critical inquiry.

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POLICY INSTRUMENTS TO REDUCE CONSUMPTION OF EXPANDED POLYSTYRENE FOOD SERVICE WARE IN THE USA

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ABSTRACT
There is global recognition that waste plastic is a ubiquitous pollutant in the built and natural environments. A component of plastic litter and debris is single-use, expanded polystyrene (EPS) food service ware. Reducing the consumption of EPS food service ware is challenging because reuse is not feasible, recycling is not economically viable, and composting is not possible. In the absence of national action to reduce EPS in the USA, local governments have taken the lead on enacting ordinances to eliminate or reduce EPS food service ware. This paper examined the variety of policy instruments that can or have been used by local governments in the USA to reduce EPS food service ware. Because of the inabilities to reuse, recycle, and/or compost EPS, the most frequently used policy instrument has been a ban. As of December 2019, there were 249 local bans in the USA covering 12.85% of the nation's population: of these bans, 9.6% were partial bans restricting distribution only on government and public property, 65.9% were narrow bans that ban distribution by restaurants and food providers, 8.8% were full bans that include the narrow ban and also ban using EPS food packaging by grocery stores, and 15.7% adopted an expanded ban that includes the full ban and also baned other single-use plastic food ware related items including the selling or distributing of EPS coolers and single-use plastic utensils, straws, stirrers, lids, cups, plates, and containers.

1. INTRODUCTION

Continuous images of plastic waste, litter, and debris have established that plastic waste is ubiquitous in the built and natural environments. While there has been recognition for decades that plastics were a major component of land-based, surface water, and costal/beach litter, its prevalence in seemingly unexpected areas have raised the public's alarm.

In 2010 alone, an estimated 4.8 to 12.7 million MT of plastic waste entered the ocean from land (Jambeck et al., 2015). In the open oceans, the worldwide marine distribution of plastic debris is mostly accumulating in the five ocean gyres (Cózar et al., 2014). Plastics and microplastics occur on ocean surfaces, in the water column, in sediments (Galgani, Hanke, & Maes, 2015), and in marine organisms (Law, 2017). Humans are exposed to waste plastics through the consumption of contaminated seafood (Smith, Love, Rochman, & Neff, 2018; Cox et al., 2019). Microplastics have even been found in remote mountain areas as a result of wind transport (Allen et al., 2019) and are entering the environment through organic compost (Weithmann et al., 2018). Collectively these occurrences firmly establish that plastic waste is a ubiquitous environmental pollutant.

Each year more than 300 million MT of plastics are produced worldwide (Law, 2017); the majority of these plastics are not recovered. As shown in Figure 1, in 2014 only 9.5% of post-consumer plastics were recovered for recycling in the USA (US EPA, 2018), which was the highest rate recorded. Since 2014 the rate has decreased; in 2017, the recycling rate dropped to 8.4% (US EPA, 2019) and will likely have decreased in 2018 and 2019 because of the implementation of China's National Sword Program as discussed below. As a consequence, non-recovered plastic waste is disposed or becomes litter thereby potentially entering the environment.

This pervasiveness of plastic waste in the environment has prompted the media, concerned citizens, environmental groups, corporations, and governments to reassess our seemingly insatiable consumption of plastics, specifically single use plastic (SUP) products. This concern is further exacerbated by the predicted growth of the SUP industry fueled by population growth, increased urbanization, and growth in middle-class income (Plastics Insight, 2019). National, sub-national, and local governments have begun to act to reduce the consumption of SUPs, especially plastic bags. One particular category of SUPs of increasing con-

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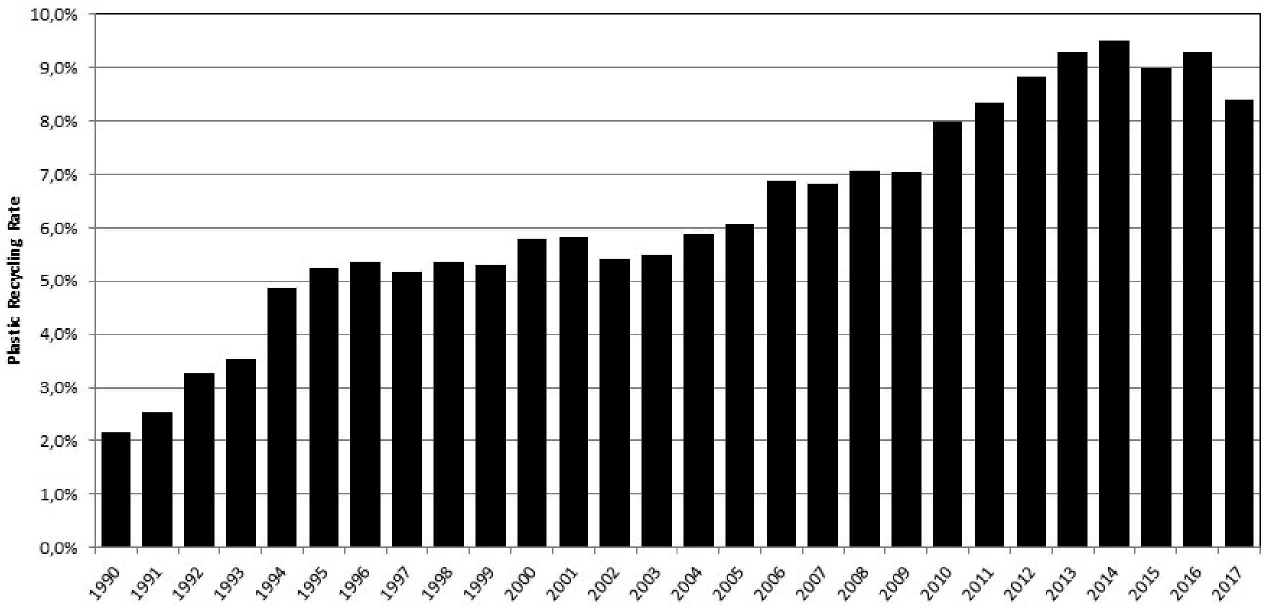


FIGURE 1: Estimated annual plastics recycling rate in the USA, 1990-2017 (US EPA, 2015; US EPA, 2019).

cern is expanded polystyrene (EPS) food service ware. EPS food service ware includes products used by food vendors (e.g., restaurants, food trucks, coffee shops, grocery stores, etc.) to serve, transport, and package prepared and ready-to-consume food and/or beverages or to transport leftovers. By design, EPS food service ware is intended to be disposed of after a single use and includes cups, plates, bowls, trays, hinged or lidded containers, cartons, and coolers (e.g., cool box, chilly bin, esky).

This article first describes EPS food service ware, which includes the use and estimated consumption of EPS food service ware. This is followed by a discussion of the presence and economic impacts of EPS litter and debris. The article then discusses the lack of an economically viable recycling market. The next section discusses a sustainability approach to solve the problem of EPS. Finally, the article examines the various policy instruments available to governments to reduce consumption of EPS with a focus on government actions undertaken thus far in the USA designed to eliminate or reduce the consumption of EPS food service ware.

2. EPS FOOD SERVICE WARE

EPS food service ware is used by a variety of commercial and institutional organizations including restaurants, coffee shops, food trucks, grocers, convenience stores, hospitals, cafeterias, prisons, universities, and schools. It is also sold by various retailers for home use including department, grocery, convenience, and office supply stores; pharmacies; membership warehouses, and online vendors.

EPS is commonly used for food containers by restaurants for takeout and takeaway because of its thermal insulation and moisture resistant properties (Barnes, Chan-Halbrendt, Zhang, & Abejon, 2011). The historical popularity of EPS is a function of its positive characteristics including resistance to heat allowing warm food and beverages to remain warm and cold food to remain cold (Heverly, 2017). It is waterproof, sturdy, and flexible and is also inert thus it is nonreactive with its contents (Heverly, 2017). EPS is convenient because it does not need to be washed, it is sanitary, and it is inexpensive to purchase. The intended purpose and design of EPS is to use it only once before it is discarded; thus, it is considered disposable. The designation itself, disposable, signals to the user the intended disposition after a use — disposal.

The cost advantage of EPS products compared to similarly functional recyclable and compostable products makes EPS products the rational choice for profit-maximizing firms. As to be expected, the cost of EPS substitutes generally is higher with an approximate increase of about 85% (MB Public Affairs, 2017), but this figure represents a relatively very small operating cost for restaurants. According to an economic analysis conducted for the City of San Jose, CA, “to go” containers used by full service restaurants represent about 0.3% of total sales revenues and for takeout restaurants, “to go” containers represent about 1.3% of total sales revenues (EPS, 2012). Retail prices of compostable coffee cups were 37% higher than EPS and large hinged compostable food containers were 58% higher (CCEAC, 2018). Single-use EPS cafeteria trays cost $0.065 each and reduce costs associated with non-disposable trays including labor, water, energy, detergent, and equipment maintenance (CCEAC, 2018). In a study in Hawaii (Barnes et al., 2011), researchers found that consumers had an increased willingness to pay for more sustainable food containers, especially if they were made with local resources (e.g., sugar cane) and produced locally, compared to EPS containers.

The primary material for EPS food service ware is polystyrene (PS, resin identification code #6). Polystyrene is a fossil fuel-derived plastic made by the polymerization of a styrene monomer that is formed by a reaction between ethylene and benzene. It is one of the most common types of plastics for food service ware, which includes cups, lids, bowls, and takeout/takeaway containers, trays for prepared foods, straws, and utensils. The major types of PS are extruded, molded, and expanded. Extruded PS, which contains less air than expanded PS, is a smooth material used primarily for displaying and packaging food including "foam" trays for meat, produce, bakery items, deli, seafood, and eggs. Another version of extruded PS is oriented PS, which is a clear, stretched PS film used in bakery, deli, and freezer applications. Clear extruded PS is also used for clear hinged food containers ("clamshells"). Molded PS is used to produce a more rigid material for such products as disposable cups and lids, containers for dairy products, cutlery, straws, and containers used for salad bars and produce. Expanded polystyrene (EPS) is typically produced by expanding polystyrene beads that contain pentane in the presence of steam; the greatly expanded pellets are subsequently steam-fused and molded. EPS is used to produce take-out food and beverage containers primarily cups, bowls, plates, hinged clamshell containers, platters, and cafeteria trays and is also used for disposable coolers and packing material. EPS coffee cups first entered the market in 1960 (Smith, 2017). EPS is also used as an insulator to hold and ship perishable goods (e.g., disposable coolers and mail order foods) and to protect contents from damage (e.g., packing "peanuts" and formed blocks). Styrofoam™, a commonly used but incorrect generic term for EPS food service ware, is actually the Dow Chemical Company's trademarked name for closed-cell extruded PS used for thermal insulation in buildings.

Regarding consumption data, the only national consumer data for some EPS food service ware in the USA is from 2004 and 2008, but only for certain EPS products. In 2004, Americans consumed 56.73 billion EPS cups, bowls, plates, clamshells, and trays, which equated to 193.2 items per person (Keybridge Research, 2009). In 2008, consumption increased slightly to 59.04 billion items, or 194.2 items per person (Keybridge Research, 2009). As shown in Table 1, if historical trends were to continue, extrapolating linearly to 2019, Americans will consume 63.628 billion SUP food service ware items. (This amount does not include the consumption of molded and extruded EPS food service ware.) It is recognized that at best, the extrapolation is only an indicator of potential consumption due to a variety of factors. For example, between 2008 and 2018, there was an 8.7% increase in real personal income in the USA (USA Census Bureau, 2019). The reliance on take-out food, and thus SUP food service ware, has increased. Between 1997 and 2017, American's increased their expenditure on take-out food by 8.4%; take-out food now comprises 36.1% of all food away from home expenditures (USDA, 2018). However, voluntary and government actions (discussed later in this paper) have reduced consumption in some areas.

3. EPS LITTER AND DEBRIS

Litter is generally defined as waste items that have been disposed of improperly at an undesirable location. Litter is also generated through the mismanagement of waste including its spillage and escape during collection and transportation for recycling as well as escape as disposal as trash. A study in Florida (FCSHWM, 2003) found that overflowing and uncovered dumpsters were major causes of solid waste collection-generated litter. Collection-generated litter is especially pronounced with automated collection as Schert (2000) found a 71% increase in litter with automated collection of trash. SUPs collected from curbside recycling or trash receptacles or public space receptacles can spill during the manual or automatic transfer of materials to collection trucks. Litter is commonly generated from the curbside collection of recycling and trash because of the design of some collection containers (open-top bins), wind, wind generated from passing vehicles, the physical transfer of contents to collection vehicles, and from scavenging by humans and animals (Wagner & Broaddus, 2016).

When littered, EPS is problematic because of its longevity, design, and density. EPS is lightweight making it susceptible to dispersion by wind and stormwater. EPS is especially problematic as litter and debris because it is about 95-98% air with a bulk density of only 0.05 g/cm3 and thus highly buoyant allowing it to float; its lightweight design allows it to be windblown (natural wind and wind produced by passing motor vehicles). The buoyancy aspect also makes littered EPS transportable by stormwater resulting in its transfer from land to surface water to ocean and stormwater collection systems to surface water and ocean. Because EPS is not biodegradable, when littered, it will break down into macro, micro, and nanoparticles through a variety of processes including abrasion, embrittlement, wind, wave and tidal action, and photooxidation (Wang et al., 2016). This results in increased and wider dispersion and accumulation of EPS in the environment.

3.1 Prevalence of EPS Litter

Plastics have consistently dominated the top items collected around the globe based on item counts of coastal litter. Each year, the Ocean Conservancy conducts an International Coastal Cleanup event. In the 2018 global cleanup event, 122 countries participated (Ocean Conservancy,

TABLE 1: Estimated annual USA national consumption of EPS food service ware, 2019.

	Cups	Plates, bowls & platters	Clamshells	Trays	Total Consumption
Mean per capita	110.6	36.5	36.5	9.8	193.4
Mean per household+	279.8	92.3	92.3	24.8	489.3
Total consumption	36.387 billion	12.008 billion	12.008 billion	3.224 billion	63.628 billion

+The US population in 2019 was 329 million; mean individuals per household in 2016 was 2.53 (US Census Bureau, 2016).

2019). The 2017 cleanup event was the first year that the top-ten most commonly found items were all made of or included plastics, which was repeated during the 2018 cleanup event (Ocean Conservancy, 2019). In the 2018 coastal cleanup, EPS food service ware and packaging was the 8th most commonly found item in the USA (Ocean Conservancy, 2019).

Baltimore, MD, has three water-wheel trash interceptors, which are stationary, solar and hydro powered vessels that intercept and remove debris from tributaries of Baltimore's Inner Harbor. The first wheel was installed in 2014, the second in 2017, and the third was installed in 2018. Between 2014 and November 2019, the three wheels collected and removed 1,478,787 EPS containers (Clearwater Mills, 2019).

A trash survey of the Anacostia River in the State of Maryland (USA) found that on a per item count basis, EPS food service ware was the fourth largest category of trash behind plastic bags, the largest category; food packaging and plastic bottles were the second and third highest count categories respectively (MWCOG, 2015).

EPS has been found to be a significant component of trash collected in stormwater drains. In the San Francisco Bay area of California, trash in stormwater capture devices was collected and characterized. EPS food service ware accounted for 6% by volume of the waste characterized; however, on a per-item count, EPS was 262% greater than recyclable beverage containers and 87% greater than SUP bags (EOA, 2014). In San Jose, CA, EPS was found in stormwater drains at volumes ranging from 7.8% to 10.8% of trash collected that was consistent with findings of the California Department of Transportation, which found that EPS constituted 15% by volume of the trash in statewide sampled storm drains (Romanow, 2012). In a sampling event in Santa Cruz County, CA, EPS constituted 12.66% by count of all trash found in the stormwater system (Romanow, 2012).

3.2 Economic Impacts of Plastic Litter

As found by Beaumont et al. (2019), the direct and indirect costs of marine plastic debris are significant and include the loss of seafood as a human food source, a negative impact to heritage through the loss of culturally significant and iconic marine megafauna, and reduced marine ecosystem services. The authors postulate that marine plastic debris causes an annual loss of $500–$2,500 billion in reduced marine ecosystem services, which equates to $3,300 - $33,000 per MT (Beaumont et al., 2019).

The presence of litter, especially in tourist-heavy recreational areas, also has a negative economic impact in part because of its adverse aesthetics prompting some visitors to avoid littered beaches (Leggett et al., 2018). Based on an analysis of the economic impact of marine debris during the three-month prime beach season for 31 beaches in Orange County, CA, the economic benefits of a 25% reduction in marine debris were valued at $29.5 million with a per capita seasonal value of $12.91 (Leggett et al., 2018)

Cleaning-up marine litter from the open ocean is not currently feasible as prevention is the only successful approach to manage the problem (Jambeck, 2015). In contrast, cleaning up coastal and land-based litter is feasible, but can be a significant expense for a local government. The annual cost of beach and waterway cleanup in New York City was $2,719,500 (Columbia University, 2015). In a comprehensive marine debris cost study by Stickel, Jahn, and Kier (2012), the authors found that on average West Coast cities in the USA annually spent $56,688 on beach and waterway cleanups, $664,580 for street sweeping, $165,811 for stormwater capture devices, $294,935 on storm drain cleaning and maintenance, and $304,545 for manual cleanup of litter. In a study on the impact and cost of litter from curbside recycling collection, the estimated labor cost to collect each visible piece of litter ranged from $0.17 to $0.79 (Wagner & Broaddus, 2016). San Francisco estimated the clean-up cost for each littered plastic bag to be $0.052 (Pender, 2005; Burnett, 2013).

4. NO ECONOMICALLY VIABLE MARKET FOR RECYCLED EPS

Currently, the recycling rate for EPS food service ware in the USA is insignificant. Although no national recycling rate of EPS food service ware is available, in Los Angeles County, CA, the EPS food service recycling rate in 2011 was about 1% (LA County, 2011). The statewide recycling rate of EPS food service ware and packaging in California in 2001 was 0.2% (IWMB, 2004). In Baltimore, MD, a free EPS food service ware drop-off recycling program operated from 2011 to 2018, but collected only about 4 tons per year (Weigel, 2018), which equates to about 0.2 ounces per person per year or about the equivalent to 4 EPS coffee cups per year.

The city of San Diego, CA found that it was not economically viable to continue collecting EPS in curbside collection systems based on the high collection and recycling costs. The city estimated that the collection and recycling of 105 tons of EPS over a 7-year period would cost approximately $900,000 or $8,570 per ton of EPS (San Diego, 2019). Based on a study by the New York City Department of Sanitation (NYC Department of Sanitation, 2017), over a 30-year effort, even when subsidized by the plastics industry, recycling EPS food service ware was not economically feasible

EPS is extremely lightweight (about 95%-98% air) meaning that a large volume is required to produce sufficient marketable quantities for recycling. For example, weight per volume for EPS is about 9.6 pounds per cubic yard. In comparison, whole unflattened plastic bottles are 36 lbs/yd3, corrugated cardboard is 100 lbs/yd3, and newspaper is about 433 lbs/yd3. (It should be noted that the bulkiness of EPS is also a negative when landfilled and/or transported because it consumes a disproportionate amount of landfill and collection vehicle space in relation to its weight requiring the vehicle to take more trips.) Thus, collection and transportation costs are comparatively very high and inefficient without significant compaction or densification. Compaction and densification requires special equipment, such as grinders and densifiers, to compact EPS into dense blocks for transportation or storage prior to recycling, which are both labor intensive. And, only clean EPS can be densified. For example, based on a limited cost analysis conducted by Sedona Recycles (Sedona Recycles, 2015); there was a net loss of $725 in recycling an 837-lb pallet of EPS, which was due primarily to the labor involved in densifying the material. Because of its high volume to weight ratio, considerable space is also required to store EPS prior to densification. In addition, an EPS food service ware recycling program would require a separate collection system.

EPS cannot effectively be collected in a single-stream system because it breaks apart easily. EPS in tiny pieces is too difficult and not cost effective to sort and segregate for market (City of Portland, 2013). Because of the difficulty and cost of segregating out EPS, allowing it to remain with other recyclable materials can contaminate the target material thereby further reducing the target material's post-consumer market value; the lower the contamination of a commodity, the higher value. For example, at materials recovery facilities, EPS fragments can blow throughout the facility often entering the paper stream and causing contamination of segregated paper (City of Portland, 2013). EPS food service ware, especially food and beverage containers, are often contaminated with food, oils, grease, and other materials reducing recycling efficiency and its acceptance as a post-consumer commodity. In addition, food residues on containers can also contaminate other materials in single stream systems (FPI, 2014). Consequently, EPS food service ware is generally prohibited from municipal curbside and drop-off recycling programs because it is viewed as a contaminant.

China, which has been a major importer of post-consumer plastics for recycling, has enacted a series of initiatives (i.e., Green Fence, National Sword, and Blue Sky) that have essentially banned the import of post-industrial recyclables beginning in 2018. China's import policies banned certain recyclable materials, particularly mixed plastics, and instituted exceptionally strict standards on allowable contaminants while increasing compliance inspections of recyclable commodity imports. The ban has caused major upsets for communities that had collected and segregated low grade plastics (RIC #3-7), which include EPS, and now have to pay to have these materials recycled or have chosen to instead, dispose of this material. As noted by Brooks, Wang, and Jambeck (2018), "89% of historical exports of plastic waste consist of polymer groups often used in SUP food packaging…".

As China's market has closed, industrialized countries, including the USA, have sought other markets for low grade plastics including Bangladesh, Laos, Ethiopia, Senegal, and Vietnam (McCormick et al., 2019). In response, the unregulated dumping of low grade plastics was addressed at the Basel Conference of the Parties in May 2019. The Basel Convention on the Control of Transboundary Movements of Hazardous Wastes and Their Disposal is designed to promote the environmentally sound management of wastes and prevent dumping in less developed countries. The Convention was amended to include plastic waste in a legally-binding framework for Parties to increase the transparency and regulation of the global trade of plastic waste (Basel, 2019). The USA, one of the largest producers of plastic waste, is not a Party to the Basel Convention.

5. SOLVING THE PROBLEM

The environmental problem presented is the ubiquitousness of EPS litter and debris in the environment. In constructing a policy goal to solve this specific problem, sustainable materials management provides guidance. Sustainable materials management is a principle strategy adopted by the Organization of Economic Cooperation and Development OECD Council in 2008 regarding resource productivity (OECD, 2008). As defined by the US EPA (2015), "Sustainable Materials Management (SMM) is an approach to serving human needs by using/reusing resources productively and sustainably throughout their life cycles, generally minimizing the amount of materials involved and all associated environmental impacts." A principle philosophy of SMM is to reduce disposal and environmental impacts by using less of the material.

By applying the principle philosophy of SMM, a series of hierarchical management approaches can be employed as shown in Figure 2. The preferred approach of SMM is source reduction, or avoidance. Source reduction is crucial because often the greatest environmental and resource demand impacts are at the materials acquisition and manufacturing stages, the upstream, as opposed to the use and post-consumer/end-of-life (EOL) stage, the downstream. Thus, reducing the creation of a material not only reduces the generation of EOL waste, but also reduces the upstream generation of wastes and associated environmental impacts. Source reduction can also include product substitution where less harmful materials are used as a substitute for the target materials or recovered post-consumer materials are used to reduce the amount of virgin materials consumed. If, however, source reduction is not appropriate or achievable, and the material is created and used, the priority is to reuse the item. If reuse is not feasible or possible, recycling is the next priority, which seeks to maximize the recovery of the material's values. Composting can be considered a form of recycling, but it is debatable as to whether composting, hierarchically speaking, is superior to recovery of materials through recycling. Next in the hierarchy is to extract the energy value from the material, generally with waste-to-energy facilities. Finally, the least desirable action is disposal through landfilling, or incineration without energy recovery.

Based on SMM's hierarchy, the policy goal proffered here is to reduce or eliminate the consumption of EPS. Because EPS is not reused, unless EPS is replaced with reusable food service ware items, the ability to successfully recycle or compost alternative food service ware items is crucial (some compostable products have fossil fuel-based coatings rendering them non-compostable in many compost systems). The actually recyclability and compostability of each product, however, are a function of the capabilities, budgets, and preferences of the local solid waste management program. Thus, within the SMM framework, it is insufficient that a product be "recyclable" or "compostable," which is theoretical, but that it is actually recycled or composted locally. Alternative products that are potentially recyclable include polyethylene terephthalate (PET, RIC #1), polypropylene (PP, RIC #5), aluminum,

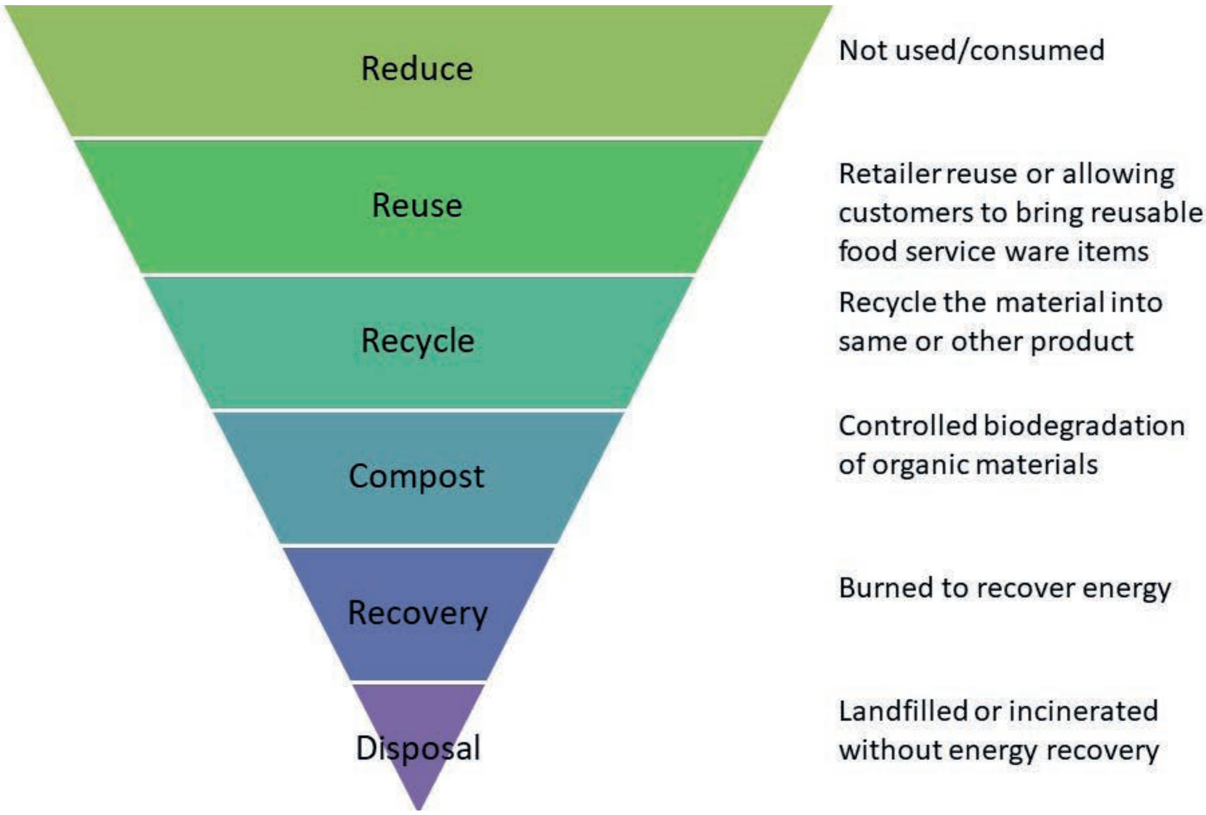


FIGURE 2: SMM minimization hierarchy for EPS and EPS substitutes.

and paper and paperboard. Alternative products that are potentially compostable, depending on the composting operation, include plant-based products made from bagasse, cassava starch, potato starch, wheat straw, and palm waste and the bio-plastic polylactic acid (PLA) made from corn or sugarcane, which is also used to coat paper and paperboard products.

One of the major issues with compostability is that most compostable plastics do not readily biodegrade in the natural environment; they require high heat and moisture, which are used in specialized industrial composting facilities. According to Goldstein (2019), in the USA, there are only 185 full-scale food waste composting facilities. Based on a survey of these compost facilities, of the respondents (N=95), 37% do not accept Biodegradable Product Institute (BPI) certified compostable paper products and 55% do not accept BPI-certified compostable bioplastic products (Goldstein, 2019).

When focusing specifically on the post-consumer/end-of-life stage of alternatives to EPS, the ability and practice of a product to be effectively recycled or composted are crucial determinants in addition to its fate as litter. In examining EPS's contributions to land-based litter and marine debris, this is reasonable. Thus, an alternative material that is recycled would have a reduced disposal rate; however, if it not recycled and becomes litter, it can be as problematic as EPS. An additional important comparison is based on the life cycle impacts of each product, which examines the environmental impacts throughout the products' life cycle from the cradle to the grave. However, there are significant challenges and limitations to life cycle assessments (LCA) making direct comparisons difficult and potentially misleading. For example, there can be significant data gaps, what impact categories are selected, energy type used, spatial variation, land use, resource depletion, uniqueness of local environments, environmental dynamism, treatment of subjective values, and consumer behavior (See for example, Vendries et al., 2018; Reap at al., 2008). Moreover, LCAs are a snapshot in time to assess current conditions and impacts, which can easily change due to a multitude of factors such as energy source, transportation distances, and local conditions.

Recognizing the limitations of specific LCAs, various studies have been conducted in an attempt to compare EPS and SUP products to available substitutes. For example, compostable packaging is the best material for single-use applications when it can be composted locally at EOL (Davis & Song, 2006). In Greece, recycled paper egg cartons had an overall lower environmental impact than polystyrene egg cartons (Zabaniotou & Kassidi, 2003). Based on the LCA, the authors found that polystyrene egg cartons contributed more to acid deposition and photochemical ozone by generating seven times more NOx and 16 times more SOx than paper egg cartons while recycled paper egg cartons generated twice as much heavy metals. In contrast, in Thailand, PLA thermoform boxes had a slightly higher environmental impact than polystyrene boxes when the indirect land use change emissions from growing corn and cassava were included (Suwanmanee et al., 2013). The intent here is not to present an exhaustive assessment of the various comparisons, but to present an indication of the limitations of comparisons from a LCA perspective while simultaneously heralding the importance of LCAs in guiding the selection of preferable substitutes.

The power of the market is that more environmentally preferable substitutes can be produced if sufficient demand exists. While demand can be pushed by increased government actions, especially targeted product bans, voluntary actions can be powerful in increasing the demand for preferable alternatives. For example, Starbucks coffee chain committed US$10 million for the NextGen Cup Challenge to produce a preferable substitute to the disposal cup. Igloo Products Corporation created a reusable and biodegradable cooler (the Recool) designed to replace the disposable EPS cooler.

6. POLICY INSTRUMENTS TO REDUCE CONSUMPTION OF EPS FOOD SERVICE WARE

As previously discussed, reuse, recycling, and composting of fossil fuel-based EPS food service ware are currently not technically feasible and/or economically viable. Reduction, which includes product substitution, is the most preferred goal from a SMM perspective. While the goal of reduction is obvious, with regards to product substitution, the goal is to foster the replacement of EPS with reusable products that contain recycled material or replacement with products that are recycled or composted locally. Again, this does not mean theoretically recyclable or compostable, but products that actually can be or are recycled or composted locally.

To achieve the overall policy goal of reduction and preferable product substitution, we look to policy instruments, which are methods of government interventions designed to achieve a specific desired outcome. As shown in Table 2, the major policy instruments to encourage reduction and product substitution are command-and-control approaches (i.e., bans and mandated design specifications), market-based approaches (i.e., extended producer responsibility, deposit-refund, taxes/fees, and grants/credits), and voluntary and public educational programs.

6.1 Command-and-Control Approaches

Command-and-control approaches use policy instruments designed to prescribe allowed and prohibited actions or products. These instruments seek to provide a clear requirement to meet and are comparatively easier to monitor compliance such as with bans. The primary command-and-control policy instruments available for EPS food service ware include bans/prohibitions and mandated design specifications.

6.1.1 Bans/Prohibitions

The intent of a ban is to prohibit a certain action, or, in the case of EPS food service ware, ban the use, distribution, or sale. The intended outcome of a ban is an overall reduction in consumption while also, if necessary, encouraging the consumption of environmentally preferable substitutes.

While the European Union proposed the Single Use Plastics Directive to reduce the 10 single-use plastic products in May 2018, including EPS items, there are no national laws in the USA that ban (or for that matter, seek to reduce consumption or use) the distribution or use of EPS. The first ban of EPS in the USA occurred in 1987; the State of Maine banned the distribution of food and beverages in EPS containers but only at state facilities and functions (Wagner, 2016). As of December 2019, there have been three additional subnational bans on EPS food service ware (two states and the District of Columbia). Washington, DC's ban, enacted in 2014, was part of a comprehensive ban on disposable food service, which included products made with EPS as well as other food service products (i.e., straws, stirrers, paper bags with plastic windows, aluminum-coated paper take out containers, and foil-coated food wrap paper) that cannot be recycled or composted. As of 2019, Washington, DC's Department of Energy & Environment reported a 97% compliance rate with its EPS ban (Crunden, 2019). The US states Maine and Vermont both enacted comprehensive bans in May 2019, which apply to private retail distribution and use of EPS food service ware.

While these state-level bans are very recent, historically, local governments have taken the lead on reducing EPS with various types of bans. The first EPS food service ware

TABLE 2: Primary policy instruments to reduce consumption of EPS food service ware.

Category of Policy Instrument	Policy Instrument	Summary
Command and Control	Ban/Prohibition	Prohibit consumption, use, or sale of EPS
	Mandated Design Specification	Mandate use of only compostable or recyclable items or mandate that items must contained certain percent recycled content
Market-Based	Extended Producer Responsibility	Impose end of life economic, management action, or logistical responsibility onto the producer
	Deposit/Refund	Items have a front-end monetary deposit that is refunded when customer returns item
	Tax/Fee	Levy tax/fee to increase price of targeted item to discourage consumption, reduce/eliminate any tax on preferable item to encourage consumption
	Grants/Credits	Provide grants to retailers to offset higher cost of preferred alternatives or provide credits to customers to bring reusable items
Voluntary & Public Education Programs	Voluntary Reduction Campaigns	Voluntary levy of a fee on using disposable item or provide credit for using reusable item
	Social Marketing	Zero Polystyrene Campaigns

ban in the US was adopted in March 1988 by Suffolk County, NY, followed by a similar ban adopted by Berkeley, CA, in October 1988. Between 1988 and 2004, 19 local bans were adopted. Many of these early bans were due to the then concern with the use of chlorofluorocarbons (CFCs) to produce EPS foam, which was based on the role of CFCs in depleting stratospheric ozone. (CFCs are no longer used for that purpose.)

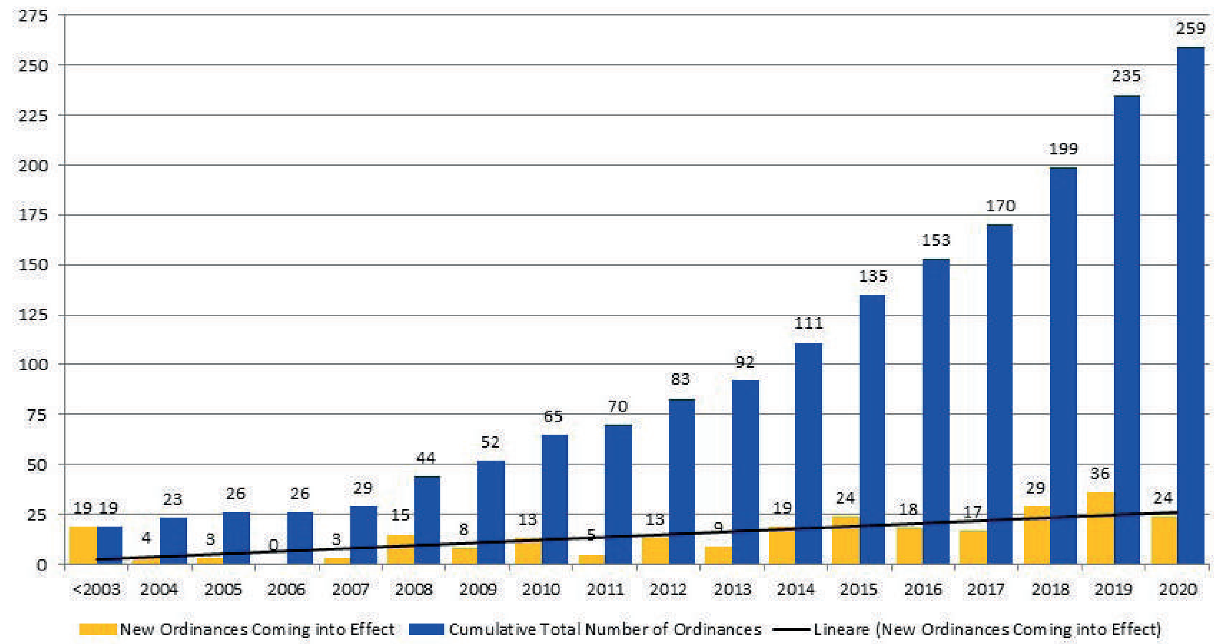
As of December 2019, in the USA, there were 249 local-government ordinances enacted in 20 states, including the District of Columbia, and 2 state laws, which combined, cover 12.85% of the USA population. (See Appendix 1 for a list of bans including the type of ban and the effective date of the ban.) As shown in Figure 3, which depicts the annual and cumulative total of local ordinances by effective date, the number of local bans began to increase in 2004 and especially starting in 2008. As the trend line indicates, there also has been a gradual increase in the number of ordinances coming into effect each year.

As of December 2019, all 249 local government ordinances in the USA were variations of a ban on the distribution, sale, or use of EPS food service ware. These bans are categorized as partial, narrow, full, and expanded as described in Table 3. As shown in Table 3, the vast majority of bans have been narrow bans (65.9%) followed by expanded bans (15.7%), full bans (9.6%), and partial bans (8.8%). Table 3 also notes the approximate population size affected by each ban with narrow, partial, expanded, and full in descending order of populations affected.

The primary differences in the categories of EPS bans (i.e., partial, narrow, full, and expanded) are the systematic broadening of where, who, and what are subject to the ban. Some local governments ban only take-out EPS food service ware whereas others include sit-down restaurants, grocery stores, and/or all food vendors. There are also examples of limited or directed bans such as banning the use of EPS only at special events that require a municipal permit. For example, in San Jose, CA, as a condition of an event permit, EPS food service ware is banned at special events with more than 1,000 attendees. Miami Beach, FL, banned single-use EPS but only in public outdoor areas such as public parks and beaches. Kiawah Island, SC, adopted a unique application of an EPS ban that prohibits the possession of EPS including polystyrene/plastic foam products but only on the town's public beach. The most comprehensive ban was adopted by Berkeley, CA. In addition to banning EPS, starting in 2020, all dine-in restaurants are required to use only durable reusable plates, cups, and utensils for dine-in meals. Take-out restaurants are required to use only compostable food service ware certified by the Biodegradable Products Institute. Finally, Berkeley's ordinance allows for customers to bring their own cups or they must pay $0.25 for a disposable, compostable cup.

Most bans are phased in over time giving retailers time to use up any existing stock. Another common element in bans is an economic hardship provision, which allows establishments to obtain a conditional, time-limited exemption if switching to more expensive reusable, recyclable, or compostable alternatives would cause economic hardship.

One of the untended consequences of narrowly worded, specific bans on EPS is that some communities found that a small number of retailers simply switched to extruded polystyrene food service ware products, which were not explicitly banned when the word "foam" was explicit-



* The data is current as of December 2019 and includes only local bans. This figure includes 19 ordinances enacted in 2003 or earlier. This data also includes the 10 local ordinances enacted in Florida vacated by state law, which are currently under court challenge.

FIGURE 3: Cumulative and yearly municipal EPS food service ware ordinances by effective date, 2004-2020 (N=259).*

TABLE 3: Categories, prevalence, and populations covered by EPS food service ware bans, USA (N=249).

Ban Type	Description	Prevalence	Affected Approximate Population
Partial Ban	Partial Bans apply only to (1) EPS sold by local governments or in public buildings, facilities, and public property (e.g., parks and beaches) and/or (2) sale or distribution of EPS special events (e.g., festivals, parades, concerts, etc.) requiring a local government permit.	8.8%	13.52 million
Narrow Ban	Narrow Bans ban retailers that sell or provide prepared food, including full-service and take-out restaurants, from using or providing EPS food service ware. Some Narrow Ban ordinances include or specifically exclude food trucks. Narrow Bans do not include packaging (e.g., meat and deli trays and egg cartons) for foods sold by grocery stores.	65.9%	21.43 million
Full Ban	Full Bans include Narrow Bans on EPS, and also prohibit all polystyrene (i.e., foam and extruded) packaging used for meat, poultry, fish, produce, deli, and bakery products and egg containers at grocery stores.	9.6%	1.1 million
Expanded Ban	Expanded Bans include Full Bans and also (1) prohibits retail sale or distribution of EPS food service ware, (2) sale of single-use EPS coolers, and/or (3) sale or distribution of single use plastic (regardless of resin identification code) food service ware related items including utensils, straws, stirrers, lids, cups, plates, and containers.	15.7%	6.09 million

The 249 ordinances are local, municipal ordinances only and do not cover the state-level bans. They do not include the 10 municipal bans in Florida affected by the state preemption law

ly used in the ordinance's definition or applicability. This is a common challenge of narrowly and explicitly worded bans.

Public school systems, which often operate as a somewhat separate municipal authority within local governments in the USA, have imposed their own bans, primarily on single-use EPS cafeteria food trays, applicable only within the school system. For example, EPS food trays have been banned and replaced with compostable fiber-based trays in the USA's six largest school districts with 2.9 million students: New York City, Los Angeles, Chicago, Miami-Dade, Dallas, and Orlando (Layton, 2015). These six school districts collectively serve about 2.5 million meals per day resulting in an annual reduction of approximately 225 million EPS trays. The state of Oregon passed a law in 2015 that banned school districts from using EPS food service ware unless the school district recycles EPS.

6.1.2 Mandated Design Specification

Another approach to eliminate EPS food service ware is to mandate that only recyclable or compostable food serve ware items may be sold or used, which acts as a de facto ban on EPS. Product specification can mandate that items be recyclable, biodegradable, or compostable or that the product contain a specified amount of recycled content. Biodegradable generally means that a product can degrade by naturally occurring microorganisms (e.g., bacteria, fungi, and algae) whereas compostable is biodegradable in a composting environment, which means controlled conditions. However, the designations of recyclable and compostable are theoretical; another approach is to specify that a product is recycled or composted locally, which in addition to technical feasibility, also incorporates economics and infrastructure.

For example, the US State of Maine has proposed the term "readily recyclable," which means a material can be sorted by a Maine-based materials recovery facility and "there has been a consistent market for the material for the previous two calendar years" (Staub, 2020).

Portland, Oregon's original ordinance of 1989 banned all food vendors from using EPS that was manufactured using chlorofluorocarbons (CFC). The ordinance was subsequently modified to eliminate the CFC component. Portland's definition of EPS Foam Food Containers means any material composed of EPS and having a closed cell air capacity of 25% or greater, or a density of less than 0.787 grams per cubic centimeter based on an average EPS density of 1.05 grams per cubic centimeter, as determined by an analytical testing laboratory. As argued by Hardy and Charles (2007), Portland's approach can encourage the use of EPS that is denser. This is another example of the challenges and potential negative unintended consequences when proscribing narrowly and explicitly defined actions in municipal ordinances.

6.2 Market-based Approaches

A primary intent of market-based policy instruments is to raise the price of non-desirable products and/or lower the price of desirable products. This approach allows for consumer choice, but modifies the price signal with regard to that choice. The primary market-based policy instruments used for EPS are the imposition of extended producer responsibility, deposit-refunds, taxes/fees, and grants.

6.2.1 Extended Producer Responsibility and Product Stewardship

In North America, the primary management responsibility for municipal solid waste (MSW) resides with local governments funded by local taxpayers. Because producers are able to externalize costs related to end-of-life (EOL) management of EPS (e.g., collection, recycling, disposal, etc.), the price is distorted through indirect subsidies from local taxpayers. The intent of extended producer responsibility (EPR) and Product Stewardship (PS) is to shift the economic cost and/or logistical responsibility of their products at EOL from local governments and onto producers and other responsible parties. EPR seeks to require manufacturers of specific products to take environmental responsibility for the management of the product throughout its lifecycle with special emphasis on the post-consumer stage. In contrast, PS is a strategy to share the economic and/or logistical responsibility of a product among all responsible parties including the manufacturer, retailer, consumer, and government (Curtis et al., 2014). Both programs seek responsible parties to internalize the costs, which theoretically would encourage them to incorporate sustainable practices into product design including environmental and economic EOL considerations (McKerlie, Knight, & Thorpe, 2006) and consumption. EPR and PS have been adopted by national governments and sub-national governments in North America, Europe, and Asia for multiple product categories including electronics, product packaging, beverage containers, fluorescent lights, batteries, cell phones, mercury-thermostats, architectural paint, mattresses, and carpet.

In the State of Maine, PS for electronic waste was implemented in 2006, which required waste electronics to be certified that they were recycled in an environmentally sound manner. Households had to deliver their electronics to a designated municipal or retail collection point where they may have to pay a fee. Local governments had to collect, document, and prepare for shipment all household electronic waste. Consolidators collected, transported, and recycled the waste. Producers were required to pay the consolidators for the transportation and recycling costs, which shifted the costs away from local governments (and in most cases, households) and shared the costs among all parties, with producers paying the majority of costs. In the first three years of the program there was a 221% increase in the number of electronic items collected and properly recycled (Wagner, 2009). The upsurge in collection rates was a result of the dramatic increase in convenient drop-off locations at municipalities and non-profits, a reduction or elimination of fees previously charged to households due to producer financing, public education and outreach, and a subsequent disposal ban on electronic waste starting in 2007 (Wagner, 2009).

While the electronic waste PS program in Maine has been successful, in contrast, two other PS–based programs in Maine, covering mercury thermostats and mercury-containing compact fluorescent lamps, have not been successful. The mercury thermostat program requires producers to pay a US$5.00 bounty for each wall-mounted thermostat it collects, but the recovery rate has never exceeded 13% in any year (MDEP, 2019). Similarly, the collection and recycling of mercury-containing compact fluorescent lamps are funded by the lamp industry, but the lamp recovery rate has been only 10.15% (MDEP, 2019). Some of the primary reasons why these programs have not been as successful as the electronic waste program include the number of collection sites, public awareness, and most importantly, the funding responsibility. While all three programs are funded by the producers, only the electronic waste program is basically operated by an independent third party, the consolidators, which are incentivized to collect as much electronic waste as possible with the costs passed onto the producers. In contrast, there is no independent third party running the collection and recycling in the thermostat and fluorescent lamp programs. Consequently, there is an inherent economic disincentive for producers of thermostats and compact fluorescent lamps to collect and recycle their products. This disincentive manifests itself in a lack of education and public outreach, fewer collection sites, and imposition of burdensome paperwork.

While EPR and PS-based programs are an option for states in the USA, local governments charged with the responsibility to manage MSW lack the legal authority to adopt EPR. Regarding EPS food service ware, thus far, no state (or local governments) in the USA has adopted EPR.

6.2.2 Deposit/Refund

A type of PS-based program is the deposit/refund system, in which the manufacturer, retailer, and consumer typically share some responsibility to implement the system. Deposit/refund programs have been used for beverage containers, used motor oil, tires, and automotive lead-acid batteries. According to the Container Recycling Institute (2013), USA states with beverage container deposit-refund programs achieve container recycling rates of between 70% and 95% while states without deposit-refund programs achieve container recycling rates of about 22%. The application of a deposit-refund system can be applied to select, reusable food service ware where the meal or beverage is intended to be consumed onsite such as the informal pfand (deposit) system in Germany. In addition to Germany's official pfand system covering beverage containers where deposits and refunds are managed at retail beverage and grocery stores, there is also an informal pfand system widely used for reusable food service items (plates, cups, glasses, mugs, and utensils) at festivals and food and beverage kiosks. The system works by charging a customer a separate pfand (e.g., 1€) above the price of the food or beverage. The customer is typically given a plastic or wooden token that when returned along with the covered food service ware item, receives the deposit back. This demonstrates that such a system can be adopted on a voluntary basis by any establishment, but it could also be adopted as a local ordinance for appropriate food and beverage service operations.

6.2.3 Taxes/Fees

The primary goal of levying taxes and fees is to directly raise the price of a targeted product to discourage its consumption. Taxes/fees do not prohibit the use or sale of a product or behavior so it maintains consumer choice. Highly visible, small taxes/fees are often referred to as nudges as they are intended to encourage a specific behavior or action rather than impose an economic hardship (Rivers, Shenstone-Harris, & Young, 2017). To be most effective, fees and taxes need to be levied at the point of sale. As noted by Bury (2010), a separate, visible point-of-sale tax or fee can prompt the customer to internalize the cost. In response, overall consumption will decrease when customers are presented with a higher price because of the tax or fee. Taxes/fees for EPS food service ware can be defined as a litter, eco, or disposal tax/fee signaling the negative impact that is being taxed.

Local government fees have been very successful in reducing the consumption of SUP shopping bags (Wagner, 2017). To support France's goal of recycling 100% of its plastic waste by 2025, starting in 2019, plastic items not containing recycled plastic will be taxed. Similarly, starting in 2020, the UK will impose a plastic packaging tax for items containing less than 30% recycled materials. The UK also has proposed £0.25 tax on all disposable cups (e.g., the "latte tax").

Thus far, there have been no national or sub-national laws in the USA that tax EPS food service ware and only two local ordinances thus far have adopted a tax/fee. In 2019, Berkeley, California's EPS ordinance adopted a disposable cup tax: restaurants and coffee shops are required to charge customers $0.25 when they do not bring a reusable cup for their beverage. Similarly, Watsonville, California requires retailers to charge customer $0.10 for a disposable cup.

A variation of the tax/fee strategy is for local governments to authorize private establishments to charge fees rather than to impose them for all establishments. For example, Sebastopol, California's polystyrene ordinance authorizes retailers to charge customers up to $0.10 for takeout food packed in disposable packaging.

6.2.4 Grants/Credits

Grants are economic tools to provide direct economic incentives to an entity to engage in an action or to purchase a good that may not otherwise be done or purchased because of higher costs. Grants can be used to reduce costs of the preferable product. There have been minimal efforts in the USA to provide grants to reduce EPS food service ware. In one case, Encinitas, CA, offered a $400 grant for switching from EPS food service ware prior to the citywide ban, which was designed to offset the initial higher cost of purchasing EPS substitutes.

Monetary credits can be offered to engage in a preferred action such as for a customer utilizing their own reusable containers. Sebastopol, California's polystyrene ordinance encourages retailers to provide a credit of up to $0.25 to customers who bring their own reusable takeout containers.

6.3 Voluntary and Public Education Programs

Voluntary programs seek to challenge or foster private entities to engage in a preferred action or behavior such as offering substitutes to EPS food service ware. Voluntary programs for reduction of EPS consumer products tend to be local or regional in nature, are adopted by companies supporting corporate responsibility, or can be adopted as a means to reduce or obviate the need for potential regulation. While not EPS, in 2018, Starbucks in the UK voluntarily imposed a £0.05 fee on paper take away cups. Prior to this self-imposed fee, Starbucks was offering a £0.25 credit for customers who brought their own reusable cup; however, the usage rate of reusable cups throughout UK's Starbucks was only 2.3% (HUBUB, 2018). A pilot study was conducted at Starbucks' 35 London, UK stores. The study found that a £0.05 fee on paper cups, coupled with a £0.25 discount for bringing a reusable cup increased the proportion of hot beverages sold in reusable cups from the baseline of 2.2% to 5.8% (HUBUB, 2018). The control group had a 1.1% increase in reusable cup usage during the same time period likely from company-sponsored education and outreach (HUBUB, 2018).

Another example is a voluntary monthly subscription service for closed-loop reusable bowls at take-out restaurants (Fassler, 2019). As described by the author, with this model, subscribers pay a small monthly fee, which allows them to "check-out" a single reusable take-out container from the restaurant and return it to be replaced with a clean reusable take-out container.

Based in Atlantic Beach, FL, Girls Gone Green is a regional non-profit group that has created the "Hang Up The Foam" campaign. The group uses online media to encourage customers to request non-EPS food service ware, they distribute a brochure to restaurants to help them voluntary reduce EPS, and they offer technical assistance to restaurants to find viable substitutes. Governments can promote voluntary programs through information sharing and cooperative purchasing assistance to increase access and reduce costs of EPS alternatives (CCEAC, 2018).

Another approach to educate the public is to focus on the negative effects of EPS, especially litter. Litter-based education is used to increase awareness of the negative impacts of litter while increase awareness as to the proper placement of EPS in trash receptacles to prevent litter. The expectation is that the target audience will then not litter. This is often done through social marketing, which uses traditional commercial marketing approaches to change human behavior for social good. West Coast cities in the USA spend an average of $80,927 on public education to discourage littering (Stickel, Jahn, & Kier, 2012). Unfortunately, knowledge is not the sole determinant in an individual's proper waste management actions as cost and convenience are far more influential (Wagner, 2013). If education alone was successful, ordinances would be unnecessary; however, education has had only limited success with regards to reducing EPS (Wagner, 2016). To be sure, education is essential in bringing to light and understanding the problem and providing baseline knowledge to promote and support an adopted solution.

Similar to voluntary campaigns, education campaigns are often undertaken by local or regional groups. For example, the 5 Gyre Institute has adopted the "Nix the 6" social marketing campaign asking individuals to pledge to avoid polystyrene products (RIC #6). The Universiti Teknologi Malaysia adopted its Zero Polystyrene Campaign to encourage students to bring reusable containers while simultaneously introducing a biodegradable container made from oil-palm waste produced in Malaysia. The city of Manhattan Beach, CA, started its "Bring Your Own!" campaign to encourage the public to bring reusable containers, cups, straws, and utensils whenever possible.

7. CONCLUSIONS

The widespread dispersal of plastic waste throughout the environment has captured the world's attention.

It is a symptom of a global problem: seemingly insatiable consumption of plastics coupled with poor EOL management of plastics. One of the contributors of this problem is EPS food service ware. Its consumption continues to increase in spite of the lack of environmentally preferable EOL management. Moreover, the use of EPS food service ware presents an economic burden to local governments' solid waste management efforts because of its limited recyclability and the costs of litter and stormwater system clean-up. There are also economic and social costs included in the negative impacts of EPS litter to local governments from reduced tourism and blocked storm drains. Given that reuse, recycling, and composting of EPS are not currently viable options, based on SMM, the primary policy goal should be to reduce the consumption of EPS food service ware. In the USA, this goal has been the focus as 249 local governments and 2 states (and the District of Columbia) have banned EPS to varying degrees. While most of the bans have focused on EPS food service containers, increasingly, bans have included EPS packaging (e.g., deli and food trays, egg cartons) and other materials (e.g., EPS coolers and packing material and SUP utensils). An intended consequence of these bans has been to foster the switch to environmentally preferred substitutes that are locally recycled and/or composted. The adoption of fees for single-use cups has also recently been adopted as means to retain customer choice while achieving the goals of reduction and preferable product substitution.

To be sure, reducing EPS and increasing the use of environmentally preferable food service ware substitutes is a complex challenge. Until recently we have relied on private markets to solve the problem by producing a viable and cost-effective substitute to EPS. However, as evidenced by the continued build-up of plastic waste, litter, and debris and the increasing consumption of EPS with no economically viable post-consumer market, government intervention to solve the problem has increased. In the absence of national and state actions, and given that local governments shoulder the economic burden of MSW management in the USA, they have taken the lead on adopting ordinances to reduce their costs and reduce local environmental impacts primarily through bans.

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APPENDIX 1

List and type of municipal bans on EPS food service ware in the USA as of December 2019 (N=259).

City	State	Ban Type	Effective Date
Bethel	AK	Partial	2010
Cordova	AK	Narrow	2016
Seward	AK	Narrow	2019
Fayetteville	AR	Narrow	2019
Alameda city	CA	Expanded	2018
Alameda County	CA	Narrow	2008
Albany	CA	Narrow	2008
Aliso Viejo	CA	Narrow	2005
Arcata	CA	Narrow	2015
Arroyo Grande	CA	Narrow	2016
Avalon	CA	Narrow	2018
Belmont	CA	Narrow	2012
Berkeley	CA	Narrow	1988
Brisbane	CA	Narrow	2014
Burlingame	CA	Narrow	2012
Calabasas	CA	Narrow	2008
Campbell	CA	Narrow	2015
Capitola	CA	Narrow	2009
Carmel	CA	Narrow	1989
Carpentaria	CA	Expanded	2009
Colma	CA	Narrow	2013
Concord	CA	Narrow	2019
Contra Costa	CA	Narrow	2020
Costa Mesa	CA	Partial	2019
Culver City	CA	Expanded	2017
Cupertino	CA	Narrow	2014
Daly City	CA	Narrow	2012
Dana Point	CA	Narrow	2012
Davis	CA	Expanded	2017

City	State	Ban Type	Effective Date
Del Rey Oaks	CA	Narrow	2010
El Cerrito	CA	Narrow	2014
Emeryville	CA	Expanded	2008
Encinitas	CA	Narrow	2017
Fairfax	CA	Narrow	1993
Fort Bragg	CA	Narrow	2015
Foster City	CA	Narrow	2012
Fremont	CA	Expanded	2011
Gonzales	CA	Narrow	2015
Greenfield	CA	Narrow	2015
Grover Beach	CA	Expanded	2018
Half Moon Bay	CA	Narrow	2011
Hayward	CA	Narrow	2011
Hercules	CA	Narrow	2008
Hermosa Beach	CA	Expanded	2020
Huntington Beach	CA	Narrow	2004
Imperial Beach	CA	Narrow	2018
Lafayette	CA	Narrow	2015
Laguna Beach	CA	Narrow	2008
Laguna Hills	CA	Partial	2004
Laguna Woods	CA	Narrow	2013
Livermore	CA	Narrow	2011
long Beach	CA	Expanded	2018
Los Altos	CA	Expanded	2014
Los Altos Hills	CA	Narrow	2012
Los Angeles City	CA	Partial	2008
Los Angeles County	CA	Partial	2010
Los Gatos	CA	Narrow	2015
Malibu	CA	Narrow	2005

City	State	Ban Type	Effective Date
Manhattan Beach	CA	Full	2013
Marin County	CA	Narrow	2010
Marina	CA	Narrow	2012
Martinez City	CA	Narrow	1995
Mendocino County	CA	Narrow	2014
Menlo Park	CA	Narrow	2012
Mill Valley	CA	Narrow	2009
Millbrae	CA	Expanded	2008
Milpitas	CA	Narrow	2018
Monterey City	CA	Narrow	2009
Monterey County	CA	Narrow	2010
Morgan Hill	CA	Narrow	2014
Morro Bay	CA	Expanded	2016
Mountain View	CA	Expanded	2014
Newport Beach	CA	Narrow	2008
Novato	CA	Narrow	2014
Oakland	CA	Narrow	2007
Ojai	CA	Narrow	2014
Orange County	CA	Partial	2005
Pacific Grove	CA	Narrow	2008
Pacifica	CA	Narrow	2010
Palo Alto	CA	Expanded	2010
Pasadena	CA	Narrow	2017
Petaluma	CA	Partial	2020
Pismo Beach	CA	Narrow	2016
Pittsburg	CA	Narrow	1993
Pleasanton	CA	Narrow	2013
Portola Valley	CA	Narrow	2012
Rancho Cucamonga	CA	Partial	1989
Redwood City	CA	Narrow	2013
Richmond	CA	Expanded	2009
Salinas	CA	Narrow	2012
San Bruno	CA	Narrow	2010
San Carlos	CA	Narrow	2012
San Clemente	CA	Narrow	2011
San Diego	CA	Expanded	2019
San Francisco	CA	Expanded	2007
San Jose	CA	Narrow	2014
San Juan Capistrano	CA	Narrow	2004
San Leandro	CA	Narrow	2012
San Luis Obispo	CA	Narrow	2016
San Luis Obispo County	CA	Expanded	2020
San Mateo	CA	Narrow	2013
San Mateo County	CA	Partial	2008
San Pablo	CA	Narrow	2015
San Rafael	CA	Narrow	2013
Santa Barbara	CA	Narrow	2019
Santa Clara City	CA	Narrow	2014
Santa Clara County	CA	Narrow	2013
Santa Cruz City	CA	Expanded	2008
Santa Cruz County	CA	Expanded	2008
Santa Monica	CA	Narrow	2007

City	State	Ban Type	Effective Date
Sausalito	CA	Narrow	2008
Scotts Valley	CA	Narrow	2009
Seaside	CA	Full	2010
Sebastopol	CA	Full	2019
Solana Beach	CA	Narrow	2015
Sonoma City	CA	Narrow	2015
Sonoma County	CA	Narrow	1989
South Lake Tahoe	CA	Expanded	2018
South Pasadena	CA	Full	2018
South San Francisco	CA	Narrow	2008
Sunnyvale	CA	Expanded	2014
Ukiah	CA	Narrow	2015
Union	CA	Narrow	2017
Ventura County	CA	Partial	2004
Walnut Creek	CA	Narrow	2014
Watsonville	CA	Expanded	2009
West Hollywood	CA	Narrow	1990
Yountville	CA	Narrow	1989
Hamden	CT	Narrow	1990
Norwalk	CT	Expanded	2020
Westport	CT	Narrow	2019
Washington	DC	Expanded	2017
Alachua County*	FL	Narrow	2020
Bal Harbour*	FL	Narrow	2014
Bay Harbor Islands*	FL	Narrow	2015
Coral Gables*	FL	Narrow	2016
Deerfield Beach	FL	Partial	2017
Gainesville*	FL	Full	2019
Hollywood*	FL	Narrow	1996
Key Biscayne*	FL	Narrow	2014
Miami Beach	FL	Partial	2014
Miami-Dade	FL	Partial	2017
North Bay Village*	FL	Narrow	2015
Orlando*	FL	Narrow	2019
St. Augustine Beach*	FL	Narrow	2020
St. Petersburg	FL	Partial	2019
Surfside	FL	Narrow	2015
Hawaii County	HI	Narrow	2019
Maui County	HI	Narrow	2018
Abington	MA	Narrow	2018
Amherst	MA	Narrow	2014
Andover	MA	Expanded	2018
Arlington	MA	Full	2020
Brookline	MA	Narrow	2013
Cambridge	MA	Narrow	2016
Chelmsford	MA	Partial	2019
Concord	MA	Narrow	2017
Denis	MA	Narrow	2016
Falmouth	MA	Partial	2018
Georgetown	MA	Narrow	2018
Gloucester	MA	Narrow	2019
Great Barrington	MA	Narrow	1990

City	State	Ban Type	Effective Date
Greenfield	MA	Narrow	2017
Hamilton	MA	Narrow	2016
Ipswich	MA	Narrow	2017
Lee	MA	Narrow	2017
Lenox	MA	Narrow	2017
Lexington	MA	Narrow	2019
Manchester by-the-Sea	MA	Full	2018
Marblehead	MA	Narrow	2015
Nantucket	MA	Narrow	1990
Newton	MA	Expanded	2020
Northborough	MA	Narrow	2020
Orleans	MA	Narrow	2010
Pittsfield	MA	Narrow	2016
Provincetown	MA	Expanded	2019
Saugus	MA	Narrow	2020
Shrewsbury	MA	Partial	2018
Somerville	MA	Narrow	2014
South Hadley	MA	Narrow	2015
Stockbridge	MA	Narrow	2018
Wayland	MA	Narrow	2018
Wellfleet	MA	Narrow	2018
Westfield	MA	Narrow	2016
Westford	MA	Narrow	2017
Williamstown	MA	Narrow	2015
Winthrop	MA	Narrow	2017
Annapolis	MD	Full	2019
Anne Arundel County	MD	Full	2020
Baltimore	MD	Narrow	2018
Gaithersburg	MD	Full	2016
Montgomery County	MD	Expanded	2016
Prince Georges County	MD	Partial	2016
Rockville	MD	Full	2019
Takoma Park	MD	Full	2015
Bar Harbor	ME	Full	2019
Bath	ME	Narrow	2018
Belfast	ME	Narrow	2018
Bethel	ME	Narrow	2019
Blue Hill	ME	Full	2018
Brunswick	ME	Narrow	2016
Camden	ME	Full	2019
Cape Elizabeth	ME	Full	2017
Freeport	ME	Full	1990
Mount Desert	ME	Narrow	2020
Portland	ME	Full	2015
Rockland	ME	Narrow	2019
Rockport	ME	Narrow	2019
Saco	ME	Narrow	2016
South Portland	ME	Narrow	2016
Southwest Harbor	ME	Full	2019

City	State	Ban Type	Effective Date
Topsham	ME	Narrow	2017
Minneapolis	MN	Narrow	2015
St. Louis Park	MN	Narrow	2019
Portsmouth	NH	Narrow	2020
Avalon	NJ	Narrow	2019
Bergen County	NJ	Narrow	2020
Cranford	NJ	Narrow	2020
Fair Haven	NJ	Narrow	2019
Hawarth	NJ	Narrow	2019
Hoboken	NJ	Narrow	2019
Lambertville	NJ	Full	2020
Little Silver	NJ	Narrow	2019
Monmouth Beach	NJ	Narrow	2018
Ocean Gate	NJ	Narrow	2019
Paramus	NJ	Narrow	2020
Rahway	NJ	Expanded	1997
Red Bank	NJ	Expanded	2019
Bernalillo County	NM	Partial	2020
Albany County	NY	Narrow	2014
Duchess County	NY	Narrow	2019
East Hampton	NY	Expanded	2018
Glen Cove	NY	Expanded	1988
Hastings-on-Hudson	NY	Full	2015
Patchogue Village	NY	Narrow	2018
Putnam County	NY	Partial	2015
Suffolk County	NY	Narrow	1988
Ulster County	NY	Narrow	2015
New York	NY	Narrow	2018
Ashland	OR	Full	2010
Florence	OR	Full	2018
Medford	OR	Narrow	2015
Milwaukie	OR	Partial	2019
Portland	OR	Narrow	1990
Barrington	RI	Narrow	2019
Acadia Lakes	SC	Full	2020
Charleston County	SC	Full	2020
Edisto Beach	SC	Expanded	2020
Folly Beach	SC	Expanded	2016
Isle of Palms	SC	Expanded	2020
James Island	SC	Expanded	2020
Kiawah Island	SC	Narrow	2019
Mount Pleasant	SC	Expanded	2018
Sullivan's Island	SC	Expanded	2018
San Marcos	TX	Partial	2012
Issaquah	WA	Narrow	2010
Port Townsend	WA	Narrow	1990
San Juan County	WA	Narrow	2010
Seattle	WA	Narrow	2009

** Florida municipal ordinance that may be vacated by the state prohibition on local EPS bans.*

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ASCENDING THE WASTE HIERARCHY: RE-USE POTENTIAL IN SWEDISH RECYCLING CENTRES

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ABSTRACT

According to the waste hierarchy principle, which constitutes the basis of European waste legislation, waste prevention and re-use are considered – most of the times – better waste management options than recycling. However, prevention and re-use activities are difficult to operationalise and measure, without a monitoring framework in place. This contribution investigates the potential of re-using end-of-life products that have been disposed at recycling centres in Sweden. Recycling centres receive a wide variety of materials for recycling, of which a portion could be re-used instead. The aim is to identify what product groups can be re-used, the share of these potentially re-usable products in the recycling centres, and under what conditions their re-use is feasible. A literature review of similar studies, site visits at recycling centres in Sweden, and semi-structured interviews with relevant stakeholders were used to analyse the potential for re-use in private recycling centres in Sweden. The most suitable product groups for re-use identified are building materials, furniture and electrical equipment (mainly white goods), as other material types are mostly handled by charity organisations (e.g. textiles). There is significant potential for increasing re-use operations in recycling centres, but in order to be economically profitable it is important to identify the most suitable material fractions (or product groups) and engage in strategic partnerships that will allow more effective organisation of re-use processes.

1. INTRODUCTION

Waste management in the European Union (EU) is defined in the Waste Framework Directive (2008/98/EC), as amended by Directive (EU) 2018/851, which outlines the rules and conditions by which all waste management operations and planning is taking place in the EU Member States. It is complemented with a number of Directives setting the rules of managing separate waste streams (e.g. packaging waste, electronic waste etc.). The central principle of EU waste management, as it is expressed in Article 4 of the Waste Framework Directive, is the so-called "waste hierarchy". The waste hierarchy addresses the prioritisation of waste management options according to environmental and resource efficiency aspects. According to this hierarchy, waste management operations with negative environmental impacts are considered undesirable and should progressively be limited, and ultimately substituted by waste management operations that are considered more resource efficient and environmentally sound (European Commission, 2008).

The waste hierarchy includes the following waste management operations: (a) waste prevention; (b) re-use and preparation for re-use; (c) material and biological recycling; (d) energy recovery from waste; and (e) disposal to controlled or uncontrolled landfills, land or water. It is worth noting that although the hierarchy is addressing waste management, step (a) and partially step (b) of the hierarchy deal mainly with non-waste. Waste that is prevented is waste not generated, and re-use of a product means that the product did not become waste in the first place. However, step (b) might indicate that a product first became waste and then brought back to a suitable condition for re-use.

The term "re-use" is defined in the Waste Framework Directive as 'any operation by which products or components that are not waste are used again for the same purpose for which they were conceived.' (European Commission, 2008; Article 3). Product repair, refurbishment, and remanufacturing are all considered to be re-use operations (Ijomah and Danis, 2012), and are often environmentally preferable to material recycling and manufacturing of new products as they save material resources and energy, reduce greenhouse gas emissions, and lead to safer handling of potential toxic substances in products (Sundin and Lee, 2012).

Waste management in the EU has moved steadily up-

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wards the waste hierarchy, prioritising options considered as best alternatives. Indeed, the recycling share of municipal solid waste has increased from 30% in 2004 to nearly 44% in 2014 (European Environment Agency, 2017). However, there is an apparent lack of information concerning the performance and progress of Member States in re-use operations. Considering that preparation for re-use is an immediate step up from recycling in the waste hierarchy, it would be evident for municipalities and regional authorities following waste legislation (and for their subcontracted private enterprises) to strive to get to the re-use stage of the waste hierarchy.

In Sweden, more than half of all municipal recycling centres include the possibility to receive materials for re-use, such as clothes and furniture, often in collaboration with charity organisations. There are also recycling centres with adjacent recycling parks that have extended operations, such as repairs and sales of second hand goods. The volume of waste submitted to the municipal recycling centres is steadily increasing, and so are the possibilities for re-use of a variety of products and materials (Avfall Sverige, 2018). Moreover, in addition to the municipal recycling centres, private enterprises have capabilities of collecting a variety of waste from municipal or private actors within their own facilities, with a good potential for re-use (STENA, 2016).

In this contribution, we are investigating the potential of re-use and the possibility of ascending the waste hierarchy in commercial recycling centres in Sweden. Similar to previous studies that have analysed the re-use potential in municipal recycling centres (Ljunggren Söderman et al., 2011; Hultén et al., 2018a; Hultén et al., 2018b), for this study we conducted a qualitative analysis of the different waste streams treated in two sorting facilities of the largest recycling operator in Sweden. The analysis aimed at assessing the type of waste streams and the quality of waste, and to examine if the waste could have been re-used instead of recycled. In this study, we did not consider the types of material such as plastic, metal or wood, but product groups such as furniture, building components, etc. Analysis of re-use potential at product level allows better understanding of where and when re-use is feasible, which could facilitate new business models for re-use involving the recycling centres (Zacho et al., 2018).

In the following sections, the main characteristics of re-use concerning environmental, economic and social aspects are presented, as well as the legal implications of re-use in Sweden. Also, previous experiences with re-use in municipal or private entities at various EU Member States are detailed to provide background context for our current study. Then, the methodological approach of this contribution is presented, followed by the integrated results and discussion. Finally, this contribution ends with presenting the main conclusions and future research opportunities.

2. CHARACTERISTICS OF RE-USE

This section outlines the sustainability characteristics of re-using end-of-life (EOL) products and presents the associated legal and organisational implications of re-use. Additionally, previous studies on Swedish and other EU re-use centres are presented, highlighting important conditions of re-use relevant to the present study.

2.1 Environmental benefits of re-use

Waste prevention and re-use is generally considered a better environmental option than other treatments of waste. For instance, preventing the generation of one kilogramme of textile waste can potentially reduce carbon dioxide (CO2) emissions by 15 kg, while the amount of reduced emissions is 8 kg of CO2 if that one kg of textiles is re-used and approximately 0-3 kg emissions reduction if textile waste is recycled (Avfall Sverige, 2015). A deviation from this general principle can occur when products have been designed to be much more energy-efficient in the use phase of their life-cycle. Generally, products that have a large energy consumption after manufacture are not favourable for re-use (Gutowski et al., 2011). Another example of products that are less suitable for re-use are products that contain hazardous substances which, when re-used, persist in the product stock in use and are not phased out (Eriksen et al., 2018).

A previous study by Ljunggren Söderman et al. (2011) measured the environmental impacts of re-use by life cycle assessment (LCA) methodology, using as case study the recycling centre Alelyckan situated in Gothenburg, Sweden. Contrary to a "traditional" recycling centre, where received waste are destined solely for recycling, the Alelyckan recycling centre offers the opportunity to collect waste for re-use before they reach the recycling bins. In 2010, the recycling centre prevented 358 tonnes of waste, which corresponds to 5.6 per cent of the total weighted waste received at the centre (Ljunggren Söderman et al., 2011). Figure 1 shows the amount of waste collected for re-use (tonnes) and the amounts that were actually re-used in each product category. The figure shows that the product groups with the largest amount re-used were textiles and metal products. Books were also commonly collected, but these proved difficult to re-use. Furthermore, the figure shows that all wood construction products collected were eventually re-used.

The study investigated, among other things, the environmental impacts (in the form of greenhouse gas emissions, acidification and eutrophication) due to avoiding new production and waste transport, and the increase or decrease in energy recovery (as in Sweden, most waste that is not re-used, composted or recycled is burned in municipal incinerators connected to district heating systems). The LCA calculations for the 358 tonnes of waste that could be re-used confirmed that the greatest environmental benefit is due to replacement of new production. The carbon dioxide savings that could be made in one year by the introduction of Alelyckan, compared to a conventional recycling centre were, 1,300 tonnes of carbon dioxide equivalents (Ljunggren Söderman et al., 2011). This amount is equal to the total emissions of 120 Swedes over one year, including private and public consumption in and outside Sweden, i.e. about 11 tonnes per person per year (SEPA, 2018a). It is primarily textiles and small electric and electronic equipment (EEE) that have a major impact on avoided greenhouse gas emissions during re-use. For textiles, the result is largely

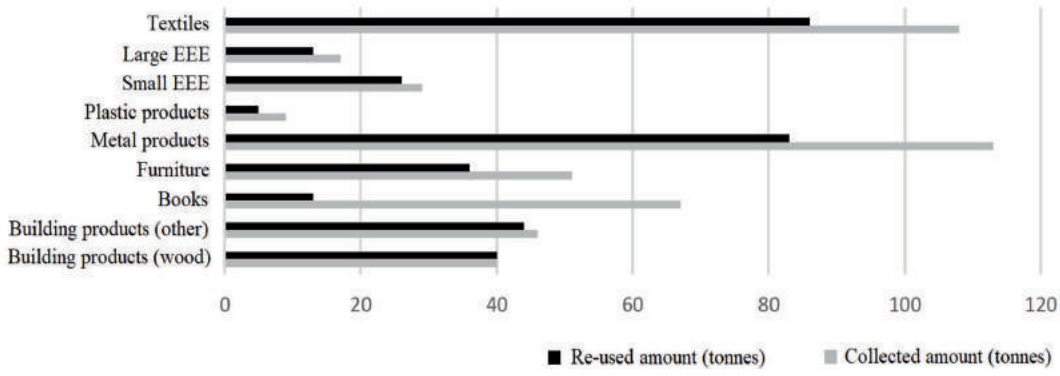

FIGURE 1: Collected and re-used amounts of waste at Alelyckan recycling centre in 2010 (Ljunggren Söderman et al., 2011). Note: EEE stands for Electric and Electronic Equipment.

influenced by the fact that large amounts of textiles were collected at the recycling centre, while small EEE have a greater environmental impact per tonne collected (Ljunggren Söderman et al., 2011).

The study also showed that in 2010 the recycling centre prevented emissions of substances with acidifying and eutrophication effect by 10 and 1.5 tonnes respectively, which is in the order of the annual emission of 400 Swedes. In conclusion, the study noted that if all recycling centres in Sweden were rebuilt with a re-use concept similar to Alelyckan, then about 80,000 tonnes of waste could be prevented annually (Ljunggren Söderman et al., 2011), which is as much waste generated as in a medium-sized Swedish city, taking into account that a Swede generates on average 473 kg of household waste per year (Avfall Sverige, 2018).

2.2 Economic aspects of re-use

The economics of a business, whether private, non-profit or public, plays a central role in determining its feasibility and long term sustainability. Unlike conventional recycling centres, the operating cost of a centre with a re-use focus is higher. Personnel costs increase as more staff is required to sort the incoming waste, inform visitors, and label waste that has been prepared for re-use. Higher premises costs arise as a result of an additional sorting station, storage facilities and more sorting containers (WSP, 2012). There are various alternatives to cover the increased costs. One example is to regulate the municipal waste tariff. This is possible since the preparation of waste for re-use can be classified as a recycling activity. There is therefore no legal obstacle to using income from the waste tariff to finance such activities (Avfall Sverige, 2014). In addition, it is commonly observed that re-use organisations in several EU countries have been receiving state support, directly or indirectly, to maintain their operations (Zajko and Hojnik, 2014).

Furthermore, the waste collected for re-use has an economic value as a product. It can therefore be assumed that re-use of products can also have higher economic benefits, in contrast to recycling (Avfall Sverige, 2015). The waste streams entering a recycling centre can be quite heterogeneous and can also vary depending on the season. This means that even the commercial value of the waste may vary, thus also the interest from external actors. The types of products received also affect the value of the waste. Products with a high commercial value will most probably not be left at a recycling centre (although this is not always the case). Products that end up in a recycling centre are most often things that households do not consider sufficiently valuable to divert in a second-hand market outlet. However, there is still some residual value in them, which can be harnessed if the recycling centre is connected to repair services (Hultén et al., 2018a).

2.3 Social aspects of re-use

The social effects of re-use concern mainly increased employment and inclusiveness. Traditionally, non-profit second-hand businesses provide a workplace opportunity for many people who find it difficult to enter the labour market in any other way. A study conducted to investigate social benefits related to second-hand activities (Jannesson and Nilsson, 2014) concluded that a large proportion (74%) of people employed in a second-hand business, through an internship or work training (subsidised by public unemployment services), experienced increased meaningfulness, reduced stress, increased participation in society and improved social relations. There are also social effects for customers who buy second-hand products. A growing secondary market would have a positive effect on households as they gain greater access to affordable products. Other positive effects for the consumer may be the feeling of acting environmentally conscious or that the money from purchasing second-hand goes for a charitable purpose (Shaw and Williams, 2018).

2.4 Legal aspects of re-use

An important parameter when designing a re-use centre is the embedded legal framework concerning the management of waste and associated re-use activities. In Sweden, according to the Environmental Code (SFS 1998: 808), each municipality is responsible for disposing or recycling household waste. Part of the municipal responsibility is therefore to establish recycling centres where citizens can leave waste that are not collected from households. When a product has been submitted to a recycling centre, this is transferred to the municipality's ownership. The right of ownership means that the municipality has the exclusive right to decide on how the waste is to be treated, taking into account national guidelines such as the waste hierarchy (Avfall Sverige, 2014).

For businesses that handle products submitted with the explicit purpose of being re-used, and thus will not be classified as waste, a waste management permit does not have to be applied, according to the Environmental Assessment Ordinance (SEPA, 2017).

For re-use to be possible, in some cases it is required that the submitted waste is in some way processed or prepared for re-use. This may, for example, involve checking, repairing or cleaning. In the most recent amendment of the Environmental Assessment Ordinance (SFS 2013: 251), a point of appeal was added, namely para. 47 "Preparation for re-use" with business code 90.29. This facilitates the operations of businesses that work with preparation for re-use, as such activities are classified obligatory for notification and therefore do not have to undergo an authorization process, but only a notification to the municipality is required (SEPA, 2017).

An important aspect to take into account in municipal sales of recycled goods is the Competition Act (SFS 2008:579). This is because the state, county council and municipality must not conduct a sales activity that can distort or impede private competition. This may be the case if waste at a recycling centre is pre-treated and sold under municipal auspices. Other recycling operators do not have the same opportunity and can be disadvantaged. One possible interpretation, however, is that other actors are not able to run collection, sorting and preparation for re-use to the extent required to fulfil the municipal responsibility, and therefore there is no competitive advantage for the municipality. Regardless, the municipalities must take into account the Local Government Act (SFS 1991: 900) which states that a municipality must not conduct activities with a profit interest and that all activities within municipal operations must have a public interest purpose (Avfall Sverige, 2014).

To circumvent potential distortion in competition, and legally uncertain practices, municipalities could cooperate with private actors (Hultén et al., 2018a). The contractual form between a municipality and private actors affects the legal framework of the collaboration. The dividing line is defined by who pays and for what. Hultén et al. (2018a) provide several examples to illustrate such inconsistencies. For instance, if a municipality gives collected products to private operators, this can be seen as unauthorized individual support for traders. Transparent selection processes among players are needed and at least local rent or similar should be paid by the private parties. In another example, a municipality could pay private actors to receive collected products, and this could be considered as the provision of service to the municipality, and in that case formal public procurement processes must be applied. Finally, a municipality could sell collected products to private operators. Procurement or selection among relevant actors is not required formally if sales are made at market prices. On the other hand, if the sale is made by renting an area or at a discounted price, it may be an unauthorized individual support for the trader and the selection process becomes necessary.

2.5 Re-use potential in municipal recycling centres in Sweden

To identify the potential of re-use in a recycling centre, it is important to identify the type of products that are submitted and the condition they are in. A recent study conducted in two municipal recycling centres in Sweden (Norra Hamnen in Malmö and in Örkelljunga) quantified the received waste for re-use and assessed its quality and re-use potential (Hultén et al., 2018a; Hultén et al., 2018b). In total, 15.5 tonnes of waste was examined through composition analysis, and the re-use potential of nearly 17,000 products was assessed.

About a quarter of the waste examined was considered commercially or functionally re-usable (Figure 2). A very small percentage was judged to be commercially re-usable after repair (3%), but many of the products that were considered to have a resale value in their existing condition would have a higher value if they were also repaired. Among all items deemed re-usable, a further distinction was made between commercially re-usable and functionally re-usable products. The latter, although retain functional capability (i.e. product can be used for its intended purpose), they have negligible economic value and are not commercially viable. That waste category constituted five percent by weight, including products such as used socks and plastic pots. More than two-thirds of the waste that was investigated consisted of other waste types such as packaging and garden waste or products in poor condition that could not been re-used.

The largest amount of commercially re-usable products did not belong to product groups traditionally handled by charity organizations, as for instance clothes. Building products, furniture, pallets and tools were commonly the ones that were identified with a high re-use potential. Repair work was not judged to be able to increase re-usable amounts to any notable extent, even if it was performed with no cost. Repair work could, however, increase the value of products already considered to be commercially re-usable. A simple cleaning would increase the value of the majority of products.

Hazardous substances were found in re-usable products, but not at alarming levels. Non-metallic toys and household utensils contained lead and nickel in 10 to 35 per cent of the products. Among toys and household utensils made of plastic or textile, bromine was detected in 20 and 25 per cent of products respectively. Laboratory tests revealed negligible levels of Polycyclic Aromatic Hydrocarbons (PAHs) and Perfluorinated Alkylated Substances (PFASs) in all samples. Phthalates were detected in half the samples but at levels below current legal limits. These findings cannot be used as general conclusions whether any specific kind of product should be re-used or not. The reason for this is that two very similar products may contain different levels of hazardous substances (Hultén et al., 2018a).

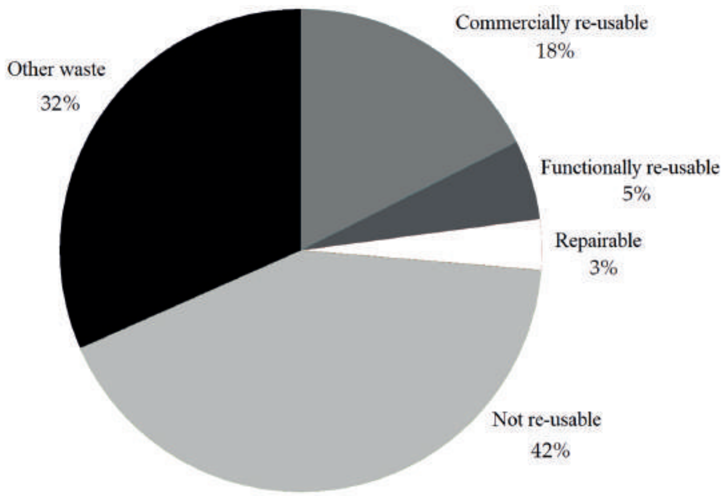

FIGURE 2: Results of the composition analysis of waste at two recycling centres (Norra Hamnen in Malmö and Örkelljunga), showing re-use potential expressed by weight of received waste (Hultén et al., 2018a).

3. RE-USE EXPERIENCES IN EU MEMBER STATES

Several re-use initiatives have sprung across the EU in recent years, with varying levels of success and organisational structures. In this section, following the detailed case study in Sweden (section 2.5), we present briefly a few initiatives that have provided increased re-use potential, as documented in literature. We take a closer look at the case of Flanders' KOMOSIE (Re-use network) and at re-use case studies in Austria, Denmark and Spain.

3.1 Re-use network in Flanders

The Flemish re-use network is represented by the non-profit organisation KOMOSIE, which stands for Federation of Environmental Entrepreneurs in the Social Economy. It is an umbrella organisation consisting of all re-use centres in Flanders, which have been accredited by the Public Waste Agency of Flanders (OVAM). The main pillars of success for the establishment of the re-use network in Flanders include: (1) the integration of re-use activities with the regional employment policy; (2) the close collaboration with municipalities by forming inter-municipal partnerships; (3) the pursuit of professionalization in operations and the constant monitoring and quality control; and (4) a carefully planned and communicated marketing policy, assisted by the support of the umbrella organisation KOMOSIE (Vandeputte et al., 2015).

By the beginning of the 1990s, the Flemish Waste Management Plan introduced the mandatory door-to-door collection of bulky household waste at least twice a year by municipalities, and the sorting of any recyclable materials. This requirement drove the municipalities to enact inter-municipal partnerships and to redesign their municipal solid waste policy. Re-use centres responded to this policy-driven opportunity by rapidly profiling themselves as an indispensable actor in the household waste collection and thus received a complementary role in the municipal waste policy (Vandeputte et al., 2015). Therefore, the Flemish re-use centres could sign agreements individually with OVAM, and with this they would receive annual subsidy for four successive years, in line with the duration of the Waste Management Plan. To be eligible for this subsidy, the re-use centres were required to participate in supporting the Flemish prevention and recycling policy and to report annually their activities to OVAM. As a result, the re-use centres became strongly embedded within the Flemish waste policy and started to gain greater momentum (Vandeputte et al., 2015). Moreover, the re-use centres were integrated into the legal take-back obligation of waste electric and electronic equipment (WEEE), by the ordinance of the Flemish Government of 17 December 1997 on the establishment of the Flemish regulations regarding the prevention and management of waste (section 3.5 Brown and White goods Art. 3.5.2). With this decision, the collected amounts of WEEE by the Belgian Producer Responsibility Organisation (PRO) Recupel would be redirected first to re-use centres for potential re-use and preparation for re-use, before reaching the subsequent recycling stage within the waste hierarchy policy framework (Vandeputte et al., 2015).

The professionalization of the re-use centres was an important step to consolidate the position achieved within the waste policy. To increase the profile of re-use centres and raise the confidence and sales of re-used products, it was necessary to develop a strong brand with a far-reaching communication strategy. Moreover, to boost confidence and quality in their products, re-use shops associated with the network (De Kringwinkel shops) introduced periodic external auditing that ensured a set of quality standards, in line with international practice (Vandeputte et al., 2015). The re-use network went even a step further, by introducing their own quality label for re-used goods named "Revisie", mostly in relation to re-used electrical equipment. The "Revisie" quality label reassured the customers that an electric device from the re-use shop (De Kringwinkel) would work properly and safely. Every device was subjected to thorough technical inspections, and if needed it would be professionally repaired and tested (Gåvertsson et al., 2018).

Profit generation is not a goal in itself for the re-use shops, but healthy financial performance is a necessary condition for sustainable operations and employment opportunities. Re-use shops that mainly rely on subsidies or on the goodwill of volunteers for daily operations cannot lead sustainable business (Vandeputte et al., 2015). The total turnover of re-use centres in Flanders is made up from two main sources, namely the shop-generated sales and the subsidies. Shop revenue is generated from the sale of re-used goods, which constituted 39% of the revenue in 2014. Other revenues are generated from the sale of materials to the recycling sector and tonnage fees for the collection (14% of the total revenue). The rest and most important share of the revenues was attributed to subsidies, mostly given for employment (Vandeputte et al., 2015).

There are 31 re-use centres in Flanders which are categorised by OVAM in 2 groups, the centres that operate on a broader scope (22) and the traditional ones (9). Traditional re-use centres collect only EOL products that have a re-use potential (not waste), while the broader scope centres have the possibility to collect larger amount of bulky waste (including also non-reusable items). Of the total quantity of collected products, approximately half of the goods redirected for re-use in 2014 were furniture and textiles, both in quantity (kg) and value (EUR). In the case of WEEE, just 12% of the collected amount was re-used, while the rest was diverted to recycling. Only one out of four books and multimedia (record vinyls, compact discs, etc.) could be re-used through the re-use shops in Flanders. Approximately 55% of the total goods inflow was non-reusable (Vandeputte et al., 2015).

3.2 Re-use case studies in Denmark and Spain

There are very few case studies in scientific literature investigating the potential for re-use in recycling centres and/or through separate collection of reusable EOL items. One case study refers to a project in a municipal waste management company in Northern Denmark (Zacho et al., 2018) and another study analyses the findings of a pilot study of separate collection for re-use in Spain (Bovea et al., 2016).

In the case of the municipal waste company in Denmark, the revised configuration of the regional recycling centre included a re-use shop with a workshop where the preparation for re-use processes were conducted, employing nine contracted full-time employees and six employees on special conditions who might otherwise have been outside the labour market. The latter have been employed through collaboration with the municipality's employment office and they had a mentor in their job training (Zacho et al., 2018).

After the introduction of the re-use centre, the amount of reusable items that were collected out of the combustible waste stream doubled and reached 3.23% of the total in 2016. The labour and logistical input to the process of sorting out and processing EOL products for re-use was so costly that the revenues from the sales of re-used products just covered the expenses. This means that the value of items for re-use at the recycling centre do not result in economic profits, but the benefits are mostly concentrated in the local employment opportunities that re-use provides. The economic value of EOL products is at the lowest point at the stage of collection. If the preparation processes result to a deficit, the expenses will ultimately be charged to the citizens that are serviced by the municipal waste organisation (Zacho et al., 2018).

Bovea et al. (2016) proposed a general methodology for assessing the potential re-use of small WEEE, focusing on devices classified as household appliances. The study presented a first approach to the "preparation for re-use" strategy that the EU WEEE Directive (2012/19/EU) advocates. The case study covered a selective collection campaign of small household WEEE in Castellon de la Plana (Spain) from March to June 2015. The campaign was carried out in collaboration with a social enterprise which was authorised for the management of WEEE. The collection points were located in different educational centres located across the town. After assessing a sample of 87.7 kg (96 units) from the collected small household WEEE, it was calculated that 30.2% of that sample were redirected to recycling, 67.7% had a potential for re-use, and 2.1% could be re-used directly (Bovea et al., 2016).

3.3 Re-use operations in Austria

Austria has a well organised waste system that enables the collection of bulky waste from households, including bulky waste wood, household scrap metal (excluding packaging) and WEEE among others. In 2010, it was reported that 601,700 tonnes of re-usable items have been collected in Austria, which means approximately 72 kg per capita of formally collected bulky waste (BMLFUW, 2010). On top of that, Ramusch et al. (2015) investigated that additionally up to a further 12 kg per capita might have ended up as waste taken care of by the informal or second-hand sector, which exported them for re-use in nearby countries. This resulted to an estimation of approximately 100,000 tonnes of additional reusable items informally collected in Austria.

A widely publicised example of re-use in Austria refers to the Repair- and Service Center R.U.S.Z (Reparatur- und Servicezentrum R.U.S.Z) in Vienna. Founded in 1998 as a non-profit organisation, the centre pursues economical, environmental, and social outcomes, as documented by the recovery and repairing of about 8,000 used products per year, and the reintegration of long-term unemployed persons (Lechner and Reimann, 2015).

R.U.S.Z receives used products from different sources. The main ways to collect products is either through direct delivery to the store by the customer, or through a collection service at the customer's home. Acquisition quantities coming from both sources of supply can be actively influenced by a fee for the pick-up service or by efforts for advertisement and information. More than half of product acquisitions is through delivery by customers to the shop. Since 2012, R.U.S.Z charged a fee of EUR 9 for the pick-up service, compared with EUR 24 in the year 2011. The price reduction was due to the public support of the municipality of Vienna which subsidised the pickup service (Lechner and Reimann, 2015).

All products that are processed in R.U.S.Z can be sold relatively easy, as the demand is – on average – greater than the number of processed items. Finished re-used goods are not always sold instantly but there is a certain delay between the end of processing and the actual sale, so the time when a product is finished may not necessarily coincide with the demand for a product. Regarding the sale prices of re-used goods, a rule of thumb by R.U.S.Z is to charge one third of the price of a new product for the equivalent re-used one. In case of high quality refurbished products, in an "as-new" condition and those associated with a prestigious brand image, the sale price can be up to half of the price of a new product. Moreover, following legal requirements, R.U.S.Z can also offer a guarantee of one year for repaired products (Lechner and Reimann, 2015).

R.U.S.Z has been supported widely by recurring media campaigns, assisted by the municipality of Vienna. However, this case study also indicated that there was a lack of collaboration between manufacturers, retailers, and the re-use sector, despite the fact that national and regional legislation aimed at boosting preparation for re-use according to EU Directives. Large retailers in Austria take back customers' used white goods when selling a new one but they are not willing to cooperate with the R.U.S.Z by providing the collected items, as they fear that the re-used white goods could affect the volume of their own sales. According to R.U.S.Z, the retailers' superior market position in collecting used items was one of the main reasons for low supply of used products at the re-use centre (Lechner and Reimann, 2015).

In a study concerning the replacement, repair and re-use of mobile phones in Austria (Wieser and Tröger, 2018), a different perspective is presented on why supply of used goods might be shrinking. Austrian consumers are generally avoiding to dispose of their old phones. The authors suggest that offering warranties for used phones may be the most effective way for establishing a viable domestic re-use market. A functioning repair system and ease of disassembly of EOL products are essential components for a sustainable re-use sector. However, it is observed that users do not repair broken mobile phones, partly due to the high cost associated with repairs. Therefore, a combination of tax cuts for repair services and more information about the reparability of mobiles could encourage people to repair defective devices (Wieser and Tröger, 2018).

4. METHOD

A qualitative analysis of the different waste streams in two sorting facilities of the largest private recycling operator in Sweden was conducted for this study. The analysis aimed at assessing the type of waste streams and the quality of waste, to examine if the waste could have been re-used instead of recycled, and to assess the potential for re-use of the identified waste streams. The approach in this contribution followed previous studies on the analysis of re-use potential in municipal recycling centres in Sweden (Ljunggren Söderman et al., 2011; Hultén et al., 2018a; Hultén et al., 2018b) and Denmark (Zacho et al., 2018), however, without quantitatively analysing the samples due to confidentiality issues (business competition).

The scope of the study derives from the strategic ambition of Sweden to transition to a resource efficient low carbon economy and to sustain growth and jobs. In this effort all economic actors have particularly important roles to play, and besides the public waste management authorities a number of private actors are actively involved in recycling and re-use operations. Specifically, as the resource efficiency agenda is advancing within a circular economy paradigm, more and more private enterprises are requesting re-use solutions for their discarded equipment. Therefore, private actors primarily active in recycling so far, have been particularly keen to explore new ways to increase the re-use potential of EOL equipment.

To collect empirical evidence on the potential of re-use, on-site field investigations were conducted in two recycling centres of the largest private recycler company in Sweden (henceforth Recycling company A). The visits were complemented by ten interviews with relevant stakeholders in the recycling and re-use sector in Sweden, both public and private. The majority of stakeholders interviewed were connected to the Recycling company A, holding various positions, from branch manager to research and development coordinator. Further interviewees were identified through non-probability sampling method (Bryman, 2016), following suggestions or direct collaborating stakeholders of the company. The full list of interviewees is presented in Table 1. The examined recycling centres are located in Malmö and Kristianstad, both in the Scania region of South Sweden.

TABLE 1: List of interviews.

#	Stakeholder type	Interviewee role	Sector	Type
1	Recycling company A	Head of sustainability	Recycling	Private
2	Recycling company A	Branch manager (Stockholm)	Recycling	Private
3	Recycling company A	Branch Manager (Malmö)	Recycling	Private
4	Recycling company A	R&D project manager	Recycling	Private
5	Recycling company A	Branch manager (Kristianstad)	Recycling	Private
6	Recycling company A	Branch manager (Linköping)	Recycling	Private
7	EEE producer responsibility organisation (PRO)	Vice CEO - Business development	Recycling / Re-use	Private
8	Re-use municipal company	Work supervisor	Re-use	Public
9	Re-use municipal department	Project manager	Re-use	Public
10	Re-use company	Marketing director	Re-use	Private

The design of the semi-structured interviews purposefully included open-ended questions in order to capture the diversity of opinion among the different stakeholders in the company and the associated stakeholders outside the company. Although semi-structured interviews are sometimes criticized for lack of generalizability, they are beneficial at providing in-depth exploration of the subject of interest and for seeking new insights (Bryman, 2016). Individual interview guides were used, as the interviewees are engaged in different departments and positions in relation to potential re-use operations, both internally and externally. Due to geographic disparity of the interviewed stakeholders, the majority of interviews were conducted by telephone.

5. RESULTS AND DISCUSSION

5.1 Site visits at two recycling centres in South Sweden (Scania)

For the purposes of this study, two recycling centres belonging to the largest recycling company in Sweden were visited (in Malmö and Kristianstad), and their organisation and operations is presented in this section. The recycling centre in Malmö mainly handles iron, other non-ferrous metals, hazardous waste, electronics, paper and plastic. It also handles a small portion of wood waste and rubber. Waste is transported to the Malmö recycling centre from a variety of client companies, but also from other recycling centres across the country that belong to Recycling company A. Iron, aluminium and other metals are treated by cutting and packaging into smaller parts for shipping to smelting facilities in Sweden and abroad. Also hazardous waste, electronics, paper and plastic are sent on to other recycling facilities. So, the recycling centre acts solely as a sorting and logistics facility, while the actual recycling of materials happens elsewhere.

The recycling centre in Kristianstad differs from the company's other facilities, as it includes a municipal recycling centre within its premises. Private persons come to the recycling centre, as well as companies to dispose waste for recycling. The centre handles about 70,000 tonnes material per year, focusing on so-called alternative raw materials such as building materials, pressure-impregnated wood and garden waste. Building materials are recycled and pressure-treated wood is treated as hazardous waste. The garden waste is crushed and sieved and then composted to soil or turned to biofuel for heating plants. The recycling centre receives also EEE waste, which is then forwarded to the company's major recycling facility in Halmstad.

This recycling centre is collaborating with the social services of Kristianstad municipality, which drives a second-hand store in connection to the recycling centre. There, individuals can leave everything from electronics, furniture and household items to clothes, toys and books. Individuals have the option to dispose items and materials in a container for re-use in close proximity to containers for recycling. The items left for re-use are sorted and repaired if necessary, and then the items go on for sale in the store. Cooperation with the social services means that people with special needs and people in training can receive employment, which contributes to the social aspect of sustainability. The staff working there has shown great appreciation for this collaboration, and the job promotes their creative and social development. The second-hand store, which has a large number of visitors, has also become popular among the inhabitants of the municipality and in the rest of Scania region.

5.2 Analysis of interviews

Through the interviews with key staff of Recycling company A and other relevant external stakeholders, a number of common obstacles for re-use were identified, largely consistent with what has been mentioned in literature. Due to the current legal framework, re-use of certain products is not possible. When a product is classified as waste, it is not allowed to be removed from the receiving waste facility and thus not allowed to be re-used. However, it is possible to bring waste under the condition "preparation for re-use" which would allow a product to become re-usable under certain conditions, fulfilling a number of criteria according to EU Waste Directives. The classification of EOL products as waste can be avoided by introducing separate collection of the incoming items into reusable fractions –meaning they are not waste– as in the case of Swedish recycling centres with an embedded re-use section and the collection services of re-use centres in Flanders and Vienna.

A product's design can make re-use more difficult, especially if it is difficult to dismantle and repair the product (Vanegas et al., 2018). Transport and logistics can also prevent re-use. Many products are damaged during transportation (Cole et al., 2018). For these products it is difficult to estimate any potential for re-use as they are not intact upon arrival at the recycling centre, and effective re-use practices require upstream measures. This is especially true for white goods. Further, for re-use to be economically feasible and profitable there needs to be a certain volume of a specific product (or product group) and an efficient process to prepare it for re-use. Previous experiences with re-use of ad-hoc collected products that do not form economies of scale, as in the case of the municipal recycling centre in Denmark, showed that re-use is not profitable but can marginally cover the costs of preparation for re-use (Zacho et al., 2018).

Additionally, peoples' attitude to re-used products is considered a significant obstacle. Private individuals may be negatively inclined to re-use due to lack of information about the product and its functionality, or its potential hazardousness (Ylä-Mella et al., 2015). Moreover, it is quite common that people would throw away functional products due to the desire to upgrade their products and acquire the latest version available on the market, although this is not always the case as Wieser and Tröger (2018) illustrated in a case of mobile phones re-use in Austria.

An overarching obstacle to re-use is the additional workload and time it would take to prepare a product for re-use. The product must be sorted out, checked, cleaned, repaired (if needed), quality assured, transported and finally sold to a customer. The potential product opportunities for re-use identified through the interviews include: 1) building materials, 2) furniture, and 3) consumer electronics. There is potential to re-use whole white goods, or components in white goods, but this must be done upstream before the goods arrive at the company's premises. Ultimately, the potential for re-use is largely influenced by the type of facility, as it is common that different facilities receive and handle different types of waste.

For materials such as bricks and tiles, there are clear incentives for re-use instead for recycling. This is because the former leads to significantly reduced environmental impacts compared to the latter (Nußholz et al., 2019). Moreover, these materials are relatively more expensive compared to other building materials such as wood and plaster. Increased re-use of construction and demolition waste could be achieved in different ways, for instance through material exchanges linked to either construction and demolition companies or recycling companies. There are several existing platforms that offer such services, which are mainly independent initiatives not being part of construction and demolition companies or conventional recycling companies. Furthermore, focus should be placed on demolition practices that favour separate collection of building components such as windows and doors, which will effectively increase their re-use potential. However, this can be a costly endeavour that might not be feasible by just one recycling company. Policy support in this area could incentivise the uptake of more selective demolition in the future. Differentiated fees between sorted and unsorted construction and demolition waste disposal could rationalise and balance the higher cost of selective demolition, which today directs demolition actors to prefer conventional demolition practices. Sales of re-used building materials could become a part of the recycling company's business model, for example by cooperating with actors already active in this area (e.g. Malmö Återbyggdepå).

In the case of re-using furniture, transport and warehousing can be complicated and costly, and these consist major obstacles in re-use potential (Öhgren et al., 2019). Transportation and storage of furniture for re-use requires more space and needs to be handled more carefully than material just for recycling. Local solutions for the sale of re-used furniture is an environment-friendly and cost-effective alternative, compared to a national scheme where the furniture must be transported longer distances. This point was also illustrated in the case of the re-use centre in Vienna, where finished products for re-use would be redirected back to the market in irregular time intervals and be able to cover the local demand of the citizens of Vienna (Lechner and Reimann, 2015). A large amount of office furniture with high potential for re-use usually becomes available when companies relocate or restructure. Through collaboration with other actors, Recycling company A can find opportunities for re-using such furniture before they arrive at the recycling centres, thus avoiding the large and costly transportation to and from the recycling centres. Then, there is potential to cooperate with, for example, moving companies which also transport furniture in a safer manner, avoiding damage of the product. Consumers' preference to divert old furniture for re-use is relatively high, and being offered an easy way to dispose old furniture is considered of great importance.

A major obstacle to the re-use of EEE is the desire of consumers to buy new products instead of used (Ylä-Mella et al., 2015). Newly manufactured products can be relatively inexpensive, which makes it economically accessible (Watson et al., 2017). In order to be able to increase the re-use of EEE, it is important to have secure handling already from the collection stage, as these products can be damaged easily if not handled properly. The case study of selective collection of small household WEEE proves this point and indicates that the re-use potential can be up to 70% of the collected WEEE (Bovea et al., 2016). When WEEE arrive in the recycling centres, it is usually too late to restore or dismantle them, as it is often not economically desirable. In addition, there is uncertainty about the products' performance and safety. Used EEE may have been disposed due to electrical faults, which can lead to fire risk and danger for further use. One incentive for re-using electronics is to provide a warranty or quality label on the re-used product (Gåvertsson et al., 2018). In this way, the customer can feel safe because the product is guaranteed to work properly for a certain period of time after the purchase. This is something that companies working exclusively with re-use (repair and remanufacturing services, as in the case of R.U.S.Z in Vienna) can readily offer, but it might be more difficult for a recycling company to provide – let alone costly. However, there is still potential for salvaging components of damaged EEE that are still operational and can find a second life as spare parts on the market, provided prices can match the costs of operations (dismantling, cleaning and forwarding to the market) performed by the recycling company.

5.3 Actions for increasing re-use in private recycling centres

Recycling company A, the subject of analysis in this contribution, would firstly need to focus on re-use opportunities from waste received directly through industrial partners rather than what comes from its recycling centres, which usually exhibit unpredictable and heterogeneous waste flows. Thus, it can reap the readily available opportunities and work with customers to develop business agreements that favour re-use, already before the collection of EOL products. Materials sorted out for re-use from private individuals and other customers at recycling centres are of varying quality and often require further processing, which is labour intensive.

Additionally, Recycling company A could focus on products identified in its recycling centres by this study, including bricks and tiles, furniture and white goods, to the extent possible. EEE in the company's recycling centres are difficult to re-use, however, there is some potential for white goods. Cooperation with other companies is seen as a key action, for example with other private re-use and second-hand sales enterprises or the Swedish EEE Producer Responsibility Organisation (PRO) 'El-Kretsen', to enable the re-use of white goods. Establishing an on-line platform for sales of white goods is a step in the right direction. The most important issue about both furniture and EEE is that the handling of the products is done carefully.

It can be problematic for a recycling company to handle at the same time sales, repair and storage of products and materials for re-use, as it is not included in the company's current business model. However, Recycling company A could act as an intermediary in the re-use process, by sorting out the products and materials that could be re-used and forward them to other actors who are responsible for repair and sales of second-hand goods. This requires further development of the current business model of the company and its logistics solutions. Moreover, collaboration with diverse actors is required, while also finding mutual profitability within the network and the re-use process. One barrier is that it requires time and resources to develop new business models, therefore investments must be able to be repaid through the re-use process. The operations may initially run at a loss until more efficient processes and larger volumes are established. Another difficulty is that the process involves many steps and actors with different costs for both work and transport. The profit margin can therefore be small, if there is not high enough value remaining in the products at the secondary market. Some products received at recycling centres may have a low purchase price and therefore reflect a low value in the secondary market. So, it is not economically viable to repair and sell these products. Interestingly, our interview with the PRO revealed that in the case of white goods there may be more value in single components than in the entire product. This is due to the fact that the breakdown of a white good tends to be caused by single components. Access to components can therefore support repair activities, as newly manufactured spare parts tend to be expensive.

Special potential for re-use is available for materials that appear in large volumes, as this means a more predictable flow that can be managed more effectively. Company executives in the recycling sector generally believe that re-use can become profitable in the future, but changes in society's attitude to re-used products is required, as well as policies that make re-use more attractive. Ultimately, in line with increased environmental public awareness, re-use of products will increase in the future, which means a reduction of material flows to recycling companies. For this reason it is imperative to take advantage of the business opportunities that exist within re-use, or else run the risk of lower revenues in the future.

6. CONCLUSIONS

Within a circular economy paradigm, EU Member States strive to ascend the 'waste hierarchy' and retain materials and energy in the economy by re-using products as much as possible. Many EU members have achieved relatively high recycling rates but the next challenge in waste management is how to prevent waste, including promoting re-use of EOL products.

This contribution presented a comprehensive approach in the development of re-use operations at recycling centres in Sweden and potentially internationally. It showed that there is a great potential to collect and re-use more products that currently are only recycled. The potential for increased re-use is demonstrated both in municipal recycling centres (public), by analysing previous studies, but also in private recycling centres operated by large recycling companies in Sweden. Achieving higher re-use rates is not only a public responsibility, but private enterprises have also a critical role to play. Therefore, the focus in this study was on the private sector and on how a "traditional" recycling company can find opportunities to adopt a more circular business model and include re-use in its operations.

About a quarter of the total waste collected in recycling centres can be commercially or at least functionally re-used, resulting in significant environmental and social gains with inconclusive economic benefits. Product groups with the highest re-use potential in private recycling centres are building materials, furniture and EEE. However, taping on this potential, private enterprises are required to explore new types of collaborations both with other private actors and public authorities. Increased collaboration and prioritisation of suitable product groups and market opportunities could effectively increase re-use in the future and mitigate potential risks of reduced recycling business activities, due to waste prevention and re-use upscaling.

Consequently, a more integrated investigation would be beneficial to determine the conditions of such collaborative actions. Therefore, future research could expand on this study by quantifying the flows of re-usable products in more recycling centres (both public and private) and map out the re-use dynamics at a regional or even national scale. Furthermore, a wider market investigation in the potential market demand and supply of re-used equipment both domestically and internationally could enable a more comprehensive economic analysis.

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SUITABILITY INDEX ASSESSMENT FOR COLLECTION BIN ALLOCATION USING ANALYTICAL HIERARCHY PROCESS (AHP) CASCADED TO ARTIFICIAL NEURAL NETWORK (ANN)

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ABSTRACT

Municipal solid waste is an inevitable outcome of anthropogenic activities. Proper sustainable solid waste management is the need of the hour. In this study, a Suitability Index (S.I) has been determined which can measure the relative importance of a district with regard to its necessity or requirement of collection bins in comparison to other districts in a municipality. The S.I was computed using Analytical Hierarchy Process cascaded to Artificial Neural Network. Four criteria viz. Demographic, Social, Economic and Technical considerations and seven factors viz. Population Density (P.D), Street Width (S.W), Waste Generation Rate (W.G.R), Income Group Distribution (I.G.D), Average Minimum Distance between the bins (MIN.D), Available Number of Bins (A.N.B) and Cost of Waste Bins (C.W.B) were considered for developing the model. Available Number of Bins was found to have the highest impact on the model followed by C.W.B, W.G.R, MIN D., I.G.D, P.D, and S.W. This index will particularly help developing countries with resource constraint and unskilled labor force in Solid Waste Management. It will help such countries to easily locate districts in urgent need of collection bins with an easily available set of data and will help in increasing collection efficiency.

1. INTRODUCTION

Creation of Solid Waste is an inevitable part of human activities, especially in the urban crowd. Municipal Solid Waste Management is one of the most pressing problems faced by most of the cities around the globe especially the developing countries. In India, the municipal agencies spend 5-25% of their budget on Solid Waste Management (SWM). But unfortunately, high capital investment in the SWM sector is not necessarily leading to improvements in the quality of service (National Solid Waste Association of India (NSWAI), 2008). Almost 85% of the total expenditure in SWM is spent on collection (Ghose, Dikshit, & Sharma, 2006). Similar reports of huge cost investment in the collection of solid waste have been made by other researchers (Ghiani, Laganà, Manni, & Triki, 2012; Ghiani, Manni, Manni, & Toraldo, 2014; González-Torre, Adenso-Díaz, & Ruiz-Torres, 2003; Kao & Lin, 2002) and municipalities as well.

Location-allocation modeling is the method of optimizing the location of centers or facilities and allocating consumers or demands to those centers (Valeo, Baetz, & Tsanis, 2002). In spite of being one of the significant factors in the successful achievement of SWM, location-allocation problem of sitting storage depots have achieved very less importance around the globe. When determining the type and size of these bins during system planning and design, the solid waste estimation and allocation are not adequately addressed. The vast majority of the studies mainly investigated the vehicular transportation of waste from bins to the disposal sites. Although these processes require heavy vehicles and machinery, the efficiency of these depends upon the number, location, type and size of bins as well as the frequency of waste removal required (Vijay, Gupta, Kalamdhad, & Devotta, 2005). Parrot et al. (Parrot, Sotamenou, & Dia, 2009) noted that the spatial distribution of the garbage accumulation points (GAPs) inside towns often does not take the needs of all local residents into account in terms of quantities of waste produced and distance from their dwelling. They also found that when the average distance to the closest GB is long, there is generally a low percentage (37.4) of people who dump their waste in them. The long-distance explains why households dispose

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of domestic waste in open areas. Similar concerns about the non-convenient location of the GAPs are expressed by Zia & Devadas (Zia & Devadas, 2008).

Various location-allocation modeling has been widely used for facility location planning in both the public and private sectors (Beaumont, 1987). Three different approaches have been attempted by researchers for addressing the location-allocation modeling viz. Geographical Information System (GIS) (El-Hallaq & Mosabeh, 2019; Erfani, Danesh, Karrabi, & Shad, 2017; Kao & Lin, 2002; Khan & Samadder, 2016; Nithya, Velumani, & Senthil Kumar, 2012; Vijay et al., 2005; Vu, Ng, & Bolingbroke, 2018), integer programming (Coutinho-Rodrigues, Tralhão, & Alçada-Almeida, 2012; Ghiani et al., 2012, 2014; Rathore, Sarmah, & Singh, 2019) and algorithm (Di Felice, 2014; Hemmelmayr, Doerner, Hartl, & Vigo, 2013). Some of them have tried to club GIS with the other two to suggest best possible locations of bin allocation (Arribas, Blazquez, & Lamas, 2010; Erfani, Danesh, Karrabi, Shad, & Nemati, 2018; Karadimas & Loumos, 2008; Tralhão, Coutinho-Rodrigues, & Alçada-Almeida, 2010). A detailed literature study of the Location-allocation modeling in SWM has been conducted by Purkayastha et al. (Purkayastha, Majumder, & Chakrabarti, 2015). The use of multi-criteria decision making (MCDM) approaches for location-allocation solutions in solid waste (Mondal, Speier, & Weichgrebe, 2019) has been extremely limited. Moreover, there has been no study on the location-allocation of collection bin using Analytical Hierarchy Process (AHP) cascaded to Artificial Neural Network (ANN).

In the present study, an index known as the Suitability Index (S.I.) has been developed with the application of AHP and ANN to address the location-allocation problem in SWM. The developed index can identify the area or district which is in urgent need of collection bins. This index will be most beneficial for SWM of developing countries who have resource constraints and are operating majorly with unskilled manpower. This index can also prioritize the area or district on the basis of its urgency in terms of collection bin requirement. Therefore SI can aid in providing collection bins to areas with immediate requirements under limited resource constraint conditions.

2. METHODS USED

The main objective of this model is to develop a Suitability Index (S.I.) for location-allocation of the collection bin. The S.I. is a comparative scale which can measure the relative importance of an area or district (hereafter known as a ward) with regard to its necessity or requirement of collection bins in comparison to other wards in a municipality. The study utilized two methods to develop this S.I: Multi-criteria decision making (MCDM) and Artificial Neural Network (ANN).

2.1 Multi-criteria decision making (MCDM)

MCDM methods have found a wide application in decision-making objectives over a wide decade of time. MCDM method is applied to compute the priority or weight of importance of the factors correlated to the objective of the study. There are various type MCDM techniques like Weighted Sum Method (WSM), Weighted Product Method (WPM), Simple Additive Weighting (SAW), and Analytical Hierarchy Process (AHP). In the present study, the AHP method has been used as because in this study relative importance as well as both qualitative and quantitative parameters have been considered. The other MCDM technique such as WSM and SAW doesn't incorporate pair wise comparison or relative weights of importance of criterias and alternatives whereas AHP incorporates it. Moreover they only consider quantitave variables, whereas AHP can include both qualitative and quantitative variables (Ghosh, Chakraborty, Saha, Majumder, & Pal, 2016).

2.2 Artificial Neural Network (ANN)

ANN is a computational model composed of many elements (known as neurons) connected by a variable weight. It was Warren McCulloch, a neurophysiologist, and Walter Pitts, a young mathematician, who in 1943 proposed the first ANN model known as McCulloch-Pitts (MP) Model. In the MP model, the activation (x) is given by a weighted sum of its M input values (a_i) and a bias term (θ). The output signal (s) is typically a nonlinear function f(x) of the activation value x. the objective function of the MP model or basic ANN model is given by:

$$s = f\left(\sum_{i=1}^{M} w_i a_i - \theta\right) \quad (1)$$

3. METHODOLOGY

The objective of the present study is to establish a Suitability Index which will help in determining the priority of locating waste collection bin in a geographical area. Suitability Index (S.I) is the ratio of beneficiary factors to non-beneficiary factors. Suitability Index can be mathematically represented according to equation (2).

$$S.I = \frac{\sum(W\ X\ Beneficiary\ factors)}{\sum(W\ X\ Non\text{-}Beneficiary\ factors)} \quad (2)$$

W = Weightage of the importance factor. This weight of importance is determined by MCDM techniques.

The "Beneficiary factors" are those which increase the probability of a place being suitable for collection bin allocation i.e. with the increase in the value of these factors the S.I value also increases thus increasing the suitability of a place for collection bin allocation. For e.g. population density, waste generation rate, etc.

The "Non-Beneficiary factors" are those which decrease the probability of a place being suitable for collection bin allocation i.e. with the increase in the value of these factors the S.I value decreases thus decreasing the suitability of a place for the collection bin allocation and vice versa. For e.g. cost of the bin, the number of bins already available in an area, etc.

3.1 Weightage Computation of factors using AHP technique

The AHP method requires three steps: (1) Selection of criteria (2) Selection of alternatives (3) Application of aggregation method (Ghosh et al., 2016).

3.1.1 Selection of criteria

For the present study, the weightage of all the beneficiary and non-beneficiary parameters needs to be determined. Henceforth all the beneficiary and non-beneficiary parameters were considered as alternatives. In this study, the weight of importance of the alternatives was established with respect to some criteria established from the expert survey and literature survey. In order to derive this index, a critical literature survey was conducted to find out the factors and criteria which were most significant to the placement of collection bins. The questionnaire adopted for the expert survey was prepared using Google form and is provided in supplementary materials (Annex-A). The expert survey was conducted through face to face interview and Social networking Media (Researchgate, Gmail, LinkedIn, and Facebook). Four criteria which were selected from the extensive literature study were found to be significant according to the expert study as well and are: Demographic Considerations (D1), Social Considerations (S1), Economic Considerations (E1) and Technical Considerations (T1). Four of these criteria are very inclusive in nature on a broader aspect and include all factors affecting bin allocation. According to both literature and expert study four of these criteria were suggested and no other criteria was suggested. Since all these criteria received significant importance according to both expert and literature study, four of them were considered in bin allocation problem in this study.

3.1.2 Selection of alternatives

The alternatives which were found to be most important in deciding locations for collection bin allocations based on expert and literature surveys were: Population Density (P.D), Street Width (S.W), Waste Generation Rate (W.G.R), Income Group Distribution (I.G.D), Cost of Waste Bin (C.W.B), Available Number of Bins (A.N.B) and Minimum Distance between Bins (MIN. D) (Table 1). These alternatives were further divided into beneficiary and non-beneficiary parameters. In this model Population Density (P.D), Street Width (S.W), Waste Generation Rate (W.G.R), Income Group Distribution (I.G.D) and Minimum Distance between Bins (MIN. D) were considered as beneficiary factors i.e. with increase in value of each of these factors the S.I value increased and vice versa. In this model Cost of Waste Bin (C.W.B) and Available Number of Bins (A.N.B) were considered as non-beneficiary factors i.e. with the increase in the value of each of these factors the S.I value decreased.

3.1.3 Application of aggregation method

Both the expert survey and literature survey was carried out further to estimate the importance of criteria and alternatives over each other. An expert survey was conducted on a set of a questionnaire asking to rank the alternatives with respect to each criterion on a scale of 1 to 9, 1 being very weak and 9 being extremely strong. The ranking was provided in the manner that the alternative under criteria which obtained the highest score was ranked 1, the alternative with the second-highest score was ranked 2 and so on. The rank of the criteria and alternative were established based on the literature survey and expert survey. In case if two or more alternatives under the same criteria obtained the same rating then in such cases all those alternatives were given the same rating and the in-between ranks were skipped The same methodology was adopted for other criteria too.

A 4 x 4 matrix was developed to find out the weightage of criteria.

$$c = \{n \, X \, n\} \tag{3}$$

Where, $\{n\} = \{D1, S1, E1, T1\} \in R$, where R is the set of real numbers.

Similarly, the alternatives are compared with each other based on their importance over each other according to each of the criteria 'n':

$$A = \{f_i \, X \, f_j\} \tag{4}$$

Where, $\{f_i\} = \{PD, SW, WGR, IGD, MIND, CWB, ANB\} \in R$, where R is the set of real numbers.

The hierarchy of decision making is shown in Figure 1.

In the case of AHP generally, the Saaty scale is used which was proposed by Saaty in the year of 1980 [10]. The scale utilized either even or odd number to represent the importance of the criteria and alternatives with respect to each other in the Pairwise Comparison Matrix (PCM). For intermediate importance, the rating in between the evens or odds is utilized. But still, there is a lot of confusion regarding what can be used for the representation of a high difference of importance and minor difference of importance between two alternative or criteria.

That is why, in the present study, we use the rank of the criteria and alternatives based on their magnitude or qualitative ratings and then ration of rank of the criteria/alternative compared and the rank of the other criteria/alternative with which it is being compared was found out (the rank is assigned in such a way that the relationship of the criteria with the decision objective can be reflected). The ratio is then reversed to give the exact difference of importance coherent to decision objective.

The direct use of rank to estimate the importance will ensure uniformity and remove the confusion involving the rating that can be given to depict two different levels of importance that exist between two different criteria or alternative.

3.2 Formulation of Suitability Index (S.I.)

The final S.I formula was formulated as:

$$S.I = \frac{(W_{PD} X \, PD^{\text{⋰}}) + (W_{SW} X \, SW^{\text{⋰}}) + (W_{WGR} X \, WGR^{\text{⋰}}) + (W_{IGD} X \, IGD^{\text{⋰}}) + (W_{MIND} X \, MIND^{\text{⋰}})}{(W_{CWB} X \, CWB^{\text{⋰}}) + (W_{ANB} X \, ANB^{\text{⋰}})} \tag{5}$$

Where,

PD⋰=Normalised value of PD	W_{SW}=Weightage of SW
WGR⋰=Normalised value of WGR	W_{IGD}=Weightage of IGD
MIND⋰=Normalised value of MIND	W_{CWB}=Weightage of CWB
ANB⋰=Normalised value of ANB	W_{PD}=Weightage of PD
SW⋰=Normalised value of SW	W_{WGR}=Weightage of WGR
IGD⋰=Normalised value of IGD	W_{MIND}=Weightage of MIND
CWB⋰=Normalised value of CWB	W_{ANB}=Weightage of ANB

TABLE 1: Table showing detail description of the Criteria and sub-criteria used in the AHP method for the present study.

Criteria	Alternatives	Description	Mathematical formulations	Literature reference
Demographic considerations: The demography of an area is very important while designing any waste management technology for an area. Suggested Waste management technology varies from place to place depending on the demography of that place.	Population density (P.D)	Population density is an important parameter in deciding the requirement of total number of bins in an area.	$P.D\ (per\ km^2) = \frac{Population}{Area}$	[3, 4, 6, 8, 12, 17, 19, 21, 22, 28–34]
Social considerations: Suggested Waste management technology will depend upon the social situation of the locality. For e.g. the waste characteristic will be different depending upon the income level of most of the people living in the society.	Street Width (S.W)	Width of the street where the collection bin needs to be provided decides which size of bin can be provided in an area as because each size of bin are collected by a particular collection vehicle which needs a particular width of street for collecting wastes from that bin.	-	[8, 12, 17, 19, 21, 28–31]
Economic considerations: The municipal budget is always a constrain in Solid waste management. Hence to take care of the economic viability we need to consider economic optimization and feasibility before deciding the waste management option.	Waste Generation Rate (W.G.R)	Waste Generation Rate decides the total quantity of bins needed to provide collection bin facilities for all the wastes generated in a locality.	$W.G.R\ (m^3) = [pcwg\ X\ P\ x\ \rho]$ - $A.N.B$ $Pcwg$ = Per capita waste generation $\left(\frac{\frac{kg}{person}}{day}\right)$ P = *Total population of the ward* ρ = *Density of waste* $\left(\frac{kg}{m^3}\right)$ $A.N.B$ = *Available Number of Bin*	[3, 4, 6, 12, 17, 19, 21, 22, 28, 29, 31, 32]
Technical Considerations: It includes the present status of infrastructure available and provided by the municipality.	Income Group Distribution (I.G.D)	Income level of a section of society decides the amount and kind of waste generated and the collection bin facilities needed to account all the waste.	High Income Group (Rating-3) Medium Income Group (Rating-2) Low Income Group (Rating-1)	[8, 30]
	Cost of Waste Bin (C.W.B)	In general their needs to be a compromise between economic viability and resource requirement. The number of extra bins needed in an area in addition to the already provided bins amount to a cost and Cost of Waste Bin (C.W.B) depicts that value.	$C.W.B = M - \sum_{i=1}^{n} N_i\, X C_i$ M = Maximum cost of bins if a ward is provided with actual required number of bins of any one type in "n" Number of types so that the total volume of waste produced in that ward is collected in bins and no waste is left unattended or uncollected. n = Number of types of bins in an municipality N_i = Available Number of Bins of type i (where i=1,2,3,..........,n) C_i (Currency unit as per the country) = Cost of Each Bin of type i	[17]
	Available Number of Bin (A.N.B)	Available Number of Bin (A.N.B) is the quantity of bins already provided to an area. This factor decides whether any further bins should be provided in a area in addition to the already provided bins to address resource constraint situation.	$A.N.B\ (m^3) = \sum_{i=1}^{n} N_i\, X\, V_i$ N_i = Available Number of Bins of type i (where i=1,2,3,..........,n) V_i (m^3) = Volume of Bin of type i (where i=1,2,3,..........,n)	[3, 4, 17]
	Minimum Distance between Bins (MIN. D)	Minimum Distance between Bins denotes the average minimum distance between the bins an area. This factor represents the frequency of bin placement in an area, so that places in an area with low bin frequencies can be identified.	$MIN.D\ (meter) = \frac{d_{min} + d_{max}}{2}$ d_{min} = Minimum distance between bins in a ward d_{max} = Maximum distance between bins in a ward	[3, 4, 6, 8, 19, 22, 33, 34]

3.3 ANN Model Formulation for Suitability Index (S.I.)

To predict S.I. Artificial Neural Network (ANN) was used. GMDH Shell software was used for carrying out the ANN-based prediction. GMDH stands for "Group Method of Data Handling". The main idea of GMDH is to build an analytical function in a feedforward network based on a quadratic node transfer function whose coefficients are obtained using a regression technique (Farlow, 1981). It was first proposed in 1966 by a Russian cyberneticist, A.G. Ivakhnenko.

The inputs of the model were the random normalised values of Population Density (P.D), Street Width (S.W), Waste Generation Rate (W.G.R), Income Group Distribution (I.G.D), Cost of Waste Bin (C.W.B), Available Number of Bins (A.N.B) and Minimum Distance between Bins (MIN. D). The input function of the ANN model is shown in Eqn. 4. The output was the Suitability Index (S.I). The model predicted the S.I value for 1000 data. Then a case study was conducted on Agartala Municipality to predict its S.I for the entire municipality, which has been presented in Section 4.3.

3.4 Description of the case study area

Agartala the capital city of Tripura is one of the eight North-Eastern states of India. It is situated along 23° 45'- 23° 55' N latitude and 91°15'- 91°20' E longitude, in the flood plains of the Haora River. The city has been an important border-trading town with trading linkages with Bangladesh. The National Highway (NH)-44 connects Agartala with Assam. The climate of Agartala is tropical monsoon type. The average rainfall of the city is about 220 cm.

The solid waste management of Agartala city is carried out by Agartala Municipal Corporation (AMC) established in 1871. Agartala Municipal area is divided into 4 zones and 35 wards. The overall population of Agartala Municipal area is 4, 70,190 with an overall area of 61.718 sq.km. A map of the Agartala Municipal Corporation (AMC) is shown in Figure 2.

The wastes generated in Agartala are of two types- Municipal solid Waste and Biomedical Waste. In our study we are considering only Municipal Solid Waste (MSW). The major waste generating sources of MSW are household

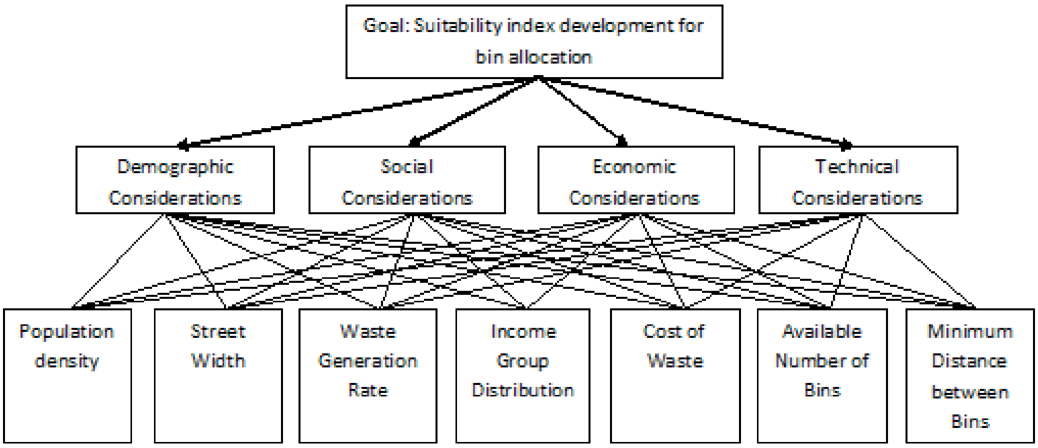

FIGURE 1: Hierarchy Structure of the AHP Modely.

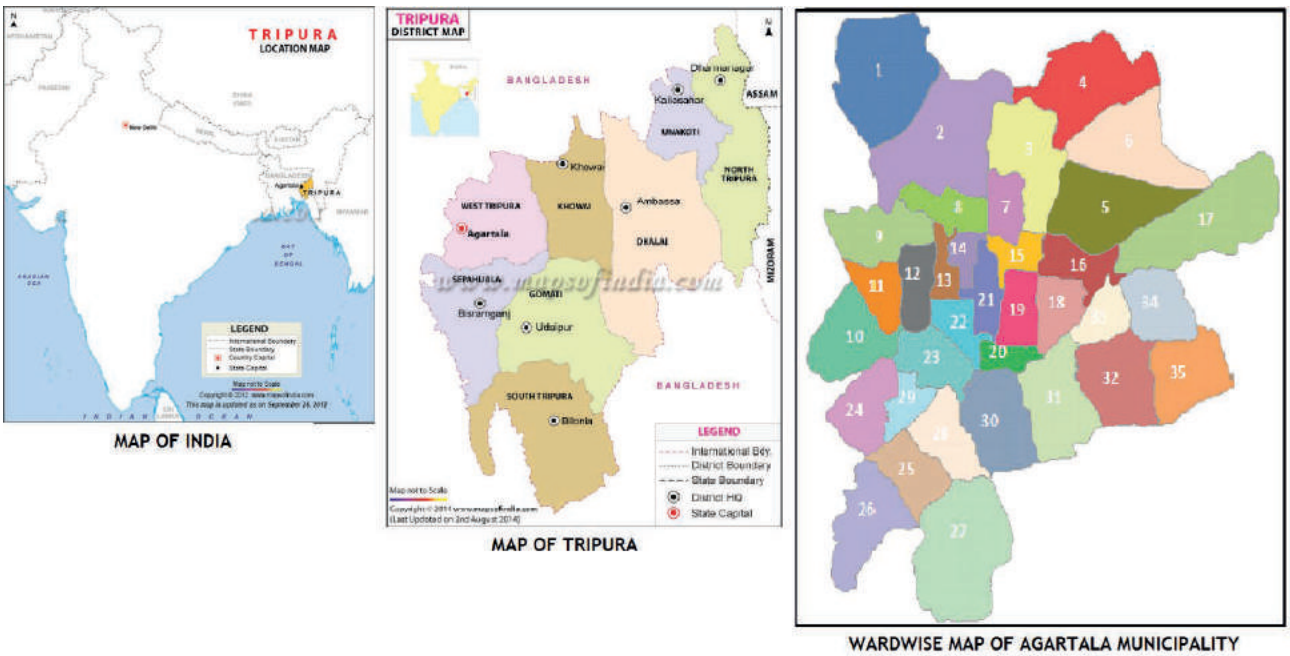

FIGURE 2: Map of the study area.

wastes, institutional wastes, market (vegetable and fish) wastes, street litters and drain silts.

Agartala Municipal Corporation (AMC) has placed more than 447 number of medium (1.1 m^3) (Figure 3a) and 55 large sizes (4.5 m^3) (Figure 3b) bins/containers by the side of major roads and in market and commercial areas.

The data table for formulating the model for AMC area is shown below in Table 2. The data of Population density (P.D), Available number of Bins (A.N.B) and Cost of Waste Bins (C.W.B) has been provided by AMC. The data on Income Group Distribution (I.G.D) isn't available with AMC, so all the wards have been assumed to fall in Middle Income Group based on the economic scenario of the state. The Street Width (S.W) data also couldn't be provided by the AMC so based on the survey of the expert views in AMC the average Street Width was assumed as 3.0 meter. Another reason behind assuming the Street Width as 3.0 metre is that the vehicles used to collect the 4.5 m^3 and 1.1 m^3 bins can't operate through roads with width less than 3.0 metre as per AMC. The cost of each 4.5 m^3 bin and 1.1 m^3 bin is Rs. 65000 and Rs. 30000 respectively as per AMC information. All data related to ward-wise Available number of bins (separately 1.1 m^3 and 4.5 m^3 bin) has been provided by AMC. The per capita waste generation rate per person has been considered as 500 gms/day as per Ministry of Environment and Forest (MOEF) and Central Public Health and Environmental Engineering Organisation (CPHEEO) guidelines.

4. RESULTS AND DISCUSSION

This section has been divided into three parts, viz., results of the AHP method to estimate the weights of importance of the parameters, results from the GMDH model to establish the suitability index and lastly the case study model of Agartala Municipality.

4.1 Weightage computation by AHP

Table 3 shows the rank of the criterias based of expert and literature survey. The rank of the factors with respect to each criteria are presented in Table 4. Technical considerations (T1) and Available Number of Bins (A.N.B) were observed to be the most important criteria and factor respectively. The weightages which were finally obtained using Analytical Hierarchy Process (AHP) are tabulated in Table 5.

4.2 ANN Model of S.I

The S.I model predicted by GMDH shell (Data Science version) software showed the correlation coefficient of the predicted model as 0.994646 which indicates that the predicted model is of good quality. The accuracy of the model is shown in Figure 4a. Figure 4b shows the correlation between S.I and the seven input factors P.D, S.W, W.G.R, I.G.D, C.W.B, AN.B and MIN.D. The highest positively and negatively correlated factor to S.I is W.G.R and A.N.B respectively

The 3D- plot between A.N.B, C.W.B and S.I (Figure 5a) shows that with the decrease in value of A.N.B & C.W.B, the S.I value increases, showing the highest peak at a value of less than 0.05 for A.N.B and less than 0.1 for C.W.B. The peak is at an area with value of A.N.B less than C.W.B because the weightage of A.N.B ($W_{A.N.B}$=0.24012) is greater than weightage of W.G.R ($W_{C.W.B}$=0.18043), so the control of A.N.B on the model is greater than C.W.B. Since C.W.B and A.N.B are non-beneficiary criterias the S.I is constricted to one corner. Both ANB and CWB has a very dominant effect on S.I model indicating that both availability and cost decided the suitability of a location for bin allocation followed by other attributes i.e. W.G.R, MIN.D, I.G.D, P.D and S.W.

The 3D- plot between A.N.B, W.G.R and S.I (Figure 5b) shows that with the constant value of A.N.B & increase in value of W.G.R, the S.I value increases, showing the highest peak at a value of less than 0.05 for A.N.B and greater than or equal to 0.5 for W.G.R. This behaviour of the graph is because of the reason that A.N.B is a non-beneficiary criteria and W.G.R is a beneficiary criteria due to which decrease in value of A.N.B increase the S.I and increase in value of W.G.R increases the S.I. This indicates that the area with higher waste generation rate and lower available number of bins will have higher S.I values.

The 3D- plot between A.N.B, MIN.D and S.I (Figure 5c) shows that at higher values the MIN.D ($W_{MIN.D}$=0.0968) the S.I value spikes up at a lower value of A.N.B. This indicates that if the minimum distance between collection bins in an area is more the suitability of that area for bin allocation increases. It was also observed that at extremely lower value of both MIN.D and A.N.B, the S.I value substantially increa-

FIGURE 3: Pictures of 2-Types of containers used by AMC for secondary collection. (a) 4.5 cu. Meter Bin - (b) 1.1 cu. Meter Bin.

TABLE 2: Data of AMC for computing S.I from the model.

Ward no.	Name	P.D.	S.W.	W.G.R	I.G.D	C.W.B.	A.N.B	MIN.D
1	Barjala	2621.25	3.00	71.00	2	1416818	8.80	277
2	Lichubagan	4341.56	3.00	61.79	2	1338791	22.30	738
3	Kunjaban	6932.91	3.00	44.95	2	879600	34.40	403
4	Chanmari	3413.73	3.00	66.41	2	1291582	13.20	383
5	Indranagar	5330.69	3.00	72.38	2	1512264	6.70	500
6	Nandan nagar	4203.40	3.00	65.13	2	1256836	14.30	199
7	Abhoynagar	17261.41	3.00	57.47	2	1163327	25.50	192
8	Radhanagar	15304.40	3.00	73.66	2	1489336	6.60	576
9	Ranjit nagar	8203.06	3.00	64.14	2	1345291	15.60	303
10	Raj nagar	7347.33	3.00	80.76	2	1682973	2.20	1434
11	West joynagar	21851.74	3.00	47.86	2	1074527	33.50	250
12	Ramnagar	20162.76	3.00	59.95	2	1173182	21.00	251
13	West krishnanagar	20065.60	3.00	49.20	2	879918	30.90	288
14	Krishnanagar	29679.67	3.00	35.28	2	673709	45.50	270
15	Dimsagar/ banamalipur	17250.91	3.00	40.54	2	788273	39.95	333
16	Dhaleshwar	11596.49	3.00	60.53	2	1189000	17.70	329
17	Khayerpur	3805.25	3.00	57.28	2	1100473	22.10	300
18	Shibnagar	9858.45	3.00	42.62	2	815918	36.60	260
19	West shibnagar	16920.22	3.00	59.78	2	1197491	18.85	210
20	Town pratapgar	18757.87	3.00	39.04	2	718473	41.00	325
21	Shantipara	11729.63	3.00	0.00	2	0	79.00	241
22	Melarmath	12704.85	3.00	34.77	2	544127	43.10	214
23	Bardowali	11992.06	3.00	74.02	2	1499182	4.40	1000
24	Bhotto pukur	9310.80	3.00	66.29	2	1403900	13.40	550
25	Arundhuti nagar	13741.97	3.00	53.81	2	1121318	24.50	623
26	South badharghat	4152.72	3.00	77.14	2	1584327	1.10	1434
27	Sidhi ashram	5080.59	3.00	45.53	2	779964	35.30	365
28	Rajlakhie nagar	5265.18	3.00	45.79	2	729245	31.90	214
29	Arundhuti nagar	11538.46	3.00	77.33	2	1589509	2.20	1434
30	Pratapgar/ west pratapgar	9616.30	3.00	67.01	2	1423564	12.30	350
31	East pratapgar	9616.69	3.00	79.35	2	1644627	0.00	1434
32	Jogendranagar	8863.54	3.00	71.49	2	1430264	7.70	333
33	North jogendranagar	9739.36	3.00	68.64	2	1410236	8.90	300
34	Aralia	4693.64	3.00	63.06	2	1200273	18.70	400
35	East jogendranagar	6450.13	3.00	78.55	2	1622864	0.00	1434

ses which means that if both the available number of bins and minimum distance between collection bins in an area is less than the collection bins need relocation and hence the S.I value of the area increases.

The 3D- plot between A.N.B, I.G.D ($W_{I.G.D}$=0.087) and S.I (Figure 5d) depicts that areas with more of high income group have highest values of S.I but the weightage of this lower have low impact on the S.I model. The 3D- plot between A.N.B, P.D ($W_{P.D}$=0.083) and S.I (Figure 5e) shows that the S.I. values slowly increases with a increase in value of P.D. i.e. the suitability of an area for collection bin allocation increases with increase in value of population density but increases at very slow rate indicating that P.D. has a very minimal influence on the S.I. model. The 3D- plot between A.N.B, S.W. ($W_{S.W}$=0.051) and S.I (Figure 5f) represents that the value of S.I for an area increases with the decrease in street width. This might be due to the fact that S.W has the

TABLE 3: Rank of the criterias based literature survey and expert survey.

Criteria	Abbreviation	Score
Demographic Consideration	D1	2
Social Consideration	S1	3
Economic Consideration	E1	4
Technical Consideration	T1	1

TABLE 4: Rank of the factors based literature survey and expert survey.

	D1	S1	E1	T1
P.D	1	3	5	6
S.W	4	6	7	5
W.G.R	2	1	4	4
I.G.D	3	2	3	7
C.W.B	4	3	1	2
A.N.B	7	3	2	1
MIN.D	6	7	6	3

TABLE 5: Weightage computation by AHP.

Factors	Abbreviation	AHP Weightage
Population Density	P.D	0.14980
Street Width	S.W	0.07578
Waste Generation Rate	W.G.R	0.15991
Income Group Density	I.G.D	0.10047
Cost of Waste Bin	C.W.B	0.18043
Available Number of Bin	A.N.B	0.24012
Minimum Distance between bins	MIN.D	0.09349

lowest weightage and hence least or negligible impact on the model especially in comparison to A.N.B, which has the highest impact on the model. Therefore might be due to this reason the S.I shows increased value with decrease in S.W.

4.3 Suitability Index for the case study

The S.I model obtained above was then applied to AMC area for finding out the ward-wise Suitability Index. The model predicted the Suitability Index (S.I) value for all the 35 wards (Figure 6).

According to the S.I Model the first five wards with highest values of S.I are Ward No. 29, 31, 23, 10 and 35 with S.I values of 2.234060498, 2.083174661, 2.028085959, 1.91108766 and 1.859772522 respectively.

The suitability index value for each and every location inside the Agartala Municipality Area can be obtained from

Postprocessed results	Model fit	Predictions
Number of observations	800	200
Max. negative error	-1.39189	-2.26909
Max. positive error	1.0747	1.25886
Mean absolute error (MAE)	0.109201	0.144452
Root mean square error (RMSE)	0.178652	0.282605
Residual sum	-2.30782E-12	-1.33941
Standard deviation of residuals	0.178652	0.282525
Coefficient of determination (R^2)	0.998214	0.989293
Correlation	0.999107	0.994646

(a)

Name	Corr.	Bars
Suitability index	1.000	
A.N.B	-0.310	
C.W.B.	-0.250	
W.G.R	0.110	
MIN.D	0.098	
I.G.D	0.087	
P.D.	0.083	
S.W.	0.051	

(b)

Variable	P.D.	S.W.	W.G.R	I.G.D	C.W.B.	A.N.B	MIN.D	Suitability index
Numeric values	1000	1000	1000	1000	1000	1000	1000	1000
Text values	0	0	0	0	0	0	0	0
Missing values	0	0	0	0	0	0	0	0
Unique values	1000	1000	1000	1000	1000	1000	1000	1000
Zero values	0	0	0	0	0	0	0	0
Most frequent								
Min. value	0.000208657	0.00352632	0.000976689	0.000370506	0.00442318	0.000199995	0.00015408	0.296348
Max. value	0.997402	0.999579	0.999919	0.999481	0.999749	0.999224	0.999228	102.936
Median	0.497878	0.508169	0.508874	0.514606	0.482223	0.500705	0.515461	1.39735
Mean value	0.496112	0.504421	0.505048	0.509758	0.482955	0.493987	0.500926	2.08774
Std. deviation	0.287683	0.291957	0.288516	0.297021	0.282179	0.293303	0.288156	3.97525
2σ outliers	0	0	0	0	0	0	0	14
3σ outliers	0	0	0	0	0	0	0	7
4σ outliers	0	0	0	0	0	0	0	4

(c)

FIGURE 4: (a) Accuracy of the Global model Predicted by GMDH Shell; (b) Correlation of the 7-factors with Sutability Index (S.I); (c) Statistics of the Global model Predicted by GMDH Shell.

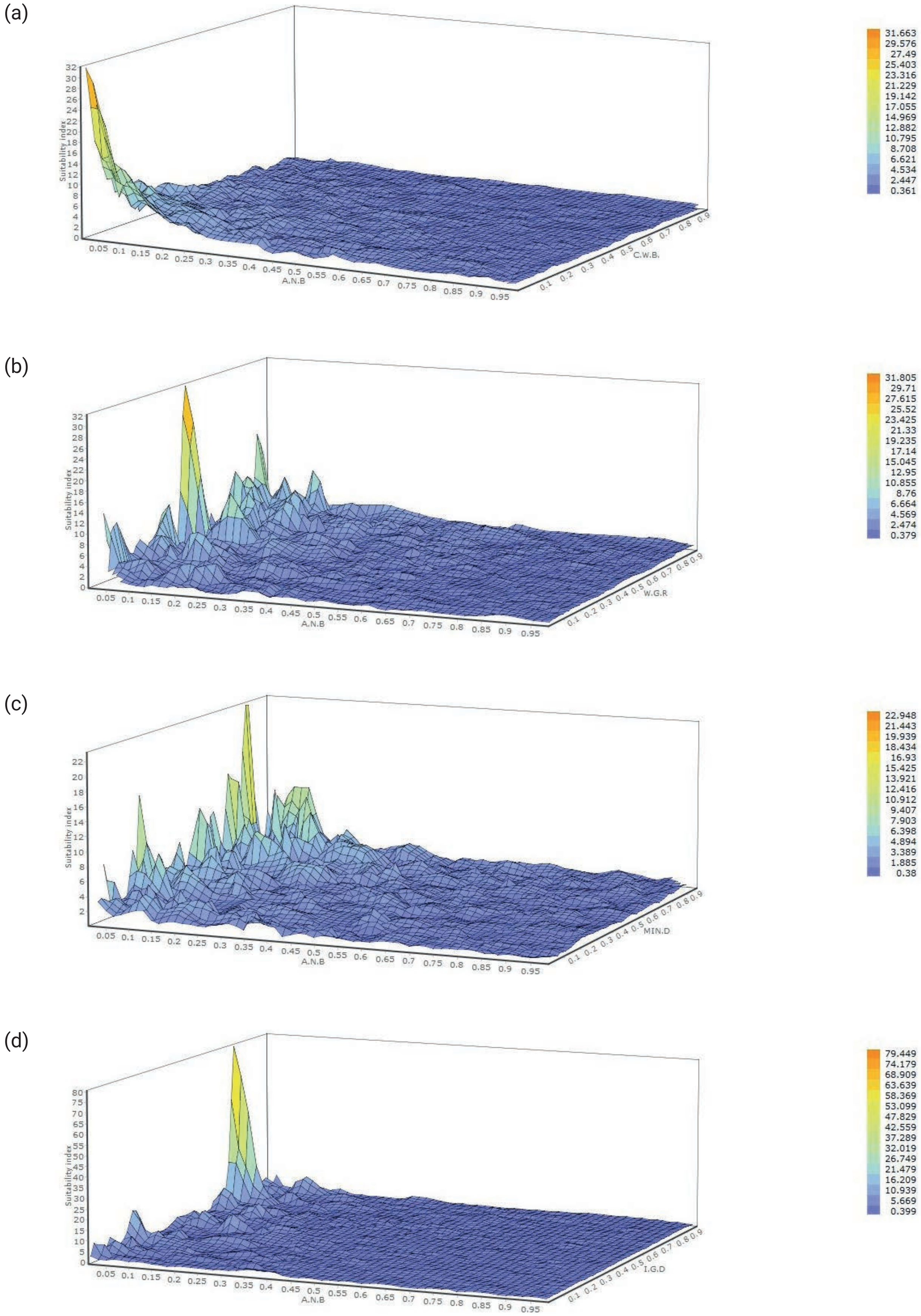

FIGURE 5a-d: (a) 3D-Plot between A.N.B, C.W.B and S.I; (b) 3D-Plot between A.N.B, W.G.R and S.I; (c) 3D-Plot between A.N.B, MIN.D and S.I; (d) 3D-Plot between A.N.B, I.G.D and S.I.

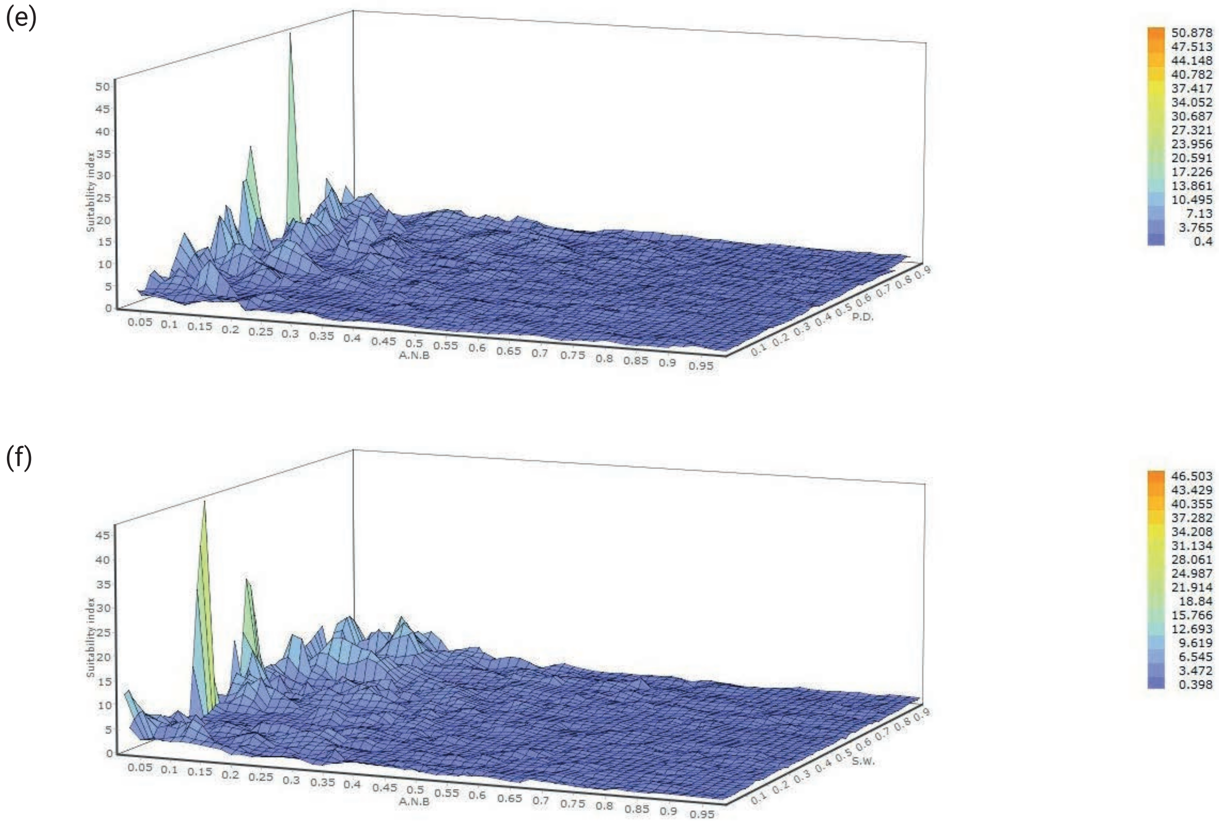

FIGURE 5e-f: (e) 3D-Plot between A.N.B, P.D and S.I; (f) 3D-Plot between A.N.B, S.W and S.I.

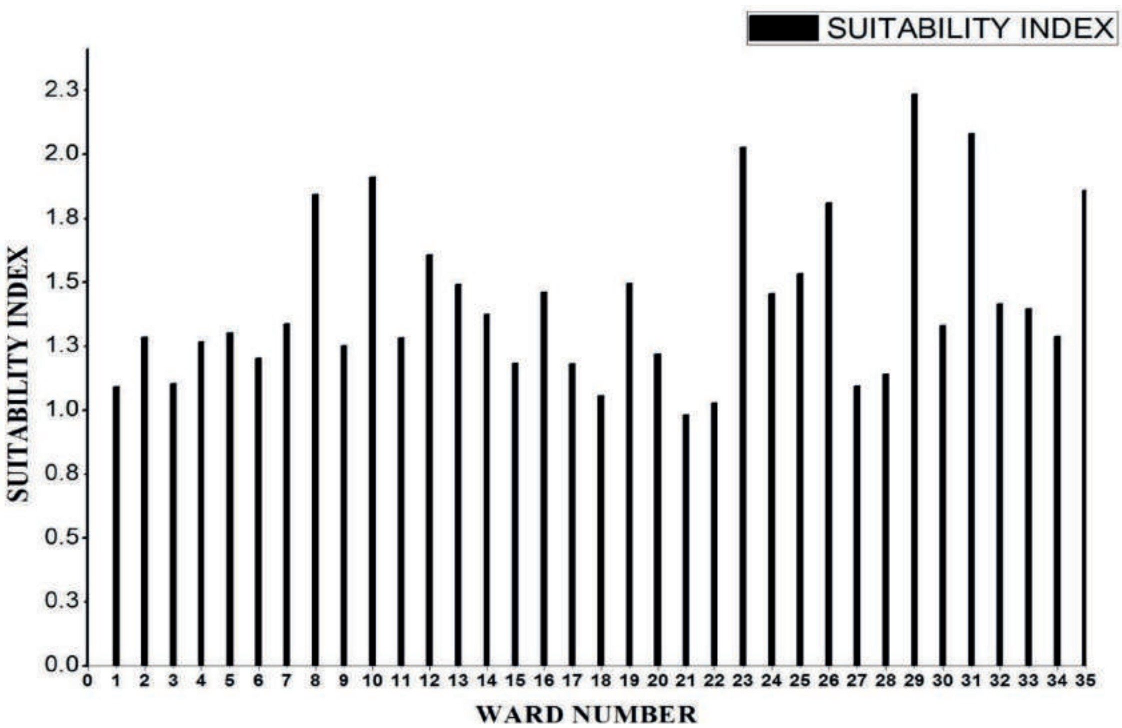

FIGURE 6: Ward-wise Suitability Index.

the contour map shown in Figure 7, which has been made using Surfer 12. Using this map, the areas in most urgent need of collection bins can be found out and the same can be provided with collection bins at the earliest. Subsequently the areas with low S.I values can be provided with collection bin facilities at the earliest convenience.

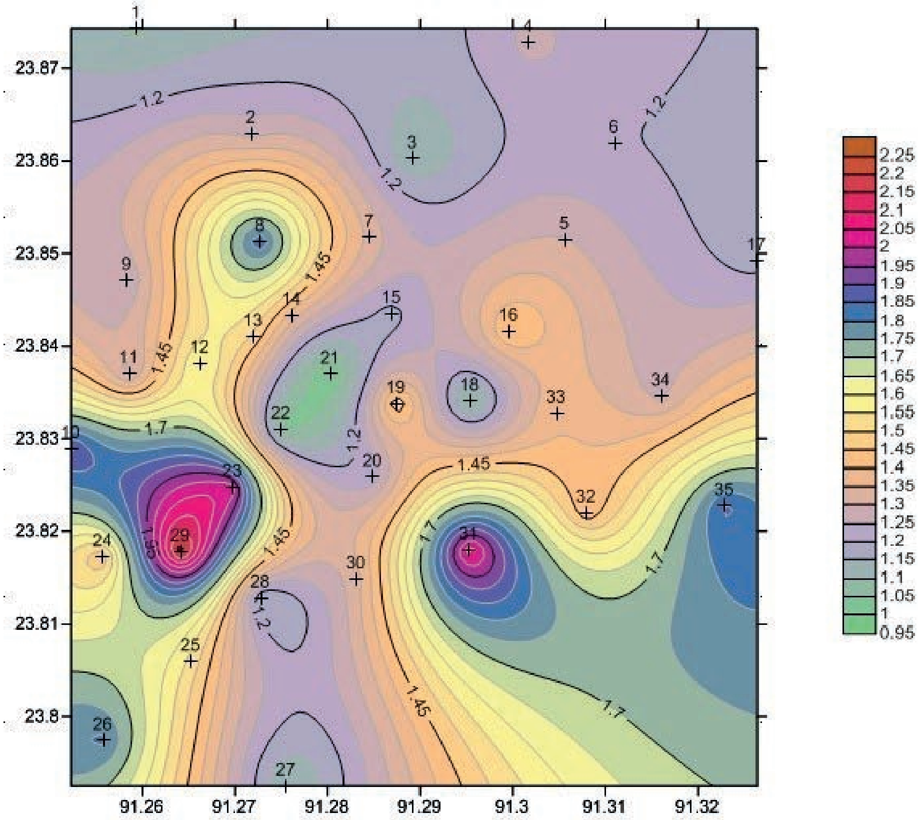

FIGURE 7: Contour Map of Suitability Index (X-axis: Longitude, Y-axis: Latitude).

5. CONCLUSIONS

From the study conducted in this research it is quite clear that the existing location of collection bins are uneven with many wards provided with absolutely no bin. Also it has been observed that there is absolutely no relation between numbers of bins in a ward and its population density. These problems are due to manual placement of bins with absolutely no use of any optimization technique. An optimization technique will help distribute the bins evenly along the wards at point where waste generation is occurring.

This work focuses on formulation and implementation of an innovative Suitability Index by using Analytical Hierarchy Process (AHP) and Artificial Neural Network (ANN). The S.I was found to depend on seven factors which were grouped under beneficiary and non-beneficiary factors. Population Density (P.D), Street Width (S.W), Waste Generation Rate (W.G.R), Income Group Distribution (I.G.D) and Average Minimum Distance between the bins (MIN D.) are beneficiary factors and Available Number of Bins (A.N.B) and Cost of Waste Bins (C.W.B) are non-beneficiary factors. The factor Available Number of Bins (A.N.B) was found to have the highest impact on the model followed by C.W.B, W.G.R, MIN D., I.G.D, P.D and S.W.

The case study conducted in Agartala Municipal area using this model showed that Ward No. 29, 31, 23, 10 and 35 are the first five wards with high Suitability Index value. These wards should be provided with collection bin facilities at the earliest. Using the contour map (Figure 7), the S.I. value at each and every location inside the Agartala Municipality can be obtained with known latitude and longitude.

This index will particularly help the developing countries with resource constraint and unskilled labor involvement in Solid Waste Management to easily locate areas/wards/districts needing most urgently collection bins with an easily available set of data and help increase the collection efficiency. The data related to the seven factors incorporated in this model for computation of Suitability Index are generally easily available with all Government bodies and hence the practical applicability of this Suitability Index is very high, easy and convenient.

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OPTIMIZED ROUTING OF TRUCKS FOR INSTITUTIONAL SOLID WASTE COLLECTION IN KUMASI, GHANA

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ABSTRACT
Solid waste collection constitutes 60-80% of the total solid waste management cost. Reduction of solid waste collection cost can be achieved through route optimization in a geographic information system (GIS) environment. The purpose of this study was to generate optimized routes for solid waste collection on Kwame Nkrumah University of Science and Technology campus in Kumasi, Ghana. The study modelled the existing routes for a complete collection cycle using travel time criteria and generated optimized routes for same using an ArcGIS software. Validation of the optimized outcome (travel distance and travel time) was done by subjecting the solid waste collection trucks to the optimized routes. The results from the study showed significant reduction in total travel time from 1,000.75 mins to 855.70 mins for existing and optimized routes respectively, translating into saving of 14.5%. Total travel distance significantly reduced from 367.30 km to 334.20 km for existing and optimized routes respectively, representing saving of 9.0%. Significant savings in travel time and travel distance have implications on reduction of fuel and maintenance cost of institutional solid waste collection trucks. The results indicate that the application of GIS-based route optimization in solid waste collection can provide significant improvement in reduction of operating cost.

1. INTRODUCTION

The rapid increase in human population, switch from traditional agricultural methods to industrialization and technological advancement have resulted in increased generation of solid waste. Management of solid waste is a major challenge, especially to developing countries partly due to lack of effective and adequate management systems. Solid waste management is a discipline that involves processes including; control of the waste generated, collection, processing, reuse and recovery and finally its disposal (Diaz et al., 2005; Tchobanoglous et al., 1993). Collection of waste is recognized as one of the most essential stages in the solid waste management process (Coffey and Coad, 2010). Efficient waste collection involves the regular collection, cleanup and transportation of solid waste to treatment or disposal sites. The process is designed and operated in an integrated way such that all aspects are selected taking into consideration: (1) Type of collection service production; (2) Type of collection system and equipment used as well as associated labour; (3) Analysis of collection systems; and (4) General methodology in collection route setting (Tchobanoglous et al., 1993).

The cost of solid waste collection is reported by previous studies to constitute about 60-80% of the total budget of solid waste management (Ansari and Pakrou, 2015; Beliën et al., 2014; Sulemana et al., 2018). According to Li et al. (2014) and Tavares et al. (2009), the huge expenditure on solid waste collection is warranted from the high cost of fuel and maintenance since the collection process involves extensive use of trucks. There is a necessity to cut down collection cost by practitioners and one of the most effective means indicated by previous studies is through route optimization. Routing involves scheduling and defining routes travelled by trucks during the solid waste collection process (Sulemana et al., 2018). Poor and expensive collection systems can come about when routes are selected without using scientific or technological methods (Tavares et al.,

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2009). Previously, the determination of routes for collection trucks was left to the drivers' discretion (Beliën et al., 2014). Such a state has often engendered remarkably high solid waste collection cost. Therefore, there is the need to scientifically generate optimized routes to cut down time and distance travelled by trucks and operating cost.

Route optimization involves scheduling collection trucks to visit all waste collection points in a defined path that reduces the total travel cost through reduction in fuel consumption and operating time (Chalkias and Lasaridi, 2009). This can effectively be achieved through significant savings in travel time and distance. Several studies have been conducted on the use of geographic information system (GIS) applications as a tool for municipal solid waste management (Ristic et al., 2015; Sulemana et al., 2019; Zsigraiova et al., 2013). GIS technology is a widely used decision support system which provides an advanced modelling framework for decision makers in order to analyze and simulate various spatial problems (Chalkias and Lasaridi, 2009). Various studies have been done optimizing solid waste collection in local authorities within GIS environment with dearth information on institutional setting. This study therefore attempts to answer the question: How does optimized routes contribute to reduction of operating cost through savings in travel time and fuel consumption? In this regard, this study sought to optimize nineteen routes for solid waste collection on the campus of a selected university in developing country setting. Implementation of the outcome of route optimization can significantly contribute to operating cost reduction in solid waste collection.

2. METHODOLOGY

2.1 Description of study area

The study was conducted at Kwame Nkrumah University of Science and Technology (KNUST) campus, Kumasi, situated in the Oforikrom Municipal Assembly of the Ashanti Region of Ghana (Figure 1). KNUST is a public educational institution established in 1950. It is situated approximately on a sixteen square-kilometer sub-urban area and located around Latitude 6°41'5.67"N and Longitude 1°34'13.87"W. The institution has a current student population of 56,403, teaching staff population of 1,074 and non-teaching staff population of 2,578. The campus is divided into a faculty area (housing the six colleges), a residential area (constituting the halls, hostels and staff bungalows) and commercial area. The collection of MSW on KNUST campus is managed by the Environmental Quality Unit of the university.

The solid waste collection systems practiced in KNUST include pegged bins collection, small bins collection and communal bins collection with bin sizes of 120/240L, 240L and 12m^3 respectively. The pegged and small bins collections are done using compactor trucks (12 tons capacity) while the communal bins are collected using skip trucks (12m^3 capacity). The pegged and small bins collections service five operational sites (routes) each in a cycle (i.e. P_1-P_5 & S_1-S_5 respectively) whereas the communal bins service nine operational sites in a cycle (i.e. C_1-C_9). The university has four compactors and three skip trucks with two of each truck type functioning at the time of data collection. Collection of MSW on KNUST campus from the

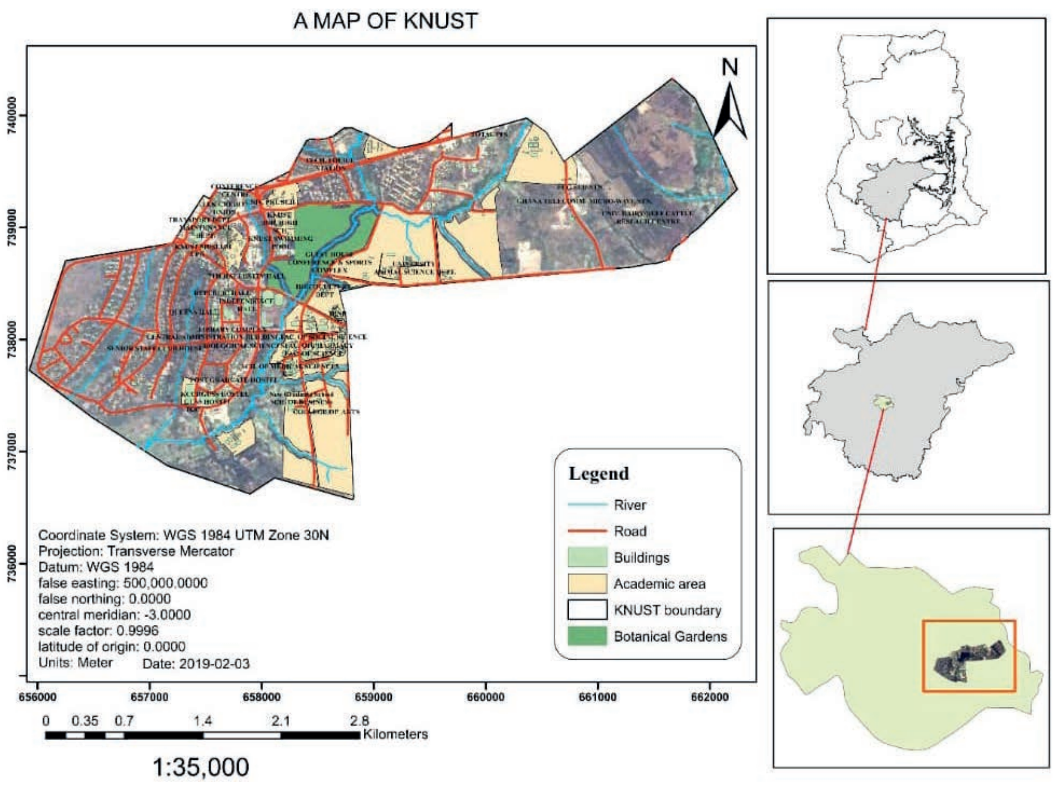

FIGURE 1: Map of KNUST campus showing road networks.

faculty, residential and commercial areas is done six days in a week (from Mondays to Saturdays). The MSW are collected without any segregation at source of generation. During collection, drivers are assigned to specific orders (collection points) and the drivers assigned to specified collection sites apply their discretion on the routes used. Each compactor truck has a collection crew, consisting of a driver and about six accompanying janitors whiles the skip trucks are operated by two people, i.e. the driver and his janitor.

2.2 System requirements and design for optimized routing

System requirements for the study were grouped into functional and non-functional requirements. Functional requirements constituted the input elements of the optimization system. These included the orders (collection points), depots (landfill site and transport yard) and road network dataset. Road network dataset consisted of a compilation of the shapefiles of all roads within the study area. The non-spatial road attributes were established through field observation and measurement. The non-functional requirements played an auxiliary role in the optimization system. These were the operational, technical and transitional aspects of the study that were essential in performing the analysis as indicated by O'Connor (2013). A computer with Microsoft Windows 8.1 operating system served the technical role. Other specifications of the computer included a 2.0 GHz processor and 8 GB RAM for boosted performance. The technical services of a 10.5 version ESRI ArcGIS software equipped with the latest features for the establishment of both existing and optimal route models was used.

A thorough understanding of the operation of the ArcGIS software extension was required to structure the components precisely and to accurately design, construct and execute the network analysis using the Vehicle Routing Problem (VRP) solver. The non-functional variables (transitional data) used in the study construction were in geodatabase format; compactible with the 10.5 version of ArcGIS software. Microsoft Excel and Word 2016 formats constituted the construction of tables and guidance documents for non-functional transitional purposes. The entire system was designed from analyzing the requirements and structured to accommodate the system's principal functional and non-functional components. The VRP solver of ArcGIS software used functional requirement inputs and parameters of travel time and distance to set up and construct optimized routes with optimal sequencing (Figure 2). Additional layers such as traffic lights, right turn, round-about, speed rams, pedestrian crossings and one-way roads were inputted to estimate time delays and restrictions on the roads. With all factors accounted for, optimized routes were generated accordingly. Route accuracy evaluation was carried out by plotting generated routes on Open Street Map and any new parameter identified and necessary adjustments re-entered into the optimization system.

2.3 Mapping and data collection

Detailed spatial information was required to generate the optimized routes. The spatial information comprised of the geographical background of the study area and the

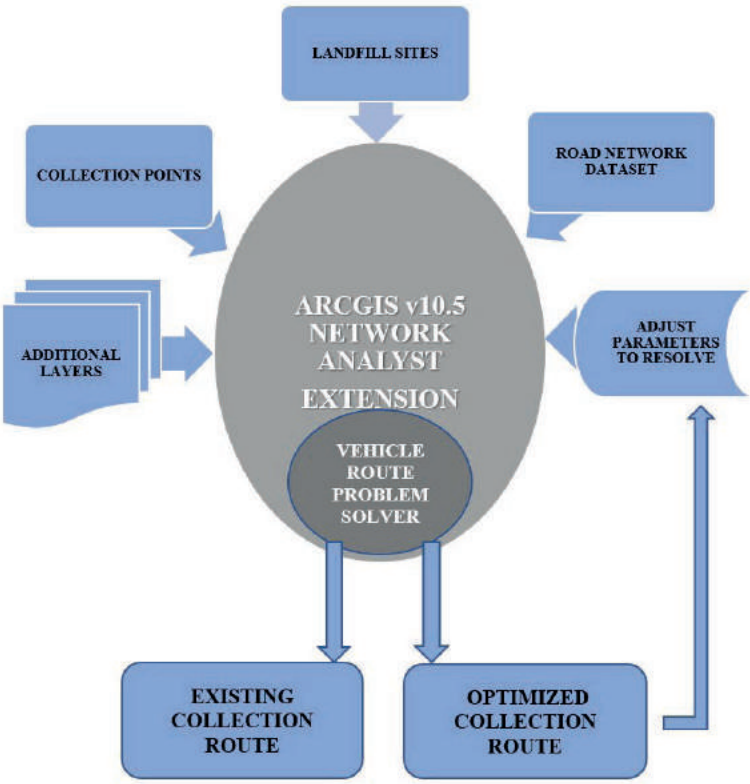

FIGURE 2: System design for route optimization in GIS environment.

spatial data related to the waste collection procedure. The static and dynamic data for the existing collection system were obtained from the KNUST Environmental Quality Unit. This consisted of: the population density; road network and attributes; position, number and type of the waste collection bins; routing system used by trucks; capacity and other characteristics of the collection trucks; and the geographic location, boarders and other characteristics of KNUST relevant to waste collection. The following data were generated for the optimization of collection routes: the boundary, detailed plan, road networks and road class information (traffic volume and road restrictions) of study area. Satellite image of KNUST was derived from Google Earth and the collection points of waste were obtained from field work using a GPS device. The schedule for waste collection in all the operational sites for the existing collection routes were obtained from the KNUST Environmental Quality Unit.

To optimize the collection routes, ESRI ArcGIS was used to generate a spatial geodatabase. This ensured congeniality with data obtained from the waste management section of the institution and access to many network analysis routines available from the software as reported by Chalkias & Lasaridi (2009). The road network's background spatial data for existing waste collection routes were obtained and updated. Non-spatial (attribute) data considered include road name, type, slope, collection time and average speed of vehicles. The road attributes such as traffic marks, topographic condition and spatial restrictions (turn restrictions) were noted and used to efficiently model conditions similar to that of the real-world road network conditions. The spatial database used are summarized in Table 1.

TABLE 1: Summary of the spatial database used.

Spatial Data	Type	Geometry
Road Network	Vector	Line
Waste Bins	Vector	Point
KNUST Urban plan	Vector	Polygon
Existing Collection Routes	Vector	Line
Street Address	Tabular	-
Road Network Attributes and Restrictions	Tabular	-
Satellite image of KNUST	Raster	-

2.4 Optimization and validation of optimized waste collection routes

Collection points and landfill sites' coordinates were taken using GPS device in a routine day-to-day collection process in the study area. With all variables in place, existing routes of the study area were generated on the digital map. A network dataset of roads was constructed to ensure that all streets were connected at junctions. Time delay variables such as traffic lights, speed rams, turns and pedestrian crossings were inputted for a better estimation of any impedance encountered during the collection process. Table 2 presents the attributes of the roads, containers, vehicles and delay parameters. These delay variables were allocated appropriate units in second(s). The route optimization process was based on travel time but travel distance was also estimated. These travel time and travel distance were measured in "seconds" (sec) and "kilometers" (km) respectively in the VRP Solver interface of the ArcGIS software. The VRP Solver constructed routes with several iterations made until optimized collection routes were identified. Comparisons of the identified optimal routes were made against the existing routes and tested on the ground to determine the credibility of the routes generated.

2.5 Data analysis

Data obtained from the existing and optimized situations were compared. Data in the form of time spent and distance travelled were measured for both situations. The data measured were tested using a paired sample t-test at a significance level of 5%. From the comparison, the difference in savings were expressed in terms of percentage.

3. RESULTS AND DISCUSSION

3.1 Existing and optimized routes for solid waste collection

When the solid waste collection trucks were subjected to the optimized routes, reductions in travel distance and travel time were observed in the operational sites of the study area. Of the nineteen routes optimized, one randomly selected route, each from small, pegged and communal bins collection services (S_3, P_2 and C_8 respectively) are presented in detail in Figures 3-5 and the rest have been included in the supplementary material. In the operational site S_3 for instance, travel distance and travel time reduced from 19.70 to 16.40 km and 67.40 to 44.97 min respectively (Figure 3). These converted into savings of 16.8% and 33.5% for travel distance and travel time respectively. The operational site P_2 recorded reductions from 23.00 to 18.50 km

TABLE 2: Determination of attributes of roads, containers, delays and vehicles.

Item	Attributes	Determination
Roads	Names, junctions, left turns, right turns and round-about	Information obtained from field visits with support from the Environmental Quality Unit of the University
Containers	Type, volume, location and lifting frequency	Information obtained from field visits with support from the Environmental Quality Unit of the University
Vehicles	Type, capacity, containers serviced and routes used	Information of the attributes were obtained from the Environmental Quality Unit of the University
Delays and impediments	Traffic light, peed rumps, right-turns, round-about, one-way traffic, etc.	Information of the attributes were obtained from field visits

and 76.20 to 58.15 min for travel distance and travel time respectively (Figure 4), translating into respective savings of 19.6% and 23.7%. At operational site C_8, reductions for travel distance and travel time were recorded as 23.00 to 20.00 km and 48.62 to 44.20 min (Figure 5), translating into savings of 13.0% and 9.1%, respectively.

The optimized routes generated were the shortest possible routes in terms of travel time criteria. This resulted in reduced travel times with savings recorded in all operational sites for the three collection systems (Table 3). Similar findings were reported by Sulemana et al. (2019) when they studied into the effects of optimal routing on travel distance, travel time and fuel consumption of waste collection trucks in three selected local authorities in Ghana. The optimization system considered road network impedances and restrictions such as traffic light, right turn, round-about, speed rams, pedestrian crossings and one-way road to generate the optimized routes which were time dependent. Although time criterion was used for the optimization, travel distances reduced in the collection systems except communal bins where optimized distances were greater than the existing distances. In such places, the quickest route, taking road impedances and restrictions into consideration were longer in terms of distance. Generally, travel distance reduced when trucks used the optimized routes.

The findings of this study are consistent with previous studies that applied GIS-based route optimization. Apaydin & Gonullu (2007) used route optimization to collect solid waste in Trabzon, Turkey and achieved reductions of 4-59% for distance and 14-65% for time. GIS was also used by Zsigraiova et al. (2013) to define new collection schedule with reductions of 62% for the total spent time, 43% for the fuel consumption and 40% for emitted pollutants and total cost savings of 57% per year. This indicates that the application of route optimization has positive implications on fuel consumption of trucks and environmental conservation. Other previous studies have also reported reduced travel distance and travel time through

TABLE 3: Travel distance and travel time recorded on the three collection systems.

Collection Systems	Travel Distance (km)			Travel Time (min)		
	Existing Routes	Optimized Routes	Savings	Existing Routes	Optimized Routes	Savings
Pegged Bins	100.20	86.30	13.90	318.45	262.20	56.25
Smaller Bins	104.30	84.30	20.00	326.00	259.20	66.80
Communal Bins	162.80	163.60	-0.80	356.30	334.30	22.00
All Systems	367.30	334.20	33.10	1,000.75	855.70	145.05

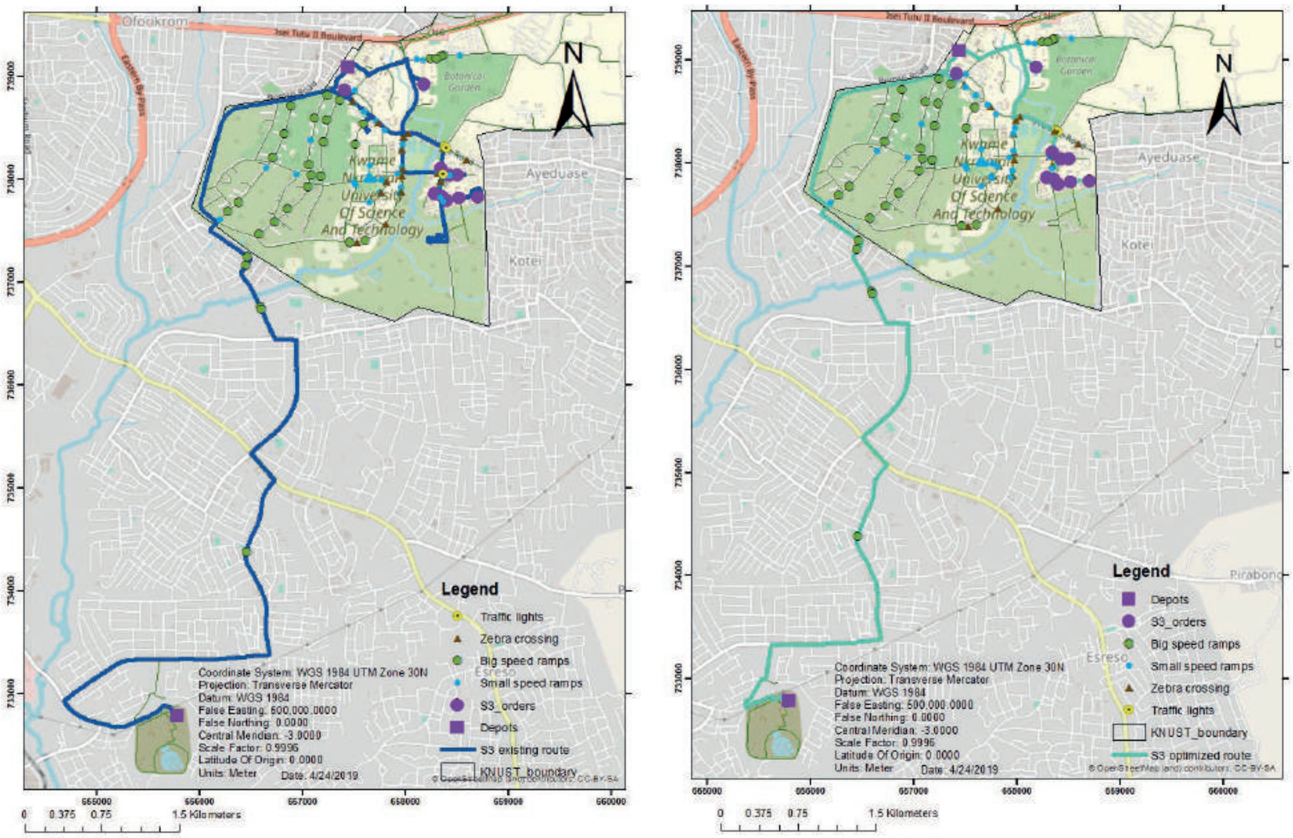

FIGURE 3: Existing and optimized routes for site S_3.

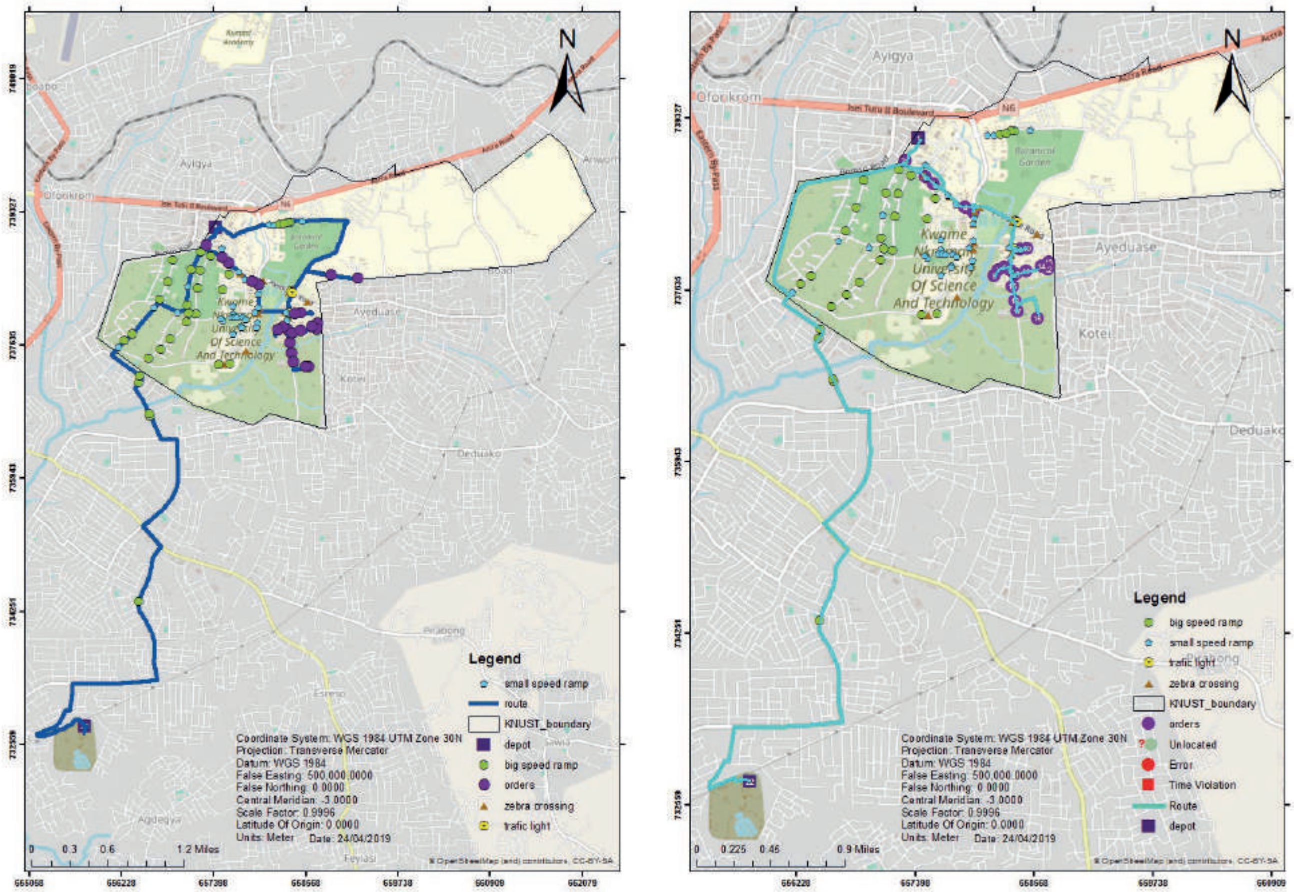

FIGURE 4: Existing and optimized routes for site P_2.

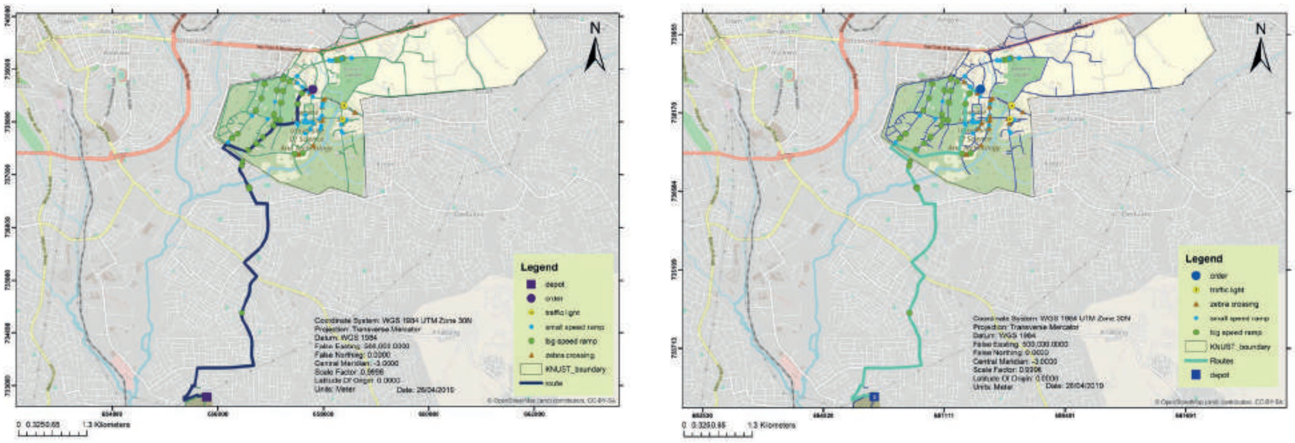

FIGURE 5: Existing and optimized routes for site C_8.

route optimization (Knobe et al., 2015; Sulemana et al., 2018; Velumani, 2013).

3.2 Travel distance on existing and optimized routes

Pegged bins and small bins recorded relative reductions in travel distance for the optimized routes by 13.90 km and 20.00 km respectively with communal bins collection system recording an increase in travel distance for optimized routes by 0.80 km (Figure 6). When all collection systems were combined, travel distance for optimized routes relatively reduced by 33.10 km, translating into saving of 9.0%. Significant reduction in travel distance was recorded in most of the operational sites although the system optimization was done based on travel time criteria. Comparison of optimized distance with existing distance showed significant differences for pegged bins ($p = 0.018$), small bins ($p = 0.041$) and all collection systems combined ($p = 0.009$), but not communal bins ($p = 0.834$) as displayed in Tables 4-7.

Significant reductions in travel distance can be attribut-

TABLE 4: Paired samples t-test for travel distance and time for all collection systems.

		Paired Differences					t	df	Sig. (2-tailed)
		Mean	Std. Deviation	Std. Error Mean	95% Confidence Interval				
					Lower	Upper			
Pair 1	Existing distance – optimized distance	2.0821	3.3226	.7623	.4807	3.6835	2.732	18	.009
Pair 2	Existing time-optimized time	429.8420	423.0970	97.0650	225.9160	633.7680	4.428	18	.000

t = t-statistic value, df = degrees of freedom, sig. = p-value, std. = standard

ed to the effective sequencing of service orders by the GIS-based route optimization system. This generated optimized routes with reduced distances as indicated by O'Connor (2013) and Sulemana et al. (2019) in their studies into route optimization using GIS-based approaches. This finding is supported by a study undertaken by Chalkias and Lasaridi (2009) which applied GIS-based optimization system to reduce travel distance, saving 5.5% in the Municipality of Nikea, Athens, Greece. Malakahmad et al. (2014) further used route optimization to obtain a 22.0% length minimization in routes. Travel distance not being significant (Table 7) in three sites of the communal collection systems (C_2, C_3 and C_6) serves as a testament to further indicate that travel distance is not a good criterion to effectively predict the operating cost of solid waste collection. This resulted from the fact that trucks spent comparatively shorter time on longer optimized routes in terms of distance due to high impedances and time delay constraints on shorter distances. These delay constraints include: speed ramps, number of turns, zebra crossings, traffic flows, etc. It is on this basis that travel distance does not prove a viable criterion for route optimization as reported by Kinobe et al. (2015) and Sulemana et al. (2019).

3.3 Travel time on existing and optimized routes

Pegged bins, communal bins, small bins and all collection systems combined recorded relative reduction each in travel time for the optimized routes by 56.25, 22.00, 66.80 and 145.05 min respectively (Figure 7). Significant reduction in travel time was recorded in each of the collection systems in the study area: pegged bins ($p = 0.031$), communal bins ($p = 0.003$), small bins ($p = 0.024$) and all collection systems ($p < 0.001$) when optimized travel time was compared with existing travel time (Tables 4-7). The system took into consideration the shortest distance to orders, taking account of road network impedances and time constraints. Significant differences in travel time were recorded for all collection systems (smaller bins, pegged bins and communal bins) when the trucks were subjected to the optimized routes. This resulted in reduced travel time for all collection systems.

Previous studies such as Ansari and Pakrou (2015) used GIS-based route optimization to obtain a travel time saving of 60.0% in Tabriz City, Iran. A similar study was conducted by Sallem and Rouis (2017) which yielded a travel time saving of 5.0% in optimization of household waste collection route in El Bousten District, Tunisia. It is

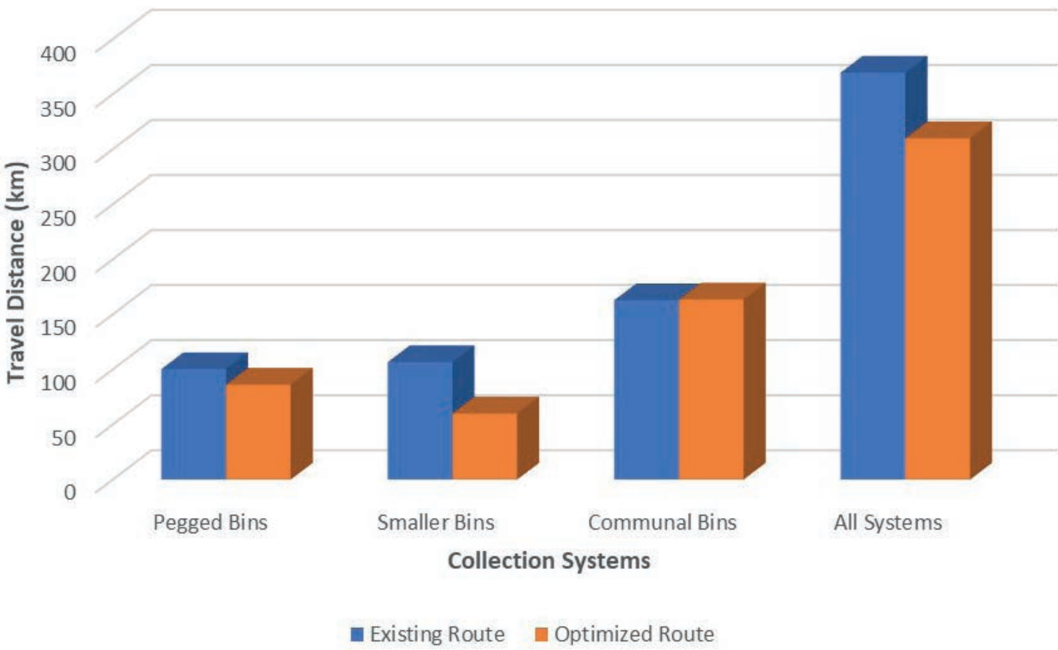

FIGURE 6: Mean travel distance on existing and optimized routes.

TABLE 5: Paired samples t-test for travel distance and time for small bins collection system.

		Paired Differences					t	df	Sig. (2-tailed)
		Mean	Std. Deviation	Std. Error Mean	95% Confidence Interval				
					Lower	Upper			
Pair 1	Existing distance – optimized distance	4.0000	3.0125	1.3472	.2595	7.7405	2.969	4	.041
Pair 2	Existing time-optimized time	801.600	506.074	226.323	173.226	1429.974	3.542	4	.024

t = t-statistic value, df = degrees of freedom, sig. = p-value, std. = standard

TABLE 6: Paired samples t-test for distance and time for pegged bins collection system.

		Paired Differences					t	df	Sig. (2-tailed)
		Mean	Std. Deviation	Std. Error Mean	95% Confidence Interval				
					Lower	Upper			
Pair 1	Existing distance – optimized distance	2.8400	1.6273	.7277	.8195	4.8605	3.903	4	.018
Pair 2	Existing time-optimized time	567.600	387.292	173.202	86.714	1048.486	3.277	4	.031

t = t-statistic value, df = degrees of freedom, sig. = p-value, std. = standard

TABLE 7: Paired samples t-test for distance and time for communal bins collection systems.

		Paired Differences					t	df	Sig. (2-tailed)
		Mean	Std. Deviation	Std. Error Mean	95% Confidence Interval				
					Lower	Upper			
Pair 1	Existing distance – optimized distance	-.0933	1.2961	.4320	-1.0896	.9029	-.216	8	.834
Pair 2	Existing time-optimized time	146.778	106.504	35.501	64.911	228.644	4.134	8	.003

t = t-statistic value, df = degrees of freedom, sig. = p-value, std. = standard

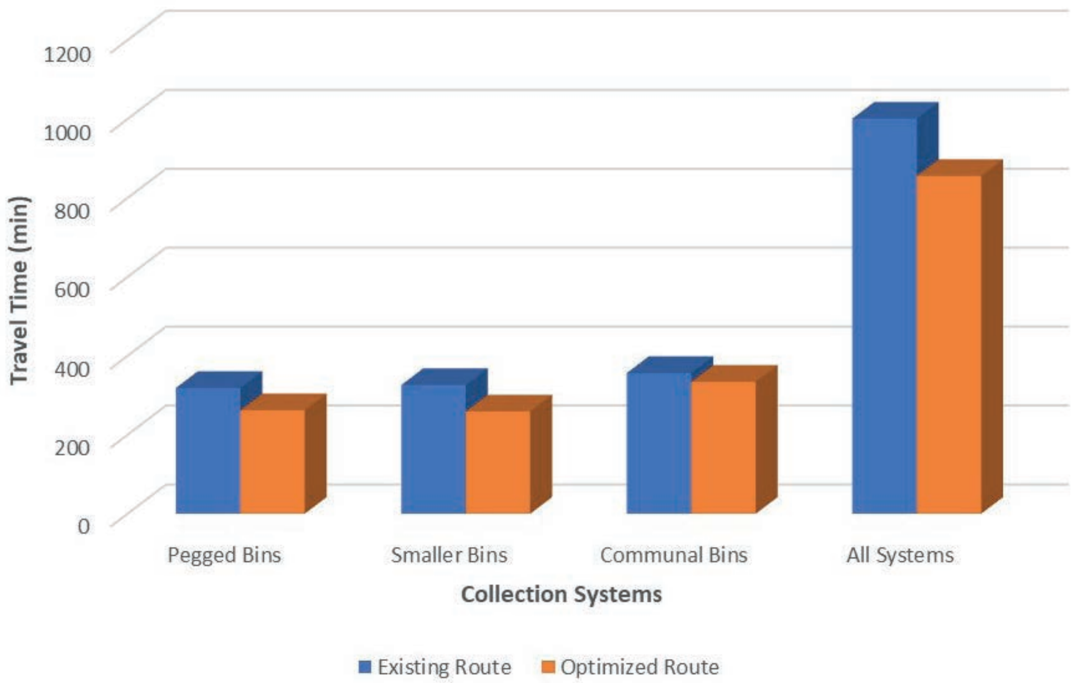

FIGURE 7: Mean travel time on existing and optimized routes.

on this basis that travel time proved the main criterion for route optimization in this study due to the fact that travel time is directly proportional to operational time with implication on cost (Kinobe et al., 2015). This supports the claim of the possibility of increasing operational coverage of waste collection and get more work done due to optimal time usage (Sulemana et al., 2019). Findings from this study indicate that the longer a truck travels on a specific route, more distance is covered, resulting in more working time as well as fuel and maintenance cost. A reduction in travel time translates into less fuel consumption and less period for servicing trucks hence, associated cost reduction (Chalkias & Lasaridi, 2009). This is consistent with previous studies that applied route optimization in a GIS environment (Risti´c et al., 2015; Sulemana et al., 2019; Tavares et al., 2009).

4. CONCLUSIONS

The study used route optimization to significantly reduce travel time and travel distance of solid waste collection on KNUST campus. Various geographical data (road networks, collection points, road limitations, etc.) were used in this method with Network Analyst extension of ArcGIS software. The road network analysis tool of the VRP solver in GIS environment facilitated the generation of routes with relatively reduced travel distance and travel time. The results from the study showed reduction in total travel time from 1,000.75 mins to 855.70 mins for existing and optimized routes respectively, representing saving of 14.5%. Total travel distance reduced from 367.30 km to 334.20 km for existing and optimized routes respectively, representing saving of 9.0%. The results show that, the optimal routes generated were more efficient in terms of travel distance and travel time. Significant savings made for all the various collection systems on KNUST campus in relation to travel distance and travel time supports the hypothesis that, optimal routes reduce solid waste collection operating cost through significant savings in travel time, thereby reducing fuel consumption. The optimized routes were not previously implemented and hence are recommended for use. This study is therefore not only a theoretical one, but it has a clear benefit, which ensures an optimization of the waste collection in KNUST. Route optimization should therefore be considered in policy provisions associated with operating cost reduction in relation to solid waste collection. There is the need to broaden the scope of route optimization in future studies to establish the effects of daily and seasonal variations on the optimal outcomes. Further study should also aim at developing decision support systems related to route optimization which are more user friendly for easy adoption and usage by practitioners.

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INFLUENCE OF THROUGHPUT RATE AND INPUT COMPOSITION ON SENSOR-BASED SORTING EFFICIENCY

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ABSTRACT

According to the Directive (EU) 2018/851 of the European Union, higher recycling rates for municipal waste will have to be met in the near future. Beside improvements to the collection systems, the efficiency of mechanical processing and sorting will have to be increased to reach the EU´s targets. Sensor-based sorting (SBS) plants constitute an integral part of today's sorting processes. Two main factors determine the sorting performance: throughput rate and input composition. To improve recycling efficiencies, especially SBS machines need to be optimized. Three evaluation criteria are used to describe the performance of these processes: recovery (content of input material – both eject and reject material discharged into the product fraction) or product quantity (amount of product generated via sorting within a specific interval – calculated by multiplying throughput rate and yield), yield (amount of eject material discharged into the product fraction), and product purity. For this study, 160 sorting experiments each with 1,000 red and white low-density polyethylene (LDPE) chips were conducted to investigate the effects of throughput rate and input composition on sorting processes. This simplified approach reduced the influence of other factors on the sorting performance, giving precise information on the effect of throughput rate and input composition. The testing results can enter process optimization. With increasing throughput rates, product quantity rises following a saturation graph (despite exponential decrease in recovery). In the experiments a higher throughput rate also resulted in an exponential decrease of the yield while a change to the input composition had no such effect. The third evaluation criteria, product purity, decreases linearly with increasing occupation density. The slope of this function depends on the input composition.

1. INTRODUCTION

16.3 million tons (170 kg/capita) of plastic packaging waste (PPW) are produced in the European Union (EU) per year, out of which as little as 42 wt% were recycled in 2016 according to Eurostat, 2019. E. g. by 2025, the EU also aims to increase the rate for preparing for re-use and the recycling of municipal waste to 55 wt% (The European Parliament and of the Council of the European Union, 2018).

PPW-recycling requires separation into individual plastic types (International Organization for Standardization, 2008). Plastics are usually separated using sensor-based sorting (SBS) (Gundupalli et al., 2017; Jansen et al., 2012). Spectral imaging techniques including NIR (near-infrared, 750-1100 nm (Workman and Springsteen, 1998), VIS (visual image spectroscopy, 380-750 nm (Workman and Springsteen, 1998) and HSI (hyperspectral imaging) are most commonly applied though laser-induced-breakdown-spectroscopy and X-ray-sorting are available as well (Table 1).

SBS techniques have been utilised by various industries during the last years. Additionally, research results were published in many papers. SBS is mostly applied in recycling (Gundupalli et al., 2017; Mesina et al., 2007; Rahman et al., 2014), mining (Knapp et al., 2014; Lessard et al., 2014; Dalm et al., 2014) and food (Alaya et al., 2019; Cubero et al., 2011; Tu et al., 2007) processing plants. Sound information on the sorting performance of such technologies is limited, however.

One key parameter found fluctuating in industrial SBS plants is their throughput rate adversely affecting their sorting efficiency (Feil et al., 2019). The throughput rate, in tons per hour, is related to the occupation density (the relative size of the detection zone in an SBS unit that is covered with particles), in %. With respect to SBS, the occupation

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TABLE 1: Sorting techniques applicable to different types of waste (mod. Gundupalli et al., 2017).

	Eddy current	LIBS	X-ray sort	Optical sort	Spectral sort
Non-ferrous metal	✓	✓	✓	✓	✓
Plastic		✓	✓		✓
Paper				✓	
Glass				✓	✓
Wood		✓	✓		

density is a better indicator of capacity than the throughput rate. The reason is that an SBS unit operates differently from other processing technology. Objects must here be presented separately to a sensor enabling a particle-specific sorting decision (eject or reject) by the computing technology. The spatial separation of objects is therefore of utmost importance. Their mass, high or low, in comparison to other particles in the material stream, is irrelevant. The performance of an SBS unit is accordingly related to the space particles occupy in the detection zone and only indirectly correlated with the throughput rate. This paper therefore applies, the occupation density rather than the throughput rate as a reference parameter to describe the capacity of an SBS machine. For industrial applications, conversion into throughput rates considering material-specific grammages is otherwise required.

Fluctuations of the occupation density mostly result from batch processes integrated into an SBS plant, e.g. opening packages or bales. A second key parameter found fluctuating in SBS plants is input composition, because of the delivery of input material from different (urban/rural) regions.

Variations in occupation density and input composition, as well as other factors like surface moisture and roughness (Küppers et al., 2019b), and mechanical stress (Küppers et al., 2019a) can affect the purity and recovery of products in two ways:

- Errors in detection, recognition and classification of particles (sensor and algorithm);
- Errors in mechanical discharge (conveyor belt, chute, pressurized air nozzle bar).

On the one hand, a high occupation density, or throughput rate, may impact the recovery and purity of the output as overlapping particles impede the analysis of the underlying material. Restrictions and a significantly reduced belt speed on standard sorting machines are required especially for sorting light and flat materials, such as films, because of their low weight and high surface coverage (Beel, 2017). On the other hand, high throughput rates are desirable to produce large amounts of products in a short time to be economically sound. No systematic study on the effects of occupation density on SBS of plastics has been conducted yet, however.

The two main factors affecting SBS efficiency, i.e. input composition and occupation density, were addressed in systematic testing series. Data hereby obtained provides insight for better understanding of the efficiency of SBS processes.

2. MATERIAL AND METHODS

2.1 Materials

1,000 rectangular LDPE chips (white and red) were used as input material, each featuring an investigated visible surface area of approx. 18.3 cm², a width of 30 mm, an average length of 61 mm and a thickness of 3 mm (Figure 1).

The grammage of these chips amounts to 0.27 g/cm² and the average particle weight is 4.9 g. In the conducted experiments, white particles were regarded as 'eject' and supposed to be discharged via air shocks while red particles were considered as 'reject' and not supposed to be discharged.

2.2 Equipment

An experimental SBS setup, engineered by Binder+Co AG, was utilized to conduct a total of 160 sorting experiments. As shown in Figure 2, this testing setup consisted of a chute sorter, of a work width and length of 500 mm and 455 mm, respectively, and an upstream vibrating conveyor to feed the sample material. The resolution of the colour sensor is to 0.523 mm x 0.473 mm/px. The valve resolution is 6.25 mm.

Once on the chute, the bulk material was detected using the built in VIS sensor and then classified by means of colour, intensity and brightness. If classified eject material, the respective object was discharged via the compressed air nozzle bar. Any detected object > 35 mm, e.g. multiple particles overlapping, was digitally divided into several objects and then classified individually to be rejected or ejected.

2.3 Preliminary Tests

Preliminary detection tests were carried out to evaluate the content of falsely classified pixels of reject (red) and eject (white) particles. A series of five trial runs was

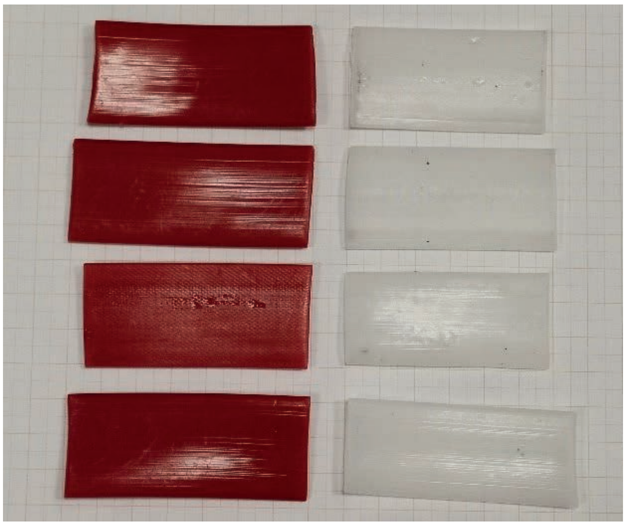

FIGURE 1: Testing material for sorting experiments - red (reject, left) and white (eject, right) LDPE chips.

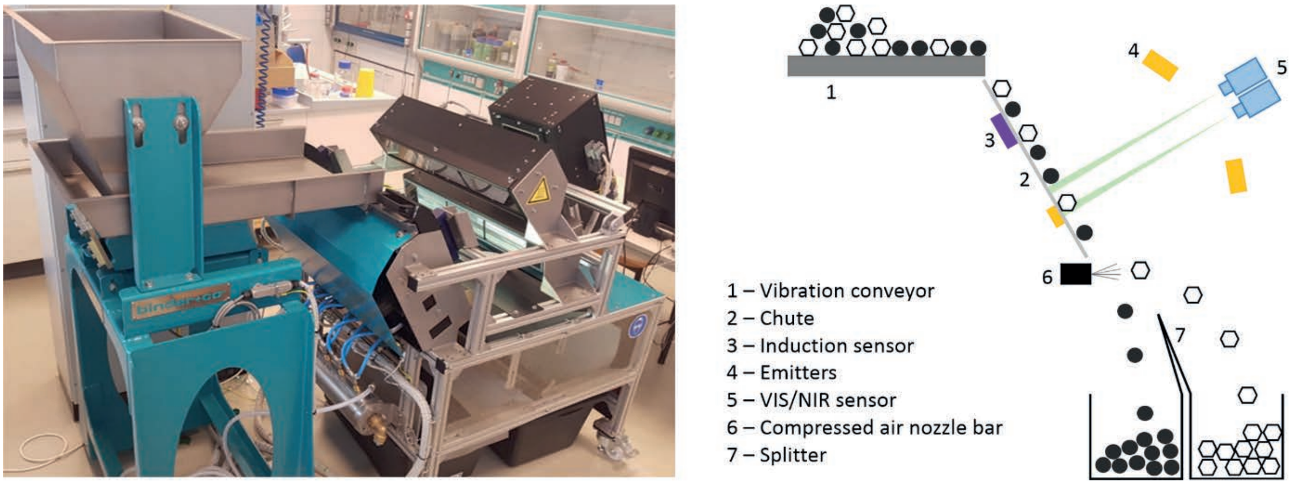

FIGURE 2: Setup for SBS sorting experiments.

conducted featuring 1,000 particles for each particle type (eject/reject). Each time, the content of correctly and incorrectly classified pixels was recorded.

Additionally preliminary ejection tests were carried out to determine the average amount of incorrect discharges based on variable sliding speeds, sideways movement, erratic bouncing, etc. on the chute. For this purpose, in five runs each including 1,000 particles supposed to be discharged, were fed to the sorting machine and overlapping of particles was avoided by feeding the particles one by one to the vibration conveyor, ensuring 100% particle separation. As a result of this approach, the occupation density remained below 2.8% for all preliminary ejection tests. This enabled assessing the effect of purely mechanically-based sorting errors. The results of the preliminary tests were evaluated by counting the number of falsely rejected and correctly ejected particles.

2.4 Main Experiments

To study the effects of input composition on sorting efficiency, eight samples of different composition were created (Table 2), each containing a total of 1,000 chips.

On average, this resulted in 5,500,000 detected object pixels per test run, with a standard deviation of about 200,000 pixels, provided that no particles overlapped. The standard deviation σ was calculated using the following equation, where x is the sample mean, $\bar{x}$ is the arithmetic mean and n is the sample size.

$$\sigma = \sqrt{\frac{\Sigma(x-\bar{x})}{(n-1)}} \tag{1}$$

TABLE 2: Generated samples and their composition.

Sample number	Sample name	Red particles	White particles
1	(95/5)	950	50
2	(90/10)	900	100
3	(85/15)	850	150
4	(80/20)	800	200
5	(70/30)	700	300
6	(60/40)	600	400
7	(50/50)	500	500
8	(20/80)	200	800

Each of the eight samples was sorted 20 times at varying throughput rates. For each experiment, the respective sample mixture of 1,000 particles was placed on the vibration conveyor and fed to the sorting machine by starting the vibration conveyor after adjusting its potentiometer to the intended testing period. After each experiment the number of ejected and rejected particles (red and white) was determined by manual sorting.

Each experiment required a steady feed, since fluctuations of the throughput rate would have resulted in shifting occupation densities, compromising analysis of the results. The testing period of each experiment could therefore deviate slightly from the intended value. The testing periods of the 20 experiments were as evenly distributed as possible for each input composition, ranging from 1.0 s to 70.6 s. Table 3 shows the relationship of testing period, occupation density and throughput rate per metre of working width.

The occupation density is defined as the ratio of detected object area and available space on the detection area for the testing period. Available space is calculated using the following equation with A=available area, υ=sliding speed of particles at the point of detection (1321 mm/s), t=testing period (cf. Table 3) and w=working width (500 mm).

$$A = \upsilon * t * w \, [m^2] \tag{2}$$

After each experiment the particles were thoroughly mixed to generate a uniform blend of all 1,000 particles designed as input material for the next experiment.

2.5 Evaluation of Results

The results were analysed based on three evaluation criteria attained at the respective throughput rate of each trial:

- Recovery (directly related to product quantity);
- Yield;
- Purity.

TABLE 3: Overview of testing period, occupation density and throughput rates.

Test duration [s]	Occupation density [%]	Throughput rate [t/(h*m)]
1	278.6	35.02
2	139.3	17.51
3	92.9	11.67
4	69.6	8.75
5	55.7	7.00
6	46.4	5.84
7	39.8	5.00
8	34.8	4.38
9	31.0	3.89
10	27.9	3.50
20	13.9	1.75
30	9.3	1.17
40	7.0	0.88
50	5.6	0.70
60	4.6	0.58
70	4.0	0.50
80	3.5	0.44
90	3.1	0.39
100	2.8	0.35

Recovery (R) is defined as the ratio of complete product mass (m_{eject}) and total input mass (m_{input}) per time unit, providing information on the product quantity generated at a respective throughput rate:

$$R = \frac{m_{eject}[\frac{t}{h}]}{m_{input}[\frac{t}{h}]} * 100\ \% \tag{3}$$

Product quantity is directly related to recovery, expressed by the mathematical product of throughput rate (Table 3) and recovery (Figure 3) for a given occupation density:

$$P = m_{input}[\frac{t}{h}] * R \tag{4}$$

The yield (R_w) is based on the amount of the desired component (eject) in the feed material and calculated from the ratio of the mathematical product of determined mass flow (m_{output}) and substance concentration (c_{output}) of the respective sorting product (output) to the respective product of mass (m_{input}) and substance concentration (c_{input}, defined as the content of eject material in the input material) of the feed material (input). It is calculated as follows (Feil et al., 2016):

$$R_w = \frac{m_{output}[\frac{t}{h}] * c_{output}[\%]}{m_{input}[\frac{t}{h}] * c_{input}[\%]} * 100\ \% \tag{5}$$

According to Feil et al. (2016), the purity (P_m) of a material is defined as the content of correctly ejected material in the sorting product. It is calculated as follows:

$$P_m = \frac{m_{recyclable\ material}[\frac{t}{h}]}{m_{impurity}[\frac{t}{h}] + m_{recyclable\ material}[\frac{t}{h}]} * 100\ \% \tag{6}$$

All three performance indicators are usually mass-specific [w%]. When evaluating SBS sorting experiments, however, particle-related [p%] information is more useful, meaning that a sorting stage is evaluated based on the number of particles contained in each output fraction (reject and eject) and not on the mass of the respective fraction. Conclusions on mass-specific evaluation criteria can be obtained by providing the average particle-specific mass. Particle-related evaluation criteria are displayed accordingly in this paper.

3. RESULTS AND DISCUSSION

3.1 Preliminary Tests

During the preliminary detection tests, an average of 0.87% of the pixels of white objects and 0.65% of the pixels of red objects were falsely classified, the edges of objects being most commonly affected. Since all objects containing > 50% pixels of the eject material are discharged and misclassification is evenly distributed among all particles, the misclassification rate is not significant for the discharge of objects.

Although misclassification did not result in a rejection of particles, 0.28% to 0.44% of all eject particles were rejected in the preliminary ejection test. These amounts of falsely rejected particles were only the result of mechanical errors, because quite low throughput rates had been chosen, overlapping cannot be a reason for rejection. The contents of incorrectly detected and incorrectly rejected particles were therefore insignificant for the used experimental setup. The subsequently conducted main experiments thus allow statements to be made about the best operation conditions of sorting stages in treatment plants applying SBS machinery that are based on input composition and occupation density of a sorting stage only.

3.2 Recovery and Product Quantity

The effects of occupation density on the recovery for different input compositions are displayed in Figure 3. Evidently the recovery decreases with increasing occupation density. For rising eject shares of input, maximum recovery increases (usually at the lowest occupation density).

Since industrial applications most often run at quite low occupation densities the graphs for recovery and product quantity in Figure 4 are displayed for occupation densities < 100%. Despite decreasing recovery, product quantity evidently rises with increasing occupation density following a saturation graph.

The slope of shown saturation graphs most often approaches zero when occupation densities reach 40% to 60%. The higher the content of reject particles in the input, the earlier saturation is reached. The slopes of well-balanced inputs (50/50 and 60/40) drop for higher occupation densities. When considering the option of increasing throughput, the result indicates that sorting at high throughput rates may only be reasonable for input material of a balanced composition with regard to product quantity. For other input compositions, high occupation densities show less benefit in this regard.

In general, there are direct correlations observed between recovery and both occupation density and input

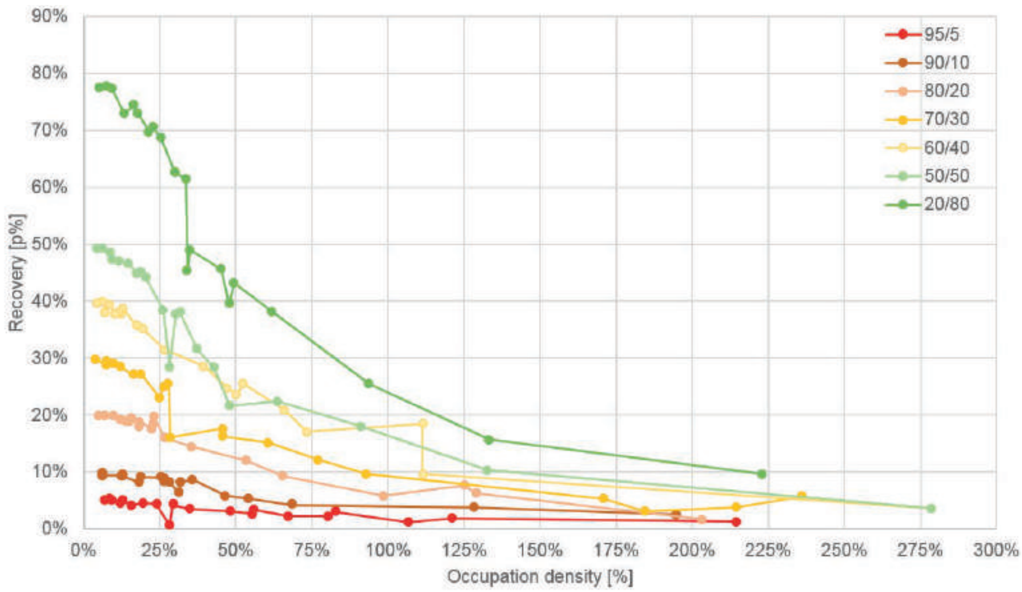

FIGURE 3: Effects of occupation density on recovery for different input compositions – occupation density < 300%.

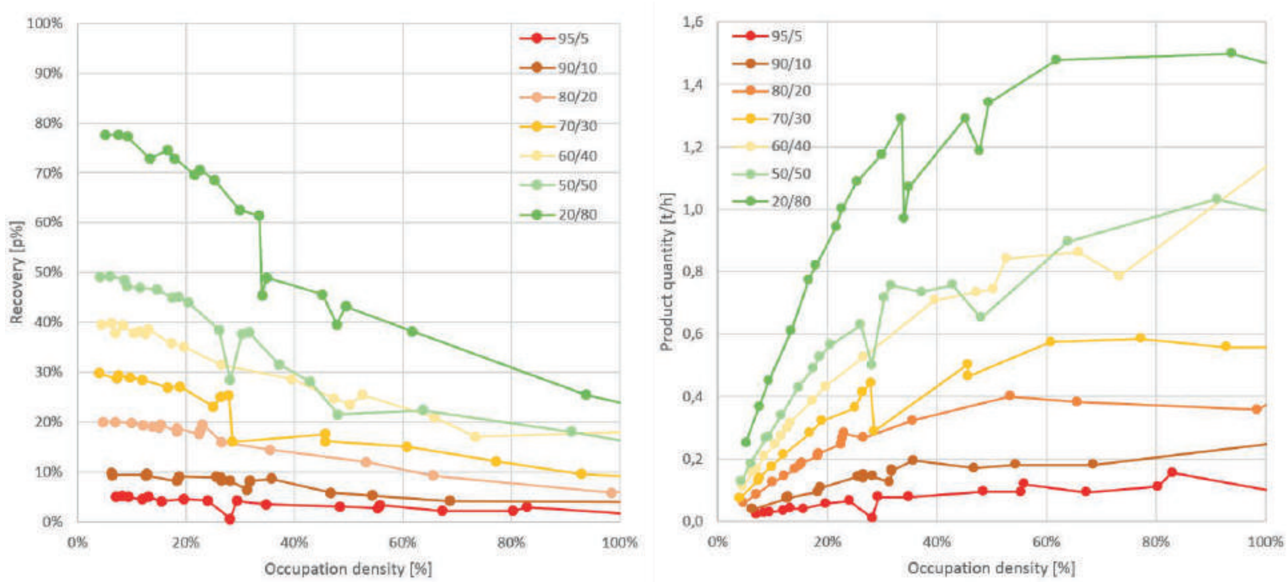

FIGURE 4: Effects of occupation density on recovery (left) and product quantity (right) for different input composition – occupation density <100%.

composition (the content of particle-related eject in input) which may permit forecasting the product quantity as a function of influencing factors. Use Figure 4 to determine the occupation density that is most suitable for a reasonable product quantity, depending on the respective input composition. Note that reject quantity increases with eject quantity.

3.3 Yield

The effect of occupation density on yield for different input compositions is given in Figure 5. With rising occupation density, the yield decreases exponentially from approx. 98 p% to approx. 10 p% for all input compositions identically.

Since industrial applications most often run at occupation densities <<100%, only the selected area shown in Figure 5 (blue frame), was taken into account for further analysis. Figure 5, therefore, gives the graph of the yield for all experiments at an occupation density <100%.

Additionally, the average yield is shown (red) as a polynomial function of the fourth degree with a coefficient of determination of $R^2=0.9417$:

$$y = -5.0564x^4 + 10.321x^3 - 6.2344x^2 + 0.2105x + 0.9784 \quad (7)$$

The inflection points of this approximation function are located at occupation densities of 27.6% and 74.5%. The first inflection point is reached at an occupation density of about 30% where its rising value impairs the yield very much. The second inflection point is reached at an occupation density of 75% when changes to its value have a much smaller impact on the yield. This is consistent with a range of occupation densities chosen for calculating the average yield. Up to an occupation density of 100%, the yield decreases constantly. Beyond, the decrease subsides. If values >100% were included for the occupation density when calculating the approximation function, there was no drop of the polynomial function at an occupation density of 100%.

Figure 6 shows that the occupation density effects the separation of eject particles while the input composition has no effect on the yield. This information helps to determine the highest occupation density while still achieving acceptable eject losses that may be controlled by, e.g., a quota, independent of the input composition.

Note that fluctuations of the yield increase significantly at occupation densities > 27%. This can be traced back to the elevated potential for overlapping as occupation densities increase since overlapping can either lead to eject losses (reject particles covering eject particles leading to reduced yield by rejection of both particles) or to a discharge of reject particles (eject particles covering reject particles leading to increased yield by ejection of both particles).

3.4 Effects on the Sorting of Rejects - Purity

The effects of changes to the input composition and occupation density on the absolute number of reject particles wrongly sorted into the eject fraction are displayed in Figure 7. Up to 35 red particles were sorted incorrectly. For clarity purposes, only trend lines (concave growth curves) are shown alongside the raw data in Figure 7.

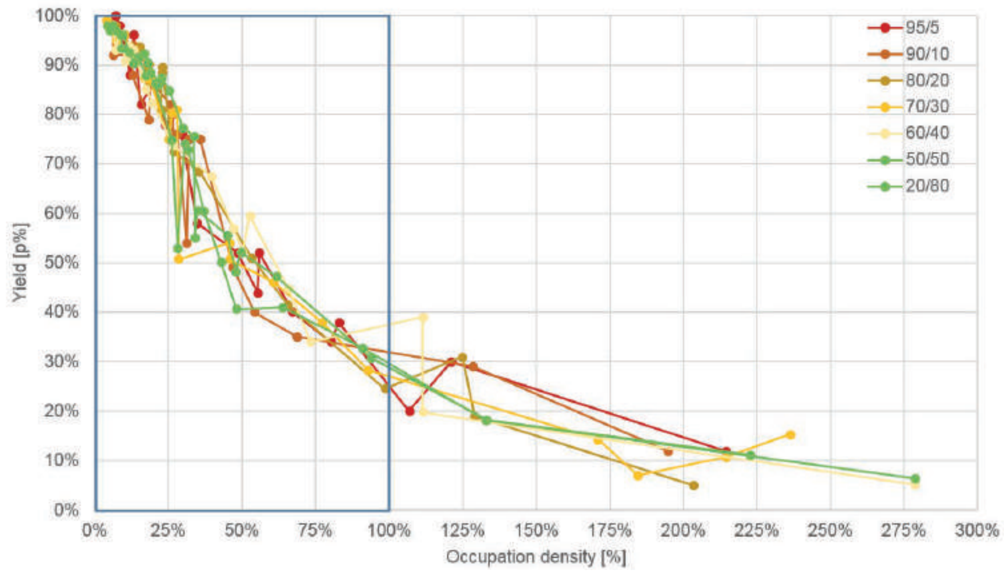

FIGURE 5: Composition-related effects of occupation density on yield - occupation density < 300%.

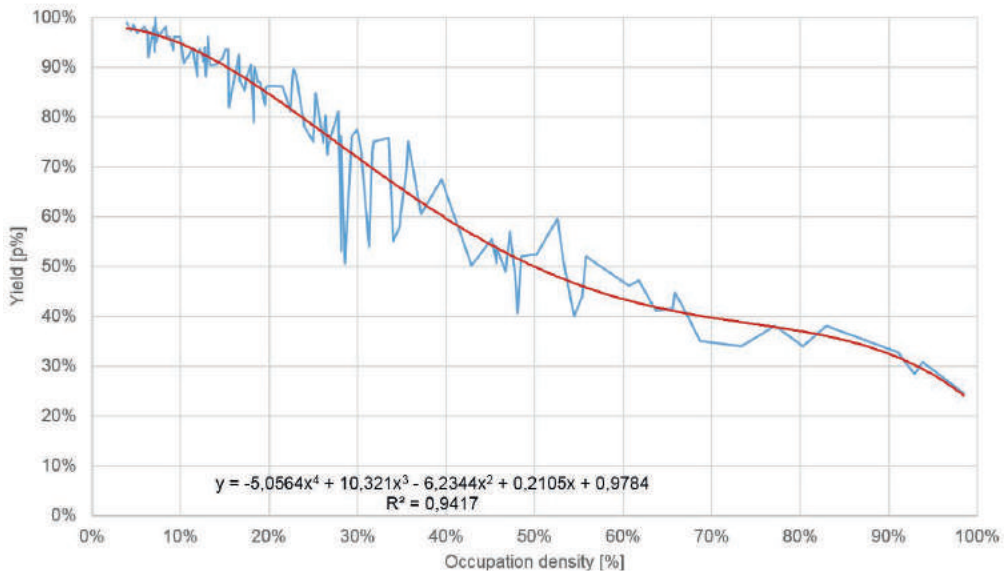

FIGURE 6: Average effect of occupation density on yield - occupation density < 100%.

The highest number of misclassifications was recorded for the quite balanced mixing ratios of 60/40, 50/50 and 70/30, while the number of falsely ejected reject particles is smaller for imbalanced input compositions. In general, the slope of all graphs is much reduced at occupation densities of around 30%. This means that the absolute number of incorrectly ejected reject particles is significantly less prone to increasing with rising occupation densities if the general level is > 30%.

As a result of the exponential decrease of recovery and the concave increase of the number of falsely ejected reject particles, a linear decrease of purity in the eject fractions can be observed for increasing occupation densities (see Figure 8, right). With decreasing eject content in the input, the decline of eject purity worsens.

A tendency to greater volatility can be observed for the input material if the content of eject particles is small. This trend is consistent with the sample size used for all experiments. For input samples with low contents of eject particles, even small numbers of falsely ejected reject particles constitute quite large amounts of impurities in the eject fraction. Fluctuations of the absolute numbers of incorrectly ejected reject particles therefore rather affect the graphs of input samples with high content of reject particles than vice versa.

Figure 8 shows that, as eject content of the input of a sorting stage increases, the slope of the resulting linear graph softens. While the negative gradient of the linear functions in principle increases with the reject content in the input, the magnitude of this change is not consistent. The graphs of the input samples 70/30 and 60/40 display an anomaly concerning the otherwise consistently increasing negative gradient of the linear graphs.

3.5 Economic Potential

This paper highlights the effects of input composition and throughput rate on recovery/product quantity, yield, and purity of the eject fraction from SBS stages. Better

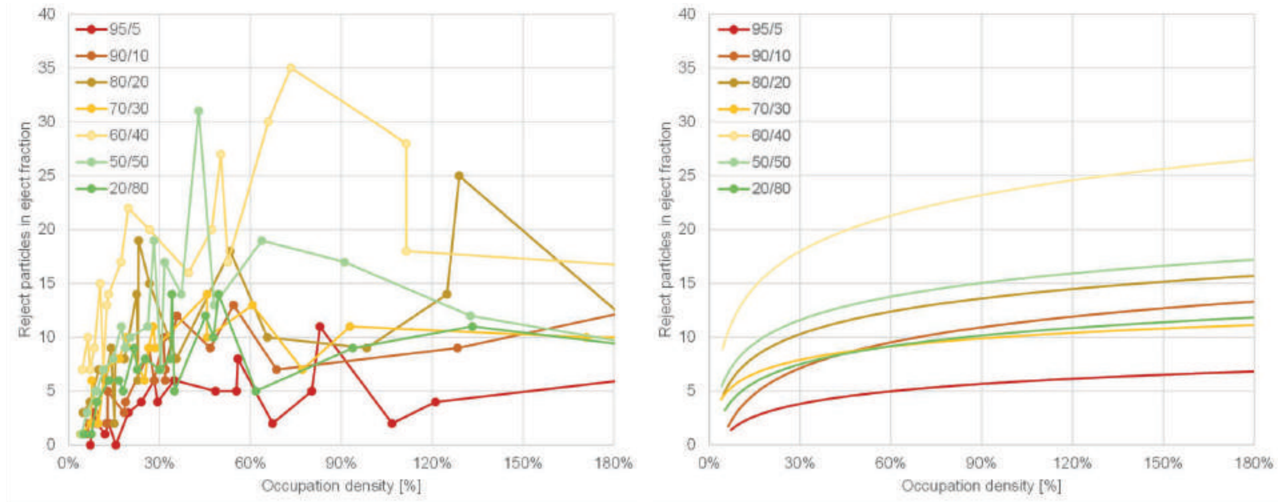

FIGURE 7: Influence of occupation density on the ejection of reject particles for different input compositions (left: raw data, right: mathematical fit) – occupation density < 180%.

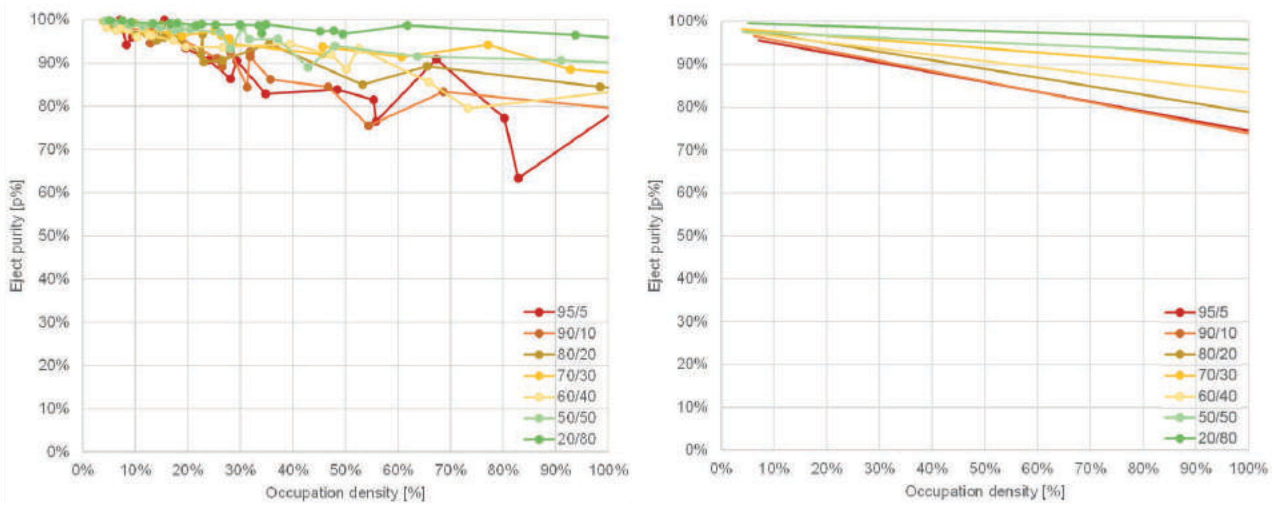

FIGURE 8: Effects of occupation density on eject purity for different input (left: raw data; right: mathematical fit) compositions – occupation density < 100%.

knowledge of the interdependence of these variables was also acquired.

When the accuracy of sensory detection and mechanical efficiency is known, both datasets can be combined to assess the efficiency of SBS steps. Each input composition imposes an upper limit on the achievable recovery, the yield, and purity. This upper limit cannot be raised by changes to the experimental setup, say, be applying a better sensor, since it is a function only of the ratio of eject to reject particles in the input stream.

To demonstrate the value of the ascertained results, Figure 9 shows a simplified sample application. The graphs (yellow= yield, red=eject purity and green=product quantity) are displayed for a certain input composition.

It was assumed that the eject fraction, produced during this exemplary sorting stage, can be sold for different prices (100/80/60 €/t), depending on its purity (95/90/85 wt%) achieved by sorting. For simplification purposes, assume that the mass balance is in accordance with particle related composition.

Purity, quantity, and possible losses of the potential product must be considered to find out at which occupation density (throughput rate) the highest profit per hour is made. For every input material whose sorting step can be described with the graphs shown in Figure 9, the price and quality of the respective maximum product quantity can therefore be located at the secondary axis of Figure 9 (green arrows). In Table 4, the best occupation densities for meeting the quality requirements are given, including the corresponding product price, yield, product quantity and arising profit per hour.

Evidently, the highest obtainable profit is found at an occupation density of 44 %, even though the generated product quantity is highest at an occupation density of 67% and the highest yield would be generated at an occupation density of 23%. Therefore, knowing the interdependencies of the described factors may help optimizing a sorting stage in the first place.

4. CONCLUSIONS

Evaluating the efficiency of an SBS machine without running sorting experiments, depends on a multitude of influencing factors. These factors can be divided into two categories that do not influence each other:

- Factors affecting the functionality of built-in sensors, reducing the content of particles that can be detected, recognised, and classified;
- Factors related to mechanical discharge issues, reducing the efficiency of an SBS machine independent of the accuracy and suitability of built-in sensors.

The scientific results shown here provide a basis for assessing the efficiency of SBS units in sorting plants. Displayed data comprises the predominant influencing factors (input composition and occupation density) affecting mechanical discharge issues.

When combined with the sensor-specific efficiency (depending on what a sensor is used for), these data can be used to predict the efficiency of a machine and maybe even the efficiency of many connected SBS machines.

The influence of input composition and throughput rate on sorting efficiency has been established using model mixtures sorted in an experimental setup at various throughput rates. The following main conclusions are based on the experimental series run with homogeneously shaped particles, evenly distributed particle weights and uniform particle size for eject and reject particles:

- Input composition does not affect the yield for any throughput rate/occupation density.
- With increasing occupation density/throughput rate, the yield decreases exponentially from approx. 98 p% to approx. 10 p%.
- The average yield, a function of the occupation density/ throughput rate, can be shown as a polynomial function of the fourth degree for occupation densities <100% (Figure 6).

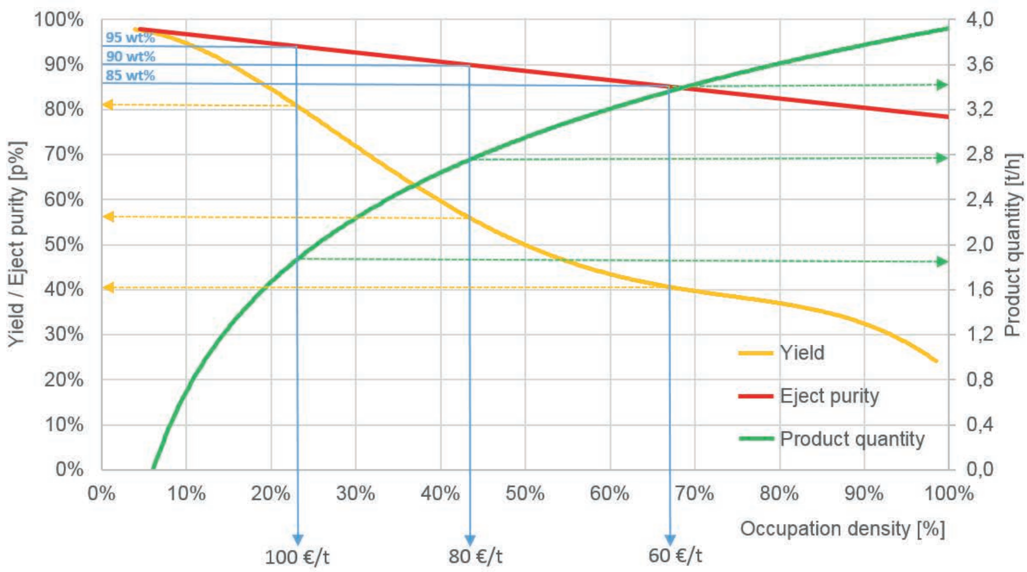

FIGURE 9: Graphs of yield (yellow), eject purity (red) and product quantity (green) for a sample case; blue arrows – material value for a specific quality; yellow arrows – obtainable yield for a specific eject purity; green arrow – obtainable product quantity for a specific eject purity.

TABLE 4: Profit per hour with respective process and product parameters.

Occupation density	Quality requirement	Product price	Yield	Product quantity	Profit
23%	95 wt%	100 €/t	81 wt%	1,85 t/h	185 €/h
44%	90 wt%	80 €/t	56 wt%	2,78 t/h	222 €/h
67%	80 wt%	60 €/t	41 wt%	3,45 t/h	207 €/h

- As occupation density/throughput rate rises, product quantity increases (despite a decrease in recovery) following a saturation curve that reaches maximum for an occupation density of approx. 60% (Figure 4).
- Eject purity can be plotted as a descending linear function of occupation density/throughput rate. The slope of this function is related to the input composition.
- The higher the eject content in the input composition of an SBS stage, the smaller the slope of the related descending linear function (Figure 8).

Profounder datasets may be obtained in large scale experimental series using stable input compositions and longer test durations are advisable. Other influencing factors like grain size distribution of the input material, particle shape, and machine design should be examined to expand the dataset presented in this paper.

Actually material flows in sorting plants are subject to powerful fluctuations due to changes to the input composition and irregular material discharge of upstream processing machinery. As has been shown here, such temporarily fluctuating throughput rates can painfully reduce the sorting efficiency of SBS stages. Choosing processing machinery to regulate input rates and to discharge the output fractions regularly can enhance the performance of downstream sorting stages at the same overall throughput rate. Another option of how to reduce fluctuating input rates is using bunkers. Depending on the scope of fluctuations, the required bunker volume may vary, causing high investment costs. This approach may not be feasible for material streams like light-weight packaging waste that, due to its non-bulk properties, could generate a blockage when stored in a bunker.

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INCREASING THE BASELOAD CAPACITY OF BIOWASTE FERMENTATION PLANTS THROUGH OPTIMISED SUBSTRATE MANAGEMENT

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ABSTRACT

This paper presents a survey to determine the influence of impurities and green waste on anaerobic biowaste treatment, comparing the current common biowaste management system with a proposed future system. The results imply that it is possible to increase the specific biogas yield and the baseload capacity by means of an adapted biowaste management system. To analyse a possible correlation between biogas yield and biowaste composition from urban or rural areas the quality and quantity of biowaste was evaluated in a long-term research programme. August was the month with the maximum percentage of 69% green waste in biowaste, compared to February with a minimum proportion of 14%. The specific biogas yield of biowaste is in the range of 91 to 160 m^3/t_{OS}. The evaluation showed that, as expected, seasonal fluctuations in weather affected the proportion of garden and park waste (green waste) as well as impurities. Moreover, the proportion of green waste and impurities affects the substrate quality and degradability of biowaste. The investigated biowaste fermentation plant is not able to generate sufficient baseload for the reasons described above. Other reasons for this limitation include maintenance work, plant operation in part-load range and plant capacity limitations concerning the CHP module, digester capacity and pipes for gas transportation. The best ratio between organic fraction and impurities is obtained from biowaste from rural areas, while gas yields from urban areas are more constant and less volatile. Based on these findings, the increasing of the baseload capacity of waste fermentation plants can be achieved by: Optimisation of process flows (substrate management, preparation, post-treatment); Use of co-substrates in compliance with legal requirements; Determination and consideration of key figures.

1. INTRODUCTION

Biomass is the most diverse of all renewable forms of energy and replaces fuels of fossil origin all over the world. Biomass is available in solid, liquid or gaseous form and can be available as an energy source to generate all kinds of energy, for example electricity and thermal energy. In Germany, the use of biomass, and especially biogas from the fermentation of biomass, plays a major role as a result of support programmes by the federal government. German bioenergy technology and knowledge has played a key role in the use and implementation of biogas projects all over the world in recent years. Compared with other countries, the use of bioenergy has been strongly associated with the use of energy crops in the past. Changes in the legal framework have however resulted in a widespread stagnation in the construction of new fermentation plants in Germany. Plant expansions, the switch to flexible operation, the construction of small-scale liquid manure plants and the use of bionic waste and residues are mainly responsible for the current increase in installed capacity. Because the temporary subsidy for the electrical energy fed will end after 20 years, many plants are likely to stop operating in the future. Today, new construction plants can only be operated economically in operating modes with waste from three different origins. Practically all low-structure organic wastes are suitable for fermentation (ATV-DVWK, 2003). The quantity of separately collected biowaste in Germany has more than doubled, from 6,554,100 t in 1996 to 15,612,800 t in 2016 (Statistisches Bundesamt, 2019).

The anaerobic digestion of organic matter is an established alternative to aerobic waste treatment by compo-

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sting. In Germany first comments relating to the collection and treatment of biowaste and the associated biological treatment could be found in the 1993 TA Siedlungsabfall (Technical Directive for Municipal Waste) (Bundesamt für Justiz, 1993). Since 1 January 2015, the comprehensive collection of biowaste has been required for German municipalities in accordance with § 11 of the Kreislaufwirtschaftsgesetz (Closed Cycle Management Act) (Bundesamt für Justiz, 2012).

The yearly in Germany collected 15,612,800 tonnes are treated aerobically or anaerobically in 1,256 biological treatment plants (Statistisches Bundesamt, 2019). The greatest advantage of anaerobic treatment compared to composting is the additional energy yield from the technical use of biogases. Central issues that need to be addressed in this field are the production of a constant biogas flow and establishing a stable fermentation process.

To promote the preferred feeding of electricity from renewable sources into the electricity grid, the Erneuerbare-Energien-Gesetz (Renewable Energy Sources Act) has been introduced in Germany. This law guarantees electricity producers a fixed feed-in tariff under specified conditions. In order to achieve the maximum fee of €0.1488/kWh, a minimum of 90% of anaerobically treated biowaste in a digestion plant must comply with the list of waste ordinance 20 02 01 (biodegradable waste), 20 03 01 (mixed municipal waste) and 20 03 02 (market waste) (Bundesamt für Justiz, 2014). The remaining 10 percent comprises the so-called co-substrates. These come from agriculture, animal husbandry, industrial processing or trade and services. Figure 1 shows the relationships between the production sectors of suitable substrates.

In the German energy system, various performance classes can be found. For example, there are bioenergy plants with electrical outputs of less than 5 kilowatts and large industrial plant complexes with outputs in the two-digit megawatt class. Different criteria are used to differentiate the used technologies of anaerobic digestion. The following criteria are suitable for classifying the different variants:

1. Dry matter content of the substrate
   a. Wet fermentation
   b. Dry fermentation
2. Feeding of the plant
   a. Discontinuously
   b. Semi-continuously
   c. Continuously
3. Process stages
   a. Single-stage
   b. Multi-stage
4. Process temperature
   a. Psychrophile
   b. Mesophile
   c. Thermophile

The definition between wet and dry fermentation is not specified according to a specific level of dry matter content because the dry matter is strongly influenced by the substrate. To simplify, it can be said that pumpable substrates are used in wet fermentation and stackable substrates in dry fermentation.Fermentation plants for renewable resources and stock-farming products and residues often work according to the principle of wet fermentation.

The continuous feeding is characterised by multiple feeding of substrate batches into the fermenter during the day. Advantages result from the continuous gas production with constant substrate properties. Quasi-continuous processes are characterised by at least one addition of substrate per day. However, gas production in this process variant is more volatile than in continuous feeding processes. Discontinuous feeding is the characteristic parameter of batch processes. The digester is loaded with new substrate and inoculated with microorganisms by digested substrate. The batch process is characterised by a time-dependent gas production rate. State of the art is the operation with several batch fermenters to produce a continuous biogas flow.

The microorganisms in the fermenter have different specific process conditions such as temperature, pH value and temperature. In the technological implementation of fermentation, the subdivision into hydrolysis phase and methane phase has established as a preferred option. The stages describe the number of process tanks independent of the process phase. If the hydrolysis and methane phases proceed simultaneously in one process tank, this is called single-stage. In the case of constructional separation of the phases, the process is named multi-stage.

The different bacteria in the fermenter have different temperature optima at which they have a high reaction speed. During fermentation, a distinction is normally made between three temperature ranges: psychrophilic, mesophilic and thermophilic. Only mesophilic (37-42°C) and thermophilic (50-57°C) fermentation is of importance in practical applications. If substrates are to be hygenised and germs killed during the fermentation process, thermophilic fermentation is a suitable method. Some biowastes and co-substrates must comply with hygienic standards before spreading to the fields. According to the Bioabfallverordnung (Bio-waste Ordinance), hygienisation can be achieved by thermophilic treatment at temperatures above 50°C during the manufacturer-specific minimum residence time in the fermenter (Bundesamt für Justiz, 1998). However, the thermophilic version is more susceptible to process problems and requires more energy to heat the fermenter. Because of these substrate-specific properties, biowaste, green waste and waste from households is mainly fermented at thermophilic conditions.

The aim of this paper is to analyse data from an existing biowaste fermentation plant to find out the percentage of green waste in biowaste for each month over an observation period of three years. The second objective is to determine the influence of the proportion of green waste in biowaste on the biogas yield.

1.1 List of abbreviations

- BG: Biogas
- OS: Original substrate as fresh matter
- IN: Inhabitant
- CS: Co-substrate
- BW: Biowaste

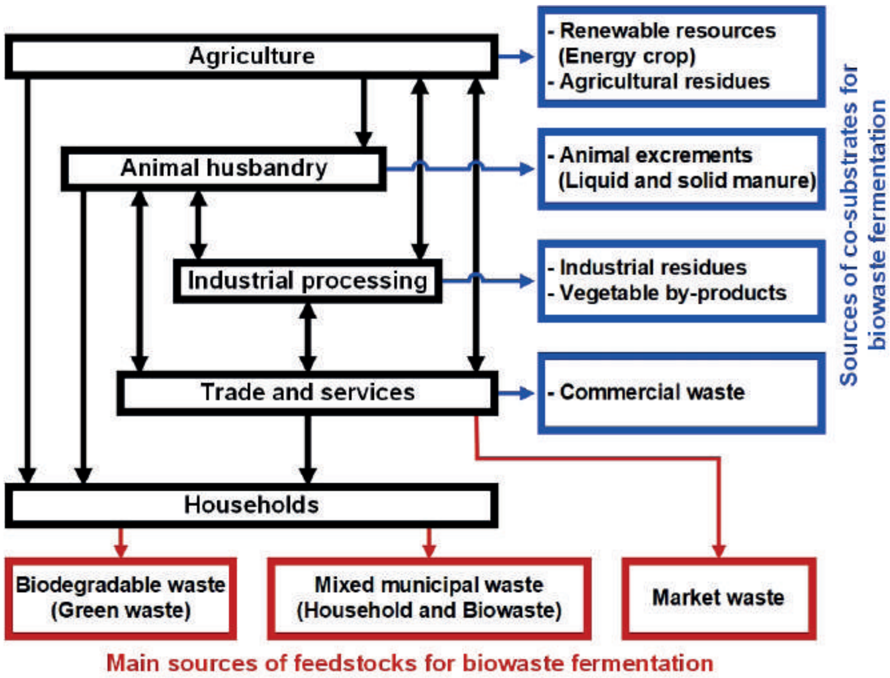

FIGURE 1: Relationships between the production sectors of suitable substrates.

2. BIOWASTE AS A SEASONALLY INFLUENCED SUBSTRATE

This section is divided into two parts. In subsection 2.1 the technical terms and the influencing factors of biowaste are explained. Subsection 2.2 clarifies the methodology of data collecting.

2.1 Biowaste and influencing factors

The definition of the term "biowaste" has not yet been standardised and varies according to its origin (Henssen, 2009). In European law, for example, biowaste is defined as:

> *... biodegradable garden and park waste, food and kitchen waste from households, restaurants, caterers and retail premises and comparable waste from food processing plants. (Europäisches Parlament und Rat, 2008)*

On the other hand, for the specific case of Germany, there is another definition of biowaste according to the Bioabfallverordnung (Bio-waste Ordinance), which is as follows:

> *Waste of animal or vegetable origin or from fungal materials for processing which can be metabolized by microorganisms, soil-borne organisms or enzymes, including waste for processing with a high organic content of animal or vegetable origin or of fungal materials... (Bundesamt für Justiz, 1998)*

As can be seen, in both definitions there is a vegetable component in the biowaste. This vegetable part in biowaste is affected by strong seasonal fluctuations, which significantly influence the quantity and quality of the biowaste. There are five main parameters that have been recognised and considered by experts and scientists: legal requirements, social factors, waste and substrate management, structural factors and weather conditions, as shown in Table 1.

From the data in Table 1, it can be seen that the influencing parameters are very diverse. For this reason, the biowaste composition is a conglomerate of all these various influencing parameters.

The three terms "biowaste", "green waste" and "co-substrate" that are used in this paper are defined as follows:

- "Biowaste" is a mixture of kitchen waste, plant residues and vegetable residues. More precisely, it can be defined as the type of waste which is derived from organic household waste, which is collected separately in organic-waste containers;
- "Green waste" is all vegetable waste (garden waste etc.) within the organic waste;
- "Co-substrate" is feedstocks that can be used in addition to the main substrate in the fermentation process. In the context of biowaste treatment, they originate from agriculture or are waste products and residues from agriculture, animal husbandry, trade and industrial processes.

From the beginning of spring until the end of autumn, organic waste is significantly influenced by green waste. When the influencing parameters are weighted, the seasonal influence and the structural factors are especially important for the quality and quantity of organic waste. A recent study has shown that green waste (garden waste) and biowaste with high proportions of food waste have different gas yields (Fricke et al., 2013). Biowaste with a high proportion of kitchen waste can produce up to 100 times more biogas than green waste.

TABLE 1: Advanced composition of influencing factors for biowaste (based on Kranert, 2010).

Influencing parameter	Examples	
Legal requirements	• Laws • Regulations	• Municipal statutes
Social factors	• Standard of living • Consumer behaviour • Environmental awareness • Sustainability awareness	• Income • Household sizes • Tourism • Education
Waste and substrate management	• Collection system • Waste bin size • Removal intervals • Charging system	• Detection systems • Public relations • Waste prevention, deposition, treatment
Structural factors	• Building structure • Sealing	• Gardens, parks and plants • Economic structure
Weather conditions	• Fair weather and frost periods • Bad weather • Clouds	• Precipitation • Air temperature and humidity • Wind

The definitions "biogas yield" and "specific biogas yield" for the gaseous products from the anaerobic treatment of biowaste are defined in this paper as follows:

- "Biogas yield" [l_{BG}/$Month_{IN}$] is the volume of biogas in litres produced by the anaerobic digestion of biowaste by an average inhabitant per month.
- "Specific biogas yield" [m^3_{BG}/t_{OS}] is the specific yield of dry biogas in m³ under standard conditions which is produced from one tonne of biowaste as original substance.

2.2 Methodology of data collection

All data were collected on an existing biowaste treatment plant in Mecklenburg-Western Pomerania. The available data covered the period from January 2016 to December 2018. The generated data encompassed information on biowaste, green waste, temperature, fill level, gas contents for methane, carbon dioxide, oxygen and hydrogen sulphide, co-substrates, heat demand, and biogas production.

3. DETERMINATION OF THE PROPORTION OF GREEN WASTE IN BIOWASTE

Having evaluated the operating data of the biowaste fermentation plant (Figure 2), we can state that the specific biowaste amount per inhabitant in the investigated region is between 2.50 kg per month in the winter months and 9.07 kg per month in summer (with a maximum green waste content of 72% in the biowaste). The following conclusions can be drawn from the data in Figure 2:

- The average amount of biowaste is highest between April and November;
- The quantity fluctuation range is most affected by the seasonal influence and structural factors during the months of March and June to August.

The influence of green waste results in a decrease in the specific biogas yield of the biowaste as well as in extensive challenges for the management of a biowaste fermentation plant.

To check if the proportion of green waste in biowaste is the relevant factor for the quantity changes, a correla-

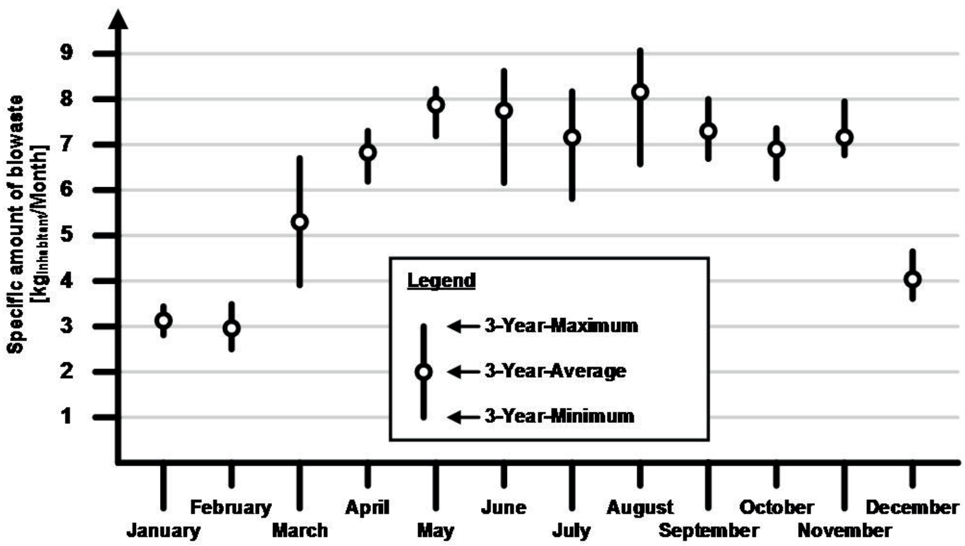

FIGURE 2: An exemplary presentation of the specific biowaste quantity of a biowaste fermentation plant in the annual cycle (2016 to 2018).

tion analysis according to Pearson was performed. The analysis with separately recorded green waste and biowaste resulted in a correlation coefficient of 0.94 (very strong correlation).

The following assumptions were made to determine the proportion of green waste in biowaste of a geographically limited region:

- No or similar co-substrates are used in the fermentation process;
- The number of inhabitants must be known for the investigation period;
- Biowaste from other regions has similar substrate characteristics;
- The observation period is three years, calculated on a monthly basis.

The following formula (1) can be used to estimate the mass of green waste in biowaste. In this determination, the month with the lowest proportion of green waste in the organic material is determined and subtracted from the month under investigation.

$$m_{G.i.B.\ (M/Y)} = m_{B\ (M/Y)} - m_{B\ min\ (BW\ 01/2016\ to\ 12/2018)} \quad (1)$$

$m_{G.i.B.(Month/Year)}$ = Mass of green waste in biowaste in the relevant month

$m_{B\ (Month/Year)}$ = Mass of biowaste in the relevant month

$m_{B\ min\ (BW\ 01/2016\ to\ 12/2018)}$ = Minimum amount of biowaste during the observation period

With the determined mass of green waste in biowaste, the weight proportion of green waste in the biowaste can be calculated by dividing it by the mass of biowaste in the month under investigation.

$$w_{G.i.B.\ (Month/Year)} = 100 \times \frac{m_{G.i.B.\ (Month/Year)}}{m_{B\ (Month/Year)}} \quad (2)$$

$w_{G.i.B.(Month/Year)}$ = Weight proportion of green waste in the relevant month

$m_{G.i.B.(Month/Year)}$ = Mass of green waste in the relevant month

$m_{B\ (Month/Year)}$ = Mass of biowaste in the relevant month

After the monthly data for green waste and biowaste has been determined, the average percentage of green waste can be determined by summing the monthly relevant masses of green waste and dividing them by the sum of the monthly relevant masses of biowaste. The formula for determining the average proportion of green waste in biowaste (Formula 3) over the observation period (2016 to 2018) is as follows:

$$w_{G.i.B.(Month/2016\text{-}18)} = \quad (3)$$
$$= 100 \times \frac{m_{G.i.B\,(x/2016)} + m_{G.i.B\,(x/2017)} + m_{G.i.B\,(x/2018}}{m_{B\,(x/2016)} + m_{B\,(x/2017)} + m_{B\,(x/2018)}}$$

$w_{G.i.B.(Month/2016–2018)}$ = Average weight proportion of green waste in the relevant month

$m_{G.i.B.(Month/Year)}$ = Mass of green waste in the relevant month

$m_{B\ (Month/Year)}$ = Mass of biowaste in the relevant month

x = Relevant month

It can be seen from Figure 3 that the proportion of green waste massively influences the composition of the biowaste. Green waste is the dominant component of organic waste, especially in the months from April to November. For example, the average maximum proportion of green waste was in August (69% green waste and 31% food waste) and the average minimum content was in February (14% green waste and 86% food waste). It can be concluded that winter is the season with the maximum specific biogas yield. The specific biogas yield can be up to 100% higher for biowaste with a high content of kitchen waste compared to biowaste with a high content of green waste (Fricke et al., 2013).

4. BIOGAS YIELD AS A FUNCTION OF THE PROPORTION OF GREEN WASTE

A central issue that needs to be addressed in this context is establishing a relationship between specific biogas yield and the proportion of green waste in biowaste. The evaluation of the plant data shows that the specific biogas yield correlates with the proportion of green waste in the biogas (Figure 4). The calculation formula (4) for the specific biogas yield resulting from the practical values is a quadratic equation. The values for the coefficients (a,b,c) were simplified, where a is -140 m^3_{BG}/t_{OS}, b is -5 m^3_{BG}/t_{OS} and c is 160 m^3_{BG}/t_{OS}. The simplification of the calculation formula (5) is intended to increase the practicability of the method for the operating personnel.

$$Y_{BG} = a \times x^2 + b \times x + c \quad (4)$$

$$Y_{BG} = -140 \times w_{G.i.B.}{}^2 - 5 \times w_{G.i.B.} + 160 \quad (5)$$

Y_{BG} = Specific biogas yield

x = $w_{G.i.B.(Month/2016–2018)}$ = Average weight proportion of green waste in the relevant month

The graph shows a slight decrease of the specific amount of biogas with increasing proportion of green waste in biowaste. In August 2017 a calculated share of 72% green waste in organic waste represents the maximum share of green waste in the observation period (three years).

To check the results of the calculation formula and to compare the practical data with the calculated values a descriptive data analysis was carried out (Table 2). The data are from March 2017 to February 2018. The results of the descriptive data analysis show that the calculated values for the specific biogas yield with the postulated methodology are comparable with the practical values of the examined plant. For instance the difference between the calculated and the plant values of mean specific biogas yield and 95% confidence interval of the mean in the investigation period is maximum 1.33m^3_{BG}/t_{OS}. Furthermore the median, minimum and maximum values are also comparable. In both analyses we find a distribution curve with a right skewness (depending on the positive value of the skewness). That indicates that the median value is lower than the mean value. The kurtosis of the practical values could be described as platykurtic, which means that the tails from the normal distribution are lighter. The tails ba-

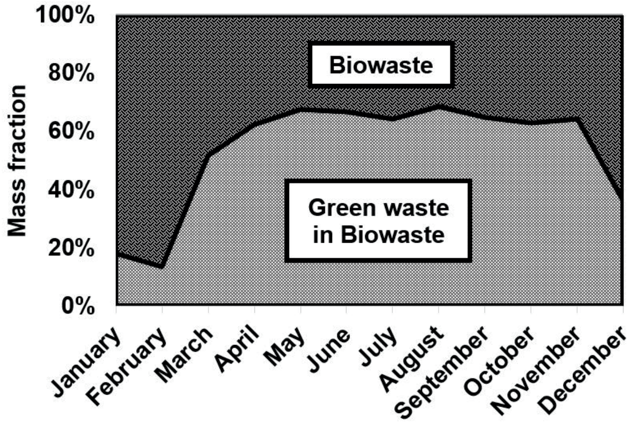

FIGURE 3: Calculated share of green waste in biowaste (2016 to 2018).

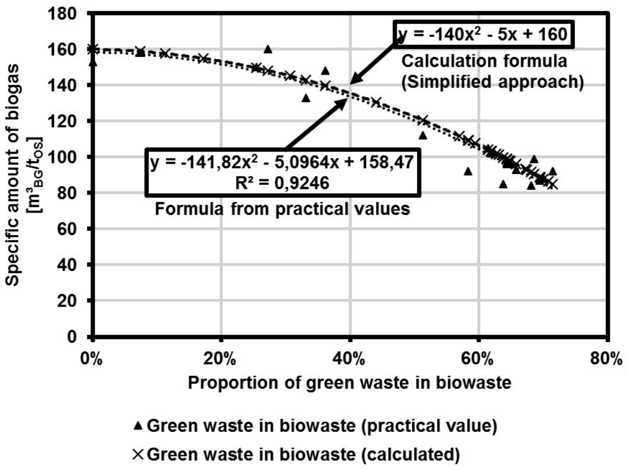

FIGURE 4: Specific biogas yield as a function of the proportion of green waste in biowaste (2016 to 2018).

sed on the data of the postulated calculation method are heavier and the curve is leptokurtic (Weaver, 2018).

The results of this research strongly support the assumption that the proportion of green waste in biowaste is the dominant influencing parameter of the specific biogas yield. Furthermore, the calculation method is suitable for and adaptable to other plants.

Figure 5 shows the regional characteristics of the studied region. As already mentioned, the specific biogas yield is strongly dependent on the proportion of green waste in biowaste. Although the substrate-specific biogas yield is lower in times with high vegetation rate, the biogas yield per inhabitant is higher due to the increasing proportion of green waste in biowaste. These key data could be used as a basis for the design of future fermentation plants.

When comparing the plant capacity and the achievable gas yield from biowaste over the year, only one possible variant with a maximum co-substrate input of 10% per day should be investigated because this variant represents the preferred variant for plant operation. If more than 10% co-substrates are used per year, the economic consequences for the operator would be negative because the feed-in tariff would decrease. The possibility of time-independent co-substrate use is not further discussed in this paper.

5. LABORATORY RESULTS

Batch tests are featured by one-time feeding of substrate and the logging of amount and quality of the biogas over the test period. Samples for the batch tests are characterised by urban and rural settlement structures and comes from the separate collection of biowaste in Schwerin (the regional capital of Mecklenburg-Western Pomerania). The test site at the University of Rostock is equipped with several batch-fermenters of 30 litres each. The tests are carried out at mesophilic temperatures (37°C). Each sample is tested in a sixfold determination. The fermenter content is mixed by magnetic stirrers. The produced biogas is stored in special gas sampling bags which are deflated regularly. While deflating, the volume and composition of the produced biogas are measured. The volume is measured by a RITTER drum-type gas meter. For the representation of the results, the gas volume is converted to volume under normal conditions (273 K, 1013 hPa). For the measurement of the gas composition, an EHEIM gas analyser with sensors for CH_4, CO_2, O_2 and H_2S is used. The biowaste sample per batch fermenter is about 600 g by using fermentation residue from a waste water treatment plant as inoculum. Due to the high sample mass of 600 g untreated heterogeneous biowaste, influences of conditioning which can cause losses of volatile substances, are excluded, as far as possible.

Batch tests were carried out according to the German directive VDI 4630 (VDI, 2016) for each season to support the thesis that seasonal fluctuations in urban and rural areas have different effects on biogas yield (Table 3).

The table clearly shows that the biogas yield from urban areas is more constant and higher. The average median value of rural biowaste is 109.79 m^3_{BG}/t_{OS} with a variance of 411 compared with urban biowaste with a median value of 134.51 m^3_{BG}/t_{OS} and a variance of 249. A biogas yield between 27 and 168 m^3_{BG}/t_{OS} can be achieved with biowaste (ATV-DVWK, 2003).

During the winter months, as can be seen from Figure 2, the proportion of green waste in organic waste decreases. For this reason, biogas production is quite similar and the gas yield is almost the same for both substrates in this period. The increasing proportion of green waste can explain the fluctuation in rural biowaste.

The methane content of the biogas is very similar for both substrates. According to ATV-DVWK (2003) the average biogas yield is between 55 and 65 m^3_{BG}/t_{OS}. The dry matter content varies between 34.3 and 49.7, which is typical for biowaste.

After reaching the abort criteria (the daily increase of biogas volume is lower than 1%) the batch test was stopped. The divergent specific biogas yield compared to the plant (92 versus 160 m^3_{BG}/t_{OS}) can be explained by the residence time of 32 to 47 days in the batch test.

TABLE 2: Descriptive data analyses of the biogas yield.

		Plant values [m^3_{BG}/t_{OS}]		Calculated values [m^3_{BG}/t_{OS}]	
		Statistic	Std. error	Statistic	Std. error
Mean		109.22	7.23	110.36	7.33
95% Confidence interval for mean	Lower bound	93.30		94.23	
	Upper bound	125.15		126.48	
5% trimmed mean		107.59		109.02	
Median		98.12		101.05	
Variance		628.11		644.18	
Std. deviation		25.06		25.38	
Minimum		87.98		84.84	
Maximum		159.91		160.00	
Range		71.93		75.16	
Interquartile range		34.66		42.48	
Skewness		1.37	0.64	1.13	0.64
Kurtosis		0.43	1.23	- 0.17	1.23

6. CONCLUSIONS

Statements about the influence of green waste in biowaste on the specific biogas yield are possible. The presented method provides a regional analysis of biowaste flows and an estimation of the expected specific biogas yield. In comparison to previously used guideline values, this method allows better planning of new biowaste fermentation plants. Based on these results, the use of co-substances in existing plants can be optimised and the baseload capacity of the fermentation plant can be increased.

Furthermore, there are several variants to reduce the proportion of green waste in biowaste. For example, mechanical treatment (sieving, sorting) and the separate col-

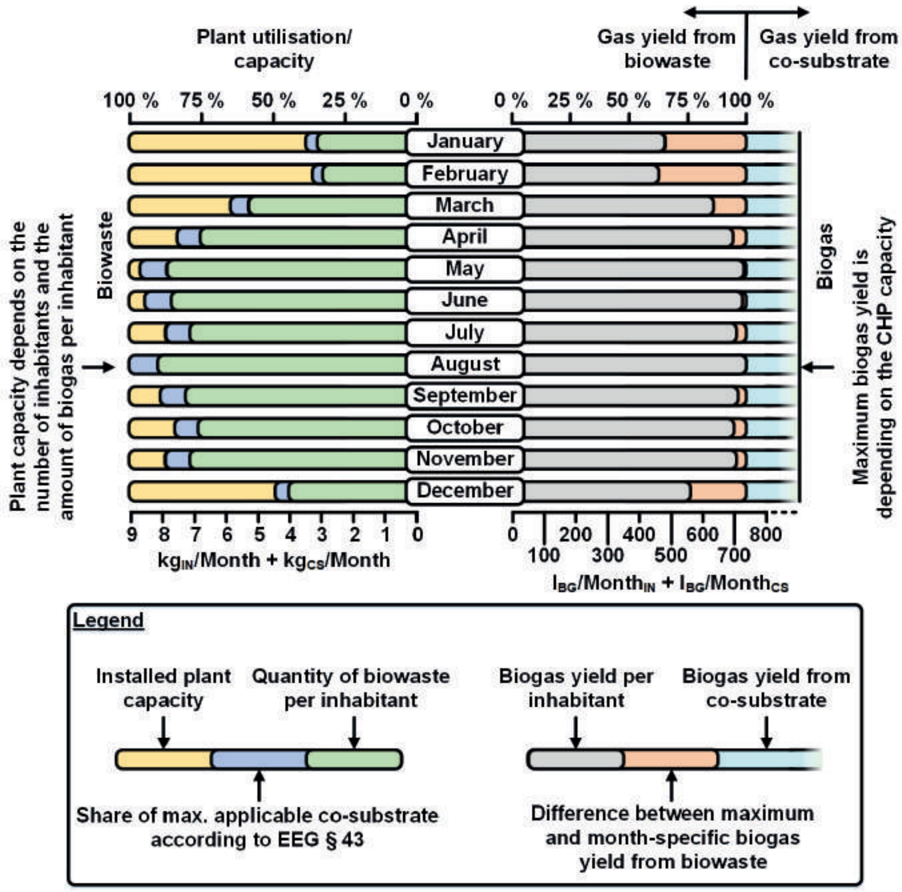

FIGURE 5: Comparison of plant capacity and gas yield of a biowaste fermentation plant (regional specific parameters).

TABLE 3: Median values for biogas yield, dry matter and methane content of urban and rural biowaste.

Season	Region	Dry matter of biowaste [%]	Biogas yield [m^3_{BG}/t_{OS}]	Methane content [Vol.-%]
Winter	Urban	34.9	137.56	59.03
	Rural	34.3	138.00	59.99
Spring	Urban	35.6	123.94	63.77
	Rural	47.8	107.80	57.48
Summer	Urban	43.5	136.07	62.96
	Rural	49.7	100.05	57.69
Autumn	Urban	40.7	135.55	57.64
	Rural	36.3	113.66	56.03

lection of green waste are conceivable. The aim of these ideas is to increase the specific biogas yield and to reduce the impact of green waste on the quantity and quality of biowaste. Benefits are efficient biogas production and constant reaction conditions. Furthermore, the microorganisms in the fermenter have a higher security against process disturbances. Moreover, with these methods, the use of co-substrate can be optimised and the efficiency of the whole process setup increase.

However, the chosen method has several limitations. Firstly, the retention time in the fermenter is assumed to be constant. Especially in the winter months, the residence time increases with decreasing quantity of biowaste. Secondly, the region-specific evaluation results from a specific mix of rural and urban biowaste. In other regions the evaluation would be different. The method refers only to dry fermentation of biowaste under thermophilic temperature conditions. In order to make statements about the influence of co-substrates on the process, further investigations are necessary.

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ENERGY SUSTAINABILITY OF SUPPLY CENTERS FROM THE CODIGESTION OF ORGANIC WASTE

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ABSTRACT

The present study evaluated the potential for biogas generation from organic waste unfit for consumption collected at the Pernambuco Supply and Logistics Center (CEASA-PE), as well as an estimation of the electric energy production from waste biodigestion generated in this plant. The biogas generation potential used BMP bench-test (biochemical methane potential) biodigesters, in which the biogas production was qualitatively/quantitatively evaluated from CEASA waste codigested with different inocula (bovine ruminal fluid, goat manure and UASB reactor sludge) under mesophilic conditions for 80 days. The laboratory test showed that the waste inoculated with ruminant manures (goat and bovine) presented the lowest net volume of generated biogas compared to the other treatments: 23.2 and 19.2 $NmL.g_{ds}^{-1}$. On the other hand, the treatments with sludge and mixture of all the inocula generated the largest biogas volumes: 37.6 and 44 $NmL.g_{ds}^{-1}$, respectively. A potential of 359 $kWh.d^{-1}$ of electric power generation was estimated from the biogas generated at CEASA from the BMP bench assay, as well as from the quantitative and physicochemical parameters of the waste generated in this unit. The results indicate high potential for energy generation in the evaluated CEASA plant, promoting the recovery of these wastes as an alternative and renewable source for sustainable energy production, transforming environmental liabilities into potentially energetic assets of aggregate economic value.

1. INTRODUCTION

The search for alternative energy sources, preferably cleaner, renewable and sustainable, has become a constant concern on the part of researchers and businesses alike (Schirmer et al., 2016). In this scenario, the use of biomass assumes a relevant role in the gradual replacement of an energy matrix based mainly on the use of fossil fuels (such as petroleum and coal) by a more environmentally friendly matrix. The use of biomass is linked to important factors such as: a) an increase in the global supply of energy; b) the valorization of organic waste, mitigating the social and economic problems associated with its disposal and treatment; c) reduced greenhouse gas emissions, since anthropogenic activities such as handling manure management, wastewater treatment, and landfill management are strongly related to greenhouse gas emission (Abbasi et al., 2012; WEC, 2013). Accompanying this scenario along with the National Policy on Climate Change itself (2009), the Brazilian federal government launched the National Biofuels Policy (RenovaBio, in Portuguese) in 2017, which aims to (among other commitments) recognize the strategic role of all types of biofuels in the energy matrix of Brazil, both in order to guarantee energy and to minimize greenhouse gases (Brasil, 2017).

Anaerobic digestion is a consolidated biotechnology in recovering organic effluents/waste and comprises the degradation of the organic matter with the consequent production of biogas, a mixture of methane, carbon dioxide, and hydrogen sulfide in percentages which can vary according to the precursor substrate and the process conditions (Cabbai et al., 2013; El-Mashad and Zhang, 2010; Lastella et al., 2002). According to Esposito et al. (2012), sewage sludge, animal manure, food waste, organic solid waste from markets and households, etc. are some of the commonly used substrates in anaerobic digestion processes.

Supply centers represent great potential for generating solid organic waste, and therefore a source of residual bi-

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omass that can be used in energy generation. Considering only the Pernambuco Supply and Logistics Center (CEASA-PE) and based on the 2017 Solid Waste Management Plan, there was a monthly solid waste production of about 1,100 tons; 90% of which were organic in nature (CEASA-PE, 2017).

The efficiency of the anaerobic digestion process depends on factors such as characteristics of the biodigested waste (volatile solids and nutrients' content), type of biodigestor and the operation parameters (pH, temperature, buffering capacity, etc.) (Alkaya and Demirer, 2011; Schirmer et al., 2014). This efficiency can be improved in terms of the generated biogas yield in the biodigestion process by using a co-substrate (Mata-Alvarez et al., 2000; Mata-Alvarez et al., 2014). Codigestion is the result of anaerobic digestion of two or more substrates with the objective of improving the efficiency of the biodigestion process, maximizing the methane production (Álvarez et al., 2010).

The BMP assay (biochemical methane potential) has been extensively used by researchers as a method to measure the biodegradability of a sample. Low operational cost and a fast response are some of the advantages commonly associated with this method (Barbosa et al., 2018; Labatut et al., 2011; Owen et al., 1979; Schirmer et al., 2014). The literature has reported the anaerobic digestion of the most varied substrates and co-substrates for biogas production: organic fraction of municipal solid wastes inoculated with wastewater sludge and agroindustrial anaerobic sludge from treatment plants (Maciel and Jucá, 2011; Oliveira et al., 2018; Schirmer et al., 2014); organic fraction of municipal solid wastes inoculated with bovine and swine manure (Barbosa et al., 2018); fresh samples of fruits and vegetables wastes (Gunaseelan, 2004), olive mill solid wastes (Rincón et al., 2013), meat-processing wastes (Cavaleiro et al., 2013), among others with high biogas generation potential.

The present work follows the premise of solid waste recovery for energy production, aiming to evaluate the potential of biogas generation from the codigestion of different substrates. The enormous amount of the waste addressed herein generated in Brazil (supply centers, sewage treatment plant sludge and animal waste) should be evaluated as an alternative and renewable source for sustainable energy production.

1.1 Abbreviations

- BMP: biochemical methane potential
- BRF: bovine ruminal fluid
- E: potential of electric power generation
- GM: goat manure
- LHV: lower heating value of the biogas
- OSW: organic solid waste
- P_{biogas}: daily biogas production
- PE: Pernambuco (State of Brazil)
- SL: sludge
- STP: standard temperature and pressure
- TCD: thermal conductivity detector
- UASB: upflow anaerobic sludge blanket
- VFA: volatile fatty acids
- VS: volatile solids content
- WWTP: wastewater treatment plant
- $\eta_{generator}$: generator yield

2. MATERIAL AND METHODS

2.1 Substrate and inocula: samples preparation

Organic solid waste (OSW, adopted as the main substrate) was collected at the Pernambuco Supply and Logistics Center (CEASA-PE) and basically consisted of fruits, greens and vegetables unfit for human consumption. Three inocula were evaluated for OSW: bovine ruminal fluid (BRF) and goat manure (GM), both collected at the Regional Abattoir of Paudalho/PE, and sludge from the UASB reactor (upflow anaerobic sludge blanket) of the "Dancing Days" wastewater treatment plant (WWTP), of Recife (PE) (BRK Ambiental). Once collected, the substrate (OSW) and sludge (SL) were preserved in plastic containers at 4ºC (CETESB/ANA, 2011), while the animal waste (BRF and GM) was kept at room temperature until the samples were processed.

The OSW was fragmented and quarantined in the laboratory according to NBR 10007 (ABNT, 2004). The OSW and inocula were then dried at 105°C in a greenhouse until no further mass variation was observed to determine the initial moisture of the waste (WHO, 1978). After drying, the waste was milled in a Wiley mill in order to guarantee homogeneity of the samples and to increase their contact surface area.

2.2 BMP assays

The relative volume of inoculum used in the experiment can vary widely, depending on the characteristics of the substrate and the inoculum itself (Angelidaki et al., 2009). In this study, each treatment consisted of 5 grams of OSW (dry and ground) plus inoculum (SL, BRF, GM) until a volume of 50 mL was reached in each biodigester. In the case of the OSW+GM treatment, distilled water was added to the mixture in sufficient quantity to guarantee sample homogeneity and a moisture content close to 80% (similar to other mixtures).

Once filled, the biodigesters (250 mL borosilicate vial) were sealed with nylon lids equipped with a manometer (to monitor biogas generation) and valves for the discharge of the biogas generated in the headspace of the vial. In summary, the following treatments were evaluated, all in triplicate: OSW+SL, OSW+BRF, OSW+GM, OSW+SL+BRF+GM and blanks (biodigesters containing only the inocula, whose biogas production was subtracted from the production of the mixtures). The anaerobiosis of the medium was guaranteed by recirculating N_2 (inert gas) for 1 minute in the headspace of each flask (Mshandete et al., 2004; Schirmer et al., 2014). The biodigesters were kept under mesophilic conditions (37°C) for 80 days. The following physicochemical parameters of the treatments were determined before and after the biodigestion period: moisture, volatile solids – VS (NBR 13999; ABNT, 2003) and pH, according to the Standard Methods for the Examination of Water and Wastewater (APHA, 2005).

2.3 Monitoring and characterization of biogas

The biogas generated in the biodigesters was monitored daily from the manometer pressure readings of each flask, as well as atmospheric pressure and incubation temperature. The pressure values were converted to the biogas volume in standard temperature and pressure (STP) (Labatut et al., 2011). The concentrations of the main components of the biogas (methane - CH_4, and carbon dioxide - CO_2) were determined in a gas chromatograph (Appa Gold) with a thermal conductivity detector (TCD) and Porapak N packed column (80/100 mesh; 3.0 m length). The temperatures of the injector, detector and furnace were 140°C, 150°C and 60°C, respectively, with a hydrogen flow of 30 $mL.min^{-1}$. The biogas samples were collected at the end of the test (80th day) in a 1 mL syringe and injected into the chromatograph for analysis.

2.4 Estimation of the electric energy production from the anaerobic biodigestion of CEASA waste

The potential for generating electric energy at the CEASA-PE plant (Recife) was estimated from the best biogas production conditions and final methane concentration during the BMP assays. Table 1 shows the quantity and characteristics of waste generated at CEASA.

Kunz and Oliveira (2006) reported a method for calculating the daily production of biogas relating the biochemical methane potential (BMP) to the concentration of volatile solids in the waste and its daily flow, according to Equation 1:

$$P_{biogas} = BMP * C_{VS} * Q_{dig} \quad (1)$$

In which:
P_{biogas} = daily biogas production ($m^3.d^{-1}$)
BMP = potential for biogas generation ($m^3.kg_{VS}^{-1}$ or $m^3.kg_{ds}^{-1}$)
CVS = volatile solids ($kg_{vs}.m^{-3}$)
Qdig = daily waste generation ($m^3.d^{-1}$)

It is possible to determine the potential of electric energy generation from the daily biogas production data, meaning the amount of energy which can be obtained by converting the biogas to electric energy using a gas generator.

According to Coelho et al. (2016), the potential electric energy generation can be determined through the daily biogas production (considering the efficiency of the biodigestor), its lower calorific value, as well as the yield of the generator that will be used (Equation 2).

TABLE 1: Quantitative and physicochemical waste parameters from CEASA (PE).

Parameter	
Daily waste generation* (ton.d^{-1})	36.7
Volatile solids ($kg_{VS}.m^{-3}$)	162.5
Total solids ($kg_{ds}.m^{-3}$)	215
Moisture (%)	78,5
Potential for biogas generation** ($NmL.g_{VS}^{-1}$ or $NmL.g_{ds}^{-1}$)	27.1 or 20.5

* CEASA (2017)
** Determined from the BMP assay

$$E = P_{biogas} * LHV_{biogas} * \eta_{generator} \quad (2)$$

In which:
E = potential of electric power generation ($kWh.d^{-1}$)
P_{biogas} = daily biogas production ($Nm^3.d^{-1}$)
LHV_{biogas} = lower heating value of the biogas ($kWh.Nm^{-3}$), obtained from NBR 15213 (ABNT, 2008)
$\eta_{generator}$ = generator yield (%); efficiency of electricity generation from biogas of approximately 40% (Scarlat, Dallemand and Fahl, 2018; Van Foreest, 2012).

3. RESULTS AND DISCUSSION

3.1 Evaluation of physicochemical waste parameters

Table 2 presents the mean values of the parameters moisture, pH and volatile solids of the evaluated treatments before and after the biodigestion period.

The moisture and pH of the waste mass are among the main parameters related to biogas generation. The literature has reported several ranges of moisture content as being more appropriate to biodegrading solid waste, depending on the conditions under which such degradation occurs (Eck, 2000). USEPA (1991) reports that high moisture contents (60-90%) tend to favor the biogas generation rate. Table 2 shows that all treatments started from values close to 80% (in the start-up of the biodigesters); the small increase in moisture content observed at the end of the process is most likely due to the conversion of biomass (solid) to gas during the waste biodigestion.

Concerning pH, literature reports a wide range of values as favorable for methane generation: between 6.7 and 7.5 (Deublein and Steinhauser, 2008), 5.5 and 8.5 (with an optimum range at 7.0-8.0) (Al Seadi et al., 2008), and so on. In a typical anaerobic digestion, the formation of volatile fatty acids (VFA - intermediate compounds such as acetate, propianate, butyrate) in the acidogenesis stage and the high carbon dioxide concentration observed in the biodigestion process determine a decrease in the pH of the medium (Al Seadi et al., 2008; O'Leary and Tchobanoglous, 2002). An increase in pH occurs in the subsequent biodigestion stages, and the system tends to return to neutrality, which in fact was observed in the final pH values reported in Table 2 after 80 days of incubation and for all the treatments, regardless the mixture OSW/inoculum. Substrates with a high proportion of animal manure (as in the treatment OSW+GM) have a greater accumulation tendency of these acids (VFA) in the mixture, implying in reduced pH values of the system (Al Seadi et al., 2008). However, this higher accumulation of VFA in the medium does not seem to have compromised the biogas production (with an inhibition properly said) but only slow down the return of medium pH to neutrality (at least within the evaluated period - 80 days), which may justify the slightly lower value (6.2) for the OSW+GM mixture (compared to the other mixtures).

All initial values of VS reported in Table 2 are higher than 60%, indicating a high biogas generation potential from the evaluated substrates. However, a low conversion percentage was observed in all treatments (16.4; 5.2; 16.5 and 20.1% for OSW+SL, OSW+BRF, OSW+GM e

TABLE 2: Physicochemical parameters of waste stabilization.

Parameter	Treatment							
	OSW + SL		OSW + BRF		OSW + GM		OSW + SL + BRF + GM	
	Initial	Final	Initial	Final	Initial	Final	Initial	Final
Moisture (%)	84.4	85.7	88.0	89.0	79.2	83.0	78.5	79.2
pH	7.2	7.6	7.1	7.5	7.7	6.2	7.5	8.2
VS (%)	67.8	56.7	63.4	60.1	78.9	65.9	75.6	60.4

OSW: organic solid waste
SL: sludge
GM: goat manure
BRF: bovine ruminal fluid

OSW+SL+BRF+GM, respectively), indicating a high residual potential of degradation, even after the 80 days of incubation. Thus, the final VS values observed were still high. According to Decottignies et al. (2005), a waste can be considered stable (degraded) when it presents VS content to the order of 10 to 17.4%. The VS reduction values reported in Table 2 are much lower, for example, than those found by Schirmer et al. (2014), which observed reductions of 43.2 and 47.4% for fresh and 1-year-old wastes (both inoculated with WWTP sludge), respectively, in the same digestion period (80 days) in BMP assays under mesophilic conditions and a substrate/inoculum ratio close to that used in the present study. Thus, although the volatile solids content is among the main parameters to evaluate the waste biodegradability, some substrates can show a very slow degradation kinetics under anaerobic conditions. For these cases, the literature has reported several methods for the pretreatment of these residues (physical, chemical, thermophysical, thermochemical, biological) in order to enhance the biogas production. However, none of these techniques were employed in the present study so that we could analyze the viability of biogas generation from the "in natura" use of all evaluated substrates and inocula, allowing a better comparison between the biogas generation potential of each one.

3.2 Biogas monitoring

3.2.1 Biogas generation

The OSW inoculated with ruminant wastes (GM and BRF) presented the lowest generated net biogas volume compared to the other treatments: 116 and 96 NmL, or 23.2 and 19.2 $NmL.g_{ds}^{-1}$ (volume of biogas per dried solid mass) for OSW+GM and OSW+BRF, respectively. Regarding to the manure treatment (GM), a low biogas production can be attributed to the large amount of ammonia, which constitutes an important anaerobic digestion process inhibiting agent present in animal waste (Yenigün and Demirel, 2013), and to the high lignin content, also present in the animal manure (a lignocellulosic substrate) and that affects negatively the biodigestion process. It is known that the presence of inhibitory agents (ammonia, sulfides, heavy metals, etc.) is strongly related to low biogas production. In turn, the rumen constitutes a complex ecosystem of microorganisms that inhabit the animal's gastrointestinal system (fungi, protozoa, bacteria, etc.) (Mackie, 2002) and the animal's own diet has a direct effect (qualitative and quantitative) in ruminal microbiology (Wlodarski et al., 2017). A feed based on silage, pasture, feed, etc. can influence biogas generation, since many of the heavy metals and sulfur compounds come from feed and/or chemical supplements administered to the animal (Barbosa et al., 2018). However, no microbiological analysis of rumen or inhibitor compounds was performed in the present study in order to justify a greater or lesser generation of biogas from this inoculum.

On the other hand, treatments with sludge (SL) and mixture of all inocula (SL BRF+GM) generated the largest biogas volumes: 188 NmL and 220 NmL (37.6 and 44 $NmL.g_{ds}^{-1}$, or 23.1 $NmL.g_{vs}^{-1}$ and 27.1 $NmL.g_{vs}^{-1}$, respectively) for the OSW+SL and OSW+SL+BRF+GM treatments, respectively. The higher biogas volume observed in the treatment with all mixed inoculants is most likely due to the synergy between the wastes and the microorganisms present in the medium promoted by the codigestion of several substrates and also the dilution of inhibitory agents.

Figure 1 presents the daily biogas generation rate ($NmL.d^{-1}$) of each evaluated treatment. Biogas generation from their respective blanks has already been subtracted from these results.

The rate of biogas production followed the curve pattern reported by Al Seadi et al. (2008) for batch tests. The biogas peaks observed in the first days of experiment are due to the hydrolysis of easily degradable compounds; in addition, the high inoculum/substrate ratio adopted in all treatments also contributed to this behavior (Parawira et al., 2004). Most of the generated biogas occurred within the first 30 days of incubation in all evaluated treatments. In this aspect, OSW treatment with all inocula was shown to be the most uniform in terms of daily generation, with a more distributed generation throughout the period of the sample digestion compared to the other treatments.

It's important to highlight that the comparison of biodegradability curves found in the literature may be a hard task (Angelidaki et al., 2009), once the specific conditions of each experiment (temperature, time of incubation, pretreatment of the substrates, pH, moisture, nutrients, etc.) as well as the kind of substrates can vary a lot. Besides the biodegradability curves, biogas generation values found in the literature can vary greatly, even for similar digestion conditions and substrates. In the present study, the biogas volumes generated for the best mixtures (OSW+SL and OSW+SL+BRF+GM) were relatively low. As already above-mentioned, we didn't employ any technique to pre-

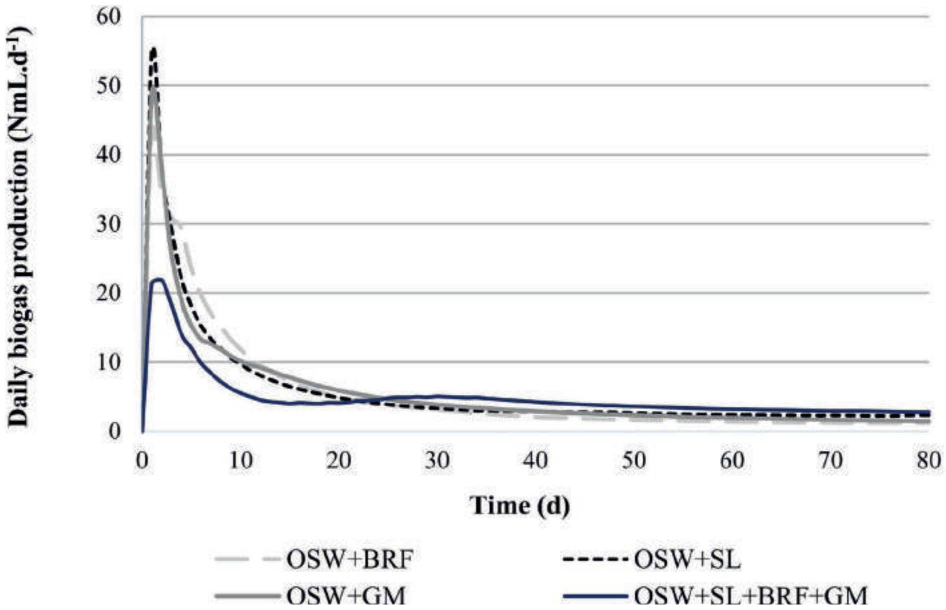

FIGURE 1: Daily biogas generation rate of the treatments evaluated in the biodigestion period.

treat these residues so that we could compare the biogas generation potential without any modification of their original features. Besides, several reasons can also be related to a "lower" biogas generation and that were not evaluated here, such as the concentration of nutrients in substrates and inocula, microrganisms population, lignocellulosic content and the presence of inhibitor or toxic agents in the biodigesters (as ammonia, heavy metals, sulphide, etc.) that, even in a very small concentration, may interfere with the quality of biodigestion.

3.2.2 Methane and carbon dioxide concentrations

Table 3 presents the biogas characterization at the end of the experiment for the four evaluated treatments.

Biogas typically presents methane (CH_4) concentrations in a range between 55-70%, and carbon dioxide (CO_2) in the range of 30 - 45% (Deublein and Steinhauser, 2008). However, Rasi et al. (2011) reported that biogas composition can vary greatly depending on the precursor substrates and the biodigestion process conditions (humidity, temperature, pH, etc.). This composition tends to vary, even during biodigestion, depending on the waste degradation phase (acidogenesis, methanogenesis, etc.). For example, in investigating the anaerobic biodegradation of potato residues alone and codigested with beet leaves inoculated with anaerobic digester sludge in different proportions, Parawira et al. (2004) obtained a very distinct behavior in terms of final methane composition, with concentrations varying between ≅10 and ≅70%.

Table 3 shows that the OSW+SL and OSW+BRF+SL+GM treatments were the only ones that presented a final methane concentration within the 'theoretical range' (55 - 70%). As observed in biogas generation, OSW inoculated with ruminant manure (GM and BRF) presented the lowest final levels of methane in the biogas. In a study on anaerobic biodigestion of manure produced by five animal species (layer poultry, cattle, goats, broiler chickens and pigs), Orrico Jr. et al. (2011) also generally found a low observed methane potential (in $L.kg^{-1}$ of volatile solids of manure) for ruminant manure (bovine and goat).

Animal manure is associated with a very low methane yield (Al Seadi et al., 2008). Besides, some of the previously mentioned factors such as the presence of inhibitory agents in animal feed, ammonia present in goat manure, sulfur compounds formed during biodigestion, etc., among other important process parameters such as medium buffering capacity and organic load control (to avoid system overload) have direct interference with the 'success' of the methanogenic biodigestion phase. The combination of some of these parameters, to a greater or lesser extent, could justify the 0.0% methane found in the biodigestion of the OSW + GM treatment.

Thus, the control of these parameters can be fundamental in cases of mixtures (codigestion) with specific substrates, such as animal waste. In this context, the literature suggests some measures of process control, and consequently optimizing the biomethanization of these wastes (Pearse et al., 2018). Although cheaper and simple technology, batch biodigesters (like BMPs, used in the present study) have lower yields in terms of biogas generation (Vandevivere et al., 2002; Ward et al., 2008), as well as not allowing operator interference in eventual corrections of process parameters. Biomethanization of organic waste comprises a series of steps (hydrolysis, acidification, etc.) for methane formation. In single-stage systems, all these reactions perform in a single digester, whereas this sequence of biochemical reactions occurs in at least two reactors in systems of two or more phases (Vandevivere et al., 2002). The biodigestion phase separation (hydrolysis and acidogenesis of acetogenesis and methanogenesis) is an important step in optimizing the biodigestion process (Ward et al., 2008), and could be applied to similar studies to that proposed herein in order to allow greater monitoring and control over the process steps, and thus a larger gen-

TABLE 3: Average concentrations of CH_4 and CO_2 at the end of the experiment.

Treatment	$[CH_4]$, in %	$[CO_2]$, in %
OSW + BRF	41.2	58.8
OSW + SL	55.7	44.3
OSW + GM	0.0	100.0
OSW + BRF + SL + GM	59.8	40.2

eration of biogas/methane.

3.3 Estimation of electric energy production from the waste biodigestion from CEASA

The estimated biogas production was 161 m³.d-1. Considering NBR 15213 (ABNT, 2008) and the methane content in the biogas experimentally determined for the best condition of the BMP assay (≅60%, in the OSW+BRF+SL+GM treatment), LHV determined for the biogas was 4,796 kcal. Nm-³ (or 5.6 kWh.Nm^{-3}). According to the scenario adopted in the present study, the OSW generated at CEASA (PE) has potential for generating 359 kWh.d^{-1} of electricity from the biogas generated in this plant. Therefore, the application of a biodigestor for energy use in a plant with large organic waste generation (as CEASA) constitutes a great alternative in relation to the reducing the expenses of this unit with electric energy.

However, in practice, a technical and financial feasibility study is needed in order to obtain more project details, such as the payback time of implementing the proposal, for example. In addition, one of the main points to be addressed in a feasibility study is biogas purification technology. An analysis of the gas concentrations which compose the biogas is essential for the intent of energy utilization (defining the biogas purification technology to be implemented). In addition to methane (the main energy component), biogas is composed of carbon dioxide, water vapor (substances that reduce burning efficiency), and hydrogen sulphide, a highly corrosive compound that implies a reduction in generator life. Two technologies commonly used in the biogas purification stage are water scrubbing and adsorption with activated carbon (Bauer et al., 2013), which could be adopted in this study (CEASA) to obtain a biogas with better quality in terms of methane and being free from sulphur compounds.

4. CONCLUSIONS

There were quite different behaviors in terms of biogas generation in the biodigestion evaluation stage of the OSW codigested with bovine ruminal fluid, goat manure and UASB sludge under mesophilic conditions. Wastes only codigested with ruminant manure (goat and bovine) had significantly lower net volumes of biogas (up to 56% lower) compared to the other treatments. Similarly, methane concentrations determined at the end of the biodigestion period were greater than 55% for treatments with sludge and all mixed inocula, remaining at 41% for treatment with bovine rumen or even 0.0% for treatment using only the goat manure as inoculum. Factors related to the presence of inhibitors or a higher lignin content in animal inocula present in feed and manure may be related to the lower biogas generations. On the other hand, the greater biogas volume observed in the treatment with all mixed inoculants is most likely due to the synergy between the wastes (carbon content, nutrients, etc.) and the microorganisms present in the medium promoted by the codigestion of these substrates, as well as to the dilution of these inhibitory agents.

About the electric energy production from the biogas volume generated on the scenario evaluated, a potential of electric power generation of 359 kWh.d-1 (or 10.76 MWh.m^{-1}) was determined from the biogas generated at the evaluated CEASA plant. It should be remembered that this study is a theoretical estimate of the potential for biogas generation and electric energy obtained therefrom, and that ideal conditions for reactor operation are adopted in bench-scale tests, which may imply higher values than those obtained in practice.

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CONVERSION OF BIOLOGICAL TREATMENT PLANT SLUDGE TO ORGANIC FERTILIZER FOR APPLICATIONS IN ORGANIC FARMING

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ABSTRACT
Conversion of the sludge generated in the biological treatment plants of glove dipping industries of Sri Lanka to a valuable organic fertilizer after removing toxic metal ions such as heavy metals and excess Zn and Al present in the sludge to allowable limits is described. In order to do so, the raw materials used were analysed for these species and for their nutritional values. Removal of metal ions by different acids such as HNO_3 and acetic acid digestion processes are revealed and the results are compared. Dilution of the metal ion-removed sludge with other raw materials used in organic fertilizer production to enable maintain right C:N ratio and the use of these materials in the fertilizer production process used are presented. Application of the fertilizer to soils of fruit and vegetable plantations and measurement of Zn, Al and heavy metals in the soil and plant parts and their crops as a function of time is also described. The quality of vegetables and fruits produced by applying this new fertilizer is compared with those obtained in the open market. It has been shown that the former contains no or much less than maximum allowable limits of heavy metals or toxic species when compared to those grown using other organic fertilizers. This study is useful for industrial biological treatment plant sludge management by converting it to a useful product.

1. INTRODUCTION

Sri Lanka has been placed one of the top-class glove manufactures of the world contributing to more than 5% of global demand (Zheljazkov and Nielson,1994). The main process used in the rubber glove manufacturing industry is called dipping. Dipping industry uses a preserved concentrated natural rubber. The dry rubber content in latex is approximately 60% and the ammonia content varies from 0.2% to 0.7% on latex by weight (Kerdtongmee et al., 2014). A latex compound is prepared by adding chemicals to the latex as a colloidal solution, dispersion or as an emulsion. The additives include fatty acid soap, zinc oxide, accelerator chemicals (zinc dibutyldithiocarbamate ZDBE), alkylated phenols and inorganic pigments. Dipping process begins with cleaning the porcelain formers and finishes with stripping of the gloves from the formers. Technology adopted for glove manufacturing is well established and widely used by latex glove manufacturers, throughout the globe, particularly in Sri Lanka. However, environmental consideration has not yet been fully incorporated in the design of the dipping process which resulting in significant environmental concerns and problems (Blackley, 1997).

In rubber glove manufacturing, there are two main sources of effluent; namely, latex compounding waste and leaching tank discharge. Latex compounding waste contains un-coagulated latex and chemical sludge (Devaraj et al., 2017). In the manufacturing process, leaching with water is used to remove water-soluble substances coated on the newly-formed gloves. On some of the lines, a post-leaching has been incorporated as it helps to further reduce the water-extractable protein on the surface of the gloves. Latex compounding effluent is treated using a flocculent to separate out latex particles from the effluent. The latex coagula thus obtained is an excellent source of fuel energy and is used as a raw material to obtain energy for the cement industry (Naqi and Jang, 2019). Leaching tank effluent, which does not usually contain latex particles, can be discharged directly into the holding tank, bypassing the rubber trap. The wastewater contains excess zinc and aluminium where zinc comes from the additives used and aluminium from the alum used as flocculent. Rubber products

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also use various colouring agents and pigments which may contain heavy metals. Finally, the wastewater is treated in the microbial biological treatment plants. As of now, there is no reliable technology for disposal of the sludge generated at these biological treatment plants.

As such, a reliable method to dispose this sludge without a significant adverse effect to the environment is essential. Hence, this study was carried out to convert the biological treatment plant sludge (BTPS) that contains aqueous dispersion of dead bacteria from rubber processing plants to a valuable organic fertilizer. In order to accomplish this goal, BTPS of the Natural Rubber Processing Plants in Sri Lanka (NRPP of SL) which has 2% Total Dispersed Solids (TDS), were collected in large scale, to meet with the requirement of 1800 kg of dry sludge. Major requirement, in this case, is the analysis to be done for (i) available metal ions such as Zn^{2+}, Al^{3+}, and heavy metal ions which have been contaminated from various dyes and pigments used and (ii) nutritional value, in terms of C, N, P and K contents and C:N ratio. The metal ions, if present in excess, are to be removed by extraction, down to below the maximum allowable limits and the C:N ratio is to be adjusted to required ranges using other raw materials used in organic fertilizer manufacturing. This was done first by the traditional leaching method, using nitric acid as the leaching agent, where the supernatant solution of the sludge then contains nitrate anion. The removal of nitrate from water requires adsorptive methods since nitrates cannot be easily precipitated out (Turhanen et al., 2015). Ion-exchange (Cummins 2019) and reverse osmosis (Pangarkar et al., 2011) are other possibilities but these techniques are expensive. In order to circumvent this problem and to make the problem advantageous to the industry, we used acetic acid to remove these contaminants (Albers et al., 2011). This was prompted due to several reasons: (i) just like most metal nitrates most of the metal acetates including lead acetate is soluble in water (solubility of lead acetate in water, at 20 °C is 0.61 mol dm^{-3}), (ii) acetic acid is a commonly used acid in glove manufacturing industries and it can be used as a food to bacteria present in the biological treatment plant thus preventing its environmental contamination, (iii) in the acidic medium prevailing in acetic acid treated wastewater, aluminium triacetate is preferentially formed when compared to Aluminium triacetate is highly soluble in water and hence aluminium present in waste sludge can be totally extracted with acetic acid (v) zinc is the next most abundant contaminant next to aluminium in the sludge and is highly soluble in acetic acid [solubility of zinc acetate in water, at 20 °C, is 2.34 mol dm^{-3}] and (iv) both nitrate and acetate are hard Lewis bases and, according to the Pearson's Hard and Soft Acid Base Principle, hard Lewis bases prefer hard Lewis acids. Both these anions prefer hard Lewis acids such as H^{+}, Li^{+}, Na^{+}, K^{+}, Be^{2+}, Mg^{2+}, Ca^{2+}, Sr^{2+}, Sc^{3+}, Ti^{4+}, Zr^{4+}, Cr^{3+}, Al^{3+}, Ga^{3+}, La^{3+}, Gd^{3+}, Co^{3+} and Fe^{3+} (LoPachin, 2011). As such, acetate ion works equally well as nitrate ion in solubilizing these cations. Having ensured that the biological treatment plant sludge is within acceptable levels of toxic components, the purified sludge was used as a raw material to produce organic fertilizer.

Production of compost from agricultural and industrial wastes, and municipal by-products is an important means of recovering organic matter and an important method of sludge disposal (Franke-Whittle and Insam, 2012). Compost is applied to cropland to maintain and improve soil structure and plant nutrition. However, the presence of heavy metals in composts is the main cause of adverse effects on animal and human health (Singh et al., 2011). Excess zinc and aluminium are also harmful to plants. Therefore, an accurate and quantitative analysis of the contents of heavy metals in composts is of primary importance for the routine monitoring and risk assessment and regulation of the environment (Smith, 2019). Elemental analysis of a compost sample requires that the organic fraction of the sample be destroyed; leaving the heavy metals either in solution or in a form that is readily dissolved. The approaches for destroying organic material and dissolving heavy metals fall into two groups: wet digestion by acid mixtures prior to elemental analysis and dry-ashing followed by acid dissolution of the ash (Enders and Lehmann, 2019). Various methods have been presented for digesting plant tissues and soil samples for metal analysis (Hseu et al., 2002). However, composts have very different physical, chemical and biological properties to those of soils and plants and hence dedicated technologies are required for compost analyses (Palanivell, 2013). Gorsuch discovered that methods of digestion that, involves mixtures of nitric, sulphuric or perchloric acids, were satisfactory for digesting mineral elements in organic and biological materials (Gorsuch, 1959). Rodushkin used two digestion methods, including open-vessel digestion with concentrated nitric acid and microwave digestion with a mixture of concentrated nitric acid and hydrogen peroxide, to analyse heavy metals in cereal and coniferous tree samples by ICP-AES (Rodushkin et al., 2004). They found that both procedures supported the fast preparation of numerous samples.

Some standard reference materials, similar to a compost matrix, have been used to elucidate the recovery of heavy metals by different digestion methods (Wheal et al., 2011). However, little attention has been paid to samples of popularly used composts. Therefore, in this research, we developed an appropriate digestion method for determining the heavy metals in the various composts and foods and the results obtained by different extraction methods are compared. We also reveal here the way the compost formulations were manufactured and their application to soils in selected vegetable and farmlands, analysis of the heavy metals and nutritional value of the raw materials, fertilizer formulations, soils and in vegetables and fruits grown using these fertilizer formulations (Jara-Samaniego, 2017).

2. EXPERIMENTAL

2.1 Materials and instruments

BTPS which has 2% TDS in selected NRPP of SL were collected in large quantities to meet with requirements. Other raw materials such as cow dung, chicken manure and goat manure were collected from the Western Province Farmlands and dry leaves of plants were collected from those accumulated in Katunayake BOI Zone of Sri Lanka. Raw materials were piled to prepare organic fertilizer. Dilina Organic Agro, SKR Watta, Vilattawa Road, Vellarawa, Sri Lanka, was selected as the agricultural field. The vegetable plant cultivations chosen were bitter guard and snake guard and the fruit cultivation chosen was their papaya plantation. All chemicals used in analytical procedures were of the highest available purity purchased from Sigma-Aldrich and were used without further purification.

2.2 Extraction Methods

2.2.1 Extraction of metal ions using conc. HNO_3 and analysis

This approach was a partly modified form of that of Zheljazkov et al, where 1.00 of sample was placed in a 250.0 mL digestion tube and 10.00 mL of concentrated HNO_3 was added (Zheljazkov et al., 2010). The sample was heated for 45 min., at 90 °C, and then the temperature was increased to 150 °C, at which the sample was boiled for 8 h, until a clear solution was obtained. Concentrated HNO_3 was added to the sample to replenish evaporation losses (5.00 mL was added three times) and digestion occurred until the volume was reduced to about 1.00 mL. The interior walls of the tube were washed down with a little amount of distilled water and the tube was swirled throughout the digestion to keep the wall clean and prevent the loss of the sample. After cooling, 5.00 mL of 1% HNO_3 was added to the sample. The solution was filtered with Whatman No. 42 filter paper followed by Millipore filter paper of pore size < 0.45 pm. It was then transferred quantitatively to a 25.00 mL volumetric flask by adding distilled water. The solution obtained was termed BTPSF-HNO_3-1 which was subjected to Inductively Coupled Plasma Atomic Emission Spectroscopic (ICP-AE) analysis. The precipitate was again subjected to the above treatments and the second filtrate thus obtained was also analysed by ICP-AE spectroscopy (BTPSF- HNO_3^{-2}). Each analysis was done in triplicate and the results obtained had very high precision in the measurements obtained for each metal ion analysis with no deviations in some cases. The average of the three repeated measurements was taken. The metal ion concentrations measured using atomic absorption spectroscopy (AAS) gave positive results only for Zn and Al, each in ppm scale. The concentrations of Zn, Cd, Cu, Ni, Co, Pb, Mn and Cr were in not detectable levels by the AAS. As such, all the measurements were done using ICP-AE spectroscopy.

2.2.2 Dry-ashing

1.00 g of the sludge sample placed in a ceramic crucible was heated in a preheated muffle furnace at 200–250 °C for 30 min, and then ashed for 4 h at 480 °C. Then, the sample was removed from the furnace and allowed to cool down to room temperature. 2.00 mL of 5 M HNO_3 was added and evaporated to dryness on a sand bath. Next, the sample was placed in a cool furnace and heated to 400 °C for 15 min, which was then allowed to cool down and moistened with four drops of distilled water. Next, 2.00 mL of concentrated HCl was added and the sample was evaporated to dryness, removed, and then 5.00 mL of 2 M HCl was added and the tube was swirled. The solution was filtered through Whatman No. 42 filter paper and Millipore filter paper with pore size < 0.45 pm, and then transferred quantitatively to a 25.00 mL volumetric flask and adjusted the volume with distilled water and homogenized by shaking.

2.2.3 Extraction of metal ions using glacial acetic acid and analysis

The solid sludge generated after removing most of the water in BTPS is termed filter cake, which was analysed for Zn, Al, Cd, Cu, Ni, Co, Pb, Mn and Cr using ICP-AE Spectroscopy. The filter cake of dry sludge (50.0 g) is completely combusted, at 600 °C, for 12 h, to remove any carbon as carbon dioxide and hydrogen as water vapour to obtain an ash coloured solid mass (10.0 g) which was powdered and homogenized. 2.00 g of this solid was reacted with 100 mL of glacial acetic acid, by stirring for 2 h, in order to dissolve any metal ion present in the powder. The solution thus prepared resembles the composition of the original sludge suspension with identical TDS and is termed BTPSF-HAc-1. The concentrations were calculated using calibration plot with respect to each and every metal ion analysed which was constructed using standard solutions provided by the manufacturer. The residue was dried and re-analysed for Zn, Al, Cd, Cu, Ni, Co, Pb, Mn and Cr by repeating the above procedure for the solution it generated which is termed BTPSF-HAc- 2.

2.2.4 AOAC Official Method 990.08

1.00 g test portion of well–mixed material was weighed to nearest 0.01 g (wet weight basis), and transferred to 250.0 mL beaker. To express results on dry weight basis, another portion of material was dried to constant weight to determine wet/dry weight ratio, but did not digest and analyse this portion because the composition of the dried portion could be different to that of TDS of wet component possibly due to evaporative losses. 10.00 mL 50% HNO_3 was added to undried test portion and mixed well. The beaker was covered with watch glass, heated to 95 °C, and allowed to digest at room temperature. 5.00 mL concentrated HNO_3 was then added and the watch glass was replaced, and the solution was refluxed for another 30 min at 95 °C. Finally, the solution was evaporated to ca. 5 mL without letting any section of the bottom of the beaker to go dry (ribbed watch glass allows evaporation and protects beaker contents from dust.) Solution was then allowed cool down to room temperature, 2 mL of water and 3 mL 30% H_2O_2 were added and the beaker was covered with watch glass, and heated slowly to initiate peroxide reaction. Heating was continued until effervescence subsides and the solution was allowed to cool, and 7.00 mL 30% H_2O_2 in 1.00 mL portions were added while heating. The solution was allowed to cool down to room temperature and 5.00 mL concentrated HCl, and 10.00 mL water were added and covered with watch glass, and the solution was refluxed for further 15 min without boiling. The solution was allowed to cool to room temperature, diluted to 100.0 mL with water, and mixed well. Any particulate matter in the digested suspension was removed by filtration.

2.2.5 Digestion of vegetable samples for heavy metal determination

The vegetable samples were weighed to determine the fresh weight and dried in an oven, at 80 °C, for 72 h to determine their dry weight. The dry samples were crushed in a mortar and the resulting powder was digested by weighing 0.50 g of oven-dried ground and sieve (< 1mm) into an acid washed porcelain crucible and placed in a muffle furnace for four hours at 500 °C. The crucibles were removed from the furnace and cooled. 10.00 mL of 6 M HCl was added. The crucibles were covered and heated on a steam bath for 15 minutes. Another 1.00 mL of HNO_3 was added and evaporated to dryness by continuous heating for one hour to dehydrate silica and completely digest organic compounds. Finally, 5.00 ml of 6 M HCl and 10 mL of water were added and the mixture was heated on a steam bath to complete dissolution. The mixture was cooled and filtered through a Whatman no. 541 filter paper into a 50.0 mL volumetric flask and made up to mark with distilled water.

2.3 Preparation of Compost Fertilizers

Two different trials were prepared by mixing in different ratios of the above raw materials as illustrated in Table 1. Biological Sludge was obtained from Natural rubber glove industry in Sri Lanka.

TABLE 1: Proportions of raw materials mixed in two trials for the preparation of organic fertilizer.

Raw Material	Mass/kg	
	Compost 1	Compost 2
Biological Sludge	35	120
Old leaves	80	25
Saw dust	0	0
Chicken manure	30	15
Goat manure	30	10
Cow dung	30	10

2.3.1 *Preparation of Compost Piles*

Five feet thick dry corn stalk was first laid on pre-cleaned ground. A mixture of crushed dry leaves and dry manure was then piled up to 8 – 10 inches of thickness on top of the corn stalk. Then, de-watered sludge, which was dispersed in a minimum amount of water, was poured on top of the layer up to 2-3 inch thickness. A layer of green leaves was then laid on top of this layer. 300 g of old compost and 100 g of urea were added to the top layer.

All the steps described above was repeated several times until the bed thickness of about 4.5 feet. The final bed prepared by completing all the steps was covered with a polythene sheet. Temperature and moisture were monitored in the daily basis. Entire bed was turned once every week in order to supply sufficient amount of air to the bed. A graphical representation of all the steps involved is shown in Figure 1. Nutritional value of the Biological Treatment Plant Waste of natural rubber and the compost 1 and 2 was analysed together with its heavy metal content, Zn and Al.

2.4 Application of the Compost Fertilizer in Agriculture

2.4.1 *Fields Selection of the sample depth*

The depth of sampling from surface sediment depends upon the purpose of the investigation. For the precise examination of soil contaminated with heavy metals, samples were collected from the top surface of the soil at depth of 0-6 inch level and also another set of from the deeper 6-24 inch level. Numbers of control and analyte soil samples were collected from the different plantations are given below in Table 2.

2.4.2 *Control and Analyte Soil Sampling*

In soil sampling, the surface of the sample point was first cleaned with a hoe. Then, using a crown bar, small pit

FIGURE 1: Graphical representation of the use of Biological Treatment Plant Sludge generated at wastewater treatment plants of natural rubber glove dipping industries together with other raw materials for the production of organic fertilizer.

TABLE 2: Proportions of raw materials mixed in two trials for the preparation of organic fertilizer.

Location	No. of Control samples	No. of Sample Analyte samples
Bitter gourd/ Ladies Fingers Plantation	02	08
Snake gourd Plantation	02	06
Papaya Plantation	02	06
Chicken manure	30	15
Goat manure	30	10
Cow dung	30	10

was created up to the depth of the pit of 0-6 inch and the soil part was obtained from the pit was thoroughly mixed. From the homogenized sample, 250.0 g of soil sample was collected for the analysis. The pit was then dug to its depth from 6” to 24” and 250.0 g of soil sample from the soil collected was also obtained as before. These samples are labelled as control samples with the depths 0-6” and 6-24”, respectively.

Control samples were collected from each of the plantation locations where the control sample points were selected as shown in Figure 2 (a). Each of the corners of the square of ...cm in length represent four adjacent plants cultivated in the field. They were watered to a circle of 12” radius every day which was designated as wet-area. Whenever fertilizer was applied it was done within this circle. The control sampling points were selected from the ... radius of circle around point that crosses the two diagonals of the square as shown in Figure 2(a). This is to ensure that there is no direct application of the fertilizer to the control points and that leakage from the adjacent sample points is also minimal since the distance between the nearest points of the circle around the plant to which water is applied daily is much higher than the radius of the circle. Measurements were done in dry season to avoid the errors due to contamination by flushing the components in the applied fertilizer with rain water to the control point. The analyte sampling points were selected from 4- 5” distance from the cultivated plant for the sampling at 0-6” and 6”-24” depth levels as shown in Figure 2(b).

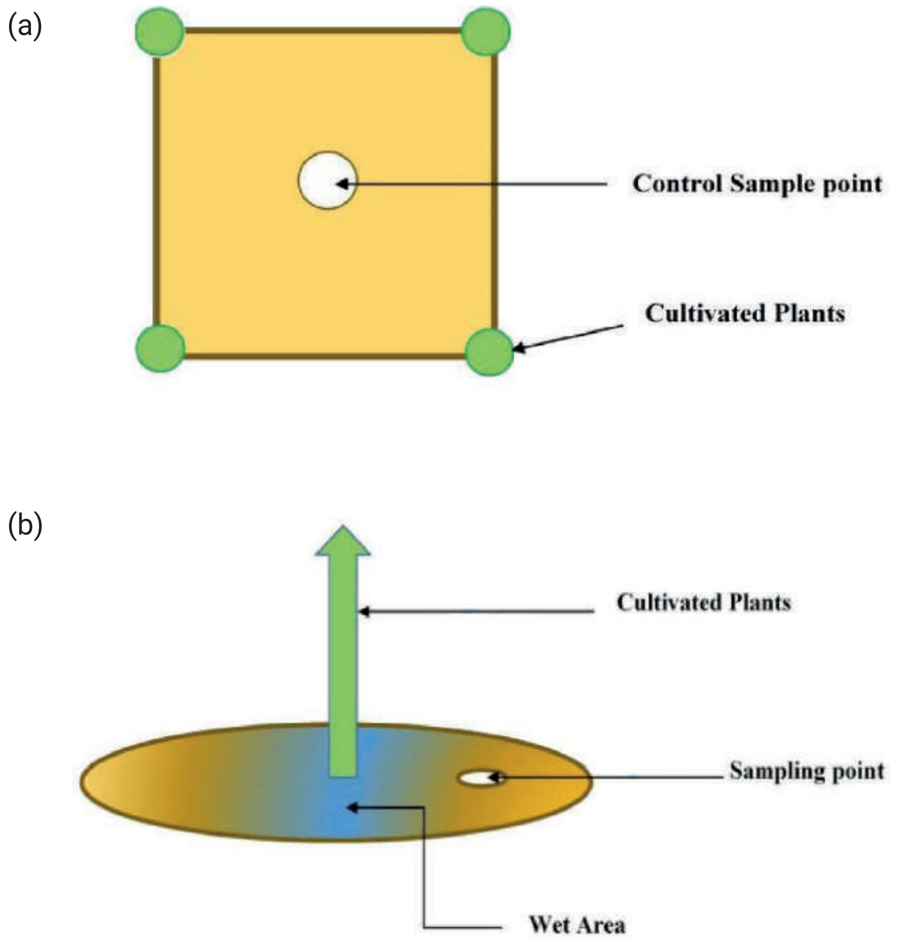

FIGURE 2: Geometric representation of (a) Control (b) Analyte Soil Sampling Points.

3. RESULTS AND DISCUSSION

Figure 3 gives the composition of the BTPS of NRPP of SL. Results obtained clearly indicate that the amounts detected depend very strongly on the method used for the extraction of the species although the same analytical method is used in the analysis (ICP-AE). During dry-ashing, most of the metal ions become their oxides while volatile components are susceptible to lose by evaporation (Jorhem, 2019). Some of the species may be present within the protein 3-D structures of dead bacteria. Therefore, acid digestion may not completely extract some of the components. It is also possible that the extraction by acid digestion may depend on the digestion time if the extraction to acid solution is slow (Karasov and Douglas, 2013). Therefore, comparison of results obtained by different methods seems a difficult task. Group 12 elements Zn, Cd and Hg are generally known as volatile metals.

However, since Zn and Al are present mainly as their oxides and ZDBC is also converted to ZnO during dry-ashing. Both ZnO and Al_2O_3 are highly soluble and react in HNO_3 giving respective ions. It could, therefore, be reliably assume that Zn and Al are completely extracted by dry-ashing followed by HNO_3 extraction. Cadmium is contaminated to the NRPP of SL from the pigments used. Generally, there are three types of cadmium pigments: Cadmium yellow which is cadmium sulphide (CdS) [C.I. Pigment Yellow 37], Cadmium sulphoselenide is a solid solution of CdS and CdSe, which depending on the S/Se ratio, different colours can be obtained [C.I. Pigment Orange 20 or C.I. Pigment Red 108]. A green pigment is obtained by mixing Cadmium yellow is sometimes mixed with viridian to give a bright, pale green mixture called cadmium green. In dry-ashing volatile organic cadmium compounds are formed and are escaped. As such, dry-ashing is not a suitable method to determine cadmium whereas wet digestion with nitric acid gives a reliable measure of cadmium while acetic acid di-

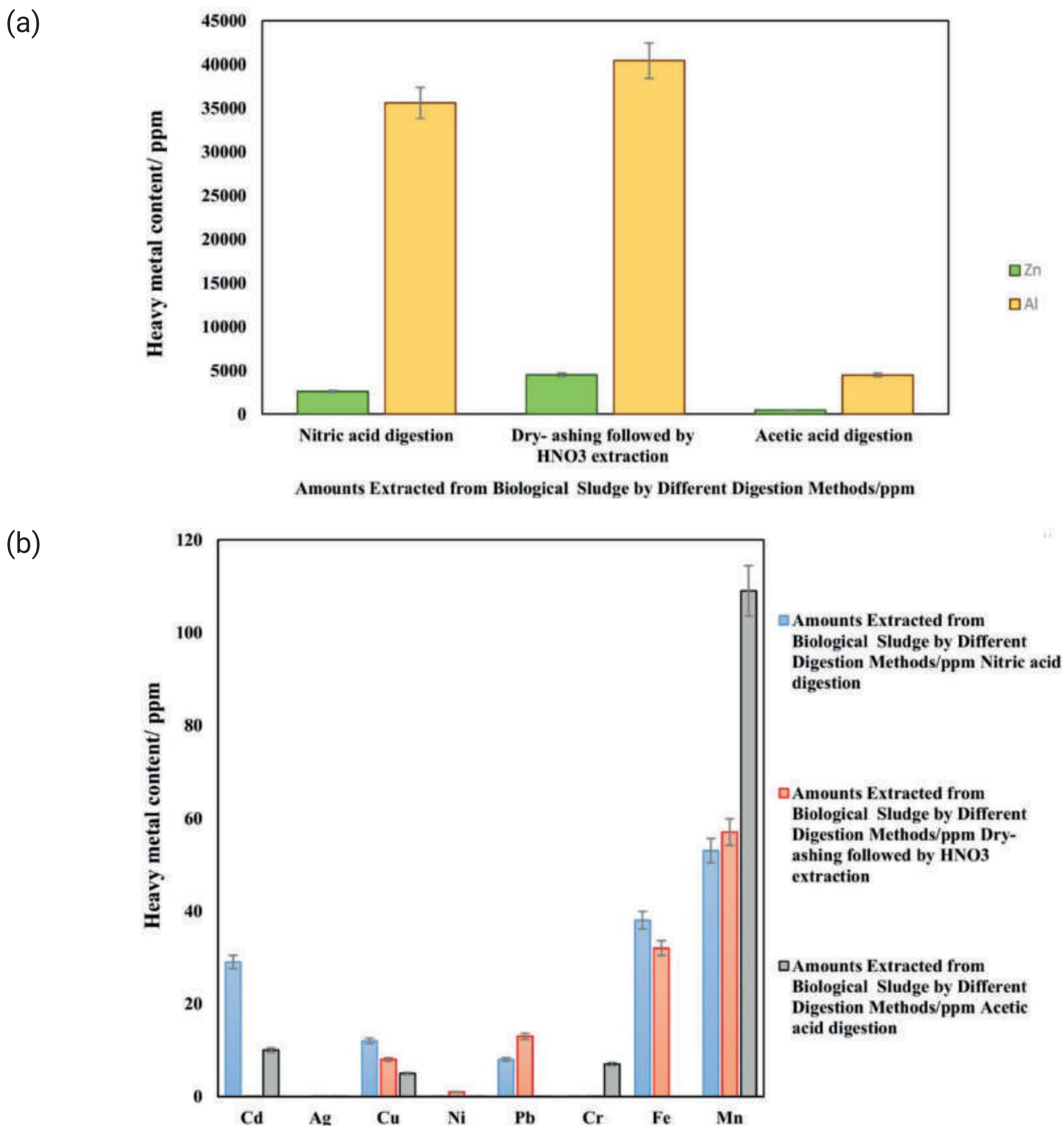

FIGURE 3: Concentrations in ppm of the metal ions analyzed in the Biological Sludge using Nitric acid digestion, Dry ashing followed by Nitric acid digestion and Acetic acid digestion methods (a) for Cd, Ag, Cu, Ni, Pb, Cr, Fe and Mn (b) for Zn and Al.

gestion tend to give lower value than that from nitric acid digestion. Although cadmium acetate is highly soluble in water, it exists in solid state as a coordination polymer where cadmium ions are coordinated by donated lone pairs of acetate ions. If such a polymer is present in the mixture of the sludge, its extraction to water could be slow whereas in nitric acid digestion cadmium exists purely as Cd^{2+} ions which are readily extracted. Extraction of all other species except manganese is preferred by dry-ashing. Interestingly, highest extraction of Mn is observed with acetic acid digestion. Mn too is introduced to the effluent and consequently to the sludge from pigments used: Manganese violet which is $NH_4MnP_2O_7$ (PV16) (Manganicammonium pyrophosphate). These pyrophosphate compounds are more soluble in acetic acid than in nitric acid giving manganous acetate which is highly soluble in water. Besides any remaining phosphate is a nutrient to a fertilizer (Razaq et al., 2017) which is an added advantage of using acetic acid. Any remaining manganeous that has not been extracted would exist as manganeous acetate which is also Mn supplement to fertilizer.

It is important to realize that composition of the industrial wastewater can be highly variable and depend on the amounts of materials used and hence contaminated to water in a given production time. It is critically important in glove industries since different types of gloves with different properties such as colour, texture, feel, abrasive resistance etc. are produced in the same factory at different times, depending upon the demand. As such, it is important to analyse water quality parameters of effluent waters as a function of time at least when different batches of gloves are produced using colorants (pigments) in different amounts. As such, chemical analysis of the BTPS was outsourced to SGS Lanka Ltd. which is an accredited analytical company in Sri Lanka. Samples collected on 27/07 2015, 24/05/2016 and 11/10/2017 were analysed in this way and the results are collected in Table 3. Although the composition is different in different samples, dangerous heavy metals are either in such low quantities not detectable by the method used or below the maximum allowable limits. As such, it is desirable to analyse the sludge that is used to manufacture organic compost fertilizer without relying on results of previous analyses.

In order to obtain reliable results, we searched for an improved method that can be used to extract metal ions present in industrial sludge samples. We then found that AOAC Official Method 990.08 is the most suitable method for digesting aqueous samples containing low TDS of ~2% w/w. As such, in our further work, we used this method for extraction of metal ions in BTPS of NRPP of Sri Lanka [Official Methods of Analysis of AOAC International, 16th Edition, 4th Revision, 1998 Volume I].

Suitability of NRPP Sludge as a raw material to produce organic fertilizer depends on its quality which includes nutritional value and toxic components. As a general rule, higher the nutritional value and lower the toxic components, better is the raw material. As such, we have compared the heavy metal content of the sludge with maximum allowable limits of heavy metals in raw materials that are used to manufacture organic fertilizer for application to vegetation soils according to the SLS 1236:2003 Guidelines which are given in the last columns of Table 4.

SLS 1236:2003 Guidelines do not give maximum allowable limits for aluminium though aluminium is non-essential for plant growth and available or soluble aluminium can be toxic to plants, whereas other forms of aluminium such as aluminosilicates and precipitates, or forms of aluminium bound up in ligands are decidedly not phytotoxic. In this sludge,aluminium exists in essentially insoluble

TABLE 3: Proportions of raw materials mixed in two trials for the preparation of organic fertilizer.

Heavy Metal	Max. Allowable Limit ppm	Quantity in ppm on 27/07/2015	Quantity in ppm on 24/05/2016	Quantity in ppm on 11/10/2017
Zn	1000	199	1617	1020
Cu	400	u	23	u
Cd	10	4.9	u	u
Pb	250	u	u	45
Hg	02	u	u	0.35
Ni	100	u	5.9	5.9
Cr	1000	u	21	9

TABLE 4: Nutritional value of Biological Treatment Plant Sludge collected on 29/07/2015.

Parameter	Protocol	Mass%
Organic C	Ref: Soil Chemical Analysis by M.I. Jackson	30.3
N as total N	SLS 645:PART 2, 1984	5.2
P as P2O5	SLS 645:PART 5, 1985	3.5
K as K2O	AOAC 3.016	0.1
Mg as MgO	Flame AAS	0.08
Ca as CaO	Flame AAS	7.1
Sand%	AOAC 3.005	2.6

forms in water around neutral pH values that are present in natural environments and hence excess aluminium would not contribute to drastic drawback. Cadmium is present at the upper limit though other heavy metals are either absent or present well below maximum allowable limits. Even then, there is a possibility to accumulate these toxic species in the soil or biota due to repeated application of the fertilizer to the soil of the vegetation (Aktar et al., 2009).

Nutritional value of the sludge which was collected at different times was also analysed by outsourcing to Industrial Technology Institute (ITI) of Sri Lanka which is an accredited laboratory for analyses. Results show nutrients such as P, K, Mg and Ca in acceptable ranges but with much lower C:N ratio of 6:1. Data obtained on the sample collected on 29/07 2015 are given in Table 4.

We believe that such low C:N ratio (i.e., high N content relative to carbon content) obtained may not be accurate and may depend on many factors such as different protocols used in C and N analysis and insufficient digestion of the sample in the carbon analysis. Repeated analyses done during 2015- mid 2017 gave similar results. It has been reported that for best performance, the compost pile, or the composting microorganisms, require the correct proportion of carbon for energy and nitrogen for protein production. (Pan et al., 2011) Fastest way to produce fertile, sweet-smelling compost is to maintain a C:N ratio between 25 to 30 parts carbon to 1 part nitrogen, or 25-30:1. If the C:N ratio is too high (excess carbon), decomposition slows down and if the C:N ratio is too low (excess nitrogen) the compost pile will be stinky (Manzoni et al., 2008). As such, searching for a reliable method for nutritional analysis became essential. Having observed the need to analyse the raw materials used in preparation of organic compost fertilizer in each and every batch manufactured, and to use most appropriate analytical method, we used AOAC Official Method 990.08 for metal ion extraction of the batch of sludge collected on 27/07/2017 and improved SLS 1246:2003 protocol for nutrient analysis. Results obtained are given in Tables 5 and 6, respectively.

As can be seen from the data given in Table 5, maximum extraction is achieved when AOAC Official Method was used. It is important to note, however, that the composition of an industrial sample is highly variable and may depend on the amounts of raw materials, dyes, pigments etc. used in a given day. However, since AOAC Method seems to be the most reliable methods, further analyses were done by extracting according to this method.

These results give an appreciable C:N ratio of 18.9:1 this is within the most suitable range required for composting. However, even if heavy metals are present in the sludge at much lower levels than maximum allowable limits the compost should not contain even those amounts of heavy metals. As such, when using BTP sludge as a raw material, the levels of heavy metals, Zn and Al should be lowered as much as possible. This can be done by two ways: (i) by diluting the heavy metal contents using other natural raw materials which have much lower or no heavy metal contents and mixing the raw materials in required proportions to pile up for composting in such a way to balance the C:N ratio also and (ii) by removing heavy metals, Zn and Al from the effluent stream at an earlier stage before directing to the biological treatment plant. We made use both strategies and in (i) BTP sludge and other raw materials such as cow dung, goat manure, chicken manure, dry plant leaves etc. were mixed in required proportions for piling up to make compost fertilizer. In (ii), we have designed and adapted an additional treatment plant in the middle of the flow process with the sequence Rubber Particle Trap - Drape - Precipitation Tank - Biological Treatment Plant. In the precipitation tank we use right amounts of K_2S so that heavy metals, Al and Zn ions precipitate as their respective sulphides while K^+ introduced mixes with the BTP sludge which also contributes to the nutritional value of the sludge. We have analysed the sludge generated this way also and used for compost-

TABLE 5: Metal ion concentrations in the Biological Treatment Plant Sludge collected on 27/06/2017, which was determined following AOAC Official Method.

Metal ion	Conc. (ppb)	Conc. (ppm)
Al	1,880,000	1880
Zn	987,000	987
Cu	98	0.98
Cd	90	0.90
Pb	50	0.50
Hg	Not detected	Not detected
Ni	5,700	5.7
Cr	9,000	9.0

TABLE 6: Analysis of Nutritional value and other components of Biological Treatment Plant Sludge collected on 17/05/2017 (Measured at SGS).

Parameter	Protocol	Mass%
Organic C	ISO 14235:1998	5.6
N as total N	SLS 645:PART 1	0.3
P as P2O5	SLS 645:PART 5	2.3
K as K2O	SLS 645:PART 4	253 ppm
Mg as MgO	SLS 645:PART 6	u
Ca as CaO	SLS 645:PART 6	5.0
Al	EPA 3051:2007	0.86
Cr	9,000	9.0

TABLE 7: Heavy metal, Zn and Al and nutritional values of compost formulations prepared. U-undetected.

Component	Compost 1	Compost 2
C%	4.8	20.7
N%	0.4	1.3
P%	0.7	2.9
K%	0.5	0.8
Mg%	1.2	u
Ca%	2.8	u
C:N Ratio	12	16
Cr (ppm)	17	u
Cu/ppm	35	u
Pb (ppm)	u	u
Hg (ppm)	u	u
Ni (ppm)	u	u
Zn (ppm)	334	u
Cd (ppm)	u	u

ing. However, we will reveal this study in a subsequent publication. Here we describe the Method (i) where the BTP sludge was diluted with other raw materials in order to prepare compost fertilizer by taking C:N ratio as the guide. The C:N ratios of the raw materials collected on 24/07/2017 which were used to make compost formulations were measured following SLS 1247: Appendix F: 2003 protocol (at SGS Ltd.). The C:N ratios obtained for sludge, cow dung, chicken manure, goat manure and dry plant leaves are 19, 22, 10, 23 and 61, respectively.They were used in the proportions given in Table 1 and the resulted compost formulations were also analysed for their heavy metal, Zn and Al contents and nutritional values. Results obtained are depicted in Table 7.

Required C:N ratio for organic compost fertilizer formulations lies between 10 and 25 and hence both formulations have the right C:N ratios. C:N ratio of urea is 2.3 and is a highly N-rich fertilizer and a C:N ratios close to 10 are highly appropriate for N-rich organic compost fertilizers. Heavy metal concentrations in each level of samples collected from 0-6 and 6-24 inch levels of soils in the snake guard and papaya plantations as detailed in the experimental section are depicted in Figure 4.

As shown in the data, all of the components in both formulations are well below the maximum allowable limits imposed by SLS Standards for Compost fertilizers. As such, this dilution method to utilize otherwise problematic BTP waste generated in large tonnage quantities in NRGD Industries of Sri Lanka is not only a convenient way of its disposal but also a useful way to convert it to highly valuable product. We have also analysed the heavy metal contents of papaya fruit grown using prepared compost fertilizer, home grown without adding any fertilizer and those bought from open market in the Western Province of Sri Lanka and the data are given in Figure 5.

Interestingly, there is hardly any difference in heavy metal concentrations of different types of papaya samples studied. One important point is that all of them had very low levels of heavy metals which are well below the maximum allowable limits in vegetables and fruits that are imposed by Joint FAO/WHO Food standards.

3.1 The Codex maximum level (ML)

The product as it should be analysed and to which the ML applies, should be clearly defined. In general, MLs are set on primary products. MLs should in general preferably be expressed as a level of the contaminant related to the product as it is, on a fresh weight basis. In some cases, however, there may be valid arguments to prefer expression on a dry weight basis (this might be in particular the case for contaminants in feed) or on a fat weight basis (this might be in particular the case for fat soluble contaminants). Preferably the product should be defined as it moves in trade, with provisions where necessary for the removal of inedible parts that might interfere with the preparation and the analysis of the sample. The product definitions used by the CCPR and contained in the Classification of food and feed may serve as guidance on this subject; other product definitions should only be used for specified reasons. For contaminant purposes, however, analysis and consequently MLs should preferably be on the basis of the edible part of the product.

4. CONCLUSIONS

In this publication, we have shown, for the first time, an economical way of managing the waste water biological treatment plant sludge of natural rubber glove dipping industries in Sri Lanka. The sludge was converted to a valuable organic fertilizer which was applied for vegetable and fruit plantations. Compositions of the sludge and other raw materials used were carefully analysed by developing the most appropriate protocol for metal ion extraction. The fertilizer thus manufactured has no significant amount of heavy metals but have excellent nutritional value. Soils to which fertilizer was applied, vegetable and plant parts of the vegetation fertilized by this fertilizer were analysed and found that their heavy metal contents are less than those available in the open market and are much less than maximum allowable limits. As such, this work shows a useful way of sludge management which is applicable in large industrial scale.

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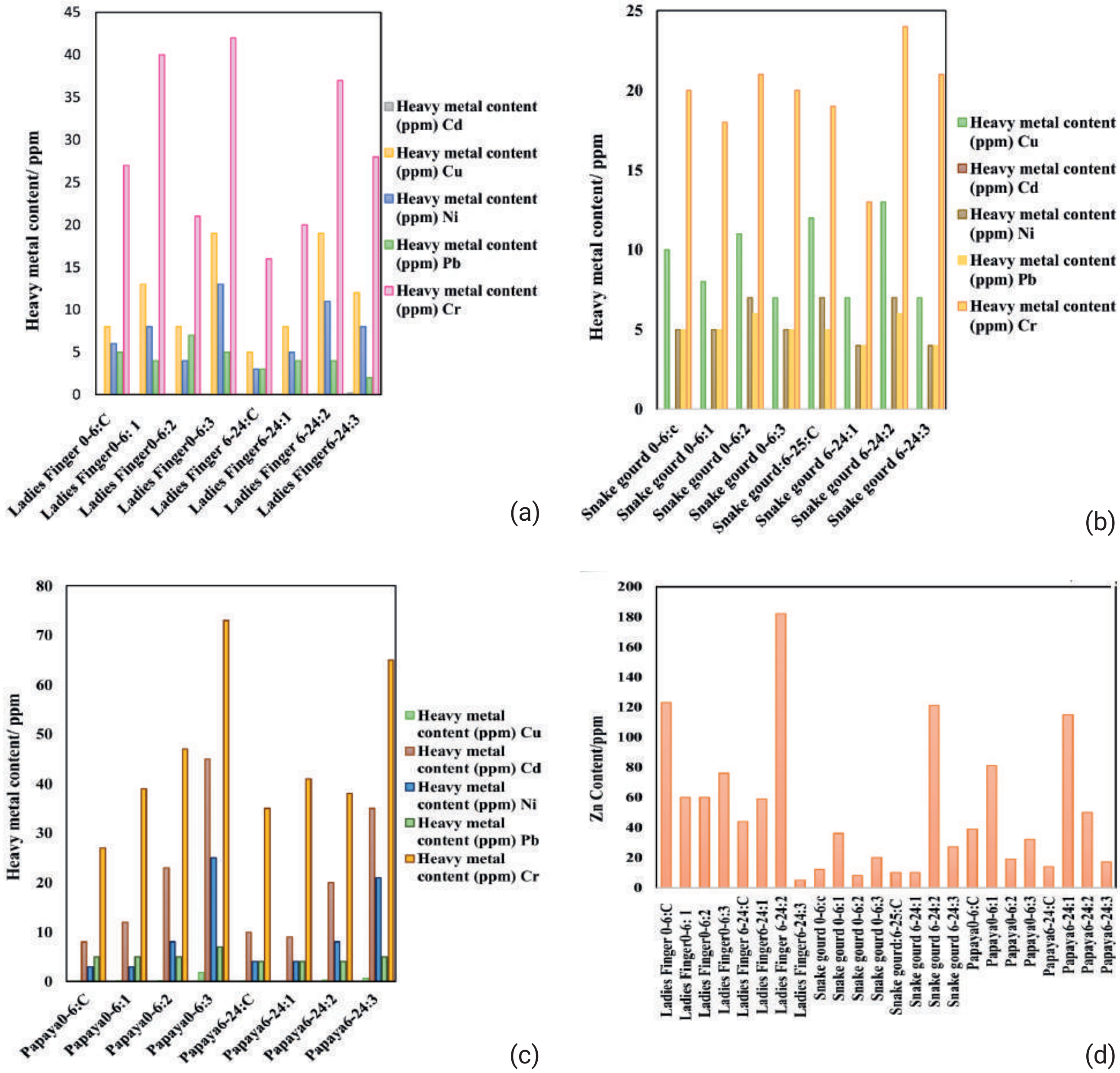

FIGURE 4: Heavy metal analytical data of soils in the vegetation extracted from control and several sample locations and depths. 0-6 and 6-24 indicate depth profiles. 1, 2 and 3 are randomly selected three sample locations inthevegetation. Different vegetable and fruit plantations used together with the depths from which soils were taken are indicated in the x-axis of the figures (a), (b), (c) and (d) whereas y-axis represents the concentrations in ppm of different types of heavy metals in (a) – (c) extracted from soils taken from different plantations at different depths and (d) the same for zinc.

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(a)

(b)

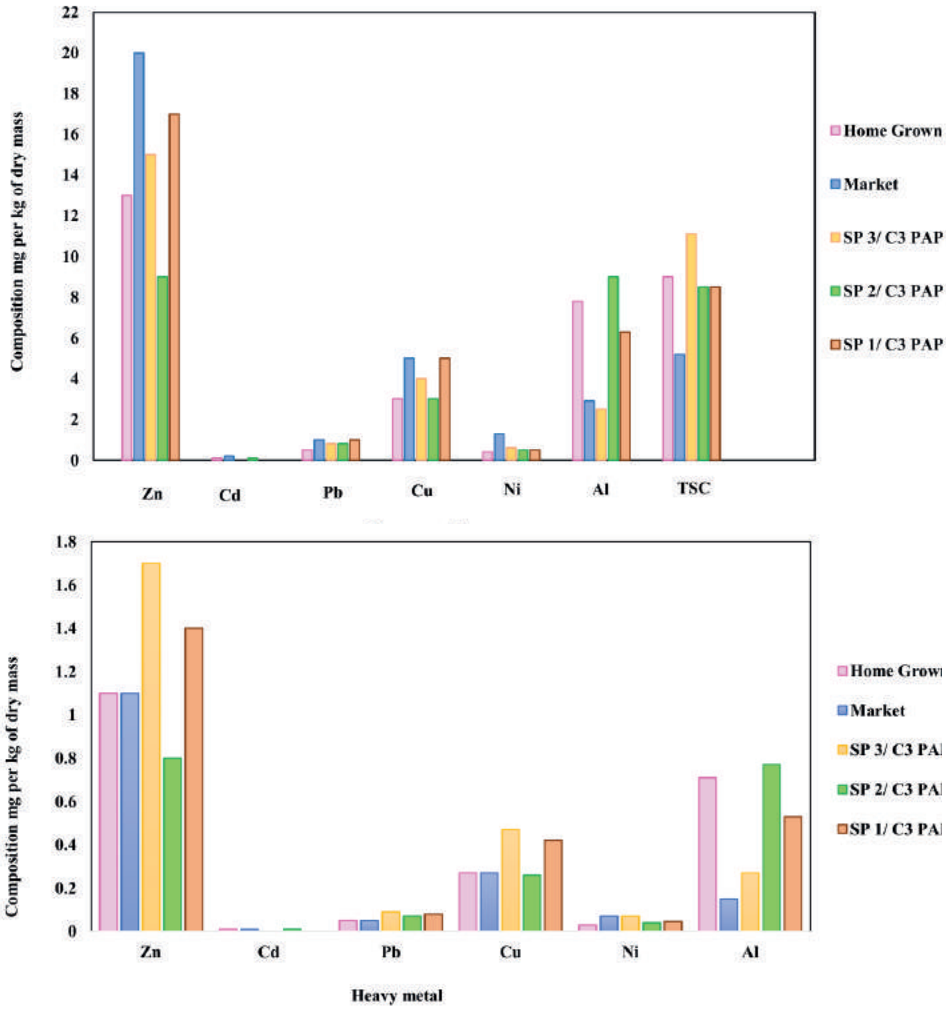

FIGURE 5: Heavy metal analysis of papaya grown by (a) applying prepared fertilizer (b) home grown without any added fertilizer and of those bought from open market for three sample points (SP- Sample Point, PAP- Papaya). Colour codes used for different samples are indicated within the figures.

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COMPARISON OF ANAEROBIC DIGESTION TECHNOLOGIES: AN ITALIAN CASE STUDY

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ABSTRACT

The present study focused on a comparison of two technologies applied in the mesophilic anaerobic digestion, namely a conventional wet one and a dry batch one. The considered substrate was the source sorted organic fraction of municipal solid waste (SS-OFMSW) and the input flow rate, for the study case, was 35,000 Mg/year. The analysed systems included the pre-treatment of the SS-OFMSW, anaerobic digestion, upgrading of biogas to biomethane and aerobic post-treatment for the purpose of obtaining compost. The comparison was made by the calculation of three indicators: net present value, total primary energy and CO_2 equivalent emissions, with the aim of providing elements for choosing the most appropriate technology for the specific case. The results obtained demonstrated the finding of worse values by the total primary energy indicator for the dry batch technology, providing a saving of approx. 21% lower compared to the wet one. In terms of CO_2 equivalent emissions, the dry batch anaerobic digestion technology provided a better indicator than the wet anaerobic digestion system. Sensitivity analysis revealed the finding of an opposite result only when high specific gas production values were assumed for wet anaerobic digestion, and low specific gas production values for the dry batch technology. From an economic perspective, the results indicated a preference for the dry batch technology due to a higher net present value and a shorter period of return of the investment. This finding was also confirmed by a Monte Carlo uncertainty analysis, showing how the dry batch system featured a 90% of possibility of achieving a higher economically sustainability versus the wet technology.

1. INTRODUCTION

The source sorted organic fraction of municipal solid waste (SS-OFMSW) is a highly biodegradable material. Thus, biological processes are the preferred methods for processing and treating SS-OFMSW, allowing subsequent material and/or energy recovery (Righi et al., 2013). Biological processes include treatments such as composting and anaerobic digestion (AD) for biogas production and they account for 95% of current biological treatment operations for organic waste (Oldfield et al., 2016). Even if composting is an energy intensive process, due to the need for forced aeration, it may provide energy savings (0.39 kJ/kg of produced compost), on a life cycle assessment (LCA) perspective, as a consequence of the avoided chemical fertilisers, with a limited increase of energy consumption compared to landfill (+20%) (Blengini, 2008). AD is the best option in terms of total greenhouse gas (GHG) emissions when compared with food waste incineration or landfill (Evangelisti et al., 2014).

Today, the installed capacity of anaerobic digestion is fast increasing (Mata-Alvarez et al., 2014). Since 2009, the number of biogas plants in Europe has approximately tripled and reached about 11,000 units at the end of 2016 with an installed electric capacity of 10,000 MW (EBA, 2017).

Italy is also following this trend. In 2016, the municipal solid waste (MSW) disposed in landfills was about 7.4 million Mg, 50% less than in 2010. Approximately 5.7 million Mg of MSW are currently processed in biological treatment plants (+10% compared to 2015). About 2 million Mg are fed to integrated aerobic/anaerobic treatment plants, and about 250,000 Mg are treated in anaerobic digestion plants. Integrated systems are spreading at the national level, showing an increase of 29% in the amount of treated waste in the last two years (ISPRA, 2017).

AD processes can be classified according to the total solid (TS) content of the substrate in the reactor. In the wet AD, the SS-OFMSW is mixed with water (or other liquid

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streams) to reach a TS content lower than 10-12%, and it is degraded in reactors whose ideal model is the continuous stirred-tank one. In a semi-dry AD (10-20% of TS) and dry AD (20-40% of TS) plug-flow or batch reactors are usually used to handle high values of TS (Mata-Alvarez, 2002). The wet AD is well known and suitable for co-digestion with high liquid content waste (such as sewage sludge), thus reducing costs for pumping and mixing systems. However, investment costs remain quite high because of the equipment required for pre-treatments and the considerable amount of process water to be cleaned. Conversely, the dry AD requires minimal pre-treatments and no mixing, reduced reactor volume and process water, but it is characterised by a lower biogas production (Cecchi et al., 2005).

In the last decades, a large number of studies were made about the AD processes and technologies, especially regarding the anaerobic treatment of the SS-OFMSW, its process design aspects and its potential for energy recovery (Chatterjee and Mazumder, 2016). Within this framework, the dry AD represents a well-established technology for the MSW treatment in Europe (Brown and Li, 2013). For this type of AD, some studies assessed the influence of the initial TS content of the organic matter in the digester and the related biogas generation (methane yield). Fernandez et al. and Forster-Carneiro et al., for a mesophilic lab-scale AD, respectively of OFMSW and food waste, operating in batch mode, evaluated a higher methane production for a process with 20% of TS rather than 30% of TS (Fernández et al., 2008; Forster-Carneiro et al., 2008). Zhang and Banks also found that, processing the same organic matter, the methane yield of a wet AD was higher than that of a dry AD (Zhang and Banks, 2013). Numerous studies focused on the operating parameters of food waste AD and on the comparison between different management strategies of MSW, among such AD technologies, have been carried out (Ren et al., 2018). However, to the best of the authors' knowledge, few works are available about the comprehensive comparison between wet and dry AD in terms of environmental impacts, energy and economics. Luning et al. made a comparison between a full-scale dry process (Valorga dry technology) and a wet process, concluding that the choice between the two technologies was highly dependent on the environmental standards of the AD products at the national level (Luning et al., 2003). Colazo et al. estimated the loss of potential biogas production due to the loss of organic matter in the pre-treatment steps of two SS-OFMSW AD facilities (BTA wet and Valorga dry technologies) and the associated environmental burdens of the rejected material disposal. They concluded that the pre-treatments required for the wet process generated a higher amount of rejected materials (161 kg per Mg of OFMSW in the dry case vs. 337 kg per Mg of OFMSW in the wet case), in which a high amount of organic fraction (about 76% of the rejected material) is lost and that the incineration disposal for such rejects was more favourable than the landfill disposal (Colazo et al., 2015). Daniel-Gromke et al., measuring the GHG emissions of twelve AD plants of bio-waste, indicated that each plant showed a very different emission rate (Daniel-Gromke et al., 2015). Only Angelonidi and Smith quantified the environmental and economic performances of eight AD plants of MSW and food waste, among which two technologies based on gas-proof box-shaped reactors working in batch mode at mesophilic temperatures, herein referred to as dry batch AD (Angelonidi and Smith, 2015). They showed how the dry batch technologies had benefits in terms of flexibility over the accepted feedstock, but worse energy and economic indicators. However, at industrial scale, dry batch AD seemed to be an appealing solution for the treatment of the organic fraction resulting from a mechanical sorting process of MSW (Di Maria et al., 2012).

Within this framework, we highlight that a comprehensive comparison of the available AD technologies considering at once environmental implications, energy and economics, and also including pre-treatments, fate of the rejected materials and effective biogas production, was not yet fully reported in previous literature. We believe that such a comparison could help in highlighting benefits and drawbacks of the AD technologies.

Global primary energy consumption grew strongly in 2017, having the fastest growth since 2013 (British Petroleum, 2018). Moreover, the energy sector still depends mostly on fossil fuels whose combustion contributes to greenhouse gas (GHG) emissions (IPCC, 2014). The calculation of GHG emissions from different plants, such as AD plants or composting plants, could help in understanding the impacts and suggesting improvements for emission reduction strategies in the waste sector (Friedrich and Trois, 2016; Marchi et al., 2017). Complementary economic and financial analysis, aimed at evaluating the costs (negative and/or positive) of a product or system during its life (Sullivan et al., 2014), would help to clarify the potential role of waste management plants.

Therefore, the main objective of this study is to analyse two possibilities for the anaerobic treatment of SS-OFMSW – the dry and the wet technologies – looking at the results in terms of costs, primary energy and CO_2 equivalent emissions. Specifically, we include within the studied systems the pre-treatments, the disposal of the rejected materials, the biogas exploitation. This type of approach is not so common in the international literature, while it can help in providing elements for choosing the appropriate technologies.

Among the dry AD technologies, the dry batch mesophilic one is selected in particular for its simplicity and ability to treat substrates with high content of dry substance (such as SS-OFMSW), avoiding potential problems related to pumps and stirrers, with expected lower energy consumption. Additionally, the dry batch technology can handle entering substrates containing undesired materials, thus not requiring intensive pre-treatments for the SS-OFMSW. As reported by Qian et al. (Qian et al., 2016), in recent years due to these advantages, this type of dry AD has gained enormously in importance in the field of waste treatment. A conventional wet mesophilic technology is considered for comparison. Furthermore, in both the cases, it is assumed to upgrade the produced biogas to biomethane. The entire plants are preliminarily sized for a study case (35,000 Mg/year).

In the following sections, after the description of the context where the case study is located, the main data about mass balances, energy consumptions and economic parameters are reported; then, the evaluation methods are explained; finally, the results are shown according to the selected evaluation methods, also analysing their sensitivity with respect to key parameters.

1.1 Abbreviations

- AD: Anaerobic digestion
- GHG: Greenhouse gas
- HRT: Hydraulic retention time
- i%: Rate of return
- LCA: Life cycle assessment
- LHV: Low heating value
- MSW: Municipal solid waste
- NPV: Net present value
- O&M: Operation and maintenance
- SGP: Specific biogas production
- SS-OFMSW: Source sorted organic fraction of municipal solid waste
- TPE: Total primary energy
- TS: Total solid
- TVS: Total volatile solid
- W: Water

1.2 Units

1 Mg = 10^6 g

2. MATERIALS AND METHODS

2.1 Case study

The analysis is carried out for a study case plant, located in Arezzo, in central Italy. At this industrial site, the company – AISA IMPIANTI S.p.A. – presently operates a mechanical sorting process of mixed MSW, a waste-to-energy plant and a composting plant. The mechanical sorting process consists of a shredder for opening the bags and reducing the size of the mixed MSW, an electromagnet for the extraction of the metals, a rotating drum screen for the separation of the above sieve fraction, mainly composed of dry materials, and the passing sieve fraction, mainly composed of biodegradable humid fraction. The dry flow, characterised by low moisture content (not exceeding 25%) and rich of materials such as plastics and paper (respectively about 30% and 25% of the dry flow), presents rather good fuel properties because of its significant low heating value (LHV), generally higher than 15 MJ/kg (Di Lonardo et al., 2012). Thus, it is sent to combustion for energy recovery through a moving-grate waste-to-energy plant. On the other hand, the biodegradable humid fraction, characterised by relevant organic matter with moisture content of 40-55% by weight (Di Lonardo et al., 2012), is processed by an aerobic biological process for biostabilisation through an aerated (turned) windrow. The biological stabilisation and decomposition of this biodegradable humid fraction produce a final stable product, free of pathogens, which can be applied as lining and capping of landfill sites. Additionally, about 23,000 Mg/year of SS-OFMSW are fed to the composting process, also present in the same industrial site. The composting of SS-OFMSW produces a nutrient-rich matter suitable for land application. The amount of SS-OFMSW is constantly increasing year after year thanks to the implementation of improved separate collection methodologies. In Arezzo, in 2017, the separate waste collection increased by 15% with respect to the year 2015 (Agenzia Regionale Recupero Risorse S.p.A., 2017). Thus, AISA IMPIANTI S.p.A. is planning to build an AD plant able to treat 35,000 Mg/year upstream of the composting process; their aim is to evaluate the best technology choice for their specific case.

Thus, the different technical possibilities for processing 35,000 Mg/year of SS-OFMSW by anaerobic digestion producing biogas, further upgraded to biomethane, are compared from the economic point of view and by performing balances of CO_2 equivalent emissions and primary energy, also comparing the results with those calculated for conventional aerobic composting. In order to carry out such analyses, a preliminary modelling of the two alternative processes is provided in the following paragraphs.

2.2 Wet anaerobic digestion process

In order to process 35,000 Mg/year, the plant is fed by a steady flow of about 96 Mg/day of SS-OFMSW. The SS-OFMSW entering the plant is originated from door-to-door collection within Arezzo Municipality. The observed SS-OFMSW still presents a large quantity of undesired materials. However, it is consistent with other common cases in Italy, as reported in Micolucci et al. (Micolucci et al., 2018). Thus, even if the entering material comes from the source sorted collection system, the quality is still not pure, and about 30% of undesired materials (mainly plastics, glasses, metals, inert) must be removed in a pre-treatment step, which uses a pulper. During such a pre-treatment process, also part of the biodegradable fraction is, unfortunately, removed. This amount is estimated as an additional 16%, with respect to the undesired materials (Khoshnevisan et al., 2018), clearly representing a loss of potential biogas production.

Thus, about 61 Mg/day of pre-treated flow is directed to the mesophilic anaerobic digestion. Table 1 reports the estimated mass flow rates before and after the pre-treatment step and their characteristics in terms of TS, total volatile solid (TVS) and water (W).

Assuming that the wet anaerobic reactor operates with a TS content equal to 10%, the amount of required water for dilution is calculated, resulting equal to about 98 Mg/day (or 98 m^3/day): this amount can be obtained downstream of the wastewater treatment of the process liquid effluent. Providing that the hydraulic retention time (HRT) equals to 20 days, the required volume for the digester is about 4,000 m^3.

According to Mata-Alvarez (Mata-Alvarez, 2002), the generated biogas volumetric composition is 60% CH_4, 38% CO_2, 250 ppm H_2S and 2% H_2O. Additionally, the specific gas production (SGP) is estimated to be 0.589 Nm^3 of produced biogas per kg of TVS supplied to the reactor (Mata-Alvarez, 2002), obtaining about 95 Nm^3 of biogas per Mg of SS-OFMSW entering the plant (i.e. referring to the entering amount before the pre-treatment). The biogas daily flow of about 9,116 Nm^3/day is sent for upgrading, producing 5,526 Nm^3/day of biomethane. For the upgrading process, a biomethane purity (CH_4 content) equal to 97% and methane losses equal to 2% (Sun et al., 2015) are considered.

The produced digestate, with 3.5% TS, is sent to a centrifuge for the solid-liquid separation step, producing about 16 Mg/day of solid part (25% TS) and 133 Mg/day of liquid part. The liquid part, rich in organic compounds and nutrients, is sent to the wastewater treatment plant (provided that it is built in the same site), and, after this process, 98 Mg/day are recirculated for the digester substrate dilution. The solid fraction is sent for aerobic composting with a specific compost production rate of 39% of the entering matter (Boldrin et al., 2010; Saer et al., 2013). Usually, to ensure the proper aeration during the composting process, lignocellulosic wastes are added to the digestate as structuring materials. However, lignocellulosic wastes are not considered within the studied system, given that they would be composted anyway (with or without the digestate).

2.2.1 Energy consumption

For the wet AD process, electricity and fuel consumptions for processing and handling the material are considered. Electricity consumption for the pre-treatment and the anaerobic digestion phase is set at 55 kWh per Mg of processed material (IPPC, 2006). For the solid/liquid separation process, an average consumption of 45 kWh per Mg of dry matter is estimated (Masotti, 2011), while for the composting of the solid fraction, discharged by the centrifuge, the estimation is 20 kWh/Mg (Torretta et al., 2014). A consumption of 0.65 kWh is assumed per each m^3 of liquid effluent treated in a wastewater process, based on primary and secondary treatments (Campanelli et al., 2013). At the time of writing, the biogas upgrading technology is not yet defined, and thus an average preliminary value of electricity consumption equal to 0.22 kWh/Nm^3 of entering biogas is estimated (Sun et al., 2015).

Diesel fuel is consumed by the operating machines used to feed the pre-treatment section as well as to the handling and movement of the heaps for the composting plant. The diesel consumption is estimated 0.78 l/Mg according to the experience of AISA IMPIANTI S.p.A.. Also, the diesel consumption for the transportation of the pre-treatment discards to landfills is considered, assuming a transportation distance of 100 km and a freight lorry transport with 26 Mg of capacity able to cover 2.8 km using one litre of diesel (Ministero delle Infrastrutture e dei Trasporti, 2009).

2.2.2 Costs

Investment, mortgage, operation and maintenance (O&M) costs are considered. Revenues from the SS-OFMSW gate fee and biomethane selling are accounted for. For the wet AD process, the investment cost for the construction of the anaerobic digestion plant (pre-treatment, anaerobic digester, wastewater treatment, biogas upgrading and additional aerobic post-treatment) amounts to approximately € 11,399,900, of which 61% is financed with an interest rate of 3.5% for 10 years (the resulting mortgage is thus equal to 839,071 €/year). Table 2 summarises the annual O&M costs and revenues. With reference to typical Italian market conditions, the diesel price, the waste landfilling cost and the gate fee specific revenue are respectively 1.51 € per litres, 90 € per Mg of waste and 62 € per Mg of SS-OFMSW treated. For AISA IMPIANTI S.p.A., the average electricity price is 63 €/MWh because the plant (see the waste-to-energy plant) is a self-producer of electric energy. According to the Italian incentive scheme for biomethane injection into the natural gas grid (GME-Gestore Mercati Energetici, 2018) and its use in the field of transportation (Ministero dello Sviluppo Economico, 2018), the biomethane sale revenues are, respectively, equal to 22.738 €/MWh and 375 €/CIC (where 1 CIC is equal to 5 Gcal of biomethane). After 10 years, the incentive due to the use of biomethane in the transport sector (CIC) will be reduced by 85%.

2.3 Dry batch anaerobic digestion process

For the dry process, we examine the technology based on gas-proof box-shaped reactors, operating in batch mode at mesophilic temperatures. The amount of SS-OFMSW entering the plant daily is the same as in the previous case (96 Mg/day). Also, the characterisation in terms of TS, TVS and W is the same as that reported in the first column of Table 1. However, in this second case, only a light pre-treatment is considered, based on waste simple shredding and producing 3% of water losses (to be processed in a wastewater treatment plant). Therefore, the stream fed to anaerobic digestion is about 94 Mg/day, according to the details reported in Table 3. In order to promote the anaerobic degradation, part of the digestate must be recirculated into the batch reactor as inoculum, in the ratio 1:1 with the fresh waste (Patinvoh, 2017). Thus, the mixture entering into the batch anaerobic digestion reactor is 188 Mg/day.

The volume of each batch reactor is 1092 m^3 and it can be filled up to a maximum height of 4 m. Therefore, the volume of the material inside the reactor can be 780 m^3 max-

TABLE 1: Entering flows and their characteristics before and after the pre-treatment step in the wet technology.

	SS-OFMSW		SS-OFMSW after pre-treatment	
	[weight %]	[Mg/day]	[weight %]	[Mg/day]
TS	26	25	26	16
TVS	84 *	21	97 *	15
W	74	71	74	45

** as percentage of TS*

TABLE 2: Summary of O&M costs in the case of the wet technology.

Costs [€/year]		Revenues [€/year]	
Staff	360,000	Gate fee	2,170,000
Maintenance	348,560	Biomethane	1,729,741
Diesel	74,274	Biomethane *	628,373
Electricity	182,180	-	-
Disposal costs	1,134,000	-	-
Other expenses	300,559	-	-
Total costs	2,325,299	Total revenues	3,899,741
-	-	Total revenues *	2,798,373

** from the 11th year*

imum, that is about 500 Mg of material loaded inside the reactor, assuming 0.7 Mg/m^3 of density for the SS-OFMSW. If we consider a process duration of 30 days, the number of batch reactors required to process the entering SS-OFMSW is equal to 10.

According to Neri et al. (Neri et al., 2018), the generated biogas volumetric composition is the same as in the previous case: 60% CH_4, 38% CO_2, 250 ppm H_2S and 2% H_2O. Furthermore, a precautionary SGP equal to 0.345 Nm^3 of produced biogas per kg of TVS supplied to the reactor (Nagao et al., 2012) is considered, resulting in about 75 Nm^3 of biogas produced per Mg of SS-OFMSW entering the plant (i.e. before the pre-treatment). The daily flow of dry biogas is actually about 7,204 Nm^3/day. Biogas is then sent for upgrading, producing 4,380 Nm^3/day of biomethane. For the upgrading process, a biomethane purity (CH_4 content) equal to 97% and methane losses equal to 2% (Sun et al., 2015) are assumed.

For the dry AD process, the produced digestate is about 179 Mg/day, of which 94 Mg/day is recirculated back to the reactor as inoculum. The remaining 85 Mg/day is sent for aerobic composting, where, after the biostabilisation, a mechanical treatment is applied to remove undesired materials (compost refining), thus generating some solid discards. The residues are therefore less than in the wet case (about 30% of the stream entering the plant), without significant losses of organic matter during the overall processes. For this reason, the discards are composed mainly of plastics, glasses, metals and inert. For the composting process of the digestate, the same specific compost production rate of 39% of the entering matter (Boldrin et al., 2010; Saer et al., 2013) is used. Also, in this case lignocellulosic wastes are required; however, they would be composted anyway (with or without the digestate) and therefore they are not included within the studied system.

2.3.1 Energy consumption

Electricity and fuel consumptions for processing and handling the material are included also in this case. The estimated electricity consumption for the light pre-treatment and the anaerobic digestion phase is now set at 30 kWh per Mg of processed material (Gunatilake, 2016). The presumed consumption for the composting section is 20 kWh/Mg (Torretta et al., 2014). The same consumption of 0.65 kWh is assumed per each m^3 of liquid effluent treated in a wastewater treatment process (Campanelli et al., 2013), yet, in this case, we assume the process to be outside the plant site. For the biogas upgrading, the same average value of electricity consumption equal to 0.22 kWh/Nm^3 of entering biogas is estimated (Sun et al., 2015).

The assumed diesel consumption for the light pre-treatment section and the composting process is equal to 0.78 l/Mg. The transportation distances of the compost refining discards to landfill and of the liquid effluent to the wastewater treatment plant are here presumed to be 100 km.

2.3.2 Costs

Investment, mortgage, O&M costs are also estimated for this second case study together with the revenues from the SS-OFMSW gate fee and biomethane selling.

The investment cost for the construction of the anaerobic digestion plant (light pre-treatment, anaerobic digesters, biogas upgrading and additional aerobic post-treatment) amounts to approximately € 9,379,500, of which 53% is financed with a rate of interest of 3.5% for 10 years (the resulting mortgage is equal to 596,135 €/year). Table 4 summarises the annual O&M costs and revenues. The specific costs and revenues are the same as in the wet case.

2.4 Evaluation methods

The two different technical alternatives are evaluated through a preliminary economic analysis and balances of energy and CO_2 equivalent emissions.

By accepting the estimated costs, the discounted cash flow analysis is calculated through the net present value (NPV) method (Equation 1). This analysis is conducted by subtracting the present values of cash outflows (including initial cost) from the present values of cash inflows over a period of 20 years. The cash flow for each year is first calculated; then, the present value of each one of them is estimated by discounting its future value at a periodic rate of return (i%), here established to be equal to 6%, according to the data provided by AISA IMPIANTI S.p.A. From the costs reported in Tables 2 and 4, the annual balance can be calculated for both the cases.

$$NPV(i,N) = \sum_{t=1}^{N} \frac{CF}{(1+i)^t} \quad (1)$$

where i is the rate of return, N is the total number of periods (20 years), t is the period (1 year), and CF is the annual cash flow [€/year].

The energy balance is carried out by considering the consumption of electricity and fuels and the energy pro-

TABLE 3: Entering flows and their characteristics before and after the pre-treatment step in the dry batch technology.

	SS-OFMSW		SS-OFMSW after pre-treatment	
	[weight %]	[Mg/day]	[weight %]	[Mg/day]
TS	26	25	27	25
TVS	84*	21	84*	21
W	74	71	73	69

** as percentage of TS*

TABLE 4: Summary of O&M costs in the case of the dry batch technology.

Costs [€/year]		Revenues [€/year]	
Staff	360,000	Gate fee	2,170,000
Maintenance	286,785	Biomethane	1,370,921
Diesel	101,394	Biomethane *	498,022
Electricity	128,558	-	-
Disposal costs	952,560	-	-
Other expenses	247,291	-	-
Total costs	2,076,588	Total revenues	3,540,921
-	-	Total revenues *	2,668,022

** from the 11th year*

duced in the form of biomethane from the anaerobic digestion. In this study, as energy indicator, the total primary energy (TPE) is considered because it accounts for different types of energy, being defined as the energy potential presented by energy flows in their natural form. To convert the electric energy to primary energy, an average efficiency value equal to 0.41 (D.Lgs 192/2005, 2005) is considered, in accordance with the common practice for all Italian cases. For the fuels, primary energy is calculated on the basis of the LHV, respectively assuming 42,900 kJ/kg for diesel and to 35,860 kJ/Nm^3 for biomethane.

The CO_2 equivalent emissions, calculated according to the method reported in IPCC Synthesis Report (IPCC, 2014), provide an environmental indicator of the human activities impact on the global climate (Wiedmann and Minx, 2007). For the CO_2 balance, the following contributions are considered: emissions due to the production of the required electricity; emissions due to the production of the required fuels and their combustion; emissions of CH_4 losses from the upgrading process; emissions from the landfilling of the separated discards and wastewater treatment; avoided emissions for the production and combustion of natural gas displaced by biomethane; and avoided emissions from the compost use. The adopted values for the specific emissions of each process are reported in Table 5.

The CO_2 equivalent emissions deriving from the diesel combustion are calculated through the stoichiometric factor equal to 3.67 $kgCO_2$/kgC. Assuming a carbon content in the diesel fuel equal to 0.86 kgC/kgDiesel and a diesel density equal to 0.85 kg/l, for each litre of burnt diesel the CO_2 emissions result equal to 2.69 $kgCO_2$/l. The emissions of CO_2 deriving from the natural gas combustion are calculated through the stoichiometric factor equal to 2.75 $kgCO_2$/$kgCH_4$. The avoided emissions for natural gas production and the emissions deriving from diesel production, final disposal of residues to landfill and wastewater treatment are retrieved from Ecoinvent 3.0 (Wernet et al., 2016).

For both the TPE and CO_2 balances, the process based on the direct composting of the same amount (35,000 Mg/year) of the SS-OFMSW is included in the comparison considering the following consumptions: 20 kWh/Mg of electricity (Torretta et al., 2014); 0.78 l/Mg of diesel according to the experience of AISA IMPIANTI S.p.A.. The compost production rate is assumed to be 39% of the entering matter (Boldrin et al., 2010; Saer et al., 2013). Residues from compost mechanical refinement are then sent to landfill, considering a transportation distance of 100 km.

TABLE 5: Assumed specific CO_2 equivalent emissions (avoided emissions have negative values).

Process	Unit	Value	Source
Electricity	$kgCO_2$/kWh	0.318	(ISPRA, 2018)
Diesel production	$kgCO_2$/l	0.50	Ecoinvent 3.0
Diesel combustion	$kgCO_2$/l	2.69	Calculated
Methane losses	$kgCO_2/kgCH_4$	28	(IPCC, 2014)
Landfill discards	$kgCO_2$/kg	0.502	Ecoinvent 3.0
Natural gas production	$kgCO_2/Nm^3$	-0.21	Ecoinvent 3.0
Natural gas combustion	$kgCO_2/kgCH_4$	-2.75	Calculated
Compost use	$kgCO_2$/Mg	-69	(Boldrin et al., 2010)
Wastewater treatment	$kgCO_2/m^3$	0.385	Ecoinvent 3.0

3. RESULTS AND DISCUSSION

For the considered case study, the results are reported in the following paragraphs showing the economic analysis, the balances of TPE as well as the CO_2 equivalent emissions.

3.1 Economic analysis results

Figure 1 shows the discounted cash flow trends over the considered time for the wet and dry batch technologies. Based on the theory of finance (Sullivan et al., 2014) and by comparing two types of investment, the process that shows the highest NPV is certainly preferable. In this instance, the discounted cash flow analysis result is more favourable for the dry batch case; at the same time, the NPV is higher.

For the first 10 years, the simple annual balances obtained for the wet and dry batch technologies (easily calculable from Tables 2 and 4) are both positive (net profit) and equal to € 1,500,168 and € 1,464,333, respectively. From the 11^{th} year, the net cash flow of the wet technology becomes significantly lower than the dry batch one, thus confirming a better economic performance of the dry batch case, which has a return of the investment of about 10 years (vs. about 19 years for the wet case). In this case study, the dry batch AD technology has a lower biomethane revenue; however, the lower costs for electricity and waste disposal to landfill provide a higher NPV than in the wet AD case.

3.2 Energy balance results

Table 6 shows the contributions to the TPE balance for the composting, the dry batch anaerobic digestion and the wet anaerobic digestion. Quite obviously, the two anaerobic digestion processes provide much better performances (negative values, meaning savings) than the composting case. This result is due to the energy recovery, in terms of biomethane. The dry batch technology provides lower savings of energy because of the lower production of biomethane. Though, at the same time, the energy required for the process – especially the pre-treatments and wastewater treatment – is also lower. Considering the overall anaerobic digestion process contribution, the dry batch AD technology requires 42% of primary energy less than the wet one. However, for the dry batch technology, the expected biomethane production is lower. This also means that its upgrading energy balance (electricity requirement + biomethane energy) gives smaller negative contribution (21%) than in the wet system. Also, the consumptions for the aerobic biostabilisation process are larger for the dry batch technology (300%) because of the larger amount of composted digestate .

Overall, for this case study, the TPE balance is worse in the dry batch case in comparison to the wet process, with savings lower by about 21%. The result is also in agreement with the assessment made by Angelonidi and Smith (Angelonidi and Smith, 2015).

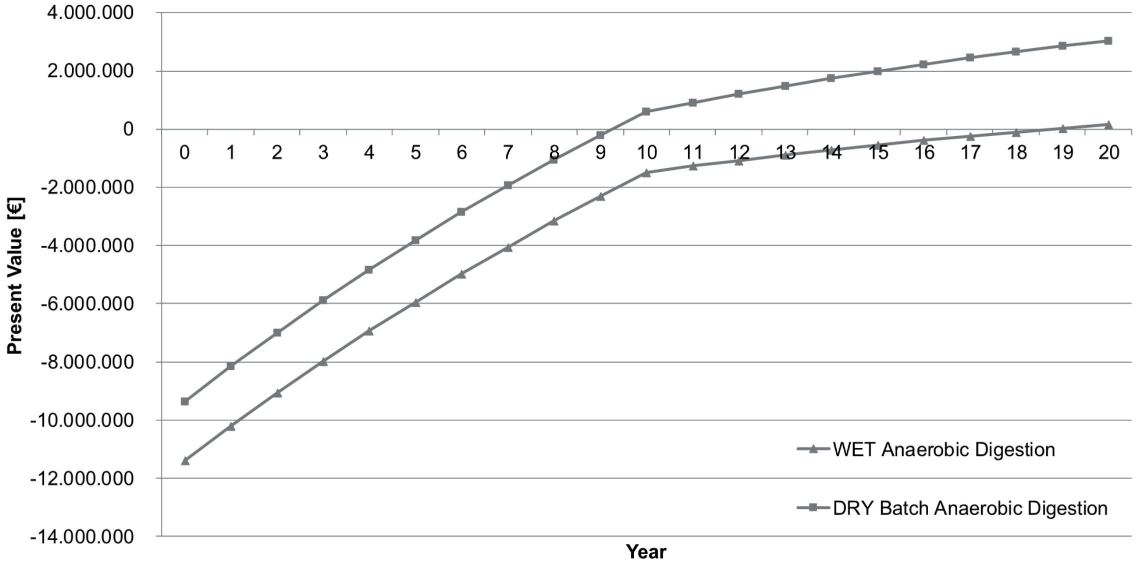

FIGURE 1: Discounted cash flow analysis for the wet and dry technologies.

3.3 CO_2 balance results

Table 7 shows the contributions to the CO_2 balance for the composting, the dry batch anaerobic digestion and the wet anaerobic digestion.

For all the cases, the balances are positive because the positive emissions are higher than the avoided (negative) ones. This result is also found in other case studies (Daniel-Gromke et al., 2015). The results are dominated by the high contribution to the CO_2 balance deriving from the landfilling of large amount of residue. However, in both the anaerobic digestion cases, the balance is better than in the composting case (which is characterised only by the avoided effects from compost usage) thanks to the avoided emissions for the biomethane production.

The use of compost in the dry batch AD technology gives greater negative contribution than in the wet case (-249%), but the biomethane production provides less avoided emissions (21%). In general, the total contribution of the AD phase in the dry batch system gives -43% Mg CO_2/year than in the wet AD. The total CO_2 emissions are lower in the dry batch AD case (-24%) because of the moderated impacts of the anaerobic digestion energy requirements, the upgrading operations and the waste disposal to landfill.

The opposite results of the CO_2 balance and TPE balance are mainly due to the CO_2 contributions not related to energy input/output, namely compost avoided emissions, methane losses and emissions from landfill discards.

3.4 Sensitivity analysis

In the field of AD processes, data on SGPs are by nature subject to variability. SGP values directly affect the production of biogas/biomethane, influencing the main contributions to the previously shown balances (energy, CO_2 and economics). Thus, a sensitivity analysis with respect to this parameter is necessary, in order to better evaluate the effective contribution of energy recovery in terms of biomethane. For this reason, the reference SGPs of both wet and dry AD technologies, respectively equal to 0.589 and 0.345 Nm^3 of produced biogas per kg of TVS (Table 8), are changed. According to the SGP values retrieved from literature (Angelonidi and Smith, 2015; Cecchi et al., 2005), a variation of +/- 20% is considered with respect to the used reference values.

TABLE 6: Total primary energy comparison of the considered technologies.

		Composting [MWh/year]	Dry batch AD [MWh/year]	Wet AD [MWh/year]
Anaerobic digestion process	Electricity pre-treatments + AD	-	1,050	1,925
	Electricity centrifugation	-	-	86
	Electricity wastewater treatment	-	0.5	31
	Diesel	-	287	277
Aerobic biostabilisation	Electricity	488	410	117
	Diesel	277	246	46
Upgrading	Electricity	-	580	732
	Biomethane energy	-	-15,924	-20,092
Transportation to landfill	Diesel	147	147	175
Total primary energy	Energy	912	-13,203	-16,702

TABLE 7: CO_2 emission comparison of the considered technologies.

		Composting [Mg CO_2/year]	Dry batch AD [Mg CO_2/year]	Wet AD [Mg CO_2/year]
Anaerobic digestion process	Electricity pre-treatments + AD	-	334	612
	Electricity centrifugation	-	-	27
	Wastewater treatment	-	0.3	19
	Diesel	-	91	87
Aerobic biostabilisation	Electricity	155	130	37
	Diesel	87	77	15
	Use of compost	-657	-551	-158
Upgrading	Electricity	-	184	233
	Biomethane production	-	-3,370	-4,252
	Methane losses	-	633	798
Landfilling of process residues	Diesel	46	46	55
	Waste landfilling	5,313	5,313	6,325
Total emissions	CO_2	4,944	2,887	3,798

TABLE 8: SGP variation for the wet and dry batch AD technologies.

		-20%	-10%	Ref	+10%	+20%
Wet AD	SGP [Nm³/kg TVS]	0.471	0.530	0.589	0.648	0.707
	SGP [Nm³/Mg SS-OFMSW]	75	85	95	105	115
Dry batch AD	SGP [Nm³/kg TVS]	0.276	0.311	0.345	0.380	0.414
	SGP [Nm³/Mg SS-OFMSW]	60	67	75	83	90

The TPE and the total CO_2 equivalent emissions of the wet and dry batch AD systems are recalculated according to the SGP variation. The modified results of TPE and CO_2 equivalent emission balances are detailed respectively in Table 9 and Figure 2.

In Table 9, "TPE wet" is the total primary energy indicator calculated for the wet case, while "TPE dry" is the total primary energy indicator calculated for the dry batch case.

Even if we consider a dry batch AD process with high performances, the "TPE dry" is better than the "TPE wet" only when the wet AD has quite low SGP values. So, in general, we expect that the TPE indicator is almost always better for the wet case.

The SGP variation corresponds to different energy yields, also leading to the modification of the CO_2 balance. In Figure 2, the modification of the CO_2 balance is reported as the difference between the CO_2 emission (ΔCO_2) values calculated for the two considered systems (Equation 2):

$$\Delta CO_2 = CO_{2\,Wet\,AD} - CO_{2\,Dry\,AD} \tag{2}$$

In Figure 2, the ΔCO_2 variation is reported as a function of the different SGPs of the two technologies.

From Figure 2, it is observed that ΔCO_2 is lower than zero (the black part of the graph) in the 2% of the cases: for these cases the CO_2 balance of the wet AD is better (lower) than the CO_2 balance of the dry AD. This only happens when high SGP values are assumed for the wet AD and low SGP values for the dry batch AD.

For the rest of the cases (98%), the CO_2 balance of the wet AD technology is worse (higher) than the CO_2 balance of the dry AD one.

The sensitivity analysis for the CO_2 balance shows that the conclusions are poorly influenced by the estimated values for the SGP, confirming that, in this case study, the CO_2 balance is in general better for the dry batch technology.

The sensitivity of the economic results to the SGP value variation is also investigated by a Monte Carlo uncertainty analysis. For both wet and dry batch AD cases, 1000 random values of SGP are generated between the maximum and minimum values reported in Table 8. For each SGP value, the NPV is recalculated. When the NPV is negative, the initial investment is not justified.

For the 1000 random SGP values generated for dry batch AD, the NPV remains positive (Figure 3b).

On the contrary, for the 1000 random SGP values generated for the wet AD, 50% of the NPV values are negative, meaning that the investment is not economically sustainable in 50% of the cases (Figure 3a).

Finally, the i-th NPV modification is calculated as the difference between the i-th NPV (ΔNPVi) values calculated for the two compared cases (Equation 3):

$$\Delta NPVi = NPVi_{\,Dry\,AD} - NPVi_{\,Wet\,AD} \tag{3}$$

quantifies the difference between the times that the NPV output of the dry batch AD system is higher than that of the wet AD system.

TABLE 9: TPE variation according to different SGPs of the considered AD technologies.

	-20%	-10%	Ref	+10%	+20%
TPE wet [MWh/year]	-12,722	-14,712	-16,702	-18,692	-20,683
TPE dry [MWh/year]	-10,117	-11,683	-13,203	-14,769	-16,290

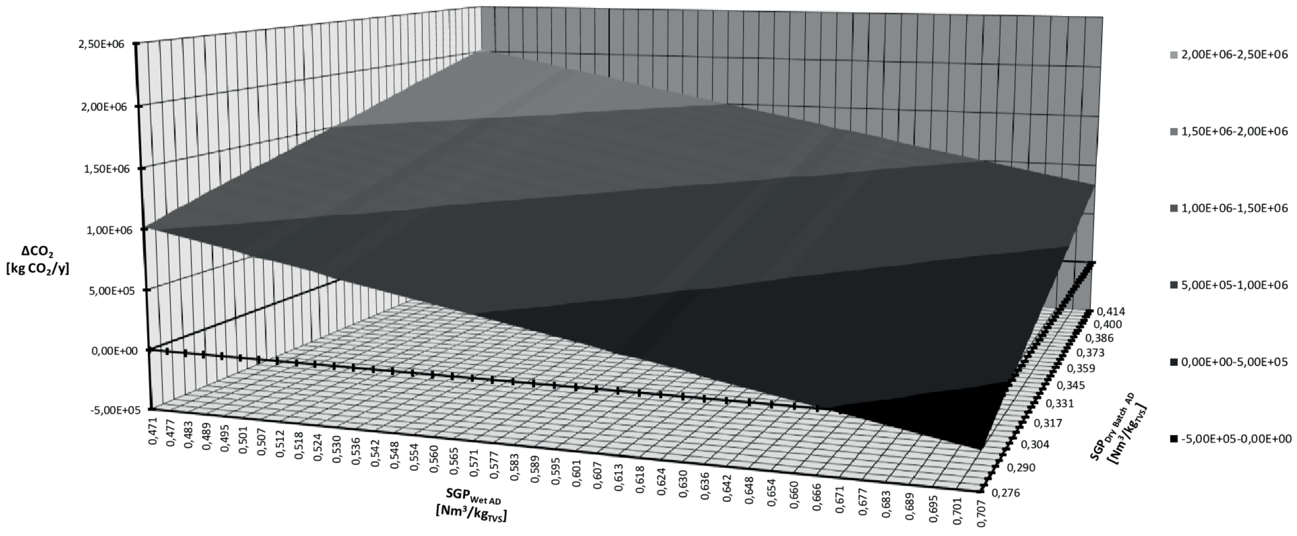

FIGURE 2: CO_2 balance variation according to different SGPs of the considered AD technologies.

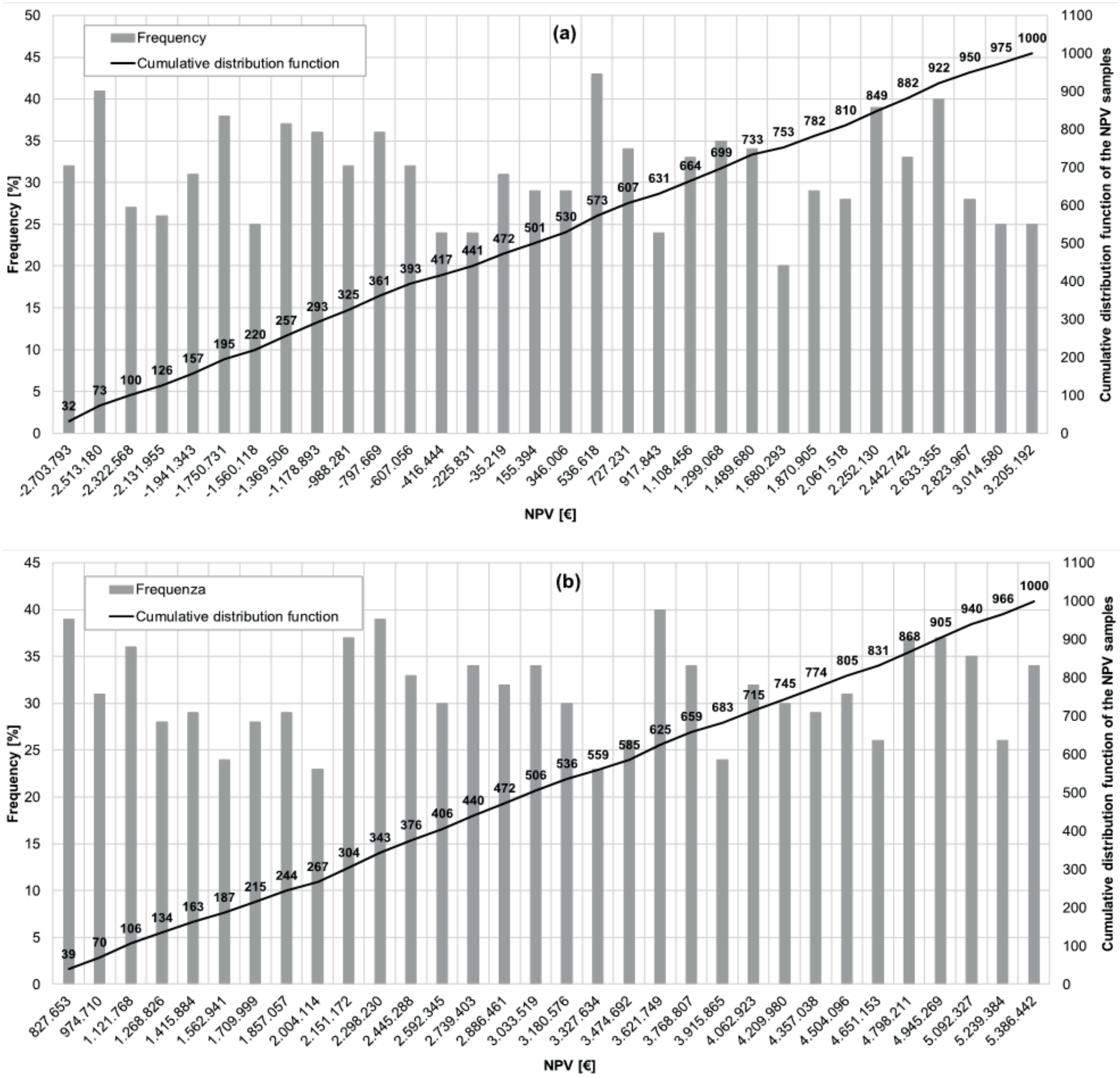

FIGURE 3: SGP influence on the NPV in wet AD (a) and dry batch AD (b) technologies.

According to this method, the dry AD batch technology may reach 90% possibility to have a higher economically sustainability than the wet AD one.

4. CONCLUSIONS

Dry batch anaerobic digestion technology and conventional wet anaerobic digestion technology were compared for the specific case: 35,000 Mg/year of source sorted organic fraction of municipal solid waste of AISA IMPIANTI S.p.A. (Arezzo, Italy).

The total primary energy indicator resulted worse for the dry batch anaerobic digestion technology in comparison to the wet one, providing a savings of about 21% less. The results were confirmed by the sensitivity analysis, showing that very high specific gas production values for the dry batch case were necessary to close the gap between the two cases.

In terms of CO_2 equivalent emissions, the dry batch anaerobic digestion technology provided a better indicator than the wet anaerobic digestion system. The opposite result was obtained only when high specific gas production values were assumed for wet anaerobic digestion and low specific gas production values were assumed for the dry batch technology.

From an economic perspective, the results indicated a preference for the dry batch technology due to a higher net present value and a shorter period of return of the investment. This finding was also confirmed by a Monte Carlo uncertainty analysis, showing how the dry batch system featured a 90% possibility of achieving a higher economically sustainability versus the wet technology.

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BENCHMARK ANALYSIS FOR PLASTIC RECYCLATES IN AUSTRIAN WASTE MANAGEMENT

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ABSTRACT

Plastic recyclates are granulates which are produced by the processing of plastic wastes. The circular economy package of the EU, especially the amendment of the Waste Framework Directive, sets a new goal for the use of different types of these recyclates. Corresponding primary raw materials can assure reliable qualities with respect to stable physical and chemical properties. Besides, the production of recyclates is often even more expensive than the production of primary raw material granulates. Several quality assurance measures are carried out along the value chain from plastic waste to final plastic products. Recyclates are evidently priced based on the price of primary raw material granulate. Pricing also correlates with different quality parameters, however, such as degree of mixing, degree of degradation and presence of impurities. This paper examines the correlation between different quality features and how they affect the pricing policy for recyclates. Experts and Stakeholders along the value chain of plastic recycling in Austria and Germany have been interviewed about the most important quality assurance parameters and how they (would) affect prices of recyclates. Therefore, quality parameters for the sorted plastic waste as an input for plastic waste recycling companies and manufactured recyclates are included in this paper. Experts from the plastic waste recycling industry confirmed that there is a profound correlation between price and quality that is presented and discussed in the paper: The higher the quality of the recyclates, the lower the level of impurities and the purer the recyclates, the higher the price.

1. INTRODUCTION

The European plastic strategy presented by the European Commission, to be implemented in the Recycling Sector Package, poses an enormous challenge for the European waste management and the plastics processing industry. The circular economy package sets a recycling rate of 55 wt.% by 2030 for plastic packaging waste (European Union, 2018). The European Commission has not stipulated a compulsory percentage of recycled plastics in the manufacturing process of new consuming products, i.e. substitution rate on a primary raw material level. Moreover, the Commission appeals to the responsibility of manufacturers to achieve its objectives regarding circular economy.

Currently, recyclates are applied with a content lower than 10% in new plastic packaging products (Reitz, 2019). This suggests that recyclates are either too expensive or of too low quality. Although scientific studies (Klumpp & Su, 2018; Martel, 2018; Pauwels & D'Aveni, 2014; Voros, 2019; Zhe Gin & Kato, 2010) have already focused on the correlation of quality and price for other goods, this paper does not only examine such correlations but also includes quality parameters for the sorted plastic waste and recyclates to provide a practical guideline for quality assurance. In the course of the applied survey for this paper, experts gave a comprehensive overview of how quality is assessed in the field and which parameters are significant for high quality material. Furthermore, this data will support assessing the economic feasibility of certain stages of plastic packaging waste treatment (European Committee, 2019).

Wide range of composited materials and problematic additives can lead to sales difficulties for recyclates too, since recycled materials from "older" waste plastics may still contain substances that are no longer permitted in new plastics due to their negative effects on the environment and health (Wilts et. al., 2014). Plastic recycling is also limited by a lack of quality and constant supply of raw materials required by the industry (Vilaplana & Karlsson, 2008). Quality criteria for recyclates for the final plastic processing companies are not standardised but defined individu-

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ally by the recycling and processing companies. Criteria include exceptionally pure colour and low content of contaminations (Vilaplana & Karlsson, 2008). Besides the lack of quality, the poor image of recycled plastics in the public also impairs plastic recycling (Moser et. al., 2016). As a result, recyclates are not used in new plastic products to the desired extent or not at all.

Despite the number of obstacles, however, recyclates are increasingly applied by the industry to pursue a sustainable strategy (Polymer Comply Europe, 2017). The market for primary raw plastics is characterized by:

- A close correlation with the price of crude oil, resulting in comparatively high volatility of prices. As a result, when the price of primary raw plastic significantly decreases, recyclates will be increasingly substituted by primary raw material granulate, as well as
- Easy substitutability of products of different suppliers and also by oligopolistic market structures, inspiring strategic behaviour of suppliers (Rothgang et. al., 2017).

The main question raised by this paper is based on these two findings and seeks to establish a correlation between the price and the quality of plastic recyclates. In addition, the quality requirements for sorted plastic waste and produced recyclates are examined. The importance of quality assurance and its practical implementation are treated in a separate section. Furthermore, the market for primary raw plastics and recyclates is examined in detail and pricing developments are analyzed.

2. MATERIALS AND METHODS

2.1 Materials

The following plastic types are being investigated in the study as they represent 57% of the demand for the plastic packaging waste processing industry in Austria (Stoifl et. al., 2017):

- High-density polyethylene (HDPE) foils and hollow bodies (emptied);
- Low-density polyethylene (LDPE) foils and hollow bodies (emptied);
- Polypropylene (PP) foils and dimensionally stable PP (bucket, canister, emptied);
- Polyethylene terephthalate (PET) bottles (emptied);
- Polystyrene (PS) foils (thermoforming film).

This paper mainly discusses recyclates since regrind materials do not undergo extensive quality assurance and, frequently, only the impurity content is of importance.

2.2 Methods

All relevant stakeholders along the value chain from plastic wastes to the finished products are shown in Figure 1. This figure also shows all the terms used in this paper along the presented value chain.

A market analysis of secondary plastic granulates was conducted to identify the quality benchmark in plastic recyclates, performed by observing the development of pricing, identifying drivers to the increase or decrease of value and verifying whether the value depends on recyclate quality or on other economic features.

To analyse the correlation between price and quality, several packaging plastic waste processing companies and plastic waste recycling companies were provided with a specially designed assessment guide. In addition to personal discussion with plastic waste recyclers and the plastic waste processing industry in Austria, the plastics recyclers and the plastics processing industry in Germany was approached with short and targeted e-mail questions. Altogether, 19 different stakeholders responded. Six phone calls were made, reaching two plastic recyclers, three plastics processing companies and one association. In addition, about 80 e-mails were sent to plastic waste collectors, plastics recyclers and plastics processing companies, resulting in a return rate of approximately 20%. Four plastics recyclers, five plastic processing companies and four other stakeholders responded. Figure 2 shows the distribution of the consulted companies by industry. 32% of plastic re-

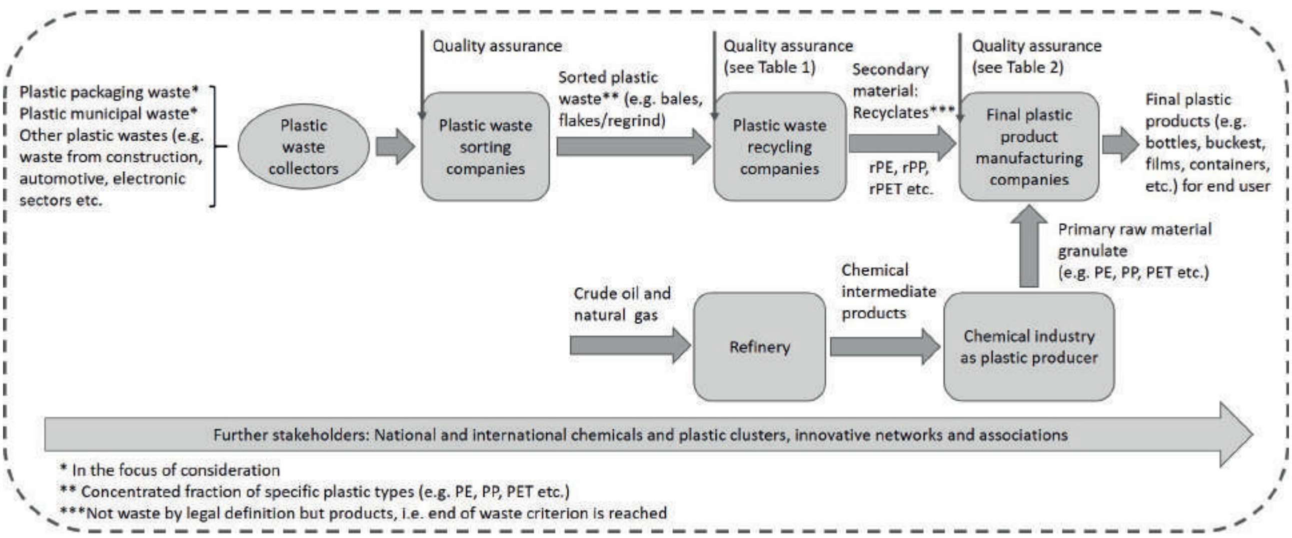

FIGURE 1: Stakeholders along the value chain from plastic waste to final plastic products.

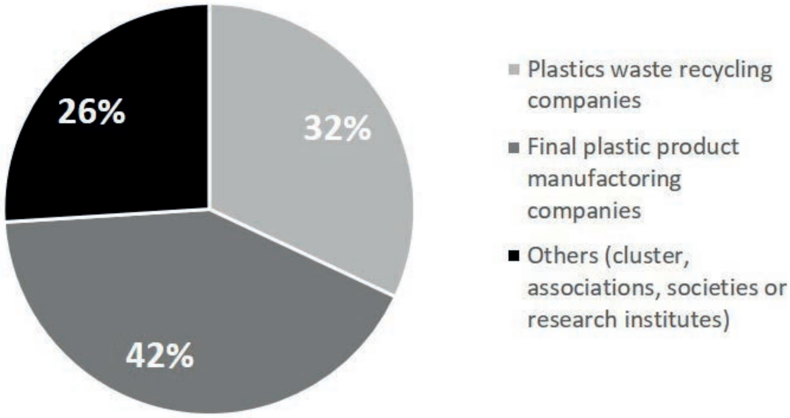

FIGURE 2: Distribution of the consulted companies by industry.

cyclers, 42% of plastic processing companies and 26% of other stakeholders along the plastic value chain participated the survey. The other stakeholders are cluster, associations, societies or research institutions operating in the field of plastics processing.

3. RESULTS AND DISCUSSION

The following section is divided into five subsections. First, the quality requirements for the sorted plastic waste and the plastic recyclates are shown. Second, the section "Quality Control" is describing, which parameters are significant for reliable quality control for the sorted plastic waste and manufactured recyclates. Additionally, price development for the polymer types mentioned above has been done. Furthermore, the most relevant questions of this study are answered in a separate section. Finally, to fulfil the titles of this paper, the quality benchmark in plastics recyclates are described.

3.1 Quality Requirements

Requirements for sorted plastic waste qualities:

In Germany, quality standards for sorted plastic waste applied in the plastic waste recycling companies have evolved within the plastic industry (Grüner Punkt, 2019), summarized in Table 1.

Quality requirements for produced recyclates:

Provided specification sheets or datasheets of produced recyclates include limit ranges (see Table 2) for the following properties:

- The density of non-cellular plastics (DIN EN ISO 1183-1)
- Melt volume-flow rate (MVR), melt-mass flow rate (MFR) and flow rate ratio (DIN EN ISO 1133-1)
- Tensile properties, in particular, modulus of elasticity (E-Modul) (DIN EN ISO 527-1)
- Notch impact strength (DIN EN ISO 179/1eA)

3.2 Quality assurance

3.2.1 Quality assurance of plastic waste

The key competence in the quality assurance process of the delivered mixed plastic waste material to the plastic waste sorting plant is found with the material acceptance staff. Based on their experience, the quality of supplied plastic waste bales can be assessed by visual inspection. Attention is paid to coarse impurities. The collective experience of the stuff is decisive. An essential part of the input control is the colour distribution of the bale because a majority of pure plastics is a requirement for the production of high-quality recyclates and their use in new products.

Furthermore, the origin of waste affects the assessment of the sorted plastic waste quality. Hence, the materi-

TABLE 1: Quality standards for sorted plastic wastes for recycling (Grüner Punkt, 2019).

Sorted plastic wastes	Metal items [wt.%]	Other plastic particles [wt.%]	Other residues 1) [wt.%]	Dimensionally stable PE articles [wt.%]	Foamed plastics incl. EPS* [wt.%]	Plastic Foils [wt.%]	PVC [wt.%]	Dimensionally stable PP [wt.%]
Plastic Foils (mostly LDPE)	< 0.5	< 4.0	< 4.0	-	-	-	-	-
Plastic hollow body (mostly HDPE)	< 0.5	< 3.0	< 3.0	-	-	-	-	-
PP	< 0.5	-	< 3.0	< 1.0	< 0.5	< 2	-	-
PET bottles	< 0.5	< 2.0	< 2.0	-	-	-	< 0.1	-
PE	< 0.5	-	< 3.0	-	< 0.5	< 5.0	-	< 3.0
PS	< 0.5	< 4.0	< 2.0	-	< 1.0	-	-	-

*Compostable waste (foods, garden rubbish). * EPS: expanded polystyrene*

TABLE 2: Physical, chemical and rheological properties of the investigated recyclates (Grüner Punkt, 2019).

Properties	LDPE	HDPE	PP	PET	PS
Density [g/cm³]	0.920 - 0.945	0.940 - 0.970	0.895 - 0.920	1.360 - 1.390	1.050 - 1.290
Melt-mass flow rate (MFR) [g/10 min]	0.5 – 0.9(1)	0.1 - 30.0(1)	0.1 - 30.0(2)	20.0 - 30.0(3)	2.3 - 8.2(4)
Tensile properties (modulus of elasticity) [MPa]	220 - 380	1 170 - 1 350	850 - 1 450	3 400 - 3 700	3 000 - 3 400
Notch impact strength [kJ/m²]	8.00 - 15.00	4.85 - 5.15	3.00 - 5.50	2.00 - 4.00	8.0 - 12.0

(1) 190°C | 2,16 kg (2) 230°C | 2,16 kg (3) 280°C | 5,00 kg (4) 200°C | 5,00 kg

al flow can be assessed using empirical values depending on the origin.

There are interesting arguments why deliveries of sorted plastic waste bales are rejected. Cartridges for sealing compounds repeatedly lead to rejection. The moisture of bales is another argument. Increased moisture can affect the surface of the particles and foaming processes during injection moulding may occur. Basically, however, non-conformity with quality requirements usually leads to a price reduction. If the content of contaminants is too high, the processing is impaired (material variations).

3.2.2 Quality control of recyclates

The quality of random samples of recyclates is controlled in a laboratory. The physical, rheological and mechanical properties of the recyclates are of great interest. The following characteristics are analysed in the course of a random sample inspection:

1. Physical properties
   a. density determination (DIN EN ISO 1183-1)
2. Rheological properties
   a. melt-mass flow rate (MFR) (DIN EN ISO 1133-1)
3. Mechanical properties
   a. tensile properties, especially modulus of elasticity (DIN EN ISO 527-1)
   b. notch impact strength DIN EN ISO 179/1eA

Frequently, further parameters of the recyclates are determined. These include:

- Melting temperature
- Colour distribution and colour composition
- Size and form of the granulated material (e.g. lenses, cylinder)
- Moisture content
- Filtration fineness
- Ash content
- Heavy metal content

In addition, there is often a continuous control of recyclates and an inspection for any specks, gas emissions, mechanical values and the colour of the recyclates.

The hardness of recyclates allows initial prediction of the foreign plastic content, the shape of the granulates and the bulk density indicating potential gas inclusions or vacuoles. The colour and odour of granulates may indicate previous thermal degradation of the material. The following devices or test methods are frequently used in quality assurance refers to the previously mentioned standard specifications: Melt index testers, differential scanning calorimetry (DSC), ash furnaces, residual moisture scales, density analysers, capillary rheometers, tensile testing and notched-bar impact test machine.

3.3 Price Development

The plastic trading market is currently shifting and, as mentioned before, increasingly developing into a buyer's market. A high dollar exchange rate (1,1008 $/€ on 24-Sept-2019) (Wallstreet-online, 2019) and weak crude oil prices (62.90 $/barrel on 24-Sept-2019) (Tecson, 2019) result in a preference for primary raw material over recyclates. Moreover, the European plastic market has changed due to the ban of exports to China that has previously been one of the largest importers of European plastic waste.56% of all plastic waste worldwide and 87% of all European plastic waste has been sent to China in recent years (Uken, 2018). The plastic waste streams, which are heavily contaminated and poorly sorted are most seriously affected. As a result, there is an oversupply of this plastic wastes in the European plastic recycling market. It follows that the plastics processing industry will favour high quality of plastics available.

Plastic wastes with low extraneous and pollutant contents and lower humidity are demanded. This oversupply of polluted plastic waste enables customers to select highest-quality plastic waste, ultimately affecting the pricing. Low-quality plastic waste losing market shares used to a great extent for thermal treatment or recovery (Sarc et. al., 2019).

3.3.1 Price development for sorted plastic waste

The price developments for HDPE and LDPE (A), PP (B), PET (C) and PS (D) regrinds and bales over the last years are shown in Figure 3. The average selling price for regrinds of commodity plastics (e.g. PE, PP, PET, PS) is about 538 €/t, varying by 92 €/t (Plasticker, 2019).

For the PE types, it is stated that the average regrind price is very similar for HDPE and LDPE with approximately 0.6 €/kg (Plasticker, 2019). The HDPE regrind price fluctuated significantly more than LDPE in the years 2014 to 2017. The LDPE regrind price is on average three times higher than the prices for the LDPE bales. This can also be observed for PP and PET. At 0.56 €/kg, the average regrind price for PP is 2.5 times higher than for PP bales, and at 0.37 €/kg, the average regrind price for PET is 1.9 times higher (Plasticker, 2019). The reason for this is the higher processing depth and the associated higher costs for the production of regrinds compared to bales. The different

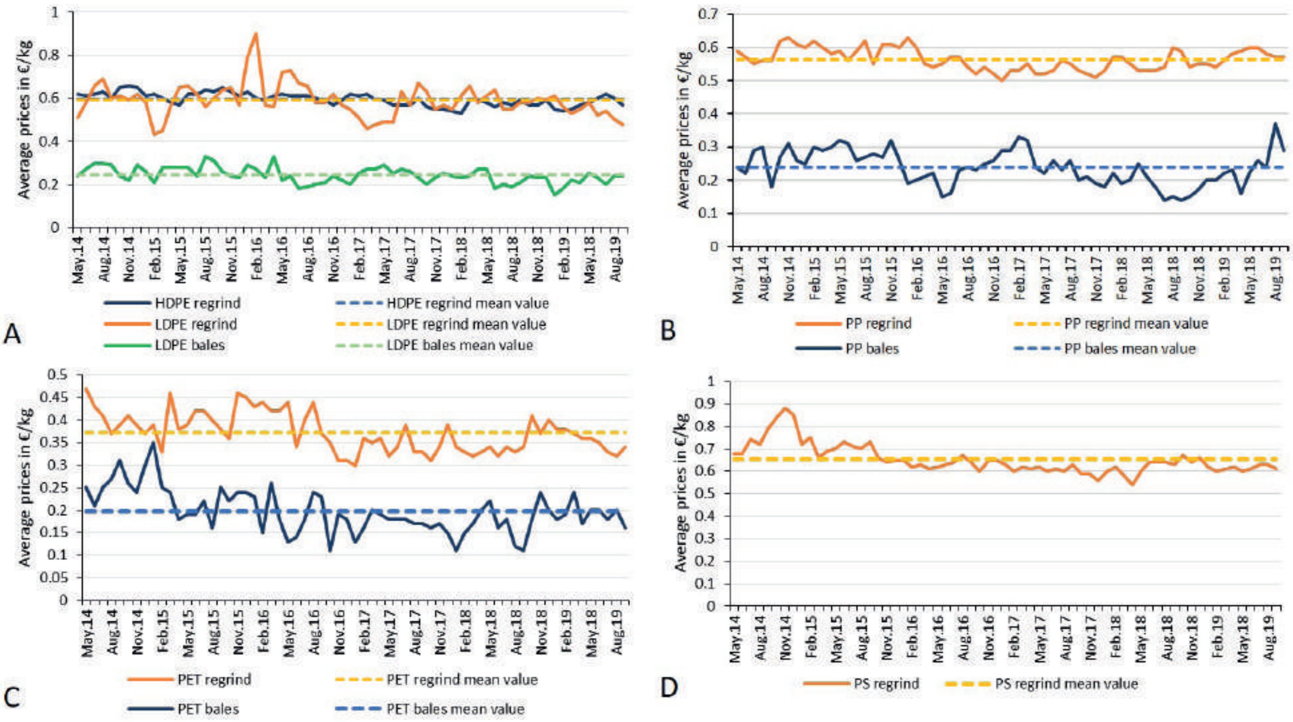

FIGURE 3: Price development for regrinds and bales of PE types (A), PP (B), PET (C) and PS (D) (Plasticker, 2019).

price differences between regrinds and bales of the plastic types can be explained by the different processing costs.

3.3.2 Price development for recyclates

The price developments for LDPE (A), HDPE (B), PP (C) and PS (D) granulates of primary raw materials and recyclates are shown in figure 4. No reliable price development could be collected for PET. The average selling price in July 2019 of primary raw material granulates of standard plastics (e.g. PE, PP, PS, PET) was around 1.17 €/t and 0.537 €/t (Plasticker, 2019) was the average selling price of recyclates of standard plastics. This means that granulates produced of primary raw material are on average twice as expensive as recyclates.

A comparison of the price developments of the primary raw materials with those of recyclates shows that there is a certain dependency between both price developments. If the price of a primary raw material rises or falls, the recyclate price of this plastic type also reacts with a price rise or fall. This fact can be seen for example well for LDPE in Figure 4 (A).

3.4 Market Study

The following section provides a summary of the most important statements:

Is there a correlation between price and quality of the sorted plastic waste?

First, the general market balance of supply and demand is pointed out. This provides the basis for any pricing. Where supply and demand meet, a corresponding market for goods develops.

The respondents 'affirm' the question, though. There is indeed a strong correlation between the quality and price of the sorted plastic waste. In addition, better application options are made accessible by purer sorted plastic waste, higher-priced. Surveyed plastic processing companies also reported the dependence of co-payments, i.e. a negative price for recyclates. If the sorted plastic waste can be purchased for a higher additional price, then the recyclates may be offered for less. When co-payments decline, however, the prices in sales have to rise. Additional payments depend primarily on the quality of the sorted plastic waste. If the material is dirty and includes high amounts of extraneous plastic, additional payments are higher. It the material is clean, on the other hand, and has a low level of extraneous plastics, additional payments will be lower.

It was also mentioned that the quality of the sorted plastic waste is primarily defined by its colour. The higher its purity, the higher the price that can be achieved on the plastic trade market. This is mainly due to its broader application range, say, in subsequent colouring, foil thickness and mechanical properties.

As mentioned above, the staff is crucial for sorted plastic waste price. They ultimately control the quality and their wealth of experience facilitates a reliable quality level and, accordingly, adequate pricing.

Is there a correlation between price and quality of the recyclates?

Regarding this question, there is again a general agreement on a higher quality of recyclates leading to higher prices. It is backed by the argument that higher quality of the recyclate reduces the risk of failures or bad batches from contamination for final plastic processing companies. Furthermore, it was mentioned that the quality of the

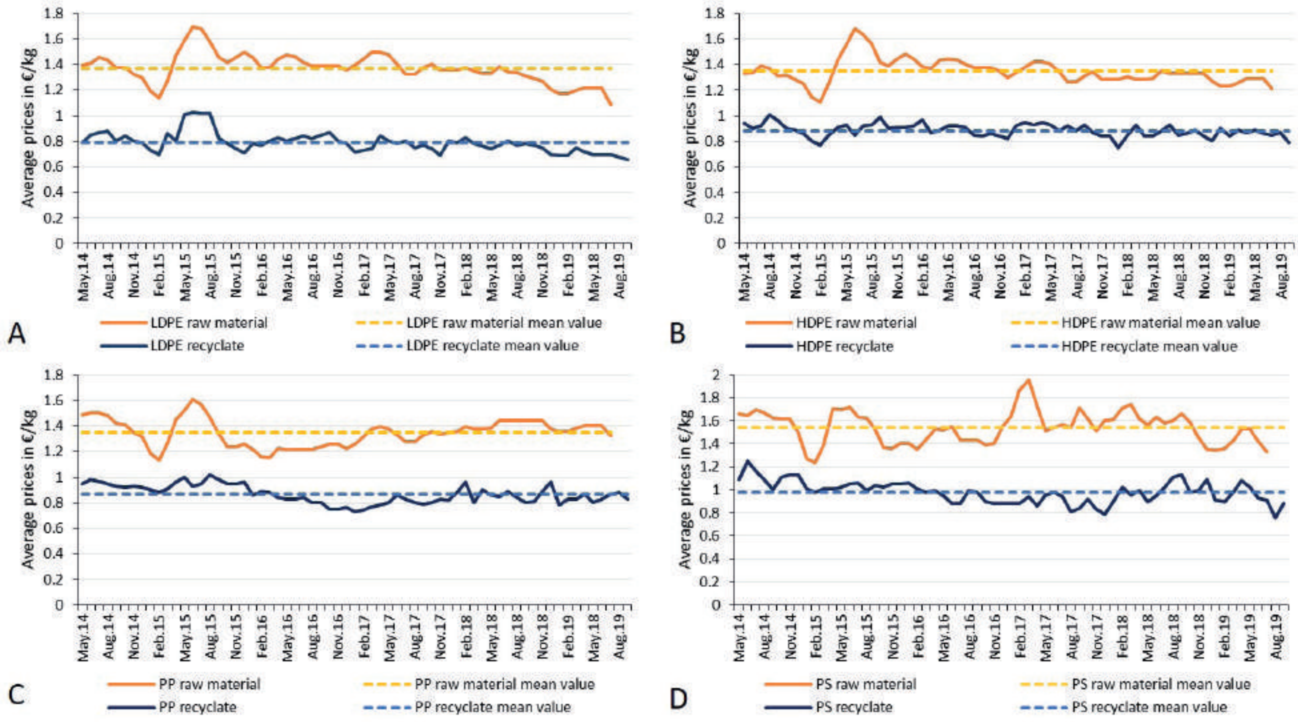

FIGURE 4: Price development for primary raw material and recyclates of LDPE (A), HDPE (B), PP (C) and PS (C) (Plasticker, 2019).

sorted plastic waste strangely affects the quality of the produced recyclates.

Pricing of recyclates

Basically, the market mechanisms of supply and demand apply. In addition, the following criteria were identified for pricing recyclates:

- Purity: the purer a material, the broader its range of application and the higher the price potentially achieved;
- Colour purity: the purer the colour of recycled material, the broader its range of application and the higher the price potentially achieved;
- A function of the primary raw material prices: Pricing polymer types is a function of the respective commodity price. If the price of primary raw material decreases, the price of polymers will drop as well. Recyclate prices are usually following the trend.

Other pricing contributors are melt filtration in the context the lower the melt filtration (measured in µm), the higher the quality and cost supplement for masterbatches. When plastic is dyed, a certain amount will be charged for this procedure, raising the price.

3.5 Quality benchmark in plastics recyclates

Market analysis has not produced any evidence for plastics recyclate benchmark. Therefore, producers of recyclates were asked to give one.

The surveys indicated that the quality standards for recyclates from Grüner Punkt (2019) are considered as a benchmark in the industry. For the recyclate quality, two levels are distinguished: mean quality for standard products like flower pots or buckets in 'standard plants' and high quality surpassing defined threshold values from Grüner Punkt (2019).

The demand for plastic recyclates is higher now than the recycling market is able to provide. For this reason, primary raw plastic granulates are mostly about 40 to 60% (see Figure 4) more expensive than plastic recyclates compared by the market data. The quality of recyclates is below that of primary raw plastic granulates regarding material properties but the consumers would tolerate it for the sake of sustainability. Better recyclability of plastics might reduce the market value of plastic recyclates. As best plastic recyclate quality, i.e. the benchmark, is met by plastic recyclates applied to food packaging like 'cap-to-cap' or 'bottle-to-bottle' production referring to the surveyed plastic processing companies.

4. CONCLUSIONS

The essential question was whether a correlation between price and quality of plastics recyclates is perceived. Experts from the plastics product manufacturing companies and plastics recyclers confirmed it unequivocally: The higher the quality of the material, the lower the impurities and the purer the material, the more applications for the material exist.

For sorted plastic waste, the plastic waste recycling companies quality standards defined by Grüner Punkt (2019) are considered a benchmark while recyclates applicable as food packaging (like cap-to-cap or bottle-to-bottle) constitute a benchmark for plastic recyclates.

In addition to the general market mechanisms of supply and demand, the pricing of secondary plastics is mainly a function of the purity of the recyclate, the purity of the colour and the respective price of raw materials. The purer and the cleaner the material, the higher the price that can be achieved on the market. The impact of respective commodity prices is also linked to the crude oil price and the dollar exchange rate.

Furthermore, the key competence of the staff in terms of quality control must be underlined. Their experience allows fast and reliable control, essential for successful further processing. For the quality control of recycled material, physical, rheological and mechanical properties are identified. In addition to density and melt flow rate, tensile properties and impact strength are identified to assure the required quality.

Plastic waste recycling companies would very much welcome a stipulation of minimum requirements for sorted plastic waste and recyclates by legislation.

Finally, it can be stated that, although the use of recyclates is facing some obstacles, many plastic product manufacturing companies are using plastic recyclates in their spite. There is a need for further changes at the political level (note: very positive example is "plastic strategy" of the EU) to help achieve a breakthrough. Many stakeholders along the plastic value chain would favour the further international introduction of quality standards. In addition, raising public awareness of the value of plastic waste is of key importance for further developments in the use of recycled plastic. Therefore, a package of measures and tools is needed to reduce obstacles and to promote high-quality plastics recycling as well as the use of recyclates.

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EVALUATION OF THE APPLICATION OF MUNICIPAL SOLID WASTE INCINERATOR (MSWI) ASH IN CIVIL ENGINEERING USING A SUSTAINABILITY APPROACH

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ABSTRACT
Incineration is regarded as one of the common methods for energy recovery as well as waste reduction, due to the high amount of waste generation in major cities; for instance Tehran (7000-8000 ton/day), and lack of sufficient landfill. The proper treatment and recycling of municipal solid waste incinerator (MSWI) residual ashes is one of the challenges which decision makers are faced with. In order to investigate the feasibility of the recycling of ashes, the sustainability index is considered. This evaluation is carried out by means of the multi-criteria decision-making approach for assessing sustainability (MIVES) and the Analytical Hierarchy Process (AHP) as a conventional decision-making tool. Six possible scenarios in Iran was determined, BA/FA landfilled with solid waste system (current scenario), Partial substitute of raw materials for cement/concrete, Ceramics and glass/glass-ceramics production, Geotechnical applications, use of BA/FA as alternative adsorbent and Fertilizers in agricultural soils. The assessment was accomplished through 25 questionnaires distributed among experts which includes environmentalists, governmental decision makers, academics, and technical groups. The questionnaires comprised of 33 pairwise comparison matrices, and the experts were asked to systematically compare elements of the constructed hierarchy in numerical terms. According to the results, reusing MSWI ash as a partial substitute for raw materials in cement/concrete scored highest in ranking among other potential MSWM scenarios (with a relative weight of 0.234). The results also reveal that the utilization of BA/FA as alternative adsorbents and as fertilizers in agricultural soils are not to be currently pursued in Iran (with relative weights of 0.117 and 0.129 respectively).

1. INTRODUCTION

On a global scale, urbanization has accelerated the generation of municipal solid waste (MSW) (Khajuria, 2010) (Talyan, 2008). Thus, disposal methods have become an increasingly important issue (Sabbas, 2003). As observable in Figure 1 and 2, an approximate trend is visible which indicates the relevancy between the amount of generated and incinerated waste. It is also noticeable that countries with higher levels of waste generation employ a wider spectrum of waste management approaches. Such diverse waste management approaches are shown in Table 1.

MSWI waste management is a subject of global debate. It has been shown globally that Waste-to-energy incineration process is a feasible management strategy for unrecyclable municipal solid waste (MSW) treatment and it is increasing by years.

The incineration process creates waste ashes; depending on the incineration process, about 80 to 90 percent of MSWI ash is comprised of bottom ash (BA), and 10 to 20 percent is fly ash (FA) by weight (Tasneem, 2014). BA is considered as non-hazardous waste; nevertheless, FA is frequently regarded as hazardous waste containing heavy metals and chloride ions, which commonly requires suitable pre-treatment prior to landfilling and reuse in construction (Margallo et al, 2015). MSWI residues are prone to recycling, given that they possess properties which are applicable in many fields (Hoornweg & Bhada-tata, 2012; Wiles & Shepherd, 1999; Forteza et al., 2004).

Due to the high cost of treatment and disposal and the imminent shortage of landfill space, the interest in potential reuses of BA and FA has been increasing in recent years (Forteza et al., 2004; Sormunen, 2016; Sun Li et al., 2016; Travar et al., 2009; U.S. EPA, 2016). In Iran, the first incinerator was employed in 2014 and currently, about 200 ton/day of 8500 ton/day generated solid waste is incinerated (Nabavi-Pelesaraei et al., 2017). Thus, Iran is a country that

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still has to pave the road of dealing with MSW incineration recycling in a noteworthy way. In terms of sustainability, there is a need to reduce consumption of global reserves of raw materials. The evolution of landfill diversion and closed-loop production models of waste as a resource material are initial steps towards Circular Economy (CE) thinking, which is increasing among European companies involved in sustainable materials management (Silva et al., 2016).

To incorporate environmental considerations in the development of MSWI ash recycling and reuse; authors employed a sustainability index (Pires et al., 2011). The success of sustainable development lies in the availability of standards by which we can compare and assess the sustainability index of any given alternative scenarios. Besides, it is very important for designing relevant policy for further improving the overall efficiency of solid waste management (Chen, 2014).

An appropriate method of assessment of sustainability is required to evaluate the management of MSWI solid residual. Many researchers have evaluated the sustainability index for solid waste management (MSW) in previous studies (Oehmig et al., 2015; Allegrini et al., 2015; Huang et al., 2015; Tasneem, 2014; Sou et al., 2016; Margallo et al., 2013; Travar et al., 2009). However; their common deficiency is the lack of emphasis on social aspects. Assessing the social aspects is of major significance since it facilitates decision-makers and the public in defining social ideals, linking them to clear objectives and targets and it eases the calculation of the impact of these activities on the environment and society.

This study attempts to address the suitable option for MSWI ash recycling in Iran with regard to technical feasibility, economy, environmental and social aspects. In addition, the multi-criteria decision-making approach for assessing sustainability (MIVES) is used to evaluate the sustainability index of MSWI ash reuse alternatives in civil engineering.

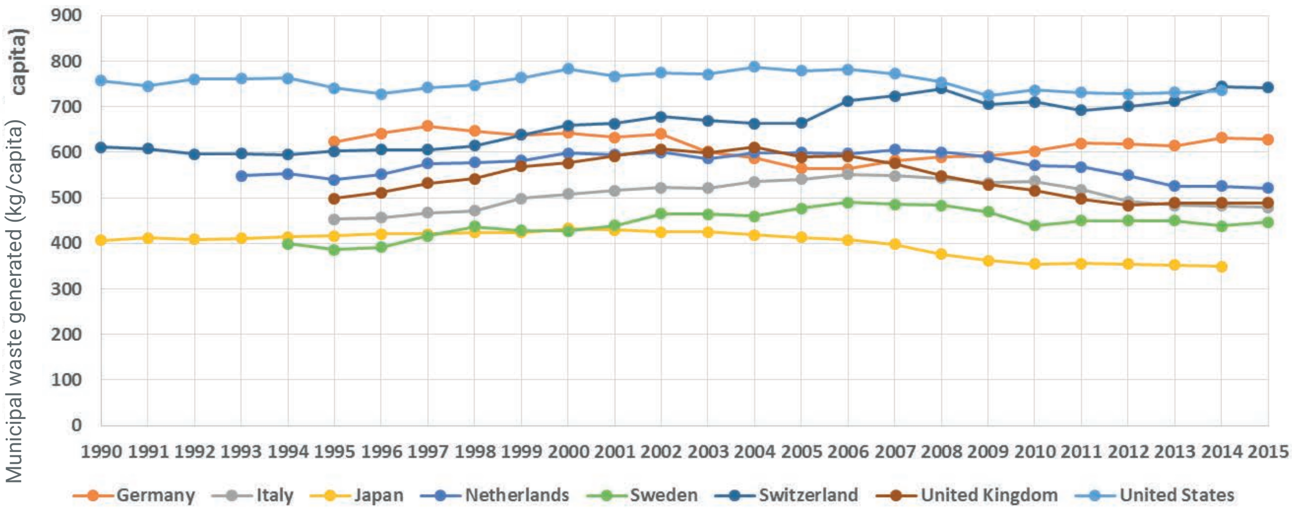

FIGURE 1: Annual municipal solid waste generation per capita in some European countries, Japan and the U.S ("http://stats.oecd.org/," 2017).

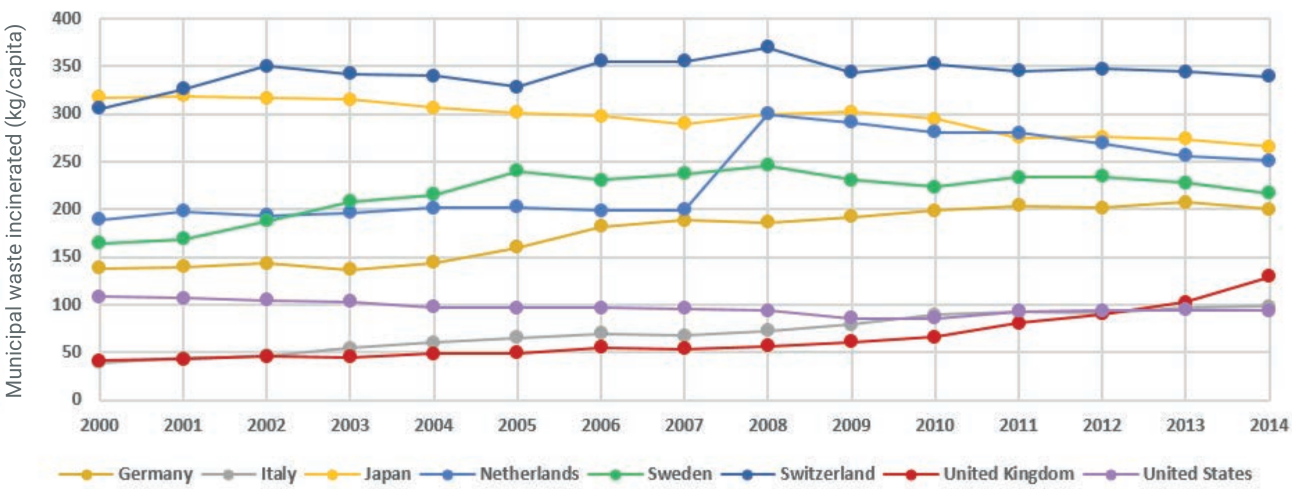

FIGURE 2: Annual municipal solid waste incinerated per capita in some European countries, Japan and the U.S ("http://stats.oecd.org/," 2017).

2. METHODS AND MATERIAL

The initial steps of this study aimed to review existing previous studies on solid residual from municipal solid waste incineration. After establishing varying viewpoints on waste incineration ashes, this paper attempts to provide assessment context for MSWI ash reuse in exemplary civil engineering fields by compiling a list of relevant sustainability criteria. Subsequently, the ideal analysis method with specific regards to the determined criteria and study goals is opted. Ultimately, the most appropriate scenario for reusing MSWI ash in Iran is proposed (Figure 3).

2.1 Applications of MSWI Ash

A considerable number of studies has implemented a systematic approach towards the beneficial utilization of MSWI ashes in European and Asian countries (Tasneem, 2014; Colangelo et al., 2012). Both BA and FA have been treated and furthermore successfully employed in civil engineering applications in varying forms and to various extents. For instance, BA is employed in road construction and embankment fill (Oehmig et al., 2015; Margallo, 2015). In addition, due to its pozzolanic reactivity, FA can be used as a substitute for lime or cement in soil stabilization (Lam et al., 2010; Tasneem, 2014). From the environmental perspective, these applications are promising and effective provided that technical feasibility is ensured. There is a broad spectrum of MSWI residue management approaches that are summarized in (Table 1) (Tasneem, 2014; Ørnebjerg, 2006).

2.1.1 Cement/Concrete Application

MSWI ash (fly and bottom ash) can be reused as a partial substitute for raw materials in cement/concrete production. Experimental results approved the possibility of casting concrete with a combined mix of MSWI bottom ash and fly ash as aggregates (Ginés et al., 2009).

The potential MSWI fly ash application in concrete is either as a replacement for cement or as a substitute for aggregate. Nonetheless, the addition of fly ash in cement will increase chloride and heavy metal concentrations(Ferreira et al., 2003). Moreover, bottom ash also possesses certain properties intended for reuse as a partial substitute for cementitious material due to its chemical components like CaO, SiO_2, Al_2O3, Fe_2O_3 that are similar to that of cement, but only after appropriate chemical treatment (Tasneem, 2014; Li, 2016).

2.1.2 Ceramic/Glass

Generally, the term 'ceramics' (ceramic products) is used for inorganic materials (with possibly some organic content), made up of non-metallic compounds which are hardened through firing processes (Barros, 2007). Consumption of silicate-based natural raw materials is predominant in the ceramic industry, which enables the consideration of MSWI ash as a surrogate, given that it is a silicate-rich substance. This fact renders the reuse of MSWI ash in a ceramic industry highly suitable(Ferreira et al., 2003). Glass ceramics are products with enhanced properties and higher market value which have unique applications, e.g. in the aeronautics industry.

2.1.3 Roads

Road construction entails vast amounts of natural aggregates. Studies demonstrated the possible reuse of MSWI ash in road construction, with BA being a more suitable candidate for its physical characteristics, whereas FA is considered as a secondary alternative due to its soluble salt and heavy metal contents (Margallo, 2015). The calculations clearly indicate that there are economic and environmental advantages in using waste in roads, including significant savings in costs, reduction in CO_2 emission, and energy consumption (Poulikakos et al., 2017). R. Forteza (Forteza et al., 2004) concluded that bottom ash is an exceptional substitute material for granular layers (bases and sub-bases) with regards to refined particle size distribution.

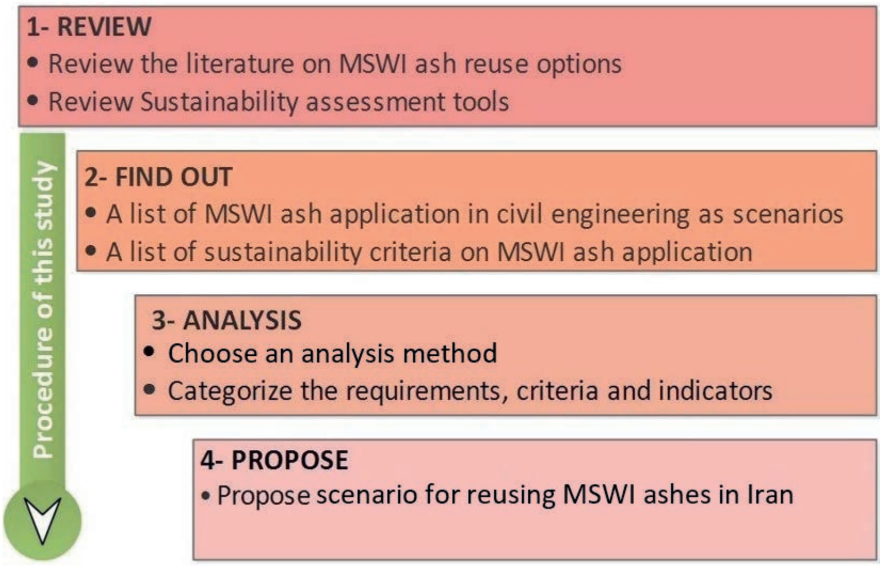

FIGURE 3: Procedure of this study.

TABLE 1: The management of MSWI ash in several countries (Ørnebjerg et al., 2006; Tasneem, 2014; Chandler, 2007; Vehlow, 2012).

Country	BA recycled	BA treatment	BA application	FA recycled	FA treatment	FA application
Netherlands	80%	ferrous and non-ferrous metal recovery, size reduction	embankment fill, road base, disposal into landfill	30%	-	filler material in asphalt as the alternative of limestone
Denmark	98%	screening, crushing, and ferrous metal recovery, gravel substitute as subbase material	building, road construction, embankment fill	exports APC residue to Norway or Germany	-	required to landfill after treatment
Germany	65%	reduction of salt content by water quenching, ferrous and non-ferrous metal recovery, 3-month maturation	road construction and secondary building materials	small quantity	stabilization	disposed into landfill
France	79%	ferrous and non-ferrous metal removal, size reduction, and sometimes cement, stabilization	civil constructions	-	cement and chemical stabilization using $NaHCO_3$, Thermal treatment	disposal into landfill designated for hazardous waste

2.1.4 Embankment

Embankments are constructed from soil or stone materials which are employed to restrain water. When soils do not display desirable geotechnical properties, it is common practice to stabilize them with lime or cement (Margallo, 2015). FA can be regarded as a replacement for lime or cement due to its pozzolanic reactivity (Garcia-Lodeiroet al., 2016), thus FA could be potentially reused as a stabilizer in embankments after treatment. R. Forteza (Forteza et al., 2004) inferred that bottom ash is classifiable as adequate soil for embankments and landfilling. The common challenging environmental issue of MSWI ash application in embankments and roads is pollutant exposure to soil and water systems (Margallo, 2015).

2.1.5 Adsorbent

The amount of colored wastewater has been increasing since industries like textile, paper, food, and cosmetics use dyes to color their products (Margallo, 2015). There are different ways to remove dyes (biological, chemical and physical), and adsorption is one of the most effective physical methods for color removal, in spite of it being a high-cost process. Instead of using commercial activated carbon or zeolites, Wang (Wang & Wu, 2006) proposed using fly ash for the adsorption of NOx, SOx, organic compounds, mercury in the air, cations, anions, dyes and other organic substances in water.

2.1.6 Agriculture

Nitrogen, phosphorus, and potassium are the three main elements commonly required for vegetative growth, hence the application of fertilizers to enrich the soil and assist plant growth is a common practice (Ferreira et al., 2003). In a general sense, MSWI BA and FA are rich in nutrients and this makes their use as fertilizers in agricultural soils possible. Agricultural utilization of MSWI ashes is an immensely controversial subject and some MSWI ashes, especially FA, are restricted to agricultural use given their heavy metal content. However, with appropriate treatment, the utilization of MSW ash will grow, which could subsequently lead to the reduction in commercial fertilizer usage (Margallo, 2015).

2.1.7 Landfill Cover

A landfill cover is a multilayer system that serves to reduce the emission of landfill gas into the atmosphere and the infiltration of water into the waste (Travar et al., 2009). Due to the conservation of natural resources, an alternative material possessing desired physical characteristics, like MSWI ash can be employed in landfill covers.

Furthermore, some studies show that FA heavy metal content improves biogas production in landfills (Marı´a Margallo, 2015). Similar to the aforementioned issues, this application too, faces serious environmental concerns due to leaching and release of contaminants (heavy metals) (Travar et al., 2009).

2.2 Sustainability tools

Due to the breadth of subject and effectiveness of various factors involved, a comprehensive assessment methodology is required. MIVES (Modelo Integrado de Valor para una Evaluacion Sostenible or Integrated Value Model for Sustainability Assessment) is a multi-criteria decision-making method (MCDM) that constructs a value function on the basis of the utility theory. MIVES offers three key advantages over other frequently used MCDMs, in that it is time-independent; it is adjustable and can be modified to take all the stakeholders' demands into account through simple adjustments to the requirements tree's items and their weights (Hosseini et al., 2018) ;and it is flexible and can be applied to myriads of fields with varying characteristics and conditions. As far as now, MIVES has already been used for the energy sector (Barros et al., 2015), urban planning (Pujadas et al., 2017; Piñero et al., 2017), buildings (Pons & Aguado, 2012; Lombera & Rojo, 2010), public infrastructures (Pardo-Bosch & Aguado, 2016), sewage systems (de la Fuente et al., 2016), wind towers (de la Fuente et al., 2017), and civil and architect projects (Pons et al., 2016). MIVES helps to consider potential nonlinearities in the evaluation and it integrates AHP (Analytical Hierarchy Process). The Analytical Hierarchy Process (AHP) is a decision-aiding method. This approach enables the decision maker to organize tangible and intangible elements and offers a structured and simple solution to the decision-making problems (Al-Harbi, 2001). AHP provides element weighting so that the system is organized according to the opinions returned by a group of experts regarding the relative importance of elements, whereas MIVES is used to produce value functions in order to transform indicators measured in different units into a value index. The diversity of these cases, some at an energy or urban level, some about building elements, some assessing broad samples in general, and some carrying out analysis in detail, etc., shows MIVES's adaptability.

Moreover, as observable in (Blanco et al., 2017; Aguado et al., 2016; Fuente et al., 2016), the most commonly used method of assessment is the AHP method and MIVES, which is multivariate which is capable of incorporating multiple sustainability indexes in the sustainability assessment and multilateral which covers a diverse variety of sustainability problems. According to (Pérez, 1995) AHP possesses a firm theoretical foundation and its viability has been illustrated through operations of all kind (Governmental agencies, Corporations, consulting firms). Besides, AHP allows the comparison of several criteria and is capable of checking the inconsistency of expert judgments. Along with intangibles such as political and social factors, tangibles like economic and technical factors can also be considered with this method.

2.3 Sustainability approach

This study employs a combination of the MIVES and AHP method to assess the sustainability of various MSWI ash management options. This Method involves plugging the environmental, economic, social and technical feasibility aspects of any scenario into an assessment evaluation framework. Six different MSWI ash management scenarios were proposed and furthermore, compared in this study, based on a broad range of beneficial utilization of MSWI ash mentioned earlier. The potential MSWI ash management scenarios are as follows:

0. BA/FA landfilled with solid waste (current system);
1. Partial substitute of raw materials for cement/concrete;
2. Ceramics and glass/glass–ceramics production;
3. Geotechnical applications including road construction, embankment and landfill cover materials;
4. Use of BA/FA as alternative adsorbent;
5. Fertilizers in agricultural soils.

According to the work of Blanco (Blanco et al., 2016), the current model also requires defining three basic terms, namely; The system boundaries, Tree requirements, The value functions that will turn Physical units or features of each index to a value between 0 and 1. Nonetheless, given that the present indexes are not of binary nature, AHP is used in order to assign a weight to each element of the requirements tree.

The analytical hierarchy is typically comprised of four levels (Goal, criteria, sub-criteria, and alternatives). The criteria are the environmental (C1), economic (C2), social aspects (C3) and technical feasibility (C4) of any given scenario.

Initially, a proper hierarchy of the AHP model containing the goal, four major decision criteria and 28 sub-criteria that have been selected due to their thematic affinity with the criteria and six goal-orientated alternative scenarios, was structured (Figure 4). Likewise, Yin (Yin et al., 2016) employed similar AHP criteria (environmental, technical, economic and social indicators) to evaluate four MSWM scenarios, including waste incineration, in the northern region of China. Dong (Dong et al., 2014) has also pursued an energy-efficient, environmentally friendly and economically affordable MSWM system via multi-criteria decision making (MCDM) using three main criteria (energy, environment and economy) based on life cycle assessment and life cycle cost inventories.

Defining the criteria is very vital in problem-solving. Sustainable strategies for MSWM entail precise objectives to be conveyed and applicable measures to be taken with regards to the political, social, financial, economic, and technical aspects of waste management (Schübeler et al., 1996).

The environmental impacts of MSWM systems should be considered because if managed inappropriately, waste poses a risk to human health and the environment.

Social acceptance is significant in all stages of waste management, from the conceptual stage to the implementation, including behavior upgrading. Social acceptance is encouraging the active participation of social actors in the decision-making process.

Technical feasibility has to be taken into attention on the basis of whether the suitable resources are existing or reasonably accessible to implement a specific alternative.

Economic aspects should never be overlooked in decision making, as the economic evaluation of the total cost of owning and operating a facility over a period of time is one of the most major criteria in engineering. The numbers in parentheses (relative weights) are obtained through the weighting process using the AHP method and are the result of the final analysis. The relative weights of the criteria represent their local priority with respect to the goal. Likewise, the relative weights of the sub-criteria are obtained with respect to their corresponding criteria and exhibit a distribution of local priority within each criterion.

Furthermore, questionnaires were distributed among individuals or organizations representing four interest expert groups. The four expert groups include:

(a) Environmentalists (environmental activists in NGO's);
(b) Governmental decision makers (local government officials);

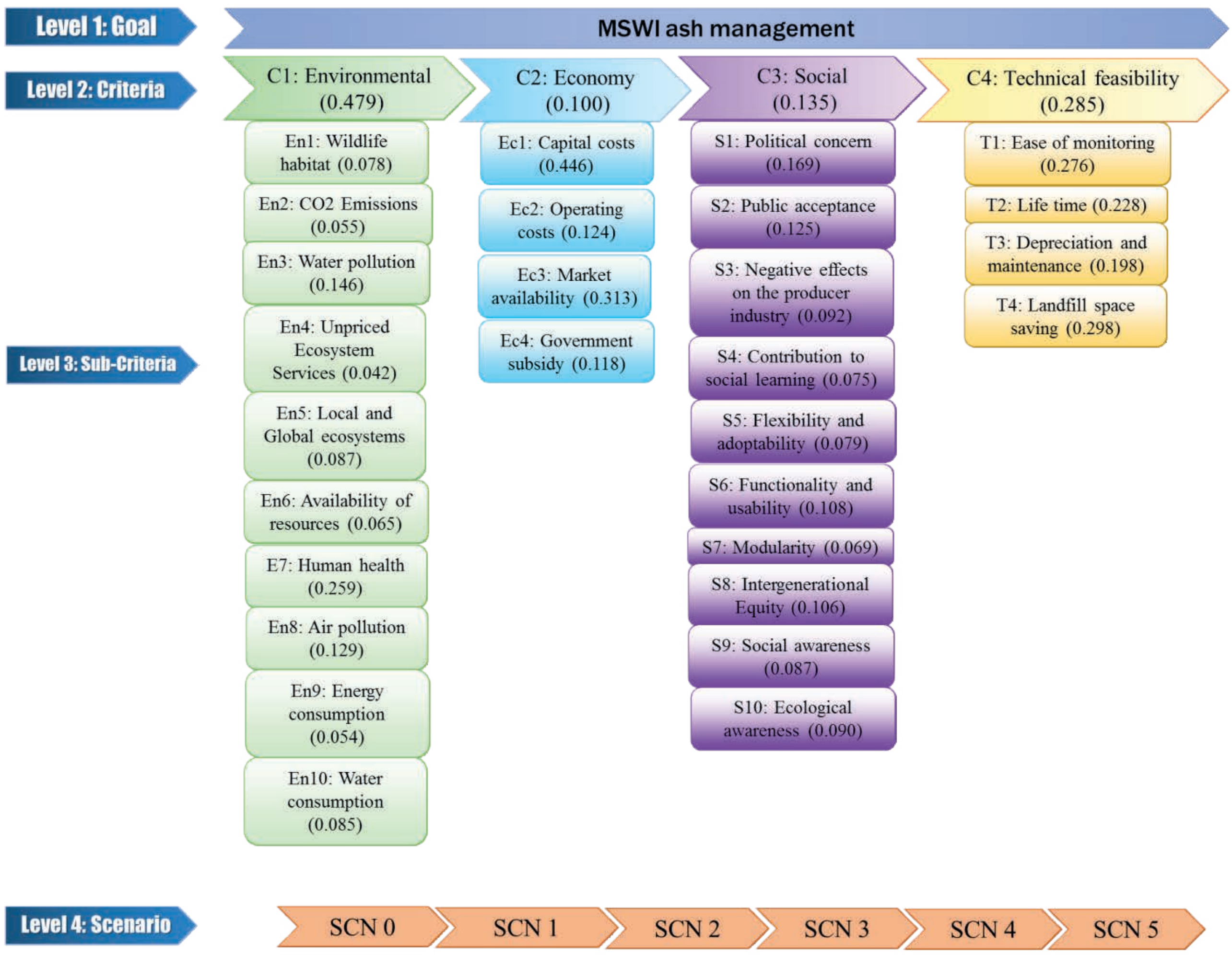

FIGURE 4: Hierarchy of the AHP model for the proposed MSWM in Iran.

(c) Academics (professors in major universities of Iran);
(d) Technical groups (operative and technical staff of waste management facilities in Tehran).

The results were calculated using the Expert Choice® software. Expert Choice® is a commercially available decision-making software which implements the analytical hierarchy process (AHP) based on the model hierarchy and pairwise comparison matrices as data inputs. Pairwise comparisons between the criteria and each set of sub-criteria were conducted in order to carry out the weighting of each component. Subsequently, the obtained weights were aggregated to establish a prioritization of alternatives, the maximum value of which is to be considered the preferred alternative. There is no need to implement analysis steps manually since Expert Choice® simplifies the implementation of the AHP steps and automates many of its computations (Al-Harbi, 2001; Ishizaka & Labib, 2009).

The main goal of the developed model is to evaluate the most sustainable MSWI ash management in Iran by calculating its sustainability index. In order to devise an optimal problem-solving algorithm, a precise definition of the sub-criteria is required. The environmental impacts of MSWI ash management options have to be carefully taken into account and if managed improperly, they may pose a threat to human health and the environment.

Economic aspects are a matter of utter importance in decision making (Hacer, 2015). The monetary issues are evaluated under the capital and operating costs sub-criteria in this study.

Social aspects of MSWM play a notable role in all stages of waste management, from the conceptual stage all the way through to the implementation (Hacer, 2015). Social aspects facilitate the active participation of social actors in the decision-making process.

Technical feasibility should also be accounted for, based on the probable availability of the required resources or reasonable accessibility to implement a specific alternative. The corresponding sub-criteria for each of the above criteria are demonstrated in Table 5. The important factor of each criterion and sub-criterion was established by the target experts through completing questionnaires. These questionnaires only comprise of pairwise comparison matrices $(n \times n)$ for each of the lower levels and one matrix for each element in the level immediately above, by using the relative scale measurement shown in Table 2 (Saaty, 2008).

The pairwise comparison matrix (PCM) for a set of n criteria $\{C_i \,|\, 1 \le i \le n\}$ is constructed as follows:

$$[PCM]_{n\times n} = \begin{bmatrix} r_{11}=C_1/C_1 & r_{12}=C_1/C_2 & \cdots & r_{1n}=C_1/C_n \\ r_{21}=C_2/C_1 & r_{22}=C_2/C_2 & \cdots & r_{2n}=C_2/C_n \\ \vdots & \vdots & \cdots & \vdots \\ r_{n1}=C_n/C_1 & r_{n2}=C_n/C_2 & \cdots & r_{nn}=C_n/C_n \end{bmatrix} = [r_{ij}]_{n\times n} \quad (1)$$

Each element of the pairwise comparison matrix $r_{ij} = C_i/C_j$ represents the relative importance of one criterion to another. Given the structure of the PCM, an intrinsic reciprocal feature of the PCM is observed. This feature is mathematically defined as following:

$$r_{ij} = 1/r_{ji} \qquad \forall\, i,j = 1,\dots,n \quad (2)$$

In the next step (Al-Harbi, 2001), hierarchical synthesis is used to weight the eigenvectors regarding the weights of the criteria and the summation of all weighted eigenvector entries corresponding to those in the next lower level of the hierarchy. Synthesizing the pair-wise comparison matrix is achieved by dividing each element of the by its column total:

$$[PCM_{synth}]_{n\times n} = \begin{bmatrix} s_{11}=r_{11}/\sum_{i=1}^{n} r_{i1} & s_{12}=r_{12}/\sum_{i=1}^{n} r_{i2} & \cdots & s_{1n}=r_{1n}/\sum_{i=1}^{n} r_{in} \\ s_{21}=r_{21}/\sum_{i=1}^{n} r_{i1} & s_{22}=r_{22}/\sum_{i=1}^{n} r_{i2} & \cdots & s_{2n}=r_{2n}/\sum_{i=1}^{n} r_{in} \\ \vdots & \vdots & \cdots & \vdots \\ s_{n1}=r_{n1}/\sum_{i=1}^{n} r_{i1} & s_{n2}=r_{n2}/\sum_{i=1}^{n} r_{i2} & \cdots & s_{nn}=r_{nn}/\sum_{i=1}^{n} r_{in} \end{bmatrix} \quad (3)$$

Moreover, the weight vector is determined by finding the row averages of the synthesized pair-wise comparison matrix:

$$[w]_{n\times 1} = \frac{1}{n}\begin{bmatrix} \sum_{i=1}^{n} s_{1j} \\ \sum_{i=1}^{n} s_{2j} \\ \vdots \\ \sum_{i=1}^{n} s_{nj} \end{bmatrix} = \begin{bmatrix} w_1 \\ w_2 \\ \vdots \\ w_n \end{bmatrix} \quad (4)$$

Given that $[w]_{n\times 1}$ represents the weight vector and λ_{max} is the principle eigenvalue of the pairwise comparison matrix $[PCM]_{n\times n}$, the following product matrix is generated:

$$[P]_{n\times 1} = [PCM]_{n\times n} \cdot [w]_{n\times 1} \quad (5)$$

$$\begin{bmatrix} p_1 \\ p_2 \\ \vdots \\ p_n \end{bmatrix} = \begin{bmatrix} r_{1,1} & r_{1,2} & \cdots & r_{1,n} \\ r_{2,1} & r_{2,2} & \cdots & r_{2,n} \\ \vdots & \vdots & \cdots & \vdots \\ r_{n,1} & r_{n,2} & \cdots & r_{n,n} \end{bmatrix} \cdot \begin{bmatrix} w_1 \\ w_2 \\ \vdots \\ w_n \end{bmatrix} \quad (6)$$

Furthermore, in order to obtain λ_{max}, the following equation is perturbed:

$$\lambda_{max} = \frac{1}{n}\sum_{i=1}^{n} \frac{p_i}{w_i} \quad (7)$$

Another feature of pairwise comparison matrices larger than (3×3) is referred to as the consistency condition. This condition is expressed as follows:

$$r_{ik} = r_{ij} r_{jk} \qquad \forall\, i,j,k = 1,\dots,n \quad (8)$$

If and only if this condition is satisfied, then the obtained weighted ranking is in complete agreement with the expressed preferences and the PCM is consistent. In addition, with respect to (Saaty, 2008), it is necessary to determine consistency by means of the eigenvalue λ_{max}, and furthermore calculating the consistency index CI as follows: $CI = (\lambda_{max} - n)/(n-1)$, where n is the corresponding matrix size. The consistency evaluation is completed by calculating the consistency ratio CR as follows: $CR = CI/RI$, where RI represents the random consistency index derived from Table 3.

TABLE 2: Pair-wise comparison scale for AHP preference (Saaty, 2008; Al-Harbi, 2001).

Intensity of importance	Definition
9	When one activity is extremely more important than the other
8	Very strongly to extremely
7	Very strong or demonstrated importance
6	Strongly to very strongly
5	Strong or essential importance
4	Moderately to strongly
3	Moderate importance of one over another
2	Equally to moderately
1	When two criteria are of equal importance to the objective

Providing CR exceeds 0.10, the judgment matrix is inconsistent (not consistent), and judgments should be reviewed and improved. All of the above mentioned steps should be taken for all levels in the hierarchy.

3. RESULTS AND DISCUSSION

As discussed above, the questionnaires distributed among four target groups were collected and were further used as a foundation for Expert Choice® inputs. As depicted in Figure 3, the lowest level of the hierarchy comprises of six proposed MSWM scenarios. As depicted in table 4, the results reveal the following ranking of the scenarios: 1 > 2 > 3 > 0 > 5 > 4 with the overall inconsistency of 0.01 that is less than 0.1, thus validating decision maker judgements. According to the analyzed data, the currently active MSWM practice in Iran which consists of landfilling has a sustainability index of (0.154). Scenario 1 proposes the function of MSWI ash as a partial substitute for raw materials in cement/concrete, which results in a remarkable change in the sustainability index (0.234). Scenario 2 envisions decreasing landfilled MSWI ash through their application in ceramics and glass/glass-ceramics production, which also improves the sustainability index (0.206). Scenario 3 suggests the use of MSWI ash in geotechnical applications including road construction, embankment and landfill cover materials, with a sustainability index of (0.159). Scenario 4 and 5 propose the utilization of BA/FA as alternative adsorbents and as fertilizers in agricultural soils respectively, which have a slight impact on the sustainability index (0.117) and (0.129).

Moreover, as seen in Figure 4, the highest weighted criteria are environmental impact and technical feasibility with 0.479 and 0.285, whereas the lowest weighted crite-

TABLE 3: Average random consistency (Saaty, 2008; Al-Harbi, 2001).

n	1	2	3	4	5	6	7	8	9	10
RI	0	0	0.58	0.9	1.12	1.24	1.32	1.41	1.45	1.49

TABLE 4: Overall priority for scenarios with respect to the goal.

Category	Sc. 0	Sc. 1	Sc. 2	Sc. 3	Sc. 4	Sc. 5
Goal	0.154	0.234	0.206	0.159	0.117	0.129
Order	4	1	2	3	6	5

ria include economy with 0.100. As for the reasons behind the evident shortcoming of social aspects in criteria rankings (0.135), the relatively qualitative characteristics and conceptual intangibility of such a novel criterion come to mind. These are precisely the reasons Hacer AK and Washington Braida asserted in their study (Hacer AK, 2015) that among the sub-criteria related to environmental aspects, the impact on human health and water pollution were of solid importance compared to other sub-criteria. Similarly, among the sub-criteria under economic aspects, capital costs and market availability were relatively more important. Furthermore, landfill space saving and ease of monitoring were strongly more important than others among technical feasibility sub-criteria. Ultimately, upon weighing social aspects sub-criteria, it is understood that political concern, public acceptance, functionality, and usability are of relative importance.

The final step of the AHP modeling involves conducting a sensitivity analysis on the final outcome, using Expert Choice® sensitivity performance graph.

The performance sensitivity plot resulted from the Expert Choice® analysis in Figure 5 depicts the relative im-

TABLE 5: Selected criteria and sub-criteria for sustainable MSWI ash management.

Criteria	Sub-criteria	Description
Environmental	Wildlife Habitat	How does the operation aim to protect, preserve and restore the habitat areas for wild plants and animals.
	CO_2 Emissions	The potential Carbon dioxide levels generated from the MSWM scenarios.
	Unpriced Ecosystem Services	Recognizing all ecosystem services including those currently unpriced (e.g. pollination, water regime maintenance, climate reliability and nutrient cycling).
	Affecting Local and National Ecosystems	How does the operation affect the capacity of the local ecosystem to deliver valued ecosystem services reliably into the future (e.g. effects on water and air quality, and wildlife habitat).
	Availability of Resources	How does the operation affect the long term availability of non-renewable and renewable resources.
	Human Health	How does the operation affect human health (including exposure to toxic substances and sanitation issues).
	Air Pollution	To what extent does this operation contribute to the air pollution
	Water Pollution	To what extent does this operation contribute to the water pollution (leachate production into groundwater and reservoir for each MSWM scenario
	Energy Consumption	Comparison of energy consumption for each MSWM scenario.
	Water Consumption	Comparison of water consumption for each MSWM scenario.
Economy	Capital Costs	The cost to design, construct and expand new MSWM facilities and providing necessary equipment, infrastructure, logistics and land.
	Operating Costs	Yearly expenditures for fuel, maintenance, fringe benefits etc.
	Market Availability	The market demand for recycled materials, compost and electricity from MSWM scenarios.
	Government Subsidy	The probability of a government subsidy for each MSWM scenario.
Social	Political Concern	The role of politics in MWSI ash management.
	Public Acceptance	The aspect of public involvement in MSWM practices.
	Negative Effects on the Producer Industry	How each MSWM scenario results in production decline in other producer industries
	Contribution to Social Learning	Is the current scenario an advocate of an eco-friendly lifestyle
	Flexibility and Adaptability	The potential of each MSWM scenario to adapt to change.
	Functionality and Usability	How functional is the current MSWM scenario.
	Modularity	Is each MSWM scenario a self-reliant system and does it avoid over-connectedness and associated relations of dependence subjected to shock
	Intergenerational Equity	How does the MSWM scenario affect potential costs and benefits for future generations.
	Social Awareness	How does the MSWM scenario affect the social awareness of citizens.
	Ecological Awareness	How does the MSWM scenario affect the ecological awareness of citizens.
Technical Feasibility	Ease of Monitoring	The act of monitoring the emissions and related impacts over time (Leaching, Strength).
	Lifetime	A lifetime of the technical systems associated with the MSWM scenario and the total required area for the MSWM technology.
	Depreciation and Maintenance	Ease of technical systems maintenance in each MSWM scenario.
	Landfill Space Saving	The ratio between waste not landfilled and total waste generated in a year.

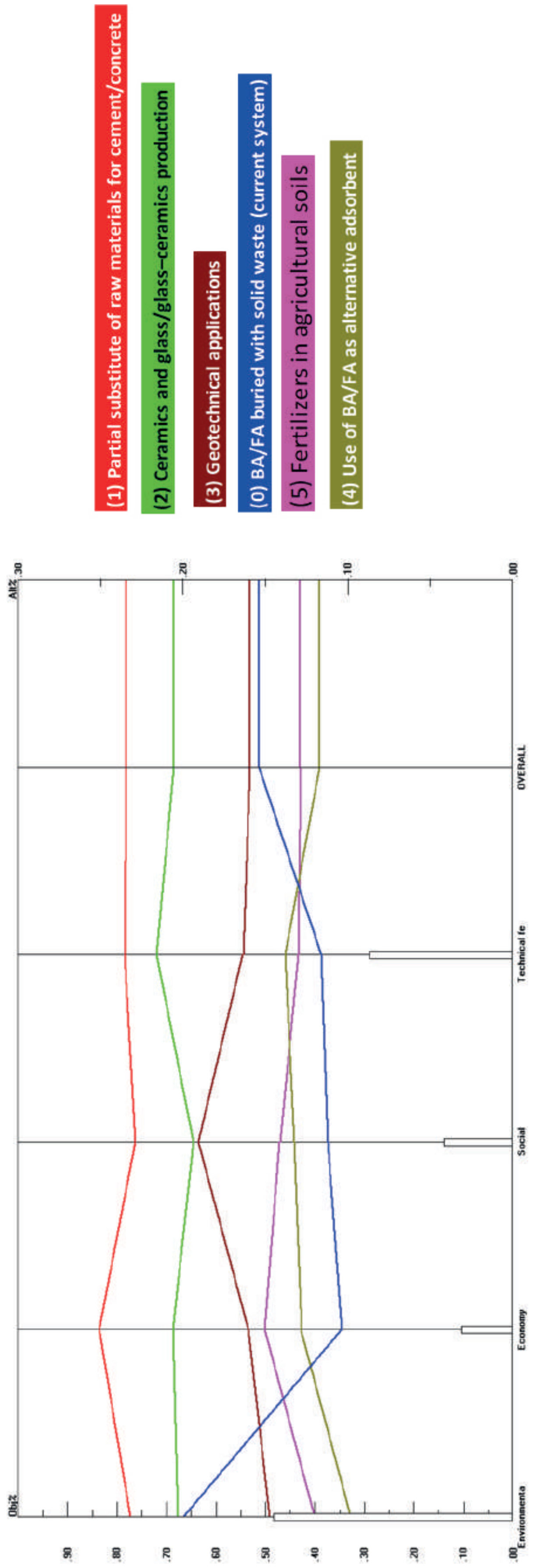

FIGURE 5: Performance sensitivity graph with respect to the goal.

portance of each criterion as bars and the relative preference of each scenario as the intersection of the scenarios curves with the vertical line for each criterion. The overall scenario importance is indicated on the right. For this purpose, the change in scenario rankings by altering the relative importance of each criterion from 0% to 100% is exhibited in Table 6 and Table 7.

By decreasing the importance of environmental, economic, social aspects and technical feasibility to 0%, scenarios 1 and 2 still rank top priorities and scenarios 5 and 4 score last. Via downgrading the relative importance of social aspects and technical feasibility, scenarios 3 and 0 swap priorities and scenario 3 drops below the currently practiced scenario 0. This ranking swap occurs by virtue of scenario 0 scoring the lowest in social and technical feasibility. Through mitigating economical aspects, scenario 0 and 3 shared a common ranking. Ultimately, it is observed that minimizing the importance of environmental aspects has no evident effect on scenario rankings.

Similarly, by increasing the importance of each criterion to 100% so as to maximize its effective impact, the following can be observed. As shown in Table 7, it is recognized that Scenario 1, with the highest overall score, is the most sustainable approach to MSWM, largely due to the fact that the economic benefits and technical feasibility of Scenario 1 were more significant than that of any other proposed scenario. Furthermore, scenario 2 is regarded as the second most sustainable approach to MSWM, with a low impact on all criteria. Scenario 4 and 5 acquired the lowest overall score and demonstrate the lowest environmental impact due to pre-treatment impacts on the environment. Hence, it is concluded that scenario 4 and 5 do not provide the credentials to be regarded as suitable MSWM alternatives in Iran. Scenario 0 is ranked the fourth suitable option, albeit scoring the lowest in all aspects. This is primarily due to its moderate to strong score in environmental impacts.

4. CONCLUSIONS

In this study, a combination of MIVES and an AHP assessment model was developed in order to assess sustainability indexes regarding reusing MSWI ash. A hierarchy of four criteria and 28 sub-criteria was defined in this study as in to establish a comparison between six proposed scenarios for choosing the most suitable MSWI ash management in Iran. These criteria include environmental, economic, social aspects and technical feasibility.

The apparent uniqueness of this current study lies in the fact that it endeavored to formulate an extensive assessment tool. This precisely justifies the incorporation of the social criterion, a factor which is often dismissed in MSWM research in Iran. It is observable that the current scenario (scenario 0: BA/FA landfilled with solid waste) is not the ideal MSWM strategy due to its inefficiency in most aspects. Landfilling MSWI ash is economically deficient and prevents the reuse of potentially recyclable materials. Another issue revolving the commonly practiced landfilling actives in Iran, is the lack of procedural monitoring and standard compliance, which renders such actives environmentally counterproductive.

TABLE 6: The relative weight by omitting factors separately.

Criteria	Change in relative importance	The weighting of each scenario						Ranking
		Sc. 0	Sc. 1	Sc. 2	Sc. 3	Sc. 4	Sc. 5	
Environment	decrease to 0%	11.20%	23.70%	20.80%	17.10%	13.50%	13.70%	1 > 2 > 3 > 0 > 5 > 4
Economy	decrease to 0%	15.90%	23.30%	20.60%	15.90%	11.60%	12.70%	1 > 2 > 3 = 0 > 5 > 4
Social	decrease to 0%	16.00%	23.50%	20.80%	15.50%	11.50%	12.70%	1 > 2 > 0 > 3 > 5 > 4
Technical feasibility	decrease to 0%	16.90%	23.40%	20.20%	15.80%	10.90%	12.90%	1 > 2 > 0 > 3 > 5 > 4

TABLE 7: The relative weight by considering one factor.

Criteria	Change in relative importance	The weighting of each scenario						Ranking
		Sc. 0	Sc. 1	Sc. 2	Sc. 3	Sc. 4	Sc. 5	
Environment	increase to 100%	19.90%	23.20%	20.30%	14.70%	9.80%	12.00%	1 > 2 > 0 > 3 > 5 > 4
Economy	increase to 100%	10.40%	25.10%	20.60%	16.00%	12.80%	15.00%	1 > 2 > 3 > 5 > 4 > 0
Social	increase to 100%	11.20%	22.90%	19.40%	19.10%	13.30%	14.10%	1 > 2 > 3 > 5 > 4 > 0
Technical feasibility	increase to 100%	11.60%	23.50%	21.60%	16.40%	13.80%	13.00%	1 > 2 > 3 > 4 > 5 > 0

The results revealed partial substitute of raw materials for cement/concrete (Scenario 1) as the top alternative solution. The main factors underlying such an outcome, are the economic and ecological advantages of BA/FA mixed cement throughout its life-cycle. The cost-effectiveness of partially replacing a high-value product like cement with treated MSWI ashes has a great impact on the cement industry, one of Iran's most vital industries.

According to the judgements, the utilization of BA/FA as alternative adsorbents and as fertilizers in agricultural soils (Scenarios 4 and 5) are not to be currently pursued in Iran given that they do not result in environmental or economic advantages, and that there are many principal difficulties involved in developing such applications and doubts about controlling their impact on human health. The common denominator in the shortcoming of either application is the leachability of heavy metals from MSWI ashes. The leaching of heavy metals in the ground water produces serious pollution loads in the already heavily polluted water regime of the country and water pollution is a subject of national sensitivity in Iran, especially in the wake of the nation's imminent widespread water shortages. Moreover, reusing waste residues as alternative fertilizers face strong public opinion, specifically due to a recent growing predilection for organic products.

Moreover, the current study only attempted to evaluate MSWM scenarios using the sustainability index and did not integrate resilience based criteria into the sustainability assessment to invoke a respect for socio-ecological system complexity. Nevertheless, resilience itself is a subject of intrinsic complexity and requires on-going implementation data and iterative operation improvement as analysis inputs, which are currently unavailable in Iran. Hence, the incorporation of resilience-based criteria and sustainability assessment demands further development and calls for extensive future work.

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PRODUCTION AND CHARACTERISATION OF SRF PREMIUM QUALITY FROM MUNICIPAL AND COMMERCIAL SOLID NON-HAZARDOUS WASTES IN AUSTRIA, CROATIA, SLOVENIA AND SLOVAKIA

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ABSTRACT

The production of Solid Recovered Fuel (SRF) and related energy recovery in the European cement industry represents the state of the art in waste management, having evolved into a highly important part of a sustainable and circular economy. This paper describes the production and quality of eight Solid Recovered Fuels (SRF) of PREMIUM quality that are produced from Municipal (Mixed) and selected Commercial Wastes (i.e. Bulky and Lightweight Fraction from Plastic Sorting Plants) in three types of treatment plants located in four European countries, namely Austria, Croatia, Slovenia and Slovakia. The investigated SRF PREMIUM Quality was produced in three different Plant Types applying various process technologies. All three types have been investigated and are described in detail (i.e. flow sheet). Eight SRF PREMIUM Qualities have been comprehensively investigated by sorting, sieving, and physical-chemical analyses. Analyses performed are in accordance with (inter)national standards (i.e. Austrian "ÖNORM", European "EN" standards and CEN TC 343 guidelines). The results gained show that all investigated SRF fulfil the Austrian quality requirements for heavy metals before co-incineration in the cement industry and it can be confirmed that SRF produced in the investigated plants in Austria, Croatia, Slovenia and Slovakia in fact may be declared as "SRF PREMIUM Quality" that can be used for energy recovery on the European SRF market and utilized in the European cement industry.

1. INTRODUCTION

This introductory chapter covers two issues, namely: "From Municipal and Commercial solid non-hazardous Waste to SRF" and "Applied technologies for production of SRF in investigated countries".

1.1 From Municipal and Commercial solid non-hazardous Waste to SRF

Waste used in resource management is based on the separate collection of valuable and suitable waste materials (e.g. glass, metals, paper, plastics) for recycling, treatment of biowaste (i.e. composting or fermentation) and highly efficient (co-)incineration of (pre-treated) residual and commercial waste. This statement is confirmed at international level too and is discussed in case studies given by Ionescu et al. (2013), Rada et al. (2018), Ranieri et al. (2017), Sipra et al. (2018) and Stępień and Białowiec (2018). Different treatment and manufacturing steps need to be accomplished before a waste becomes a waste fuel for co-incineration plants (e.g. cement industry). Common treatment and manufacturing steps include multistage shredding, classifying, separation of Fe-metals, Non-Fe-metals, and heavyweight inert materials, as well as sorting out of unwanted materials like polyvinyl chloride (PVC) plastics by using modern near-infrared (NIR)-sorting technology. In total, the waste management industry must pass different development steps, which can be generally divided into four characteristic groups (i.e. legal, material, plant, and economic) to become a fuel supplier for the co-incineration sector. Individual developments and properties of SRF are extensively discussed in Beckmann et al. (2012), Pomberger and Sarc (2012), Sarc and Lorber (2013) and Sarc et al. (2014, 2019).

In Austria, the definition of Waste Fuels or Refuse Derived Fuels (RDF) is given in the legally binding Waste Incineration Ordinance (BMLFUW, 2010) as:

"...waste that is used entirely or to a relevant extent for

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the purpose of energy generation and which satisfies the quality criteria laid down in this directive. "

Therefore, after adequate and extensive (pre-)treatment in different processing plants and applying strictly defined quality assurance measures, various non-hazardous and/or hazardous waste materials from households, commerce and industry can be used as RDF in co-incineration plants: e.g.: Sewage Sludge, Waste Wood, High Calorific Fractions from mechanical-physical (MPT) or mechanical-biological treatment (MBT) plants, Calorific Fractions of Household and Commercial Waste, Shredder Light Fractions (e.g. from old vehicles and Waste Electric and Electronic Equipment), Scrap Tyres, Waste Oil and Used Solvents, etc. In the narrow sense of the definition, only solid Waste Fuels which are prepared from non-hazardous sorted or Mixed Solid Wastes (i.e. Municipal Waste Fractions, Commercial Wastes, Production Wastes, Packaging Wastes, Lightweight Fractions from MBT-plants, etc.) including legally defined quality assurance measures and then used for energy recovery are classified as Solid Recovered Fuels - SRF (Lorber et al. 2012). Therefore, SRF is presenting a subgroup of RDF and for both waste fuels the same limit values are defined in the Waste Incineration Ordinance (BMLFUW, 2010).

In regard to the papers from Sarc et al. (2014, 2019), where the requirements for production, quality and quality assurance of solid recovered fuels (SRF) used in Austrian cement industry were presented, this paper expands the scope of investigation, besides on Austrian cement industry, also on Croatian, Slovakian and Slovenian cement plants. Selected information on Mechanical-Biological Waste Treatment (MBT) Plants and co-incineration in cement industry in Croatia have been given in Sarc et al. (2018).

The following investigated SRF that has been used in previously indicated co-incineration plants is extensively investigated, characterised and discussed:

SRF PREMIUM Quality: having particle size distribution (d_{95}) ≤ 30 (up to 35) [mm] and lower heating value (LHV) ≥ 18 ≤ 25 [MJ/kg_{OS}] (NOTE: OS=original substance) and is used for energy recovery in primary firing system of cement kiln. It is also called main burner fuel (MBF).

According to the latest published data from Eurostat in the year 2017, the following amounts of municipal waste were treated (NOTE: unit in 1,000 t): In total, Austria treated 4,944, Croatia 1,649, Slovenia 773 and Slovakia 2,057. The annual amount of municipal waste per capita [kg/capita] was 562 in Austria, 399 in Croatia, 374 in Slovenia and 378 in Slovakia. The EU 28 average in 2015 was 468 kg/capita, and 479 kg/capita in 2016 and in 2017. (Eurostat, 2019). A dynamic visualisation of the Municipal Waste management performance by applying RIL-Ternary Diagram Method was published by Pomberger et al. (2017). The RIL-Ternary Diagram method shows the performance of waste management considering the three waste treatment categories (operations): recycling & composting, incineration and landfilling. The current situation regarding waste management in the four investigated countries is displayed in Figure 1. It is noticeable that in the countries Austria and Slovenia recycling and incineration (note: in Slovenia since 2015) of waste is targeted, whilst in the other two countries Croatia and Slovakia the focus still is on landfilling of waste. The data shows that a decrease of the landfilling rate in all the mentioned four countries over the last twenty years is visible.

In Table 1 the main input waste materials for SRF production are listed and described by their waste code in accordance with the European List of Waste (Environmental Protection Agency, 2002). The three different production plants described and reported in this paper (Type #1, Type #2, Type #3) are characterised in the following sub-chapter.

1.2 Applied Technologies for SRF production in investigated countries

Before a plant operator is ready to opt for co-incineration of SRF, the plant specific technologies applied have to be checked and three fundamental conditions must be fulfilled (Pomberger and Sarc, 2012, Lorber et al., 2012):

- (Inter)national legal compliance, but also legal validity of the operating license,
- A guarantee of supply and of sufficient quantities with the required chemical and physical properties of SRF and
- The quality assurance concept (sampling plan, sampling, and analysing procedure and assessment).

In the following chapter, the three different types of production plants which are relevant for this study and have been investigated are presented with a technical description and an exemplary process flow scheme. For selected SRF production plants, mass balance is given and discussed in Sarc et al. (2018).

1.2.1 SRF production Plant Type #1

This type of plant uses Mixed Municipal and Bulky Waste as its main input material (further details shown in chapter 1). A brief technical description is given below, additionally, the process scheme for an exemplary production Plant Type #1 is shown in Figure 2.

Main features of this type of plant are:

- Waste receivement and storage:
  After weighing and documentation checking, trucks with accepted waste material are directed to the receiving pit of MBT plant.
- Pre-processing before biological treatment:
  Previous processing involves shredding of the received waste prior to its biological treatment. Waste from the receiving pit is transferred into a shredder, where it is shredded up to 200 mm and prepared for following biological treatment.
- Biological treatment (bio drying):
  The targets of bio drying are: 1) stabilisation and hygienisation of organic matter, 2) water removal, and 3) increasing calorific value of remaining waste. When biological drying is complete after approximately 1 week, the bio dried material is transported to the attached mechanical treatment section in the plant.

- Pre-procedures/processing before disposal/recovery: In the mechanical section, several devices/machines are used to separate and remove different fractions.

The first step is screening to separate the fine fraction (< 20 mm), which is furthermore treated under anaerobic conditions to produce methane and a fraction that

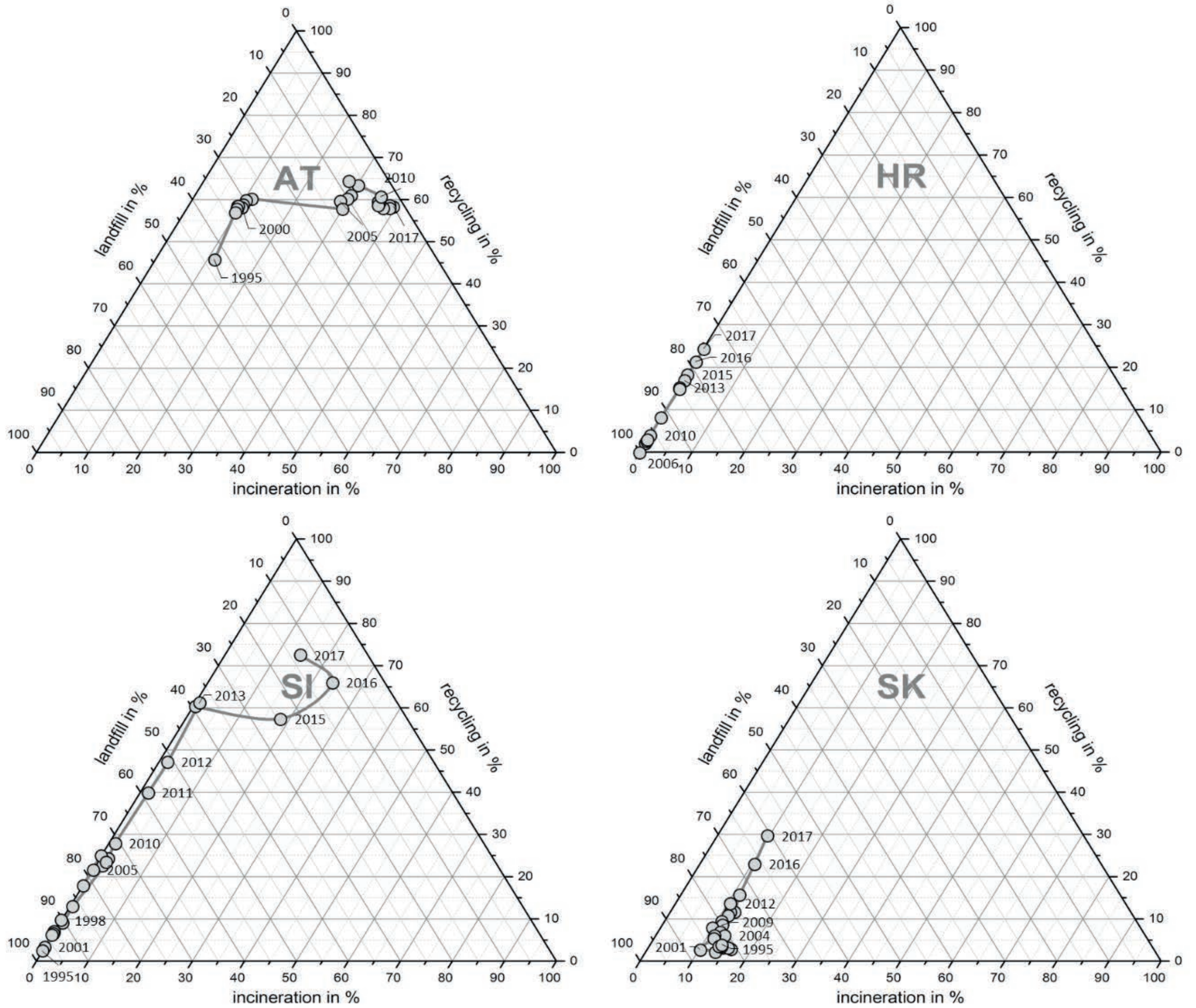

FIGURE 1: RIL-Ternary Diagram Presentation of municipal waste management performance development, data for period 1995-2017 in the investigated countries (updated/modified from Pomberger et al., 2017). AT = Austria, HR = Croatia, SI = Slovenia, SK = Slovakia.

TABLE 1: Main input waste materials for SRF production in each Plant Type described by their waste code in accordance with the European List of Waste.

Input		
SRF production Plant Type #1 20_03_01: Mixed Municipal Waste 20_03_07: Bulky Waste	**SRF production Plant Type #2** 20_03_01: Commercial Waste 20_03_07: Bulky Waste 20_01_39: Plastics 15_01_02: Plastic Packaging 15_01_06: Mixed Packaging	**SRF production Plant Type #3** 20_03_01: Commercial Waste 20_03_07: Bulky Waste
Various other municipal and industrial waste types used for SRF production: 04_02_09: Wastes from Composite Materials (Impregnated Textile, Elastomer, Plastomer) 07_02_13: Waste Plastic 15_01_01: Paper and Cardboard Packaging 17_09_04: Mixed Construction and Demolition Wastes other than those mentioned in 17_09_01, 17_09_02 and 17_09_03 19_12_12: Other Wastes (including mixtures of materials) from mechanical treatment of wastes other than those mentioned in 19_12_11 20_01_01: Paper and Cardboard		
The **output**/produced SRF legally is not considered as a product but still as a waste and is classified by two waste codes (cf. Sarc et al., 2014): 19_12_10: (quality assured) combustible waste (i.e. SRF) 19_12_12: other wastes from mechanical treatment of wastes (i.e. RDF)		

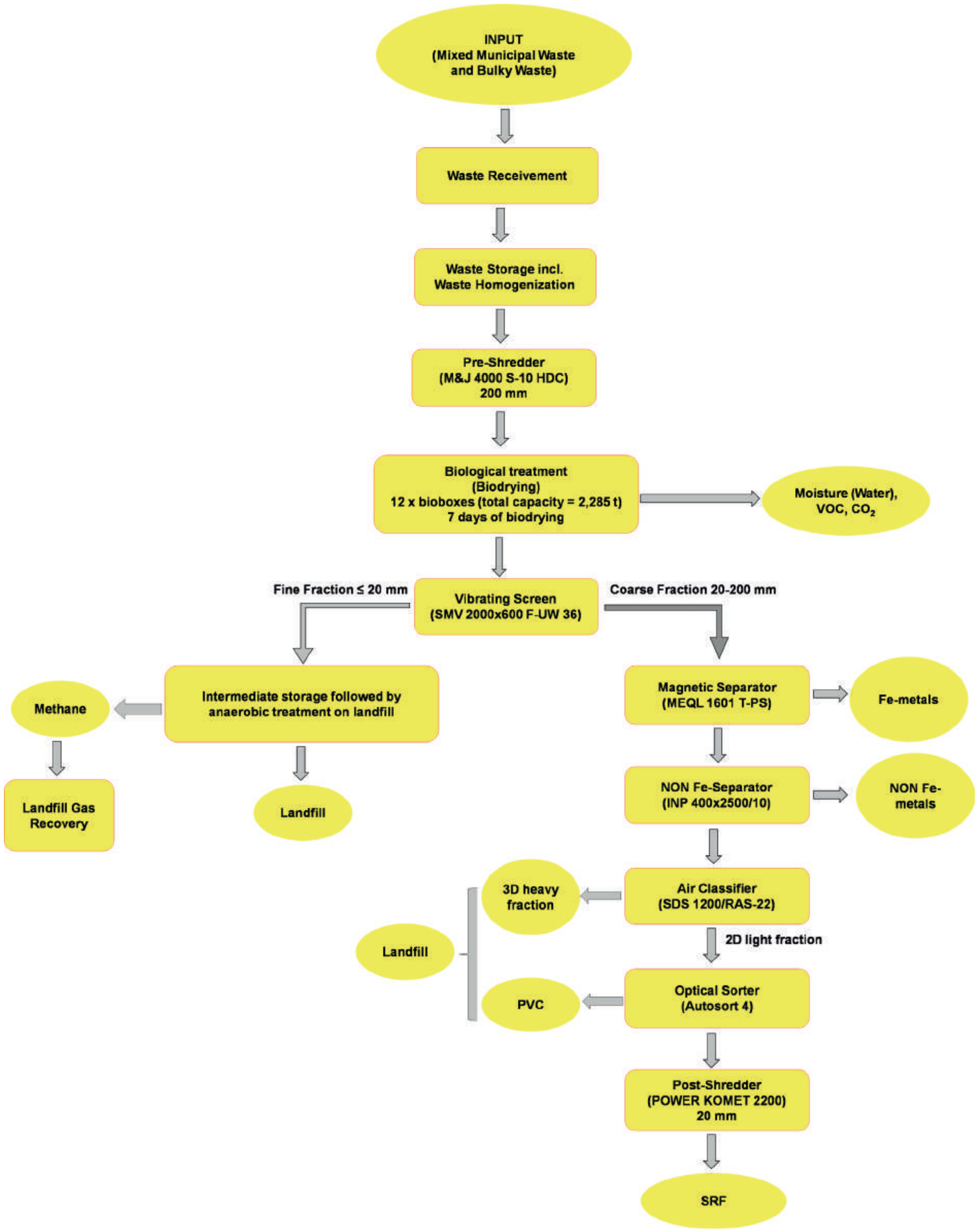

FIGURE 2: Multistage processing scheme as an example of a SRF production Plant Type #1.

is landfilled. The coarse fraction undergoes Fe- and Non-Fe-separation to remove metals and an air classifier to separate the light (2D) and heavy (3D) fraction. An optical sorting system is installed to sort out PVC plastics and finally followed by a post-shredder (20 mm), to guarantee the required quality of the produced SRF-.

1.2.2 SRF production Plant Type #2

In the production Plant Type #2, Commercial Waste, Bulky Waste and Lightweight Fractions from Plastic Sorting Plants are used to produce SRF (more details see chapter 1). Hereafter, a short technical description is given, and a scheme of an example for production Plant Type #2 is shown in Figure 3.

Main features of this type of plant are:

- Waste receivement and storage:
  After weighing, incoming waste inspection and documents checking, the delivered waste is transported to the mechanical-biological treatment (MBT) plant. There, the waste is unloaded into the waste receiving pits as it is.
- Mechanical pre-treatment:
  Pre-treatment of waste includes mechanical treatment of waste, i.e. a whole series of sieving, separation and other processing steps. In this technological process, the waste is separated into two fractions, i.e. a fine (< 200 mm) and a coarse fraction (> 200 mm). The fine fraction undergoes biological treatment, and the coarse fraction is transported to further mechanical treatment. Prior to biological treatment, magnetic materials, such as iron alloys, are separated by using a magnetic separator.
- Biological treatment (bio drying):
  After mechanical pre-treatment, the smaller particle size fractions are transferred to a section of the biological treatment plant where they are being processed (i.e. bio dried) for about 1 – 3 weeks, depending on the given input or required output parameters. By aerobic bio drying of Organic Waste Fractions, the waste becomes easier to handle (i.e. lower water content and better processing properties) for the following mechanical treatment.
- Mechanical treatment:
  In the mechanical processing area, once again potentially remaining metal portions are removed from the waste in multiple steps and the material stream is shredded to a particle size < 130 mm. The waste is transported to the Non-Fe-separating unit (i.e. Eddy-current separator) and afterwards transferred to a vibrating screen that separates the fine fraction (particle size < 25 mm) from the coarse waste. This fine fraction is either sent to a further biological treatment (i.e. stabilisation) or into a waste to energy plant applying fluidized bed combustion technology. For the final steps, an air classifier, an optical sorting system, post-shredder (30 mm) and magnetic separator are installed to remove the heavy fraction, PVC plastics and metals to ensure the guaranteed quality of SRF.

1.2.3 SRF production Plant Type #3

In type #3 of the presented SRF production plants, mainly Commercial and Bulky Waste are processed to SRF. A major difference compared to Plant Type #1 and #2 is, that this reported plant uses mechanical treatment to process SRF only, whilst the two previous types include a biological treatment (drying) step as well. A technical description of the plant is given below and as an example for this Plant Type a scheme is shown in Figure 4.

The following features of SRF production Plant Type #3 can be summarized:

- Waste receivement, input storage and pre-sorting:
  The storage of input materials takes place in receiving pits, where unwanted materials (unusual bulky parts, which might damage the plant) and recyclable materials (e.g. cardboard, metals, foils, wood etc.) are separated manually and with the help of a mobile machine (i.e. excavator). Additionally, all hazardous waste materials are sorted out.
- Mechanical pre-treatment:
  The pre-sorted material gets fed in the pre-shredding unit, which processes the waste to grain sizes < 100-200 mm.
- Mechanical treatment:
  The shredded material is then conveyed to a ballistic separator, where the material stream gets separated into three fractions. The fine fraction, which is removed by a sieve (50 mm), undergoes Fe- and Non-Fe-separation before SRF LOW Quality is produced for further waste to energy treatment. The light, 2D-fraction undergoes Fe-separation as well and is further shredded to 30 mm to guarantee fine grain sizes technically required for SRF PREMIUM Quality. The heavy 3D-fraction is used as SRF MEDIUM Quality after metals (Fe and Non-Fe) are removed.

2. MATERIALS AND METHODS

The present chapter describes the recently performed research and development approach as well as the investigation steps carried out regarding the production of premium quality SRF in four European countries (i.e. Austria, Croatia, Slovenia and Slovakia).

A comprehensive investigation was carried out on the characterization of SRF PREMIUM Quality during 4 months (March 2018-July 2018). In total, eight different SRF samples were characterized which all came from different producers (i.e. P1, P2, P3, P4, P5, P6, P6 and P8 respectively). Three of the producers were each from Austria (P1, P5, P8) and Croatia (P3, P4, P6) and one producer each from Slovenia (P2) and Slovakia (P7). All the different SRF were extensively investigated by sieving, sorting as well as physical-chemical analyses. Figure 5 shows the different Plant Types for SRF production regarding the different input waste streams. For the benefit of anonymity, the different SRF producers are named as P1-P8 and assigned to one of in total three SRF production Plant Types as seen in the Figure.

For the elaboration of a sampling concept for representative sampling of SRF from the storage depot, particle size (d_{95} in [mm]), bulk density [kg/m^3], and other parameters according to ÖNORM 15442 (ASI, 2011a) have been considered. Sampling was then carried out in every SRF production plant which was assigned to one of the three production Plant Types mentioned before (see Figure 5). During sampling procedure (cf. Figure 6), a representative field sample amount between c. 5-22 kg was taken. The samples were dried twice at 40°C (required because of Hg content) and at 105°C. Then, based on ÖNORM 15415-1 (ASI, 2011b), siev-

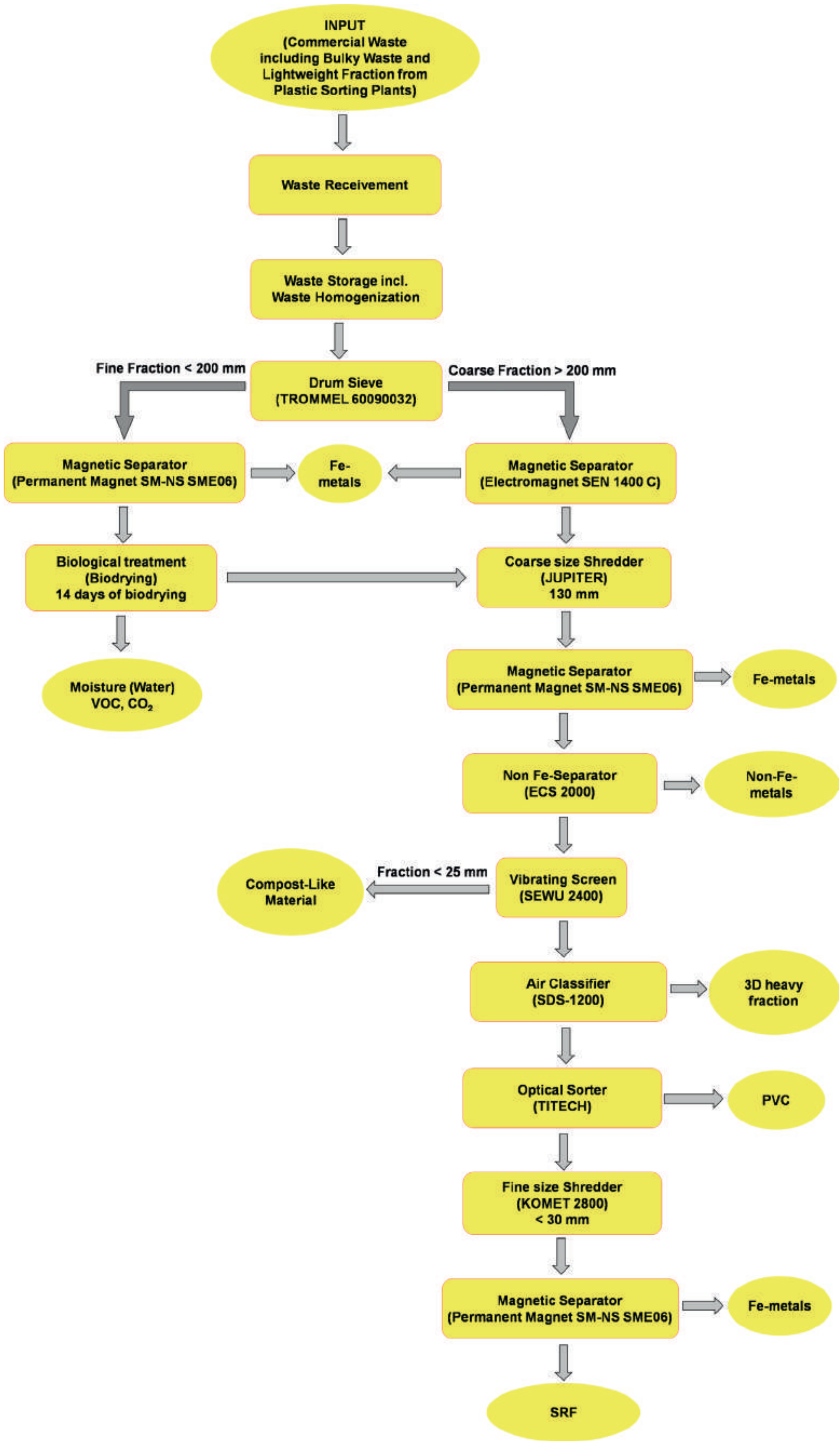

FIGURE 3: Multistage processing scheme as an example of a SRF production Plant Type #2.

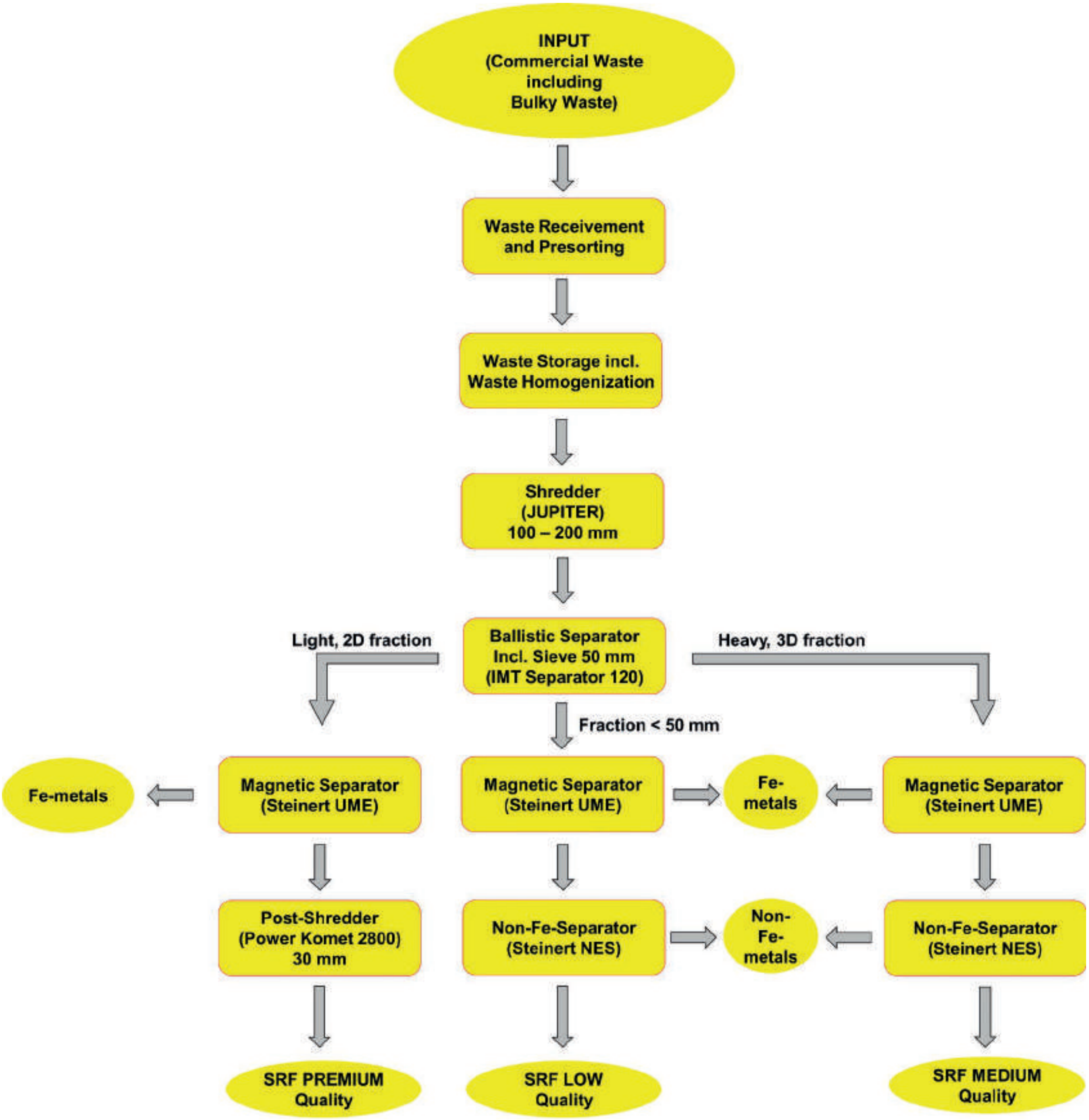

FIGURE 4: Multistage processing scheme as an example of a SRF production Plant Type #3.

ing analyses were performed on seven SRF. Additionally, sorting analyses of seven investigated samples were carried out and compared to the data published in the Federal Waste Management Plan (FWMP) (BMLFUW, 2011; BMNT, 2017) which represents the mixed municipal waste composition in Austria. Finally, extensive physical-chemical investigations were carried out for all eight SRF samples at the accredited laboratory of the Chair of Waste Processing Technology and Waste Management at the Montanuniversitaet Leoben. All investigations have been performed with dried (40/105°C) materials. Based on the data gained and other relevant data reported in the literature, statistic evaluations and comparisons have been carried out and are presented in the following chapter "Results".

3. RESULTS AND DISCUSSION

In Figure 7, the investigated SRF from all three Plant Types are depicted. As already noted in the previous section, for SRF P6, physical-chemical characterisation was performed only. For all other SRF types (P1-P5, P7-P8) additional to physical- chemical analyses, sieving and sorting analyses have been carried out too.

In the following sub-chapters, the results obtained from sieving, sorting and physical-chemical analyses are presented.

3.1 Results from sieving analyses

In total, seven SRF specimen from three different Plant Types have been sampled and analysed. The results achieved are presented in Figure 8.

The results from sieving analyses show that all three different production plants produce materials with comparable grain sizes (see Figure 8). An exception is P4 with a wider particle size distribution, which may be explained by the obvious suboptimal operation of fine shredding machines at the time of SRF sampling. Except for P4, all mate-

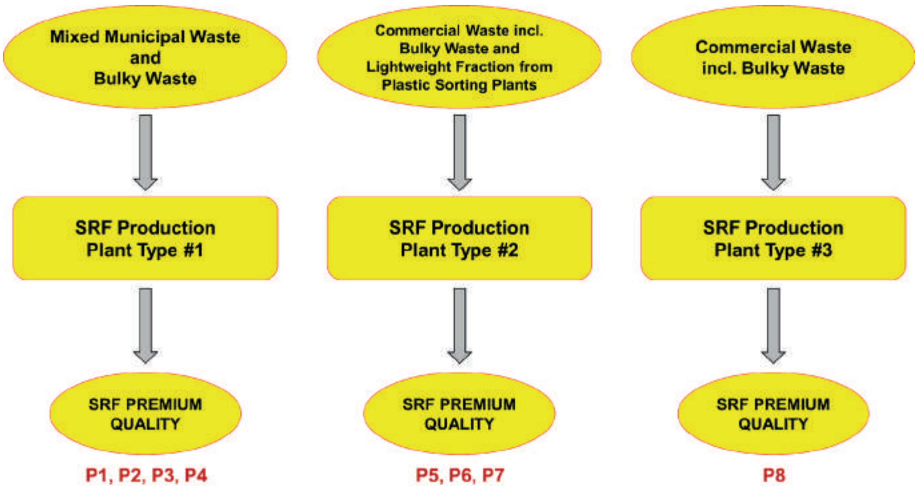

FIGURE 5: Input materials for SRF PREMIUM Quality production in three investigated different production Plant Types (#1, #2, #3) including the allocation of SRF producers (P1-P8).

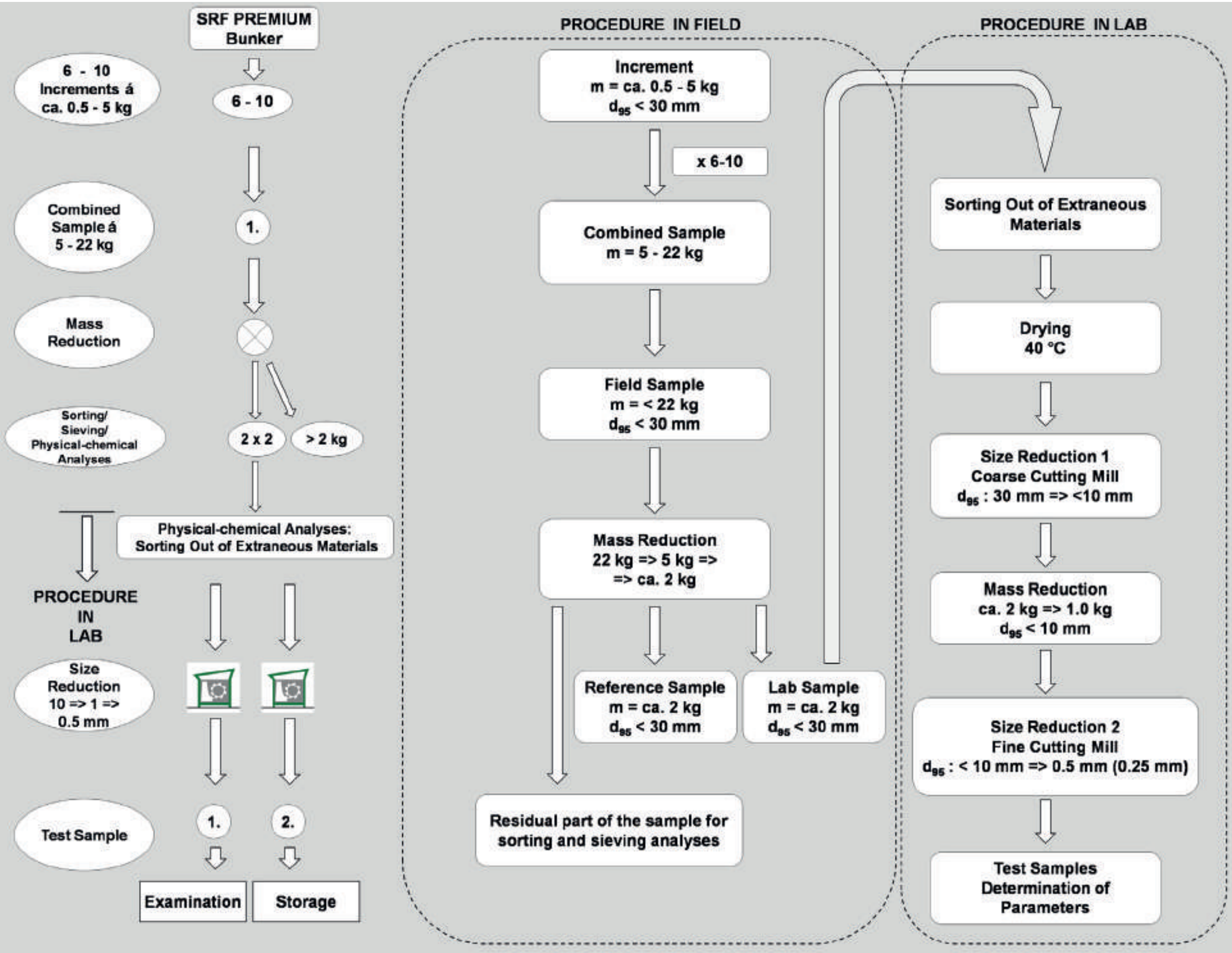

FIGURE 6: Quality assurance concept for external monitoring of SRF characteristics (reproduced from Lorber et al. (2012)).

rials fulfil the technical requirements for particle size (i.e. $d_{95} \leq 30$ (35) mm) which is required for the use in primary burners of the cement industry. In the case of P4, d_{80} can reach the required particle size of 30 mm.

3.2 Results from manual sorting analyses

In total, seven SRF supplier materials, where each material came from another SRF-production plant, have been analysed. Sorting analyses have been performed based on

FIGURE 7: Photo documentation of the investigated SRF from Plant Type #1 (left), Type #2 (middle) and Type #3 (right).

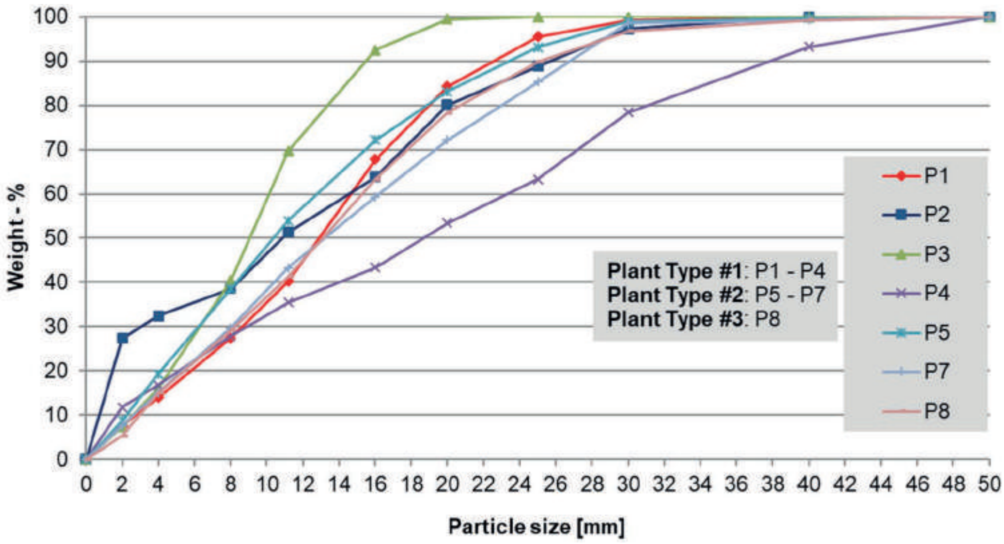

FIGURE 8: Results from sieving analyses of investigated SRF PREMIUM quality.

12 different waste fractions also used in the FWMP 2011 and 2017 (BMLFUW, 2011; BMNT, 2017). The results obtained are given in Table 2.

As shown in Table 2, comparison is done for mean value P1-P4, mean value P5 and P7, value for P8 as well as mean value P1-P8 with data on composition of mixed municipal waste, published for Austria in FMWP (NOTE: data for 2011 and 2017 is shown, as for 2017 there is no data for "fine fraction < 11.2 mm" available). Analyses results show that proportions of the fractions "Paper, Cardboard and Cardboard Packaging", "Plastics and Lightweight Fraction", "Composite Materials" and "Textiles" are in the same range in all SRF from three Plant Types (i.e. #1, #2, #3) and can be well compared to the data given in FWMP. These fractions are representing about 40% of the total amount and have a high impact on the heating value (LHV) of the SRF (cf. Table 3).

The second important information regarding SRF PREMIUM Quality is on the observed proportion of "metals, glass and inert materials". Its proportion is in the range of 0.5 – 1.7 w%$_{OS}$ only, which is much lower than in the mixed municipal waste and a good indicator that the applied technology (especially metal and heavy fraction separation; cf. chapter "Applied Technologies for SRF production in investigated countries") is doing a satisfactory job.

Finally, the proportion of "fine fraction < 11.2 mm" in SRF PREMIUM Quality shows that in all three Plant Types this amount is in the range: 42.3 – 71.7%. This result confirms that on one side the applied fine, post-shredding step is efficient and on the other side the reached particle size distribution (including d_{50} = ca. 11 or 12 mm) supports the increase of wanted SRF specific properties, especially calorific value and reaction-technical properties discussed by Beckmann et al. (2012).

3.3 Results from physical-chemical analyses

Three groups of results from physical-chemical investigations are presented in Table 3. The energy- and CO_2-emission relevant parameters mainly depend on waste input quality and then on technology applied for the processing of SRF. The procedural and mass balance parameters are relevant to the cement kiln and the combustion process (Sarc et al., 2014; 2019). Finally, mass and energy specific heavy metal content (i.e. heavy metal content [mg/MJ$_{DM}$] as a product of pollution [mg/kg$_{DM}$] and lower heating value [MJ/kg$_{DM}$]) are required for fulfilling the Austrian

TABLE 2: Results from manual sorting analyses of SRF PREMIUM Quality compared with results for mixed municipal waste published in Federal Waste Management Plan 2011 and 2017 (BMLFUW, 2011; 2017).

Fractions [$w\%_{OS}$]	Plant Type #1					Plant Type #2			Plant Type #3		FWMP 2011	FWMP 2017
	P1	P2	P3	P4	MEAN P1-P4	P5	P7	MEAN P5-P7	P8	MEAN P1-P8		
Organic/Biogenic Waste	0.0	0.0	0.0	0.0	**0.0**	0.0	0.0	**0.0**	0.0	**0.0**	20.5	17.8
Paper, Cardboard and Cardb. Packaging	9.6	5.8	6.3	25.6	**11.8**	8.9	8.8	**8.9**	8.6	**10.5**	12.4	14.0
Sanitary Articles	0.2	0.6	0.1	0.2	**0.3**	0.0	0.0	**0.0**	2.4	**0.5**	8.2	9.6
Plastics and Light-weight Fraction	24.6	17.0	18.1	9.3	**17.3**	18.0	23.7	**20.8**	28.1	**19.8**	9.7	17.6
Composite Materials	1.7	1.2	0.1	0.4	**0.8**	1.0	2.4	**1.7**	1.1	**1.1**	9.5	2.0
Textiles	15.8	6.0	3.3	15.2	**10.1**	3.5	16.5	**10.0**	4.7	**9.3**	5.8	7.8
Glass	0.0	0.0	0.0	0.3	**0.1**	0.0	0.2	**0.1**	0.2	**0.1**	4.3	4.9
Inert Materials	0.0	0.0	0.1	0.0	**0.0**	1.5	0.9	**1.2**	0.8	**0.5**	3.4	5.9
Metals	0.8	0.0	0.1	0.8	**0.4**	0.4	0.3	**0.4**	0.8	**0.5**	2.9	4.7
Hazardous Household Waste	0.0	0.0	0.0	0.0	**0.0**	0.0	0.0	**0.0**	0.0	**0.0**	1.2	1.5
Fine Fraction < 11.2 mm	42.3	52.4	71.7	45.6	**53.0**	58.1	46.5	**52.3**	43.4	**51.4**	19.6	
Other (unidentified; incl. Wood for FWMP 2017)	5.0	16.9	0.3	2.7	**6.2**	8.5	0.7	**4.6**	9.9	**6.3**	2.5	14.3
Sum	**100**	**100**	**100**	**100**	**100**	**100**	**100**	**100**	**100**	**100**	**100**	**100**

legal input-material (i.e. prior co-incineration) relevant requirements according to the national Waste Incineration Ordinance – WIO 2002 (NOTE: these legal requirements are not relevant for the Croatian, Slovenian and Slovakian SRF situation, but for the benefit of a better understanding and comparison of the SRF quality, this comparison also is shown in the present contribution, cf. Figure 9). The discussion of the results from physical-chemical analyses, which are shown in Table 3 is divided in the three groups like mentioned before.

Energy- and CO_2-emission:

The Lower Heating Value (LHV) from Plant Type #1 is significantly lower than from samples collected from Plant Type #2 and #3, which can be explained by different input materials. Whilst in Plant Type #2 and #3 Commercial Waste, Bulky Waste and Lightweight Fraction are used only, which results in a higher LHV, in Plant Type #1 Mixed Municipal Waste as input material is also used for SRF production, which leads to a lower LHV because of a reduced amount of the fractions "Plastics and Lightweight Fraction" and "Composite Materials" (see Table 2). Additionally, it is also shown that samples with higher heating values result in higher CO_2 emissions, which is also discussed in Sarc et al. (2014, 2019). It must be noted, that sample P8 in that case is an exception of this statement, which shows that the actual composition of the input material has a significant impact on SRF quality but the general correlation between Plant Type and SRF specifications is not that pronounced.

Procedural and mass balance:

The required market values for water and ash content of SRF are generally between 15 and 20%. The physical-chemical analyses show that, except for P5, all investigated samples are in that range. The comparison of the eight SRF show that even in Plant Types with the same applied technology and same input materials, the specific quality of these input materials has a significant effect on water and ash content in the SRF and a general valid statement for specific Plant Types cannot be declared for this issue. Plant Type #1 and #2 include a bio drying step during processing. Especially for samples P1-P4, where Mixed Municipal Waste is used for SRF processing, the importance of the drying process step is noticeable, because the values for dry matter in the finished SRF are in all cases higher than 80%. Samples P1-P4, which are processed from Mixed Municipal Waste show a lower chlorine content than the rest. This can be explained with the lower amount of the fractions "Plastics and Lightweight Fraction" and "Composite Materials" (see Table 2), which usually contain a higher amount of PVC plastics. This statement is based on a correlation between higher Heating Value (LHV) and higher chlorine content. Additionally to the Cl content, also the Cl:S -ratio is a crucial parameter for waste fuels, because it influences the corrosion behaviour during combustion (Lorber et al., 2011; Spiegel et al., 2012). The results obtained show significant changes in the Cl:S-ratio for the different Plant Types. Plant Type #1 shows a Cl:S-ratio of 1.8, Plant Type #2 of 2.5 and Plant Type #3 of 4.8. The chlorine content in the finished SRF is in most cases still below $1\%_{DM}$, that is required by cement industry on international market. The installation of modern NIR-sorting technology for sorting out PVC can be advisable for Type #3 production plants when the critical limit may be exceeded and Cl:S-ratio must be decreased (Kreindl, 2010; Lorber and Sarc, 2012; Pieber et al., 2012).

Legal requirements for heavy metals:

Figure 9 shows the data given in Table 2, i.e. heavy me-

TABLE 3: Results from physical-chemical analyses of investigated SRF PREMIUM Quality and limit values for SRF prior utilization in co-incineration plant (type: cement industry) according to the Austrian WIO 2002.

Parameter	Unit	Austrian Standard	Plant Type #1: P1	Plant Type #1: P2	Plant Type #1: P3	Plant Type #1: P4	**mean Plant Type #1**	Plant Type #2: P5	Plant Type #2: P6	Plant Type #2: P7	**mean Plant Type #2**	Plant Type #3: P8	**Median P1-P8**	**80th percentile P1-P8**	**Limit values acc. to the WIO 2002: Median**	**Limit values acc. to the WIO 2002: 80th percentile**
Energy and CO_2-emission relevant parameters																
HHV	[MJ kg_{DM}^{-1}]	ASI (2000)	24,9	25,1	22,4	19,9	**23,1**	20,2	30,6	23,0	**24,6**	24,8	**23,9**	**25,1**		
LHV	[MJ kg_{OS}^{-1}]		18,5	22,7	16,6	14,4	**18,1**	13,8	23,8	16,8	**18,1**	19,1	**17,7**	**22,7**		
LHV	[MJ kg_{DM}^{-1}]		22,9	23,1	20,6	18,3	**21,2**	18,6	28,1	21,2	**22,6**	22,8	**22,0**	**23,1**		
TC	[$w\%_{DM}$]	-	51,6	54,1	49,6	46,6	**50,5**	48,7	62,9	50,5	**54,0**	52,9	**51,1**	**54,1**		
X_B^{TC}	[$w\%_{DM}$]	ASI (2011c)	39,6	43,2	34,7	52,4	**42,5**	37,3	20,7	37,0	**31,7**	45,0	**38,5**	**45,0**		
X_{nB}^{TC}	[$w\%_{DM}$]		60,4	56,8	65,3	47,6	**57,5**	62,7	79,3	63,0	**68,3**	55,0	**61,6**	**65,3**		
X_B	[$w\%_{DM}$]		47,1	50,0	38,4	55,2	**47,7**	36,6	25,5	32,4	**31,5**	40,1	**39,3**	**50,0**		
X_{nB}	[$w\%_{DM}$]		40,7	37,2	35,3	22,6	**34,0**	38,0	63,1	45,5	**48,9**	43,9	**39,4**	**45,5**		
$X_{nB}W$	[$w\%_{DM}$]	calculated	31,2	30,7	32,4	22,2	**29,1**	30,5	49,9	31,8	**37,4**	29,1	**31,0**	**32,4**		
CO_2-emission	[g/MJ_{DM}^{-1}]	calculated	49,8	48,7	57,5	44,4	**50,1**	60,1	65,0	54,9	**60,0**	46,7	**52,4**	**60,1**		
Procedural and mass balance parameters																
DM	[w%]	ASI (2006)	82,6	98,6	82,5	81,3	**86,3**	77,0	85,8	81,3	**81,4**	85,2	**82,6**	**85,8**		
Ash (815°C)	[$w\%_{DM}$]	ASI (1997)	11,0	10,6	19,4	18,6	**14,9**	23,4	11,0	19,4	**17,9**	12,6	**15,6**	**19,4**		
Cl	[g kg_{DM}^{-1}]	ASI (2007)	5,5	6,1	4,8	4,9	**5,3**	11,6	7,4	8,9	**9,3**	14,4	**6,7**	**11,6**		
S	[g kg_{DM}^{-1}]	ASI (2016)	3,8	2,3	2,6	3,0	**2,9**	6,0	1,3	3,9	**3,7**	3,0	**3,0**	**3,9**		
Energy specific heavy metal content																
Sb	[mg MJ_{DM}^{-1}]	ASI (2011d)	2,5	1,1	0,9	1,1	**1,4**	1,4	2,0	1,9	**1,8**	8,7	**1,7**	**2,5**	**7**	**10**
As	[mg MJ_{DM}^{-1}]		0,1	0,1	0,1	0,1	**0,1**	0,2	0,1	0,2	**0,2**	0,1	**0,1**	**0,2**	**2**	**3**
Pb	[mg MJ_{DM}^{-1}]		3,6	7,4	4,4	4,1	**4,9**	5,3	1,7	5,3	**4,1**	16,5	**4,9**	**7,4**	**20**	**36**
Cd	[mg MJ_{DM}^{-1}]		0,040	0,012	0,026	0,012	**0,023**	0,017	0,012	0,012	**0,014**	0,364	**0,015**	**0,040**	**0.23 (0.45)***	**0.46 (0.7)***
Cr	[mg MJ_{DM}^{-1}]		3,9	2,5	2,6	1,8	**2,7**	2,5	1,0	2,4	**2,0**	3,4	**2,5**	**3,4**	**25**	**37**
Co	[mg MJ_{DM}^{-1}]		0,3	0,2	0,3	0,3	**0,2**	0,4	0,2	0,3	**0,3**	0,7	**0,3**	**0,4**	**1,5**	**2,7**
Ni	[mg MJ_{DM}^{-1}]		0,8	0,8	0,8	0,7	**0,8**	1,4	0,4	0,8	**0,9**	0,8	**0,8**	**0,8**	**10**	**18**
Hg	[mg MJ_{DM}^{-1}]		0,024	0,011	0,012	0,014	**0,015**	0,013	0,009	0,012	**0,011**	0,011	**0,012**	**0,014**	**0,075**	**0,15**

tals investigated according to the international standard and compared (i.e. expressed as exhaustion in %) to the limit values from Austrian WIO 2002 for SRF used in the cement industry.

From Figure 9, it becomes obvious that the chemical quality (i.e. heavy metals content) of currently used SRF in all the investigated four countries from the Plant Types (i.e. #1, #2, #3) presented in this paper fulfils the Austrian legal limit values for co-incineration in the cement industry and that their quality is in the same range as the SRF utilized on the Austrian market. The notable higher amount of Sb in P8 can be explained with the higher amount of plastic in the input material in Plant Type #3 (see Table 2), because Sb is frequently used as a stabiliser in plastics (Sarc et al., 2014). Also, the values for Pb and Cd are significantly higher in P8. This shows again that the specific input material is a main factor for the resulting quality of SRF and the Plant Type is – as also noted above – only of less importance compared to the input. The reported results on the physical-chemical composition of SRF are representative for the current SRF quality which is available on the waste to energy market for co-incineration plants like cement industry for all the investigated countries.

4. CONCLUSIONS

The aim of the present article was to compare the characteristics of different SRF PREMIUM Quality speci-

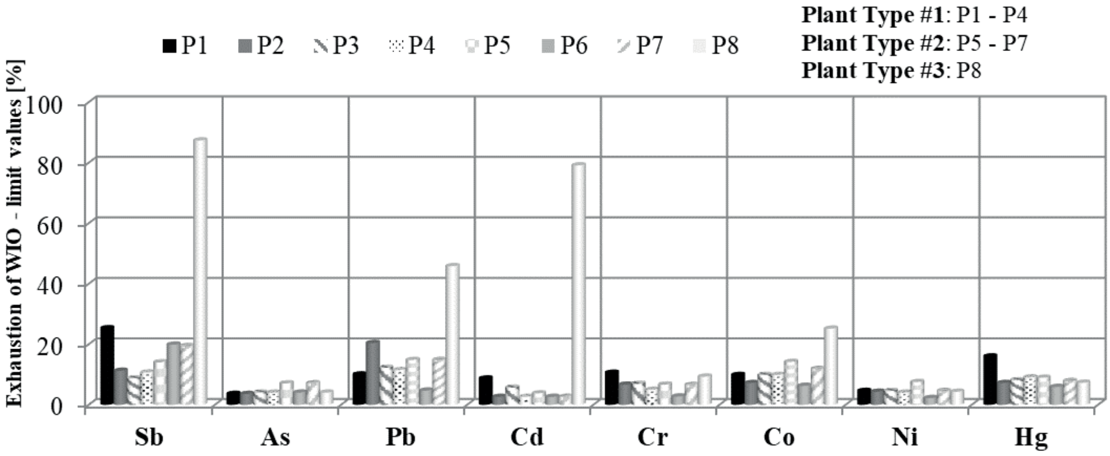

FIGURE 9: Heavy metals analysed in SRF PREMIUM Quality from eight different producers compared to Austrian WIO 2002 (BMLFUW, 2010) 80th percentile limit values that are set 100% (further interpretation of the comparison is discussed in Sarc et al. (2014)).

men which are usually utilized for energy recovery in the cement industry, including three different types of production plants in Austria, Croatia, Slovenia and Slovakia. The three investigated Plant Types (i.e. #1, #2, #3) differ in the applied processing technology (mechanical, mechanical-biological) as well as the input materials (Mixed Municipal Waste, Commercial Waste, Bulky Waste, Lightweight Fraction from Plastic Sorting Plants). To characterise the SRF, extensive sorting and sieving analyses, as well as physical-chemical analyses, were all performed in accordance with (inter)national standards (ÖNORM and EN). The results show, that all the investigated SRF meet the quality requirements for Austrian SRF before co-incineration in the cement industry and are comparable with SRF reported in Sarc et al. (2014, 2019). By considering Austrian limit values for heavy metals in SRF utilised in co-incineration plants, the exhaustion of limit values by heavy metal contents of SRF from investigated countries for all three Plant Types is within the range. Regarding the final SRF quality, the different production Plant Types show a trend in produced qualities, but also exceptions appear, which are discussed in the chapter "Results and Discussion". As stated before, it can be confirmed, that type and origin of the input waste used has more influence on the final product quality of SRF than the specific process applied for SRF production. Additionally, multi-stage processing including shredding, classifying, separation of Fe-metals, Non-Fe-metals and heavyweight inert materials, as well as sorting out of unwanted materials like polyvinyl chloride (PVC) plastics by using modern near-infrared-sorting technology is especially required for production of quality assured SRF. Finally, it can be confirmed and stated that SRF produced in all the investigated eight plants in Croatia, Slovenia and Slovakia are in fact SRF PREMIUM QUALITY that can be traded on the European SRF market and utilized in the European cement industry.

CONFLICT OF INTEREST

The authors declare no conflict of interest.

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THERMAL CHARACTERISATION STUDY OF TWO DISPOSABLE DIAPER BRANDS

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ABSTRACT

Disposable diapers have become a prominent feature of solid waste dumped in landfills. It is estimated that disposable diapers take anywhere between 300 to 500 years to decompose. Despite the associated environmental challenges, a plethora of studies show that disposable diapers have become a popular choice for parents when compared with cloth diapers. Disposable diapers are argued to be more convenient for parents because of their once-off use and super-absorbent ability, of which they are able to absorb 200-300 times the weight as compared to the cloth diapers. This study investigates thermal devolatilisation profiles of disposable diapers as well as their iso-conventional kinetic parameters, elemental and fractional composition. In this study, the two most dominant disposable diaper brands were sampled and analysed qualitatively and quantitatively. It was observed that the exterior fraction of both diapers showed a single peak devolatilisation at a temperature of around 500°C, while the interior fraction showed two distinctive devolatilisation peaks observed below 400°C and above 500°C. However, the pyrolysis heating rate produced no effect on the pyro-char fraction. Furthermore, the devolatilisation pathways of different types of disposable diaper fractions showed that there is a potential for conducting stepwise pyrolysis to promote fractional recovery of valuable products. Disposable diapers waste conversion can be better handled by separating the outer fraction (mainly fossil-based plastics and rubbery materials) from the inner fraction (mainly bio-based fibers and absorbents). It is further illustrated that the kinetic parameters, E and k are different for each disposable diaper fraction.

1. INTRODUCTION

Global economic growth and increasing population have led to a large amount of waste being generated. As a result, waste management systems have become overburdened and more complex to manage. As indicated by Chhabra et al. (2016), an approximated 1300 million tonnes of waste is generated each year globally in urban centres and this figure is expected to increase to 2200 million tonnes by 2025. The challenge with increasing waste quantities is that many developing countries, particularly in Africa, are finding it very difficult to manage their waste. Africa is estimated to generate close to 125 million tonnes of municipal solid waste every year, of which only fifty-five percent of it is collected (Oelofse and Nahman, 2019). Oelofse and Nahman (2019) further argue that there will be an estimated thirty percent increase in the waste generation rate in Africa between 2012 and 2025. The major concern is that investments in proper waste management and disposal facilities remain at the bottom of development priorities in many developing countries (Zurbrugg 2002, Marshall and Farahbakhsh, 2013, David et al.,2019 , and Moore, 2019). According to Arenas et al. (2019:01), "waste management systems in developing economies are still insufficient and in some cases, cause environmental problems because waste tends to be unseparated and disposed of in open landfill sites." Landfilling, incineration and illegal dumping are the dominant methods of dealing with waste in many African countries (Achankeng, 2003, Williams, 2005, Zoeteman et al., 2010, Kazuva and Zhang, 2019, Kyere et al., 2019, and Oelofse and Nahman et al., 2019). A plethora of scholars (McKay, 2002, Hicks, 2007, Hristovski et al., 2007, Smyth et al., 2010, Topanou et al., 2011, Badgie et

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al., 2012, and Idowu, 2019) have cited the issue of poor waste characterisation as a major stumbling-block to waste management in Africa. Topanou et al. (2011) argue that waste characterisation remains the key prerequisite for effective waste management programs. For Africa to better manage and benefit from their waste sector, waste characterisation should be highly prioritised.

Research indicates that countries worldwide should direct their waste management efforts towards creating a circular economy, and in so doing, eliminate the huge environmental burdens associated with poor waste disposal systems (UNTHA UK, 2015). Hence, the present study forms part of the paradigm shift towards creating a "circular economy" through exploring the possibility to recover valuable resources from post-consumer products. In this study, selected brands of disposable diapers are assessed in order to investigate their potential for waste-to-energy recovery. Khoo et al. (2019) indicates that globally, disposable diapers (categories under municipal solid waste) account for approximately 13% of municipal waste dumped in landfills. Disposal diapers, have by far, become the more preferred product for parents when compared to cloth diapers (Krafchik, 2016, Khoo et al., 2019, and Mendoza et al., 2019). Disposable diapers are argued to be more convenient for parents because of their once-off use and super-absorbent ability, of which they are able to absorb 200-300 times the weight as compared to the cloth diapers. A study by Sheila (2016) revealed that a child uses between 6-8 disposable diapers a day, and this accumulates to an estimated 5500 diapers before potty training (18-30 months). Despite their demand, disposable diapers are cited as a major environmental problem and burden in the waste management system, especially in developing countries (Colon et al., 2011, Remigios, 2014, Hamad et al., 2014, and Khoo et al., 2019). Disposable diapers have become a threat to the natural environment, from both a manufacturing and disposal perspective. The process of manufacturing disposable diapers further contributes to the depletion of natural resources. For instance, disposable diapers are mostly manufactured using 70% wood pulp and 30% petroleum, of which the latter is regarded as a finite resource and its combustion contributes significantly to climate change, (Khoo et al., 2019). Scholars have argued that the superabsorbent polymer (SAP) inside disposable diapers can take close to 500 years to decompose (Colon et al., 2011, Remigios, 2014, and Khoo et al., 2019).

Currently, disposable diapers are discarded mainly through landfilling and incineration amongst other methods, which include composting and recovery of materials through recycling. However, Budyk and Fullana (2019:01), highlight that, "due to the high organic and water contents of used diapers, landfill disposal and incineration are not desirable options". Similarly, Liang et al. (2008) argued that the high levels of moisture inside disposable diapers would require more time and energy to evaporate, causing the incineration process to be very slow. On the other hand, Sun et al. (2016) dismissed the application of the incineration method, indicating that it produces substances such as dioxins which have a negative impact on the environment and human health. Composting and recycling of disposable diapers has been seen as more favoured disposal options, especially in developed countries (Mihajlović, et al., 2019). Despite this, constraints such as high technological costs and complex processes (Colon, et al., 2011) are some of the drawbacks which add to the problem of managing disposable diapers. In light of the aforementioned, there is a need for researchers to look at alternative methods that are more environmentally friendly and economically beneficial in the management of disposable diapers.

In this study, kinetic pyrolysis is applied to the two most dominant disposable diaper brands in South Africa (Huggies and Pampers) in order to assess their potential of waste-to-energy production. Pyrolysis involves the thermochemical treatment of samples in the absence of oxygen at elevated temperatures, and during this process, multiple reactions occur, (Apaydin-Varol et al., 2014, Heydari et al., 2015, Perera and Narayana, 2018). In addition, pyrolysis is also known for its production of pyro-gas, pyro-oil, and pyro-char (Heydari et al., 2015). Furthermore, Aboulkas et al. (2010:1363) highlighted that, "the first step for a suitable design of any polymer reactor is knowledge of the kinetics." Hence, the present study uses pyrolysis to understand the thermal behaviour of devolatilisation patterns of disposable diapers. Sharifzadeh et al. (2019) explains that the important characteristic of pyrolysis technologies is their adaptability and flexibility to accept a diverse range of raw materials.

The present study therefore aims to use the model-free iso-conversional (Friedman and Kissinger) methods to identify a pyrolysis reaction model. Additionally, "iso-conversional methods have the potential to determine the behaviour of complex reactions because they are simpler in nature and they decrease the risks of selecting an unsuitable kinetic model and of finding the wrong kinetic parameters", (Arenas et al., 2019:02). It has been mentioned above that most of the work has focused mainly on landfilling or incinerating disposable diapers; however, little attention is given to using thermochemical treatment as an alternative method. Thermogravimetric analysis (TGA) is performed to better understand the devolatilisation patterns of disposable diapers. Therefore, this study will apply kinetics in order to pursue the possibility of using this method as a future means of safely managing disposable diapers.

2. THEORETICAL BACKGROUND

The general material devolatilisation mathematical relationship is presented as follows:

$$\frac{dm}{dt} = k(T)f(m) \quad (1)$$

where: m is a mass fraction, t is time, k(T) is reaction rate constant at temperature, T (K), and f(m) is the reaction model.

The time, t, and heating rate, β, can be related by Eq. (2).

$$\beta = \frac{dT}{dt} \quad (2)$$

where: β is the heating rate, and T is the absolute temperature.

The reaction rate constant, k, can be presented by Eq. (3).

$$k(t) = k_0 exp(\frac{-E_a}{RT}) \quad (3)$$

where: k is the reaction rate constant, k0 is a pre-reaction rate constant, Ea is the activation energy, and R is the universal gas constant.

Therefore, by substituting Eq. (2) and Eq. (3) in Eq. (1):

$$\frac{d\alpha}{dT} = \frac{1}{\beta} k_0 exp(-\frac{E_a}{RT}) f(m) \quad (4)$$

2.1 Determination of kinetic parameters

The linear mathematical analysis methods for experimental results analysis were selected to estimate kinetic parameters (Ea and k0). This approach has advantages in that it is a well-established conventional application in the analysis of thermal devolatilisation of solids. Non-isothermal pyrolysis conditions were used since the experiments were carried out at various heating rates (10, 20, 30, 40 and 50°C/min). Iso-conversional methods of Friedman (differential approach) and Kissinger (integration approach) (Friedman, 1964; Kissinger, 1957) are used along with mathematical methods for the analysis of the experimental results.

2.2 Friedman method

In order to establish a linear relationship, a natural logarithm of Eq. (4) was employed in the equation and expressed in the following equation:

$$ln\left(\beta \frac{d\alpha}{dT}\right) = lnk_0 + lnf(\alpha) - \frac{E_a}{RT} \quad (5)$$

Plotting $\ln\left(\beta \frac{d\alpha}{dT}\right)$ versus $\frac{1}{T}$ at given reaction progress, α, for various heating rates yields a straight line with slope $\frac{-E_a}{R}$. Activation energy can be obtained from this slope without knowing the reaction function f(α). The pre-exponential reaction rate constant, k_0, is the y-intercept of a straight line. Friedman method is one of the model-free iso-conversional methods (Friedman, 1964).

2.3 Kissinger method

As for the Kissinger method, temperature values at the maximum devolatilisation rate of diapers are at the various heating rates (Kissinger, 1957). In the Kissinger method, Eq. 4 is modified to Eq. (6).

$$ln\left(\frac{\beta}{T^2}\right) = ln\left(\frac{k_0 R}{f(\alpha)}\right) - \frac{E_a}{RT} \quad (6)$$

A plot of $\ln\left(\frac{\beta}{T_{max}^2}\right)$ versus $\frac{1}{T_{max}}$ at various heating rates yields a straight line that allows determination of the activation energy, Ea, from the gradient of the straight line, $\frac{-E_a}{R}$. The y-intercept can be used to estimate the pre-exponential reaction rate factor. Similar to the Friedman method, the Kissinger method is another example of a model-free iso-conversional method.

3. EQUIPMENT AND METHOD

The samples of clean disposable diapers brands were collected from the Pinetown CBD and Clermont in the KwaZulu-Natal province, South Africa. Two diaper brands, namely, Huggies and Pampers, were used for the experiments. Two or more units for each brand were used as samples for proper representation and re-testing. Additionally, the diapers were sampled by hand separation (interior and exterior fractions) and were crushed by cutting into fine particles (diameter less than 1 mm).

The proximate analysis was done using the thermogravimetric analysis, which was carried out on each diaper brand to determine the mass loss as temperature increases. The maximum temperature was set to 800°C. Nitrogen (N_2) and Air were used as a carrier gas and a combustion median to trace the devolatilisation profile and to determine ash content, respectively. A TA 60WS model thermogravimetric analyser (TGA) (Shimadzu, Kyoto, Japan) was used for all experiments. The calculations were done using the Standard Test Method for Compositional Analysis by Thermogravimetry: Designation: ASTM E 1131 – 08, (2008).

4. RESULTS AND DISCUSSION

4.1 Thermogravimetric analysis

The disposable diapers brands samples were collected from the Pinetown CBD and Clermont areas in KwaZulu-Natal, South Africa. The proximate analysis was carried out to determine the high volatile content, volatile matter, fixed carbon, and ash contents. The results of the proximate analysis are summarised in Table 1 below. Furthermore, the TGA/DTG graphs of the internal and exterior fractions of both brands are depicted in Figures 1, 4, 6 and 9.

The TGA results in Table 1 reveal that the interior fractions of both disposable diaper brands are higher in terms of the high volatile matter contents as compared to the exterior fractions. For instance, the Huggies interior fraction was at 14.74 wt. % whist the exterior had 0.22 wt.%. On the other hand, the Pampers brand had 8.09 wt.% in the interior, and only 0.67 wt.% was contained in the exterior fraction. These high volatile matter contents in the interior fraction can be attributed to the materials inside the disposable diapers. The inner core layer is made up of SAP and wood pulp which are major absorbents of water, resulting in diapers being heavier as well (Wolston, 2015). Figures 1 and 6 depict the interior fractions of both diaper brands, which is significantly different from the exterior fraction de-

TABLE 1: Proximate analysis of the interior and exterior fraction of disposable diaper brands.

Disposable Diaper Brands	Interior fraction (wt.%)				Interior fraction (wt.%)			
	High Volatile	Volatile matter	Fixed carbon	Ash	High volatile	Volatile matter	Fixed Carbon	Ash
Huggies	14.74	61.54	10.63	13.09	0.22	88.82	8.54	2.42
Pampers	8.09	67.54	11.63	12.74	0.67	94.57	3.14	1.62

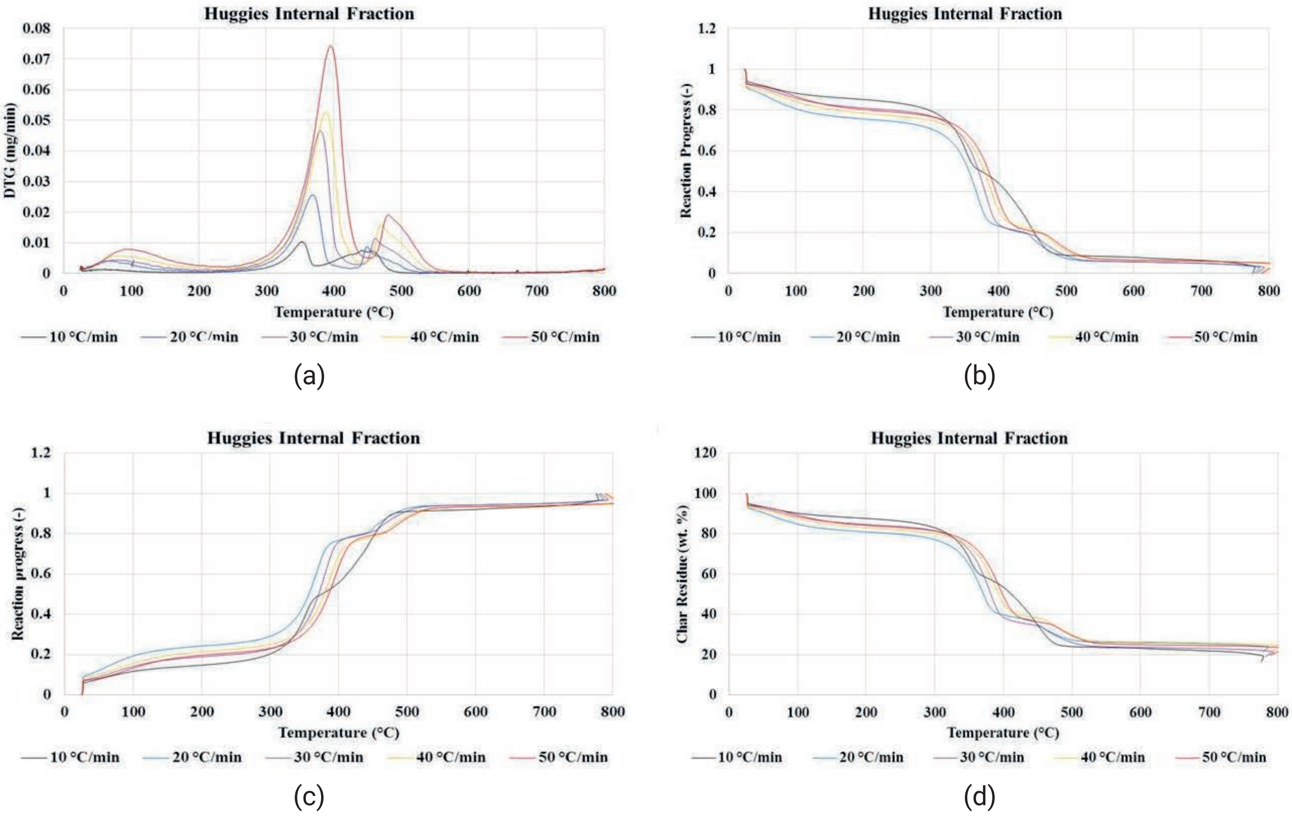

FIGURE 1: Huggies Internal Fraction derivative thermogravimetric analysis (DTG), reaction progress (α) and char residue profiles.

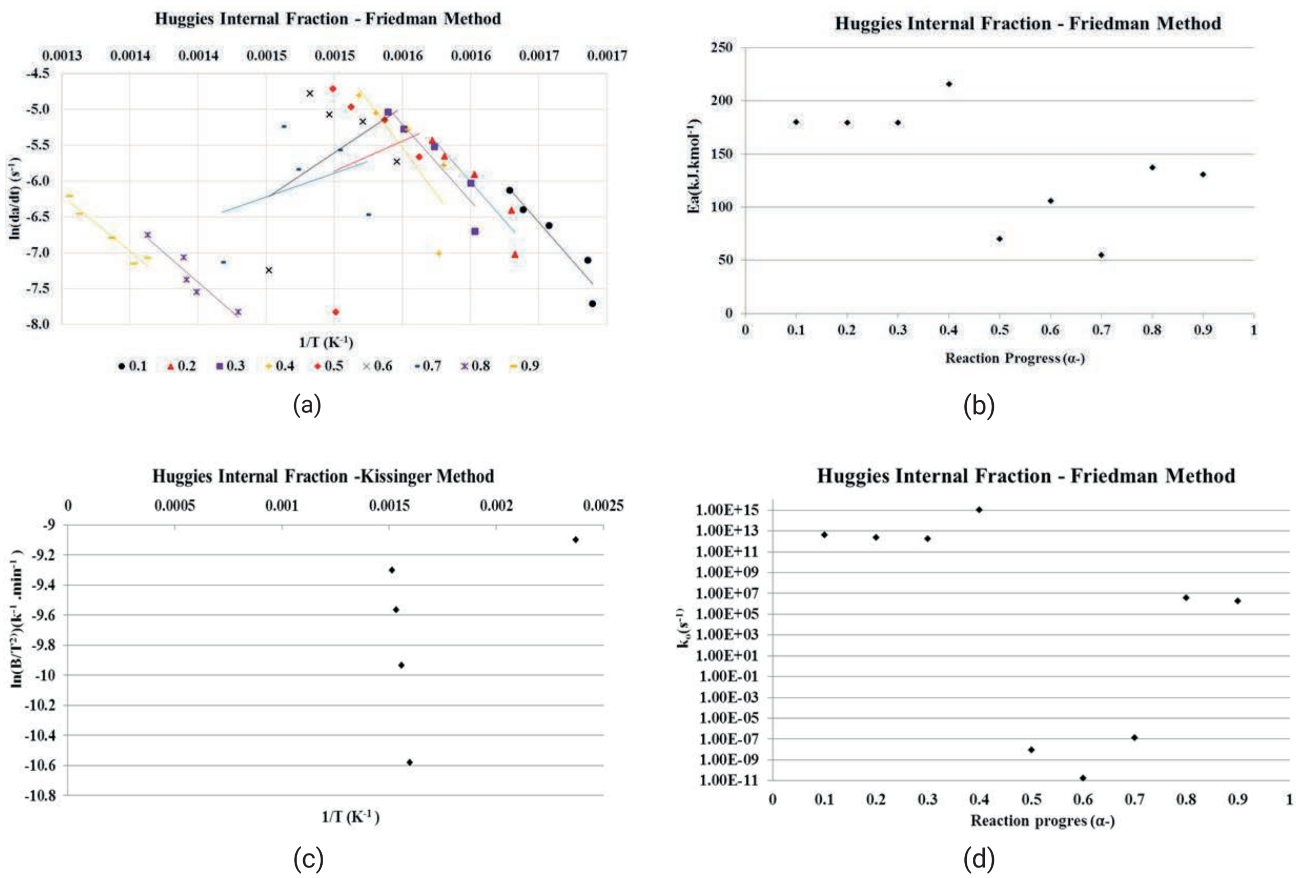

FIGURE 2: Huggies Internal Fraction kinetic parameters profiles (Friedman and Kissinger Methods).

picted in Figures 4 and 9. The latter shows each brand with two distinctive peaks, between the temperatures of 350°C and 480°C.

The exterior fractions of both diaper brands are highlighted in Figures 4 and 9, which only show single peaks, especially around the temperatures of 450°C and 500°C. This could be attributed to the fact that the exterior fraction is lighter in weight and contains mostly plastics. In addition, this is also the main devolatilisation stage, where the volatile matter is released. Thus, it is also supported by results in Table 1, which shows the exterior fractions of both brands with a higher volatile matter. The exterior fraction in Table 1 highlights that a volatile matter for Huggies is 88.82 wt.% and for Pampers it is 94.57 wt.%, which are both higher than the interior fraction. The final devolatilisation zone is observed where the fixed carbon and ash are released. According to Heydari et al. (2015), the lower devolatilisation rates observed can attribute to further gasification and char released at elevated temperatures.

4.2 Elemental analysis

The results in Table 2 below highlight the ultimate analysis, which was carried out to determine the carbon, nitrogen, hydrogen, and sulphur contents. However, only two components were obtained from the Huggies and Pampers diaper brands, namely carbon and hydrogen. From Table 2, it is observed that the Pampers diaper brand had a higher concentration of carbon and hydrogen for both the interior and exterior fraction. Table 2 notes that the exterior fraction had the highest carbon and hydrogen contents. For example, the Huggies brand had 75.89 wt.% of carbon whilst the Pampers brand had 78.29 wt.%. A similar pattern is observed with hydrogen where the Huggies diaper brand had 11.41 wt.% and Pampers had 11.83 wt.%. As a result, these observations can be supported by the results found in Figures 4 and 9 which show the major difference in the devolatilisation patterns of the exterior fractions of both diaper brands.

4.3 Kinetics analysis

The results obtained from the thermogravimetric analysis are important because they were used to obtain the activation energies for disposable diapers pyrolysis.

Hence, kinetics were used to determine the conditions at which pyrolysis of disposable diapers could take place. Kinetics aided with the investigation to establish the reactions of both diaper brands (Huggies and Pampers). Since disposable diapers contain fluff pulp polymer, super-absorbent gel crystals and plastics (polyethylene), these could react differently to experimental design. Hence, Apaydin-Varol (2014) highlights that this data, can be used to predict the kinetics of the devolatilisation patterns and hence, may give an idea about the thermal behaviour of polymeric substances during pyrolysis that may be a useful data for further reactor design.

TABLE 2: Proximate analysis of the interior and exterior fraction of disposable diaper brands.

	Interior fraction		Exterior fraction	
	Carbon (wt.%)	Hydrogen (wt.%)	Carbon (wt.%)	Hydrogen (wt.%)
Huggies	37.35	5.45	75.89	11.41
Pampers	54.12	7.93	78.29	11.83

In the current study, kinetic parameters were determined by two different methods that are explained in the above subsections (subsections 2.3.2 and 2.3.3). Both the model-free iso-conversional method, differential approach (Friedman) and integration approach (Kissinger) were used. Thus, in order to get the most appropriate thermal process (pyrolysis), different heating rates of 10, 20, 30, 40, and 50°C/min were used in the experiments.

4.4 Friedman method

The Friedman method was used to obtain the activation energy (E_a) of the thermal heating using peak values on the curves as indicated in Figures, 2, 5, 7, and 10, where the internal and external fractions of both disposable diapers brands are depicted. It is observed that the Huggies diaper brand had higher activation energies, especially in the interior fraction. For instance, the results in Figure 2 highlight that the activation energy for Huggies internal fraction ranges from 215.755 kJ/mol to 55.329 kJ/mol. However, Figure 7 depicts that the Pampers diaper has a lower range of activation from 196.556 kJ/mol to 5.068 kJ/mol. As a result, the Huggies brand in Figure 2 exhibits a non-linear pattern with conversion 0.1 to 0.4 higher than those of 0.5 to 0.9. However, the results in Figure 7 of the Pampers internal fraction show a V-shaped pattern, where conversion 0.5 is the lowest and conversion 0.8 with 0.9 is the highest. This indicates that where there is low activation energy it may be in the sensitive areas in the Friedman method.

In addition, the interior fraction has Peak 1 and Peak 2 of both diaper brands. The Friedman method shows a linear pattern, especially in Peak 1 of the Huggies internal fraction. This pattern is observed in Figure 3. In addition, Peak 2 of the Huggies internal fraction exhibits a rather non-linear pattern. On the other hand, the Pampers diaper brand Peak 1 and 2 of the internal fractions are shown in Figure 8.

The exterior fraction of the Huggies brand had higher activation energy ranging from 161.716 kJ/mol to 138.559 kJ/mol, whilst the Pampers brand had ranges of 147.357 kJ/mol to 93.360 kJ/mol. Furthermore, Figure 5 of the Huggies diaper brand shows a linear pattern of the activation energy as compared to the Pampers brand in Figure 10 which has a less linear pattern.

4.5 Kissinger method

The Kissinger method of the interior fraction of both diaper brands is not in straight lines as compared to the exterior graphs. These results are observed in Figures 2 and 7. The results are further depicted in Table 3, which shows the char residue, where the exterior fractions have smaller contents as compared to the interior fractions. In addition, Table 3 also correlates with the results observed in Figures 2, 5, 7, and 10. The results shown in Figures 3 and 8 of the interior fraction of both diapers highlight Peak

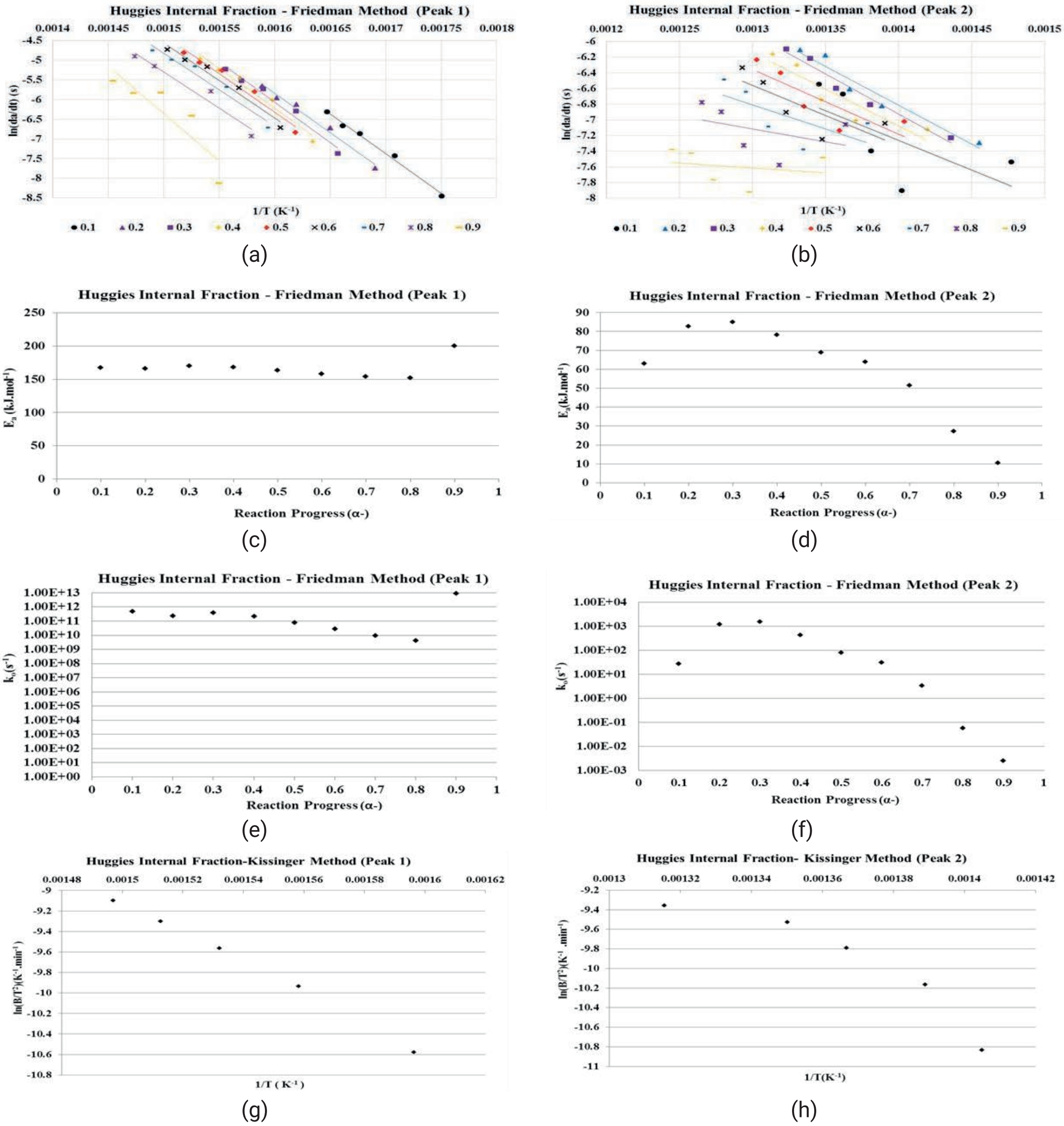

FIGURE 3: Huggies Internal Fraction two evolved peaks profiles (Friedman and Kissinger Methods).

1 with a straight line pattern as compared to Peak 2. In addition, this shows that due to the super-absorbent gel crystals and fluff pulp, the interior fraction of the diapers still need to be split further and analysed separately.

The kinetic parameters estimated values for both the Huggies and Pampers brands are illustrated in Table 4. As seen in Table 4, the Kissinger method exhibits higher activation energy, particularly in the external fraction of the Huggies diaper brand. For instance, the ranges are 164.99964 kJ/mol to 166.35620 kJ/mol. However, the external fraction of the Pampers brand shows higher activation energy in terms of the Friedman method. For example, the results in Table 4 depict the range as 159.71986 kJ/mol to 157.54123 kJ/mol whereas, in terms of the internal fraction, for both diaper brands for the Friedman method, the activation energy is lower. The Huggies diaper brand was found at 141.35382 kJ/mol whilst the Pamper brand was at 109.31620 kJ/mol.

The Kissinger method shows a dissimilar pattern from Friedman method in terms of activation energy. Table 4 indicates that the internal fraction of the Huggies diaper brand has relatively low activation energy of 6.58618 kJ/mol whilst the Pampers brand has 122.04952 kJ/mol. These results are also supported by the observation in Figure

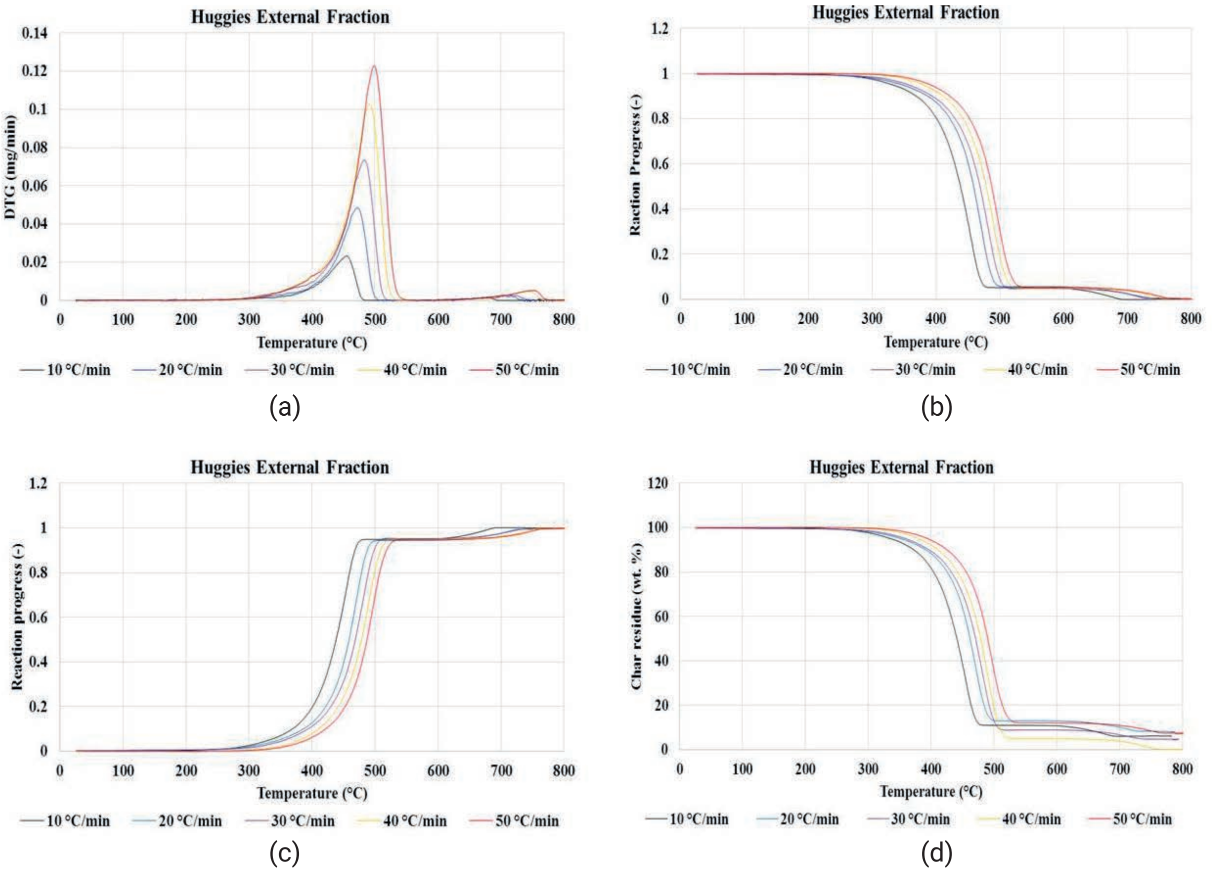

FIGURE 4: Huggies External Fraction derivative thermogravimetric analysis (DTG), reaction progress (α) and char residue profiles.

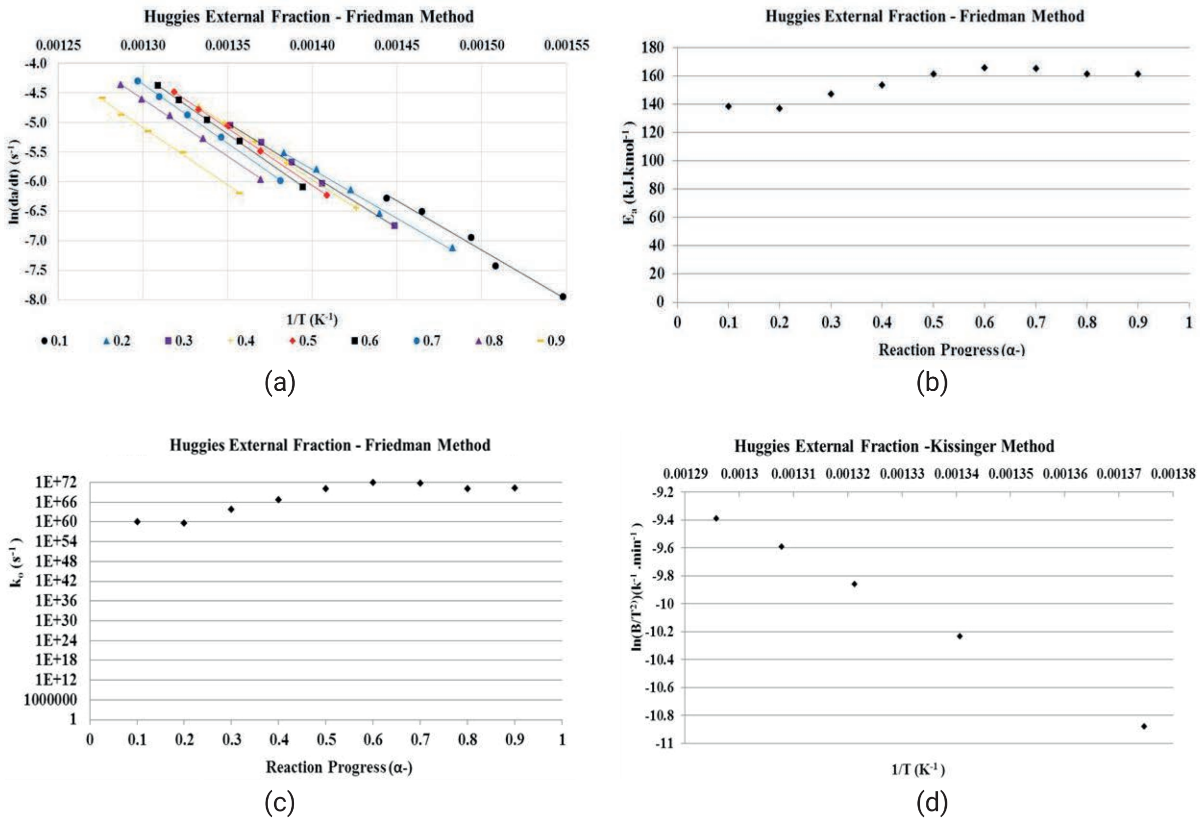

FIGURE 5: Huggies External Fraction kinetic parameters profiles (Friedman and Kissinger Methods).

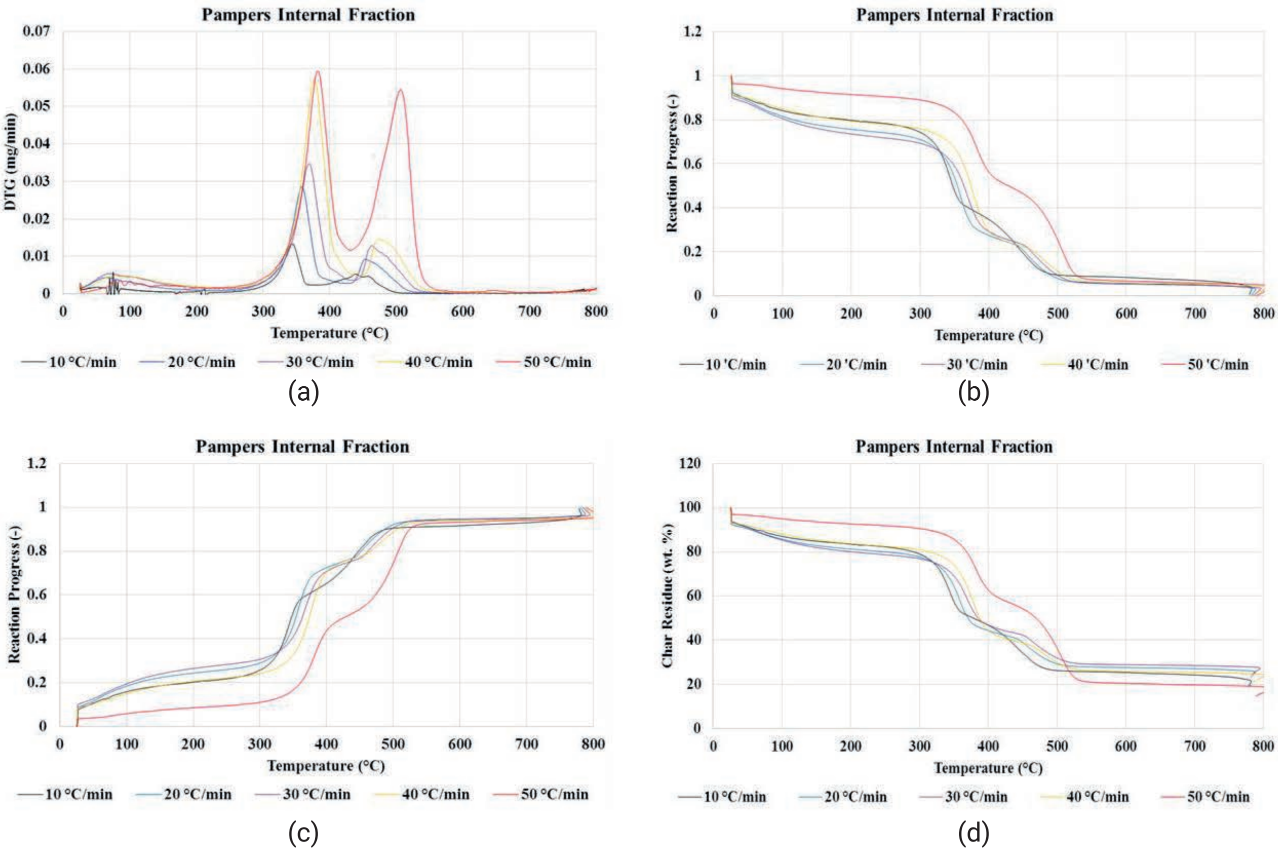

FIGURE 6: Pampers Internal Fraction derivative thermogravimetric analysis (DTG), reaction progress (α) and char residue profiles.

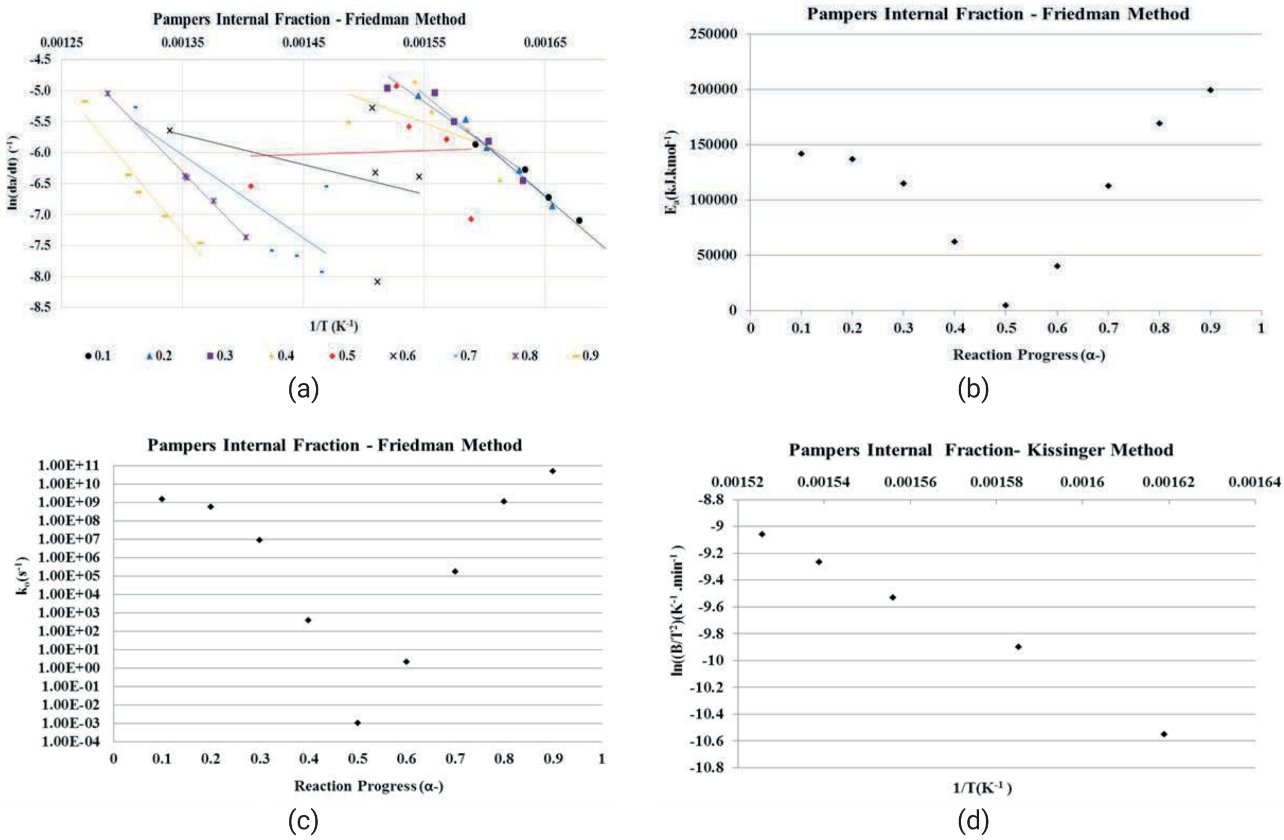

FIGURE 7: Pampers Internal Fraction kinetic parameters profiles (Friedman and Kissinger Methods).

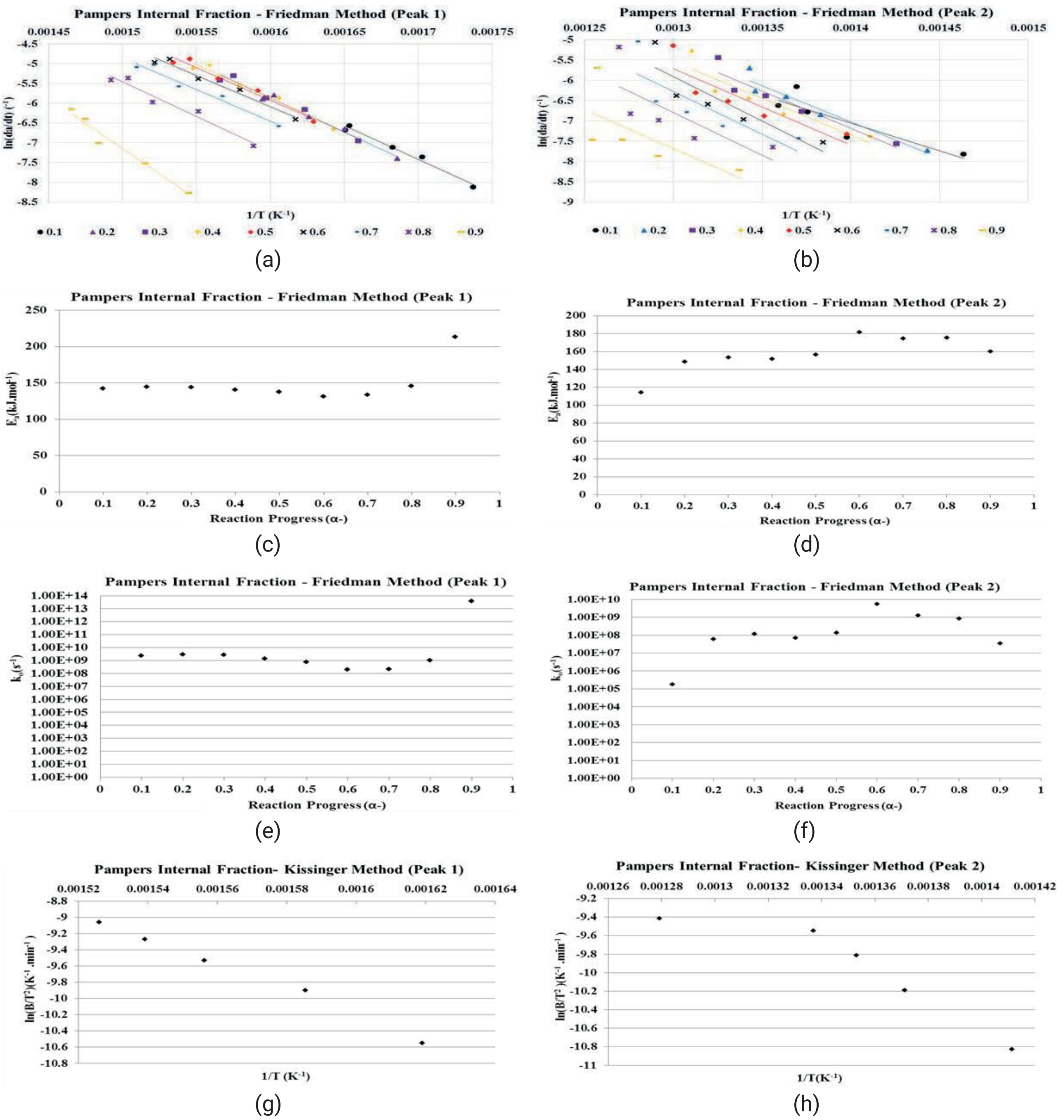

FIGURE 8: Pampers Internal Fraction two evolved peaks profiles (Friedman and Kissinger Methods).

2, where the Kissinger method of the Huggies brand depicts an unformed pattern.

Furthermore, the results in Table 4 also show the k_0 estimation for both disposable diaper brands. Using the differential method (Friedman), the internal fraction of the Huggies diaper brand has a lower estimation as compared to the Pampers brand. The results are 1.29E+14 k_0 for the Huggies diaper brand whilst the Pampers brand had 5.81E+09 k_0. A similar pattern is observed for Peak 1 in Table 4. The Huggies brand shows the estimation of k_0 ranging from 1.20E+12 k_0 to 3.82E+02 k_0. On the other hand, the results in Table 4 illustrate that the Pampers diaper brand shows Peak 1 with higher estimations of 4.55E+12 k_0 and 9.08E+08 k_0.

The integration method (Kissinger) shows the internal fraction of the Pampers brand with higher estimations of 1.85E+40 k_0 and 1.59E+05 k_0 for the Huggies brand. Finally, the results in Table 4 also depict that, the Huggies brand, in the interior and exterior fractions, has a minimal gap. For example, the Huggies brand had estimations of 1.59E+05 k_0 and 1.56E+70 k_0. However, the Pampers diaper brand had estimations of 1.85E+40 k_0 and 1.10E+07 k_0. The integration (Kissinger) method is dependent on heating rates while the other is based on differentiation (Friedman). The-

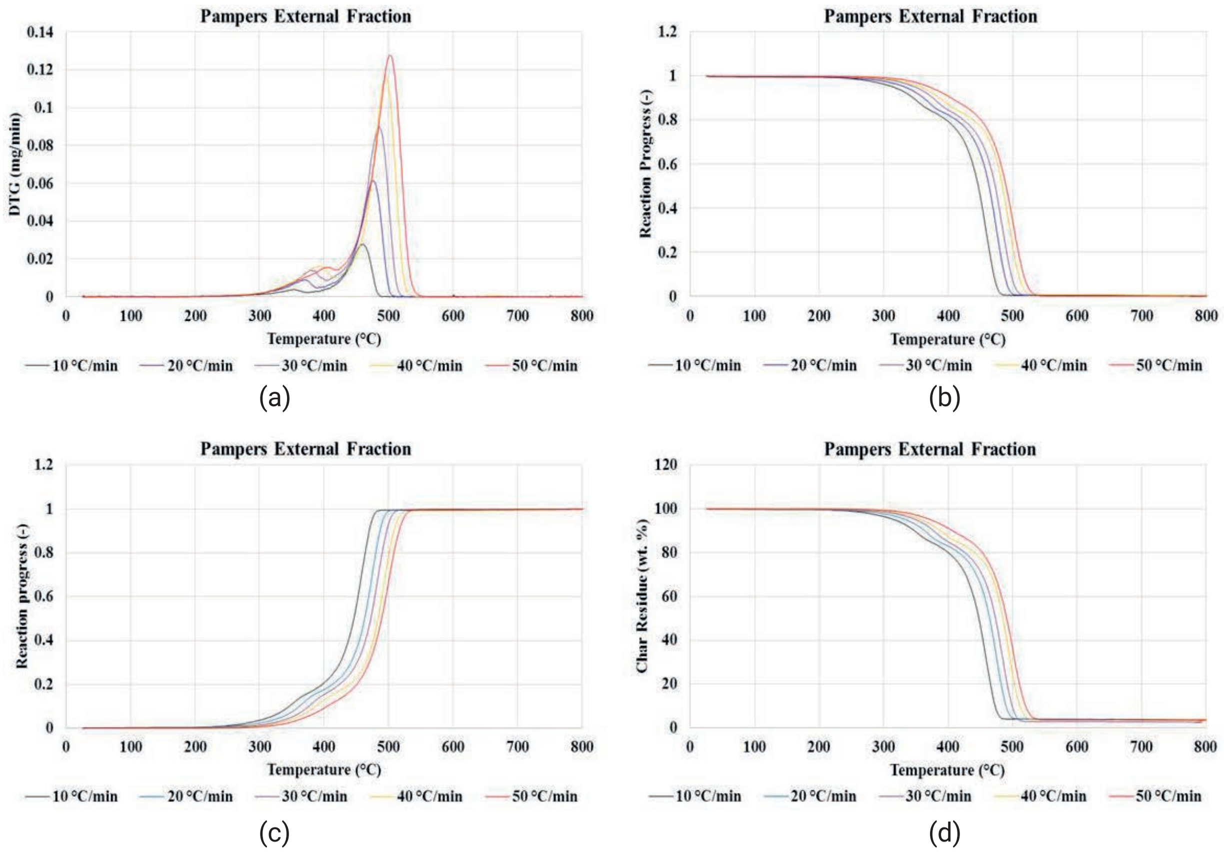

FIGURE 9: Pampers External Fraction derivative thermogravimetric analysis (DTG), reaction progress (α) and char residue profiles.

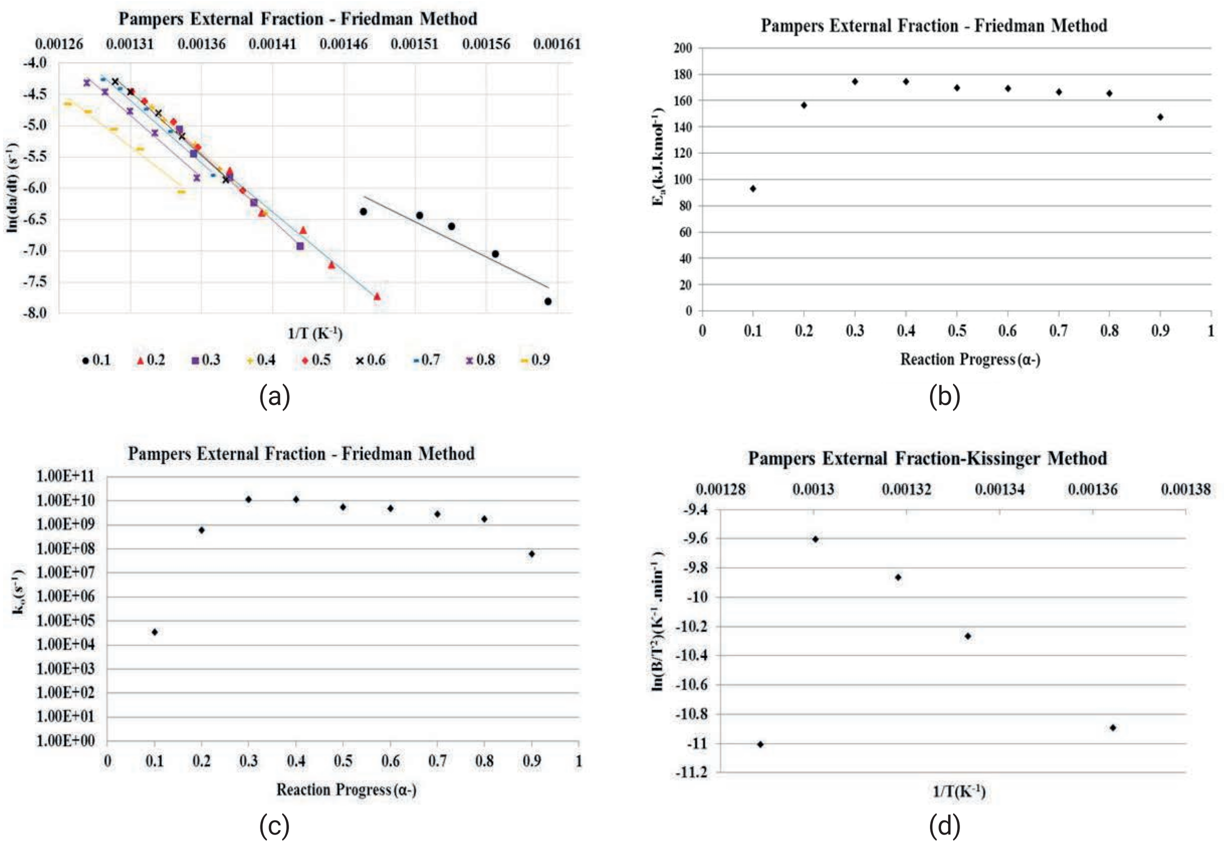

FIGURE 10: Pampers External Fraction kinetic parameters profiles (Friedman and Kissinger Methods).

TABLE 3: Char residue for the Huggies and Pampers disposable diaper brands.

Heating Rates (°C/min)	Disposable Diaper Brands - Char Residue (wt.%)			
	Huggies Interior Fraction	Huggies Exterior Fraction	Pampers Interior Fraction	Pampers Exterior Fraction
10	17.87	6.08	20.08	3.51
20	22.43	8.03	24.32	3.73
30	19.71	4.43	24.89	2.66
40	21.14	7.10	21.92	3.22
50	20.22	7.40	15.21	3.78

TABLE 4: Kinetic parameters estimated using iso-conventional methods for Huggies and Pampers disposable diapers.

Method	Parameter	Internal fraction				External fraction	
		Huggies		Pampers		Huggies	Pampers
Friedman	E_a (kJ/mol)	141.35382		109.31620		157.54123	159.71986
	k_0 (s^{-1})	1.29E+14		5.81E+09		9.56E+08	3.98E+09
Kissinger	E_a (kJ/mol)	6.58618		122.04952		166.35620	164.99964
	k_0 (s^{-1})	1.59E+05		1.85E+40		1.56E+07	1.10E+07
		Peak 1	Peak 2	Peak 1	Peak 2		
Friedman	E_a (kJ/mol)	166.93548	59.05602	147.83615	157.07943		
	k_0 (s^{-1})	1.20E+12	3.82E+02	4.55E+12	9.08E+08		
Kissinger	E_a (kJ/mol)	123.60424	160.62648	130.01433	93.03366		
	k_0 (s^{-1})	5.34E+05	1.35E+07	2.71E+06	1.77E+02		

refore, the final E_a and k_0 estimations are subsequently different. Hence, the differential approach (Friedman) is more accurate since differentiation can be carried out on every step of the reaction progress.

5. CONCLUSIONS

It has been illustrated that disposable diapers thermal treatment as a waste material recovery method is possible. Prior to processing, it is required that the diapers are separated into an internal fraction and external fraction since there are distinctive devolatilisation profiles between the two fractions. The present study also showed that disposable diapers consist mainly of volatile matter, while the ash content is relatively low. Furthermore, the interior fraction has a significant high volatile content which is a result of the materials contained in the inner layers of the diaper brands. Consequently, this further proves that disposal methods, such as landfilling and incinerating, are not suitable for disposable diapers. Subsequently, the interior fraction of the diaper can be further separated into two sub-fractions at it has two distinctive devolatilisation peaks that belong to the absorbent and bio-based material fraction. Therefore, step-wise recovery of the three main fractions, fossil-based (exterior fraction), bio-based (interior fraction fiber) and inorganic (interior fraction absorbent) need further investigation.

A precise study of the behaviour of the diapers under thermal treatment was further observed from different kinetic mechanism parameters obtained using iso-conventional methods, namely the Friedman and Kissinger methods. Since the Friedman method is independent of the heating rates as compared to the Kissinger method the Friedman method is more applicable to the present study. Therefore, the present study has not only illustrated the potential for the recovery of valuable chemical and energy waste products but further demonstrated the separation and recovery of the products during thermal conversion.

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GUIDING FUTURE RESEARCH ON THE VALORISATION OF SHREDDER FINE RESIDUES: A REVIEW OF FOUR DECADES OF RESEARCH

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ABSTRACT
Millions of tonnes of shredder fines are generated and disposed of globally, despite compelling reasons for its recovery. The absence of a review of previous literature, however, makes it difficult to understand the underlying reasons for this. Thus, this study attempts to investigate and assess what, to what extent, and in what ways shredder fines have been addressed in previous research. In doing so, guidelines are drawn for future research to facilitate the valorisation (upgrading and recovery) of shredder fines. Previous research concerning shredder fines was identified with respect to three main research topics. The material characterisation studies are predominantly confined to the occurrence of metals due to their recovery and contamination potential. The process development studies have often undertaken narrowly conceived objectives of addressing one resource opportunity or contamination problem at a time. Consequently, the full recovery (the retrieval of valuable resources and the bulk-utilisation as substitute material) potential of shredder fines has been largely overlooked. The main limitation of policy and regulation studies is the absence of in-depth knowledge on the implications of governmental waste- and resource-policies (macro-level) on actors' incentives and capacities (micro-level) for fines valorisation, which is necessary to understand the marketability of fines-derived resources. Undertaking a systems perspective is the key to recognising not only the different aspects within the individual research topics but also the inter-relations between them. It also facilitates the internalisation of the inter-relations into topical research.

1. INTRODUCTION

The shredding of end-of-life products has been gaining speed as an industrial activity worldwide, owing to the increasing consumption of goods. Waste streams such as end-of-life vehicles (ELVs) after depollution and dismantling, white goods, and industrial metallic scrap feed shredders around the world (Santini et al., 2012; Vermeulen et al., 2011). The entire process of shredding and post-shredder recovery of materials was developed primarily to recover metals.

A typical post-shredder recovery process starts by separating the shredded material into a light and a heavy fraction. What remains after recovering metals from these two fractions is known as the shredder residue. It consists of light fluff, heavy fluff, and a fine-grained material which is commonly identified as shredder fines or fines (Cossu and Lai, 2015; Vermeulen et al., 2011; Zorpas and Inglezakis, 2012). Globally, the annual generation of fines (0-20 mm) from the shredding of ELVs alone amounts to approximately four million tonnes, which is mostly shared between the regions of Europe and the US, while a relatively small proportion comes from Asia (Japan and Korea), as calculated from (Fiore et al., 2012), and (Vermeulen et al., 2011).

Even though advanced treatment processes are being employed to further recover materials (e.g. metals, plastics, and rubber) and energy (in incineration) from the larger fractions of shredder residues in certain countries, shredder fines continue to be disposed of throughout the world (Allen and Fisher, 2007; Santini et al., 2012; Singh and Lee, 2015a). Heterogeneity and small particle size are two prime factors that render the valorisation (upgrading and recovery) of shredder fines technically challenging and economically unappealing (Fischer, 2006; Vermeulen et al., 2011). Meanwhile, in regions such as Europe, decreasing availability of landfill space, stringent legislation (e.g. the EU landfill directive) and policy demands for higher re-

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source efficiency (e.g. the EU end-of-life vehicles directive) have been gradually creating stronger incentives for the valorisation of shredder fines. Given that this fine-grained material (0-20 mm) could comprise as much as 70% of the shredder residue (Cossu et al., 2014), its valorisation is deemed as essential for the fulfilment of the ELV directive targets of 95% recovery and 85% recycling (European Commission, 2000).

Although shredder fines have been studied for decades, the research is still in an early phase of development. These studies have involved their own specific objectives, and targeted different types of fines and aspects of fines handling in varying parts the world (Edo et al., 2013; Gonzalez-Fernandez et al., 2008; Mallampati et al., 2018; Singh et al., 2016b; Singh and Lee, 2015b). Therefore, a coherent overview of the current knowledge levels regarding fines management is lacking, thus preventing common knowledge building and making it difficult to identify central areas for future research. Another feature of this body of research is that most studies have only addressed one specific aspect of fines management at a time (e.g. a specific material characteristic or a particular recovery process), thereby disregarding that changing current disposal practices and achieving valorisation is a multi-faceted challenge influenced by several inter-related technical, economic, organizational, market and policy aspects (Andersson et al., 2019; Iacovidou et al., 2017; van Beers et al., 2009).

This article aims to review and assess the contributions and limitations of previous studies on shredder fines in order to provide a coherent overview of current knowledge levels, and thereby, guide future research on its valorisation. In doing so, we first identify which topics of fines management have been studied, in what ways and to what extent, and then assess how these research approaches have influenced the breadth and depth of current knowledge. Beyond specific guidance of how the research could be further improved topic-wise, an emphasis is also put on the inter-relations between the different topics of fines management and how such a systemic approach could support the development of valorisation strategies. Knowledge gaps of relevance for the valorisation of fines in terms of topics or aspects not yet addressed by the research community are also discussed.

1.1 Abbreviations

- ELVs: End-of-life vehicles
- MSWI: Municipal solid waste incinerator
- FDRs: Fines-derived resources

2. METHOD

A three-step analytical approach was undertaken in this study (Figure 1). The initial part involved the search and selection of literature concerning shredder fines based on pre-determined procedures and criteria (Cronin et al., 2018). Subsequently, the selected articles were first divided into different research topics displaying the main aspects addressed. Within each topic, the literature was reviewed in regard to the objectives, methodological approaches, and main contributions of the studies (i.e. the type of generated results). Finally, guidelines for future research on shredder fines valorisation were derived via an assessment of the limitations in scope and applicability of the reviewed literature. The relevance of the inter-relations between research topics in facilitating fines valorisation, was also investigated in order to analyse the needs for employing a systems perspective in future research.

2.1 Selection of literature

The literature search was conducted using the bibliographic database Scopus and by considering all the available literature since 1975. Scopus is the largest database of peer-reviewed scientific literature, drawing from more than 5000 publishers (Scopus, 2018). Therefore, it was assumed that limiting the search to this database would still enable us to reach an adequate domain of scientific literature. In order to obtain all the literature encompassing shredder residues including shredder fines, the following sequence of search words was used for searching within the field “Article title, Abstract, Keywords” of the database:

{shredder residue} OR {shredder residues} OR {shredder waste} OR {shredder wastes} OR {shredder fine} OR {shredder fines}.

The reason for including both shredder residues and shredder fines in the search terms was the absence of a standard demarcation for shredder fines, as it is always identified for a particular shredding facility (Morselli et al., 2010).

The search resulted in 518 articles. The titles, abstracts, and keywords of these 518 articles were subsequently read in order to screen the articles that had a relevance to shredder fines. A maximum particle size of 20 mm was used in order to demarcate fines from other shredder residues, which is apparently the most appropriate according to current literature (Allen and Fisher, 2007; Edo et al., 2013; Izumikawa, 1999; Lanoir et al., 1997; Morselli et al., 2010; Reuter et al., 1999; Santini et al., 2012; Vermeulen et al., 2011). Literature that did not specifically address shredder fines (i.e. shredder residues of a particle size <20 mm) were disregarded, and 42 articles were initially selected for the review.

2.2 Establishment of research topics and an overview of their content

The 42 selected studies were first reviewed mainly by reading through the title, abstract, and conclusions in order to identify which research topics they addressed. In total, this initial review led to the identification of three main topics, namely material characteristics, process development, and policy and regulation.

Fifteen articles were found concerning studies that have conducted different types of chemical and physical characterisations of the material. While some of these studies involve detailed characterisations of shredder fines specifically, others have characterised shredder residues as a whole and thereby only present limited information about the specific properties of the fine size fractions (<20 mm). All these studies, however, were categorised under the topic of material characteristics.

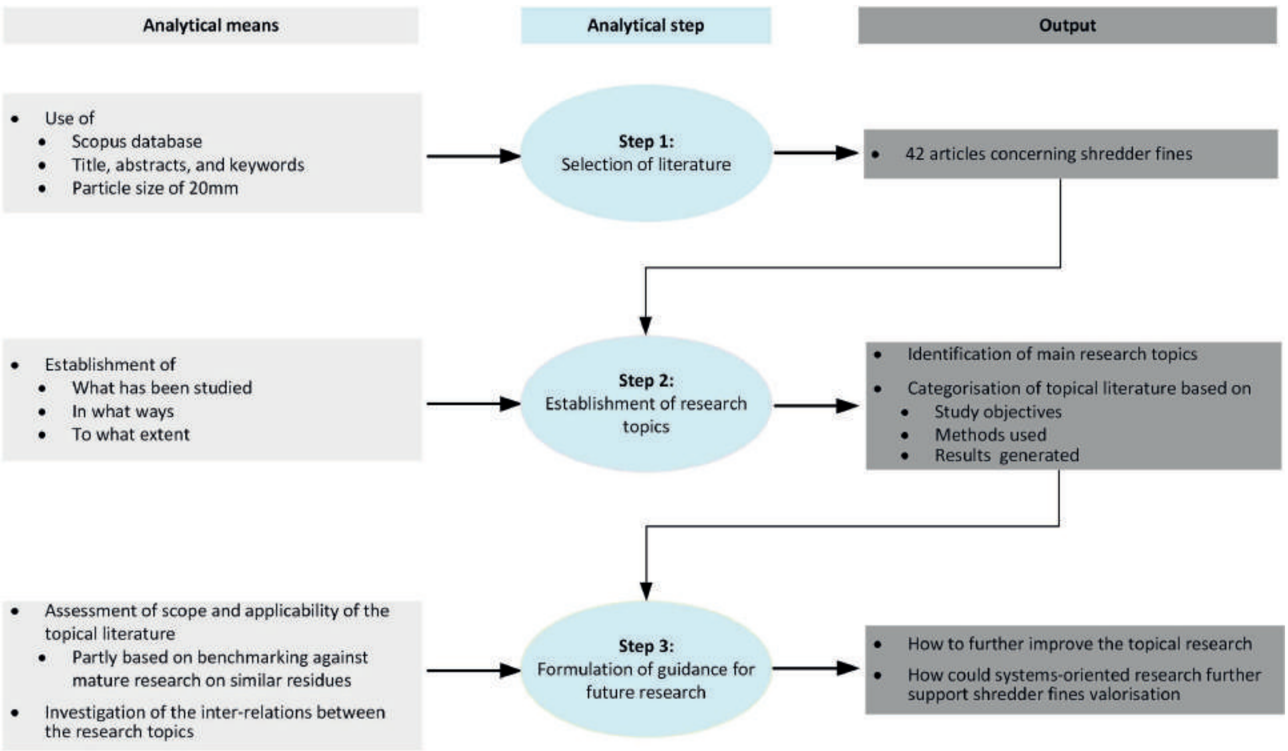

FIGURE 1: Illustration of the methodological approach of this paper.

Twenty-nine articles were found to focus on the development of technical processes for the treatment of shredder fines. Such studies investigate one or more of the following aspects: the recovery of specific materials from fines, the bulk utilisation of fines as a substitute material, and the reduction of the environmental risk of fines for disposal. This type of studies was categorised under the topic of process development. Unlike the studies in material characteristics, almost all of the literature in the process development category explicitly addresses shredder fines.

Four of the selected articles involved studies addressing the implications of governmental policy and regulation on the management and recovery of shredder fines in particular or shredder residues in general. The main reason for also including policy studies on the management of shredder residues (two studies) here was that such regulations could directly affect the conditions for the valorisation of shredder fines. These studies were categorised under the topic of policy and regulation.

It is worth noticing that some of the selected articles deal with more than one research topic. In such articles, a particular topic is always the primary focus while the others are secondary and thereby only partially addressed. For example, as part of process development (primary focus), material characterisation was performed (secondary). During the categorisation of the literature, such articles were included in both topics.

The studies within the identified research topics were then reviewed in detail to provide an overview of their different research objectives, methodological approaches, and main findings (i.e. the type of results generated). The intention behind such an overview was to facilitate the subsequent assessment of the scope and applicability of the main findings obtained within each of the research topics.

2.3 Formulation of guidelines for future research

As a means of formulating future research guidance, an emphasis was put on assessing how the scope and applicability of the provided results within each of the identified research topics could be further improved. This topic-wise assessment, however, was also complemented by a system perspective targeting the inter-relations between the research topics and how studying such inter-dependencies could further facilitate shredder fines valorisation.

2.3.1 Assessment of scope and applicability of studies within the research topics

The specific aspects of fines and their management studied under the research topics, are regarded as the scope. Thus, the assessment specifically targeted the research objectives and the breadth and depth of studied aspects within each research topic. This scope assessment was further supported by benchmarking the topical fines literature against more mature research on municipal solid waste incinerator (MSWI) bottom ash, displaying important aspects and knowledge levels for enabling the development of valorisation strategies. It is a comparable production residue to fines in terms of physical properties (e.g. heterogeneity and particle size) and chemical composition, and the fact that material characteristics can vary significantly depending on the input materials and incineration process. Review articles concerning MSWI bottom ash (Dou et al., 2017; Lam et al., 2010; Le et al., 2017; Margallo et al., 2015; Silva et al., 2017; Verbinnen et al., 2017) were primarily used as a means for conducting the benchmarking, while other literature was also used when necessary. Regarding the applicability of the reviewed topical literature, it was viewed here as the specific situations and settings under which the generated results are valid. Thus, this assessment involved a methodological review of how specific aspects of fines/fines management were studied within each research topic and in what way such procedures influence the applicability of the results.

The assessment of research in material characteristics was performed on the premise that the purpose of material characterisation would be the subsequent development of upgrading and recovery processes. There, for a heterogeneous material such as fines, a comprehensive knowledge of the chemical and physical properties is crucial (Allegrini et al., 2014; Fiore et al., 2012; Hernández Parrodi et al., 2018a). The scope was assessed in regard to the sufficiency of properties and the level of detail in characterisation. This was assisted by benchmarking against the MSWI bottom ash literature. The applicability was regarded as the undertaken sampling procedures and the material itself that was characterised (i.e. origin, quantity sampled, and particle sizes analysed). The applicability was assessed primarily concerning the ability of the sampling procedures to capture the variation of material composition, which contains crucial design parameters for the processes development (Allegrini et al., 2014; European Commission, 2004). The conditions due to materiality (i.e. origin, quantity sampled, and particle sizes analysed) were also considered.

The assessment of research in process development was performed based on the premise that the purpose of process development would be to recover valuable resources as well as avoid landfilling (by bulk utilisation as a substitute material). The scope was assessed regarding the sufficiency of the investigated aspects, i.e. the types of processes investigated, targeted resources and/or contaminants, and studied process attributes (e.g. pre-treatment, operating parameters, recovery efficiency, output characteristics, etc.) to enable full valorisation (i.e. the realisation of the full recovery potential) of shredder fines. Once again, this was assisted by benchmarking against MSWI bottom ash research. The applicability was assessed primarily in regard to the scale of operation for the developed processes, which is a direct indication of the technology readiness level (Mankins, 1995), and the material itself (i.e. origin and particle sizes) for which processes were developed.

The assessment of research in policy and regulation was performed based on the premise that the purpose of waste policy and regulation is to govern waste management, and such governance usually takes place at different socio-administrative levels (Hansen et al., 2002). The scope was assessed primarily based on the governing level on which knowledge was created, and the types of policy- and regulation-implications investigated within these levels. Such assessment presumes that typical top-down-driven policy research would not create adequate knowledge on the actor level, where the valorisation actually takes place (Rocha et al., 2007; Sabatier, 2019). Benchmarking against MSWI bottom ash research was performed to assist in this assessment as well. The applicability was assessed based on the potential of the created knowledge to facilitate future policy interventions on the valorisation of shredder fines in relation to the geographical area (spatial boundaries) of study and the material itself (shredder residue or shredder fines) for which the policy and regulation was intended.

2.3.2 Investigation of inter-relations between the research topics

The importance of employing system-oriented research for facilitating the valorisation of industrial residues is commonly recognized in the literature (Andersson et al., 2019; Iacovidou et al., 2017; Johansson et al., 2012). Here, such a perspective was used to investigate how the scope and applicability of shredder fines research could be further improved by also considering the inter-dependencies between the studied topics. Emphasis was primarily on providing guidelines on how the topical research on material characteristics, process development, and policy and regulation could be coordinated to better support the development of recovery strategies. This investigation also resulted in the identification of new, potentially important research topics that have not yet been studied for shredder fines, i.e. knowledge gaps.

3. GENERAL OVERVIEW OF PREVIOUS RESEARCH CONCERNING SHREDDER FINES

Except for a few early publications, shredder residues first started to receive noteworthy research attention in the 1990s (Figure 2). Since then, the interest in this topic has gradually increased. Although less than 10% of the research deals with shredder fines, the number of publications addressing the subject has also slowly increased during recent decades.

The research could be categorised according to three main research topics: material characteristics, process development, and policy and regulation. The research intensity is clearly prominent within the first two topics, which share 15 and 29 studies respectively, whereas only four studies are found within the policy and regulation topic (Figure 3).

In general, the pace of studies concerning shredder fines remained low globally until 2015, where a sudden increase in trend is visible. In addition, Europe accounted for more studies than the other regions. Only studies on process development could be seen as dispersed across different regions, whereas the studies on material characteristics were almost entirely limited to Europe. However, it is noteworthy that despite a sudden influx of process development research from Asia during recent years, eight of the articles from this region involve the same main author, plausibly as a consequence of mainstream research of this research group. Research in North America is only represented by four articles produced in the USA. The scarcity of research can perhaps be explained by the complex waste legislation, with more relaxed recovery objectives and availability of land for landfilling (Nayak and Apelian, 2014).

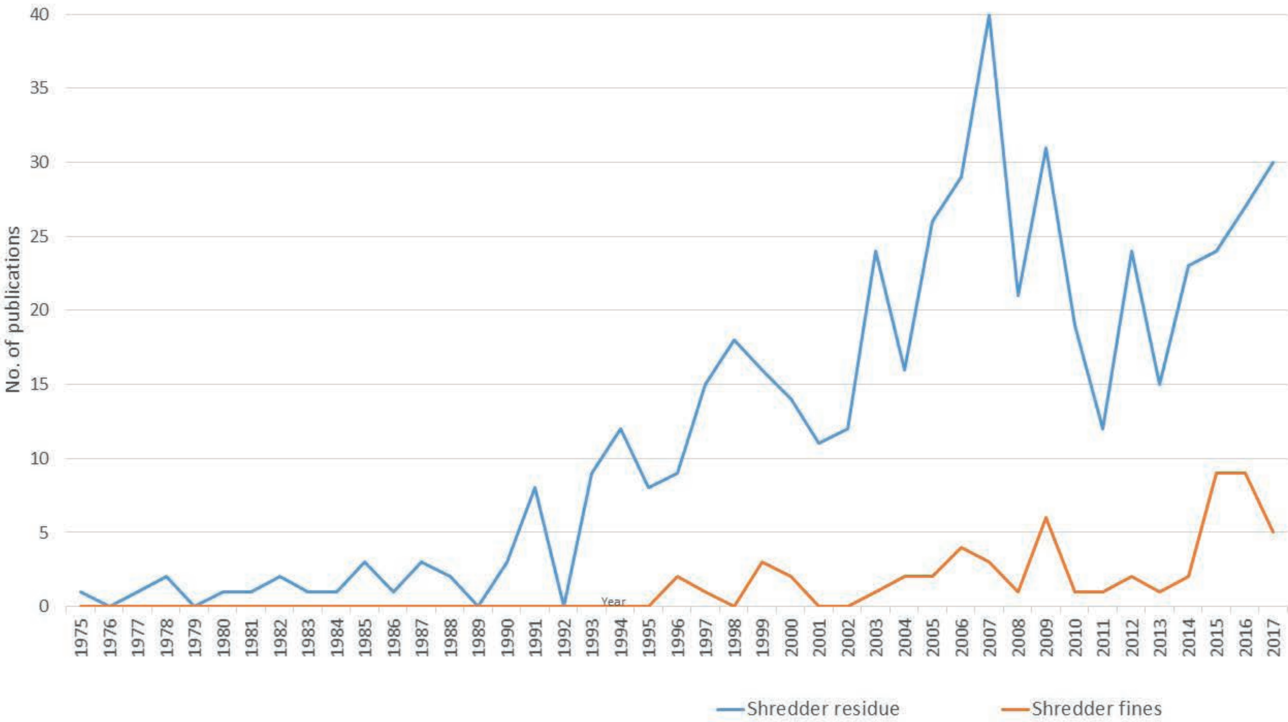

FIGURE 2: Over-time research intensity on shredder residues and shredder fines (< 20 mm) over the last four decades in terms of number of publications per year.

4. ASSESSMENT OF SCOPE AND APPLICABILITY OF PREVIOUS RESEARCH

In this section, the scope and applicability of previous research on shredder fines were assessed within the individual topics based on a review of the studies' research objectives, methodological approaches and main findings.

4.1 Material characteristics

Previous research on the material characteristics of shredder fines could be categorised in regard to two main objectives: the estimation of recovery potential and the estimation of environmental risks of disposal (Table 1). Although these objectives target different material characteristics of shredder fines, the studies within this topic have collectively established knowledge on several different types of chemical and physical properties. The characterised fines solely originate from Europe, and the research involves studies on various size fractions. In addition, the sampling procedures and type of characterised shredder fines vary significantly between the studies.

4.1.1 Scope-related limitations

One of the main scope-related issues of the research within this topic is that the established chemical and physical properties are scattered among the different studies. For instance, in studies targeting the recovery potential of fines, characterisation is done as part of developing a particular recovery process (Allen and Kolb, 2009; Konetschnik and Schneeberger, 2009; Péra et al., 2004; Reuter et al., 1999). Given the potential economic value of recovering base metals such as iron, copper, and zinc from fines, such metals have been a key focus in all of the studies. Consequently, only a few studies have established knowledge on the occurrence of other types of resources in fines such as plastics, minerals, other metals, and energy carriers. In the studies targeting environmental pollution risks (Börjeson et al., 2000; Fiore et al., 2012; Gonzalez-Fernandez et al., 2009, 2008; Lanoir et al., 1997), the characterisation focuses on base and heavy metals due to their high contamination potential and the metals-based regulation of landfill disposal in Europe. Only two studies have so far established knowledge on the presence of other contaminants in fines such as different types of organic compounds (Börjeson et al., 2000; Morselli et al., 2010). Given that the studies within this topic involve characterisation of selected chemical and physical properties for different types of fines (due to variations in input materials, shredding process, and particle size), a comprehensive understanding of both the resource and contamination potential is largely absent. As demonstrated by research on MSWI bottom ash (Dou et al., 2017; Lam et al., 2010; Margallo et al., 2015; Šyc et al., 2018), such comprehensive knowledge on the chemical and physical properties is essential for enabling effective development of resource recovery processes and facilitating the full valorisation of the material in question (Fiore et al., 2012; Hernández Parrodi et al., 2018a).

Another main scope-related issue in these studies is that the established chemical composition of fines typically limited to the total elemental concentrations. The sufficiency of such knowledge to spur subsequent process development is limited, as the form of occurrence of the different elements and materials influence both their recoverability and environmental pollution potential. For instance, MSWI

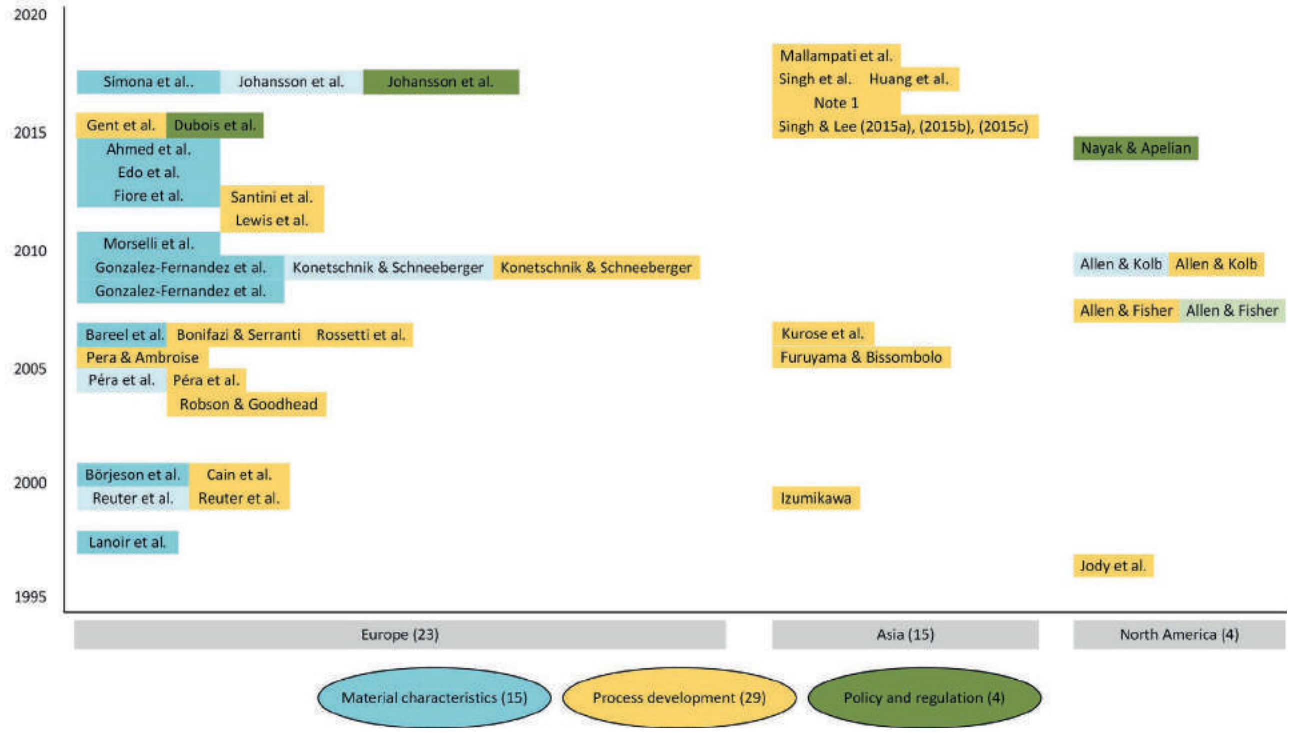

FIGURE 3: Temporal and geographical distribution of research work according to research topics. The dark colours represent literature where the respective research topic is the primary focus while the lighter colours indicate that the respective research topic has been performed with secondary focus. The number of articles is given within brackets region-wise and research topic-wise. Note 1: Mallampati et al. (Mallampati et al., 2016), Singh & Lee (2016a), Singh & Lee (2016b), Singh, Yang, et al. (2016c), Singh, Yang, et al. (2016b), and Singh, Chang, et al. (2016a).

TABLE 1: Categorisation of material characteristics literature based on research objectives (a total of 15 articles). The number of studies is given within brackets.

Scope-related attributes		Applicability-related attributes				
Research objective	**Targeted materials/ properties**	**Material origin**	**Size fractions analysed**	**Total no. of samples**	**Total qty. sampled**	**Sampling procedure**
Estimation of recovery potential (8)	Base metals [a] (7) Heavy metals [b] (7) Alkali metals (2) Alkaline (3) Metalloids (3) Inorganic compounds (3) [c] Organic compounds [d] (2) Plastics (1) Fuel properties [e] (3) Material fractions [f] (2)	Denmark: two landfills (1)	<10 mm	Two	20-40 kg	Strategic
		Italy: one plant (1)	<20 mm	One	50 kg	Non-systematic
		Belgium: one plant (1)	<2 mm	Two	Unknown	Unknown
		Sweden: one landfill (1)	<7 mm	Unknown	Unknown	Unknown
		Unknown (4)	<4 mm (2) <19 mm (1) <20 mm (1)	Unknown	Unknown	Unknown
Estimation of environmental risk (7)	Base metals (7) Heavy metals (7) Organic compounds (2)	Italy: two plants (1)	<10 mm	Two	20 – 30 kg	Representative
		Spain: two plants (2)	<6 mm	Two	16 kg	Non-systematic
		Slovakia: one plant (1)	<10 mm	One	Unknown	Non-systematic
		Italy: one plant (1)	<20 mm	Five	16 - 18 kg	Non-systematic
		France: one plant (1)	<20 mm	One	50 lit	Non-systematic
		Sweden: two plants (1)	<17 mm	Two	Unknown	Unknown

Studies focusing on the estimation of recovery potential:(Reuter et al., 1999), (Péra et al., 2004), (Bareel et al., 2006), (Allen and Kolb, 2009), (Konetschnik and Schneeberger, 2009), (Edo et al., 2013), (Ahmed et al., 2014), (Johansson et al., 2017).
Studies focusing estimation of environmental risk: (Lanoir et al., 1997), Börjeson (2000), (Gonzalez-Fernandez et al., 2008), (Gonzalez-Fernandez et al., 2009), (Morselli et al., 2010), (Fiore et al., 2012), (Simona et al., 2017).
a. Common metals that are recovered and traded in large scale. E.g. Fe, Al, Cu, Zn.
b. Metals in the periodic table ranging from scandium to bismuth, except for base metals, metalloids, Group I, and Group II.
c. Oxides, carbonates, and sulphates.
d. PCB, PAH, PCDD, and PCDF.
e. Properties that are commonly looked for in a fuel such as heat content, ash, moisture, chlorine, and sulphur.
f. Metals, plastics, and minerals (glass, stone, and sand).

bottom ash research has additionally characterised the speciation of metals, e.g. oxides, carbonates, sulphates, and chlorides (Dou et al., 2017; Lam et al., 2010; Margallo et al., 2015). The speciation largely determines the mobility of metals, which in turn influences the efficiency of any recovery process or measures to reduce environmental risks of disposal. Studies on MSWI bottom ash have even investigated the inter-relations between the chemical form of occurrence and the mobility of different components in different conditions, such as ageing and varying pH levels.

Furthermore, the physical and mechanical properties, a particularly crucial factor in the bulk utilisation as a substitute material in construction applications, is absent in shredder fines research. In contrast, a myriad of such properties (e.g. density, permeability, porosity, compressive strength, Young's modulus, deformation strength, Secant moduli, effective fringe angle, surface rupture, and global shape factor) has been investigated in the bottom ash research (Dou et al., 2017; Le et al., 2017).

4.1.2 Applicability-related limitations

Current knowledge on the material characteristics of shredder fines is not only limited by the breadth and depth of the various properties studied, but also by what material was actually addressed. The employed sampling procedure is the main issue in this regard, where most of the studies have only involved a few, randomly collected samples of fines from a particular plant. Given that the properties of fines from a shredder plant could vary significantly over hours, weeks, months and years owing to fluctuations in the composition of the feed material, such ad-hoc sampling is only capable of providing a snapshot of the material characteristics of a continuous flow. This lack of understanding about the variation in chemical and physical properties is critical since such characteristics determine the technical efficiency and economic viability of any subsequent treatment or recovery process (Allegrini et al., 2014; European Commission, 2004). Even the few studies that have employed representative sampling of fines fail to address such variation in material characteristics because the results are merely reflecting average values. A close attempt to address the variation factor has been made by Ahmed et al. (2014), using samples from previously disposed of shredder residues. There, the samples were strategically collected and characterised in order to establish the annual variation over a decade. However, results from such samples only show the long-term development of the material characteristics of fines and not the shorter-term (weekly or monthly), which is more interesting for the development of resource recovery processes. Furthermore, due to the extensive focus on the development of a particular recovery process, information regarding the employed sampling procedure, and thus, what shredder fines were actually characterised, is lacking in several studies.

In essence, current knowledge about the material characteristics of fines is constrained to a few assessed samples taken from a largely limited number of shredder plants in Europe (Table 1). In total, only a few hundred kilograms of fines have been characterised during the last couple of decades, to be compared with the approximately one million tonnes that are generated within this region annually (see Section 1). The need to develop a comprehensive knowledge on the material characteristics of fines, thus, not only calls for an analysis of a larger share of the generated amounts, but also stresses the importance of addressing the variations in physical and chemical properties on the plant, country, and regional levels. For instance, regarding MSWI bottom ash, such regional characteristics on, e.g. typical metal and metal oxide compositions have been established for European, Asian, and other countries (Dou et al., 2017), which enables the development of tailored valorisation measures in the respective levels.

4.2 Process development

The development of technical processes in previous research for managing shredder fines is categorised into three main groups: recovery of specific materials from fines, utilisation of fines as a substitute raw material, and reduction of environmental risk of fines disposal (Table 2). A majority of the studies in the first group involves the development of mechanical processes (e.g. shaker tables, jigs, eddy current separators) for the recovery of specific material fractions such as metals. These studies typically address resource recovery from the larger fractions of fines. Most of the studies belonging to the second group have investigated the utilisation of fines as a bulk material to substitute sand and gravel in construction-related applications. The dominating focus of the third group of studies is the solvent extraction of heavy metals. What the studies in these last two groups have in common is that they primarily target the smaller size fractions of fines.

4.2.1 Scope-related limitations

In general, the research has mainly developed processes focusing on one specific type of resource or contaminant. This is implied in that the studies are somewhat scattered in terms of the research objectives, types of studied processes/applications, and targeted resources (Table 2). The availability of a certain technology and required facilities at the researchers' disposal is a possible reason behind such narrowly conceived approaches to process development. Consequently, the development of integrated approaches for the full valorisation of fines has so far been largely overlooked. For instance, only 2 out of the 29 studies have investigated the integration of resource recovery and subsequent bulk utilisation of the rest as a substitute raw material (Jody et al., 1996; Konetschnik and Schneeberger, 2009). Furthermore, in studies on environmental risk reduction processes, the key target has been to enable final disposal rather than to upgrade fines to facilitate subsequent recovery. In essence, the number of studied processes and options for upgrading and recovery of shredder fines is yet limited. However, a vast domain of potential applications that could be used for integrated process development to facilitate full valorisation of fines could be identified by comparing against corresponding research on MSWI bottom ash management (Figure 4).

The mere consideration of a variety of potential processes and applications for the recovery of shredder fines would not be sufficient either. More in-depth assessments

TABLE 2: Categorisation of process development literature based on research objective (a total of 29 articles). The number of studies is given within brackets. Some studies contain more than one research objective.

Scope-related attributes				**Applicability-related attributes**		
Research objective	**Type of process / application**	**Targeted resources/ contaminants**	**Studied process attributes**	**Material origin**	**Size fraction (mm)**	**Scale**
Recovery of specific resources (16)	Mechanical separation [a] (9)	Metals as a material fraction (3)	Operating parameters (1) Recovery efficiency (3) Economic feasibility (1)	Belgium: plant A (1)	<2 mm	Lab
				Spain: plant A (1)	<10 mm	Pilot
				Italy: plant A (1)	<17 mm	Lab
		Plastics as a material fraction (2)	Recovery efficiency (1) Output characteristics (1) Economic feasibility (1)	Italy: plant B (1)	<20 mm	Lab
				South Korea: plant A (2)	<0.25 mm	Lab
				South Korea: plant B (1)	<1 mm	Lab
		Several material fractions (4)	Operating parameters (2) Recovery efficiency (3) Output characteristics (2) Economic feasibility (2)	South Korea: plant C (1)	<4.75 mm	Lab
				Japan: plant A (1)	<10 mm	Lab
				USA: plant A (1)	<20 mm	Pilot
	Smelting (2)	Base metals (2)	Pre-treatment requirements (1) Operating parameters (1) Recovery efficiency (1) Output characteristics (1)	Unknown (6)	<4 mm (1) <6.2 mm (1) <13 mm (1) <19 mm (1) <20 mm (2)	Conceptual Pilot Unknown Conceptual Lab
	Solvent extraction (4)	Base metals (4) Heavy metals (3)	Operating parameters (4) Recovery efficiency (4) Output characteristics (3)			
	Pyrolysis (1)	Fuels (1)	Operating parameters (1) Recovery efficiency (1)			
Utilisation as substitute raw material (9)	Construction applications [b] (5)	Shredder fines (5)	Pre-treatment requirements (2) Extent of utilisation (4) Operating parameters (2) Output characteristics (4) Economic feasibility (1)	Italy: plant C (1)	<4 mm	Lab
				UK: plant A (2)	<1 mm (1) <3 mm (1)	Lab
				South Korea: plant A (3)	<0.25 mm	Lab
	Moulding of thermoplastics (2)	Shredder fines (2)	Operating parameters (1) Extent of utilisation (2) Output characteristics (2)	Unknown (3)	<4 mm (1) <4 mm (1) <6.2 (1)	Unknown Conceptual Pilot
	Wastewater treatment (2)	Magnetic fraction (1)	Operating parameters (1) Extent of utilisation (1)			
		Elemental metals (1)	Operating parameters (1) Extent of utilisation (1) Output characteristics (1)			
Environmental risk reduction (9)	Solvent extraction (6)	Base - & heavy metals (9)	Operating parameters (8) Extent of utilisation (1) Output characteristics (9)	South Korea: plant A (1)	<0.25 mm	Lab
				South Korea: plant B	<1 mm	Lab
	Mechanical separation (1)			South Korea: plant C (5)	<4.75 mm	Lab
	Chemical immobilisation (1)			Japan: unknown source (1)	<4 mm	Industrial
	Physical immobilisation (1)			Unknown(1)	<4 mm	Lab

Studies focusing on the recovery of specific materials from fines:(Jody et al., 1996), (Reuter et al., 1999), (Izumikawa, 1999), (Furuyama and Bissombolo, 2005), (Bonifazi and Serranti, 2006), (Allen and Fisher, 2007), (Allen and Kolb, 2009), (Konetschnik and Schneeberger, 2009), (Lewis et al., 2011), (Santini et al., 2012), (Gent et al., 2015), (Singh et al., 2017), (Huang et al., 2017), and (Mallampati et al., 2018).
Studies focusing on the utilisation of fines as a substitute raw material: (Jody et al., 1996), (Cain et al., 2000), (Robson and Goodhead, 2003), (Péra et al., 2004), (Rossetti et al., 2006), (Konetschnik and Schneeberger, 2009), and (Singh et al., 2016a).
Studies focusing the reduction of environmental risk of fines: (Pera and Ambroise, 2005), (Kurose et al., 2006), Singh & Lee (2015a), (Singh and Lee, 2015c), (Singh and Lee, 2015b), (Singh and Lee, 2016a), (Singh and Lee, 2016b), (Mallampati et al., 2016), and (Singh et al., 2016c).
a. Consists of a series of mechanically operated unit operations that separate materials based on physical, magnetic, and electrical properties.
b. Consist of using fines as aggregate in concrete, aggregate in cement mortar, and raw material in cement production.

of the process attributes are equally important. For instance, in some of the processes displayed in Figure 4, it could be seen that both shredder fines and bottom ash have been investigated for a particular application. However, the research maturity and knowledge about the process attributes are substantially higher in the bottom ash literature. The different process attributes investigated in the fines literature are pre-treatment, recovery efficiencies, the

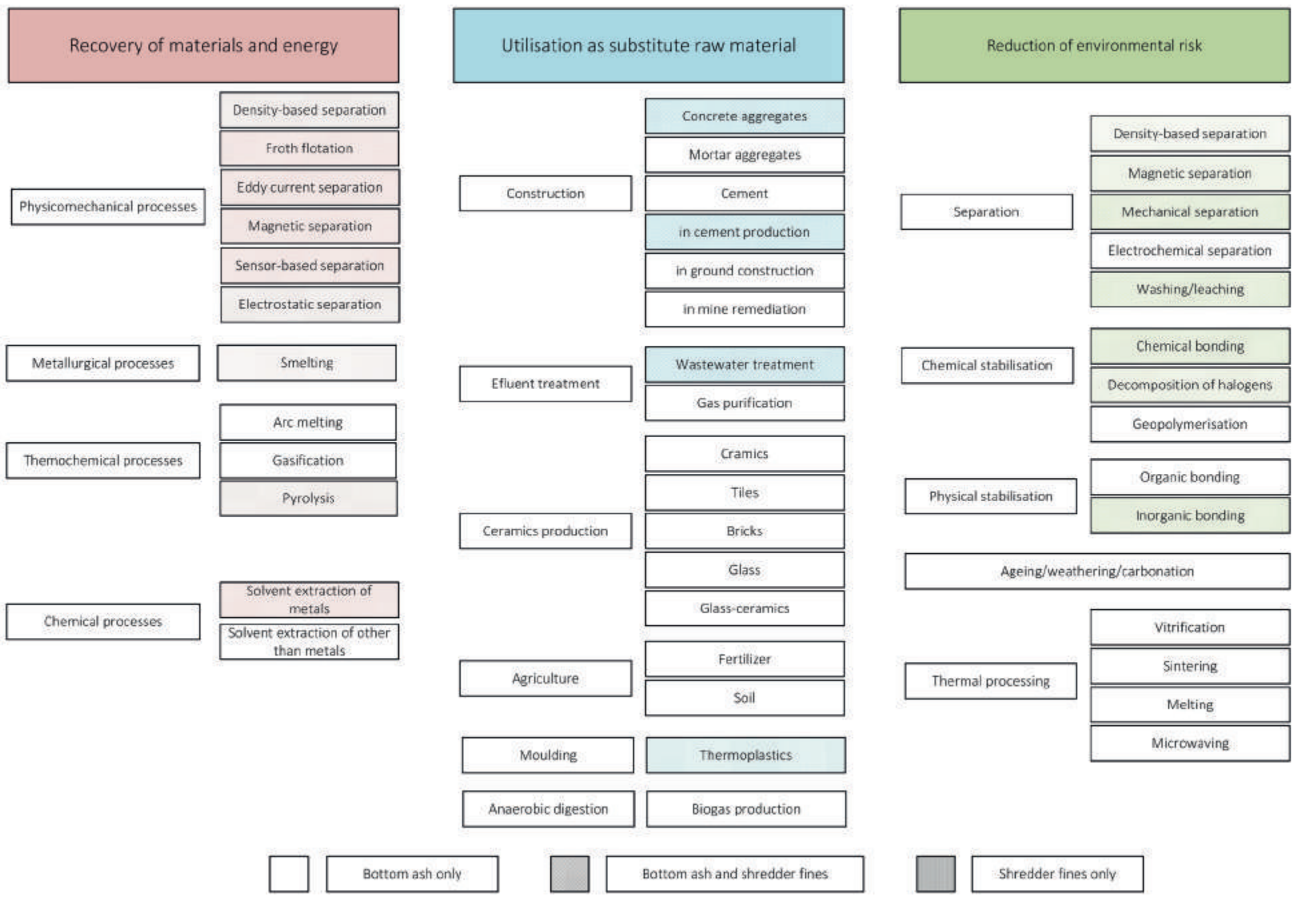

FIGURE 4: Scope of process development research concerning bottom ash management and shredder fines. References concerning municipal solid waste incinerator bottom ash research: Kinto (1996), Lam et al. (Lam et al., 2010), Rahman & Bakker (2012), Tanigaki et al. (2012), Rahman & Bakker (2013), Margallo et al. (2015), Gao et al. (2015), Tang & Steenari (2016), Dou et al. (2017), Silvia et al. (2017), Holm & Simon (2017), Breitenstein et al. (2017), and Verbinnen et al. (2017).

extent of material substitution, operating parameters, and output characteristics (Table 2). Recovery efficiencies, the extent of material substitution, and the operating parameters (temperature, pressure, flow rates, mixing rates, etc.) are the most important attributes to enable the further development and feasibility assessment of processes as they determine both technical and economic feasibilities, and therefore, featured in all the studies. Pre-treatment, owing to the heterogeneity and high contamination levels, is an important attribute that is often required to upgrade fines and make them suitable for a particular application, yet, only three studies have addressed it. Characteristics of the outputs of a process are necessary to establish due to two main reasons. The first is to determine whether the outputs meet the user and regulatory requirements; the second is in the case of the potential integration of recovery processes, where outputs from one process become the input to another. The economic feasibility has been investigated in only five of the studies, despite being a necessary condition for the application in commercial scale. Furthermore, none of the studies have addressed all these attributes. The necessity to perform research specifically focused on some of the above process attributes – especially, recovery efficiencies and operating parameters – is understandable at this early phase of research in order to closely investigate the technical feasibility of processes. However, the realisation of the developed processes would not be successful unless all these attributes are addressed.

4.2.2 Applicability-related limitations

A particularly interesting aspect of the studies within the process development topic is that a majority of them (20 of 31) investigate processes on a laboratory scale. Four out of the five pilot-scale processes are from the resource recovery category and deal with the mechanical separation of specific materials from fines (Allen and Fisher, 2007; Gent et al., 2015; Izumikawa, 1999; Jody et al., 1996), while the remaining one represents material substitution in construction applications (Jody et al., 1996). The only process that is studied on an industrial scale is the mechanical separation of heavy metals to reduce the environmental risks of fines disposal (Kurose et al., 2006). Three processes have been developed on a conceptual basis, plausibly emanating from the knowledge and experience of the authors, and the scale of operation remains unknown in two.

The applicability of the results obtained from lab-scale process development is limited for several reasons (Mankins, 1995). One is the effect of changing process conditions when the scale of operation is increased. An example of this is the metals recovery by smelting as a pyro-metallurgical process. Even though a recovery as high as 98% was reported for Cu and Fe when tested on lab scale, a dynamic system such as smelting may not reach an equilibrium between the three phases (metal, slag, and gas) on an industrial scale, thus affecting the outcome (Reuter et al., 1999). Another major limitation of process development on a lab scale is the inability to assess the investment and operating costs. The possibility of overlooking the influence of supply chain factors, which are crucial on an industrial scale, is another drawback of being limited to lab-scale tests. For example, Cain et al. (2000) have investigated the use of fines in co-injection moulding to produce composite moulds comprised of shredder residue core and polypropylene (PP) skin. However, essential physical properties such as compressive strength and thermal stability depend on the material characteristics and quality of shredder fines, which in turn depends on the source for that material. Therefore, fines from different shredder plants (instead of one single source) need to be tested because an industrial-scale operation would most likely require the supply from multiple sources.

Regardless of the studied scale of operation, a common applicability-related issue of the studies within this topic is that they typically base the process development on a specific type or even sample of shredder fines. This means that the process efficiency and performance for the upgrading and resource recovery of other fines, with significantly different material properties, are largely uncertain. Yet, in the reviewed studies, information about the origin of the fines used for process development and their specific characteristics are often scarce – something which makes it increasingly difficult to evaluate the applicability of the results.

4.3 Policy and regulation

The research within this topic is divided into two main categories (Table 3). In the first category, the focus is on how different governmental policies could influence incentives for the valorisation of shredder residues (not limited to fines) and increase the competitiveness of the secondary resources derived from this material. These studies apply a top-down and macro-level (shallow yet broader knowledge on a regional/national level) approach to the implications of policy and regulation. Research within the second category instead stems from a micro-level perspective, targeting specific shredder plants and the assessment of the marketability of fines-derived resources (FDRs) such as ferrous metals, non-ferrous metals, plastics, and minerals (sand, stone, gravel, and glass). These studies provide deeper insights on factors such as local institutional regulations for using secondary raw materials and gate requirements of potential users of FDRs.

4.3.1 Scope-related limitations

The knowledge provided by the studies on the macro level is somewhat superficial as it mainly involves anticipated implications of existing or suggested policy instruments. This knowledge is limited due to the small number of studies that could only investigate a few policy-related issues out of many. Comparing this with the MSWI bottom ash literature reveals several other macro-level implications such as policy influence on material composition (Lee et al., 2014) and triggering resource recovery (Born, 1994), and governmental decision making to facilitate valorisation (Huang et al., 2006). Regarding the micro-level studies, in addition to the small number of studies, a significant limitation is that they address the implications on secondary materials rather generally and are confined to regulatory requirements. In contrast, detailed investigations on micro-level implications, including marketability-related aspects beyond the regulatory requirements such as supply, demand, and competition in relation to specific applications of secondary materials, is evident in the MSWI bottom ash research (Vorobieff, 2010).

Additionally, the studies within this topic lack an understanding of how the involved actors interpret and react to different policy stimuli, which is paramount in putting policy into practice. Such complementation of the micro-level perspective with macro-level policy studies is evident in the MSWI bottom ash research (Born and Veelenturf, 1997; van der Zwan, 1997).

4.3.2 Applicability-related limitations

The main feature that limits the applicability of the two macro-level studies is that the focus has been on shredder residues in general rather than on shredder fines. Thereby, the understanding of the implications of current policies on the valorisation of shredder fines has become marginalised. However, the general applicability of the findings, irrespective of the region of focus in the study, is an ad-

TABLE 3: Categorisation of policy and regulation literature based on the research objective (a total of 4 articles). The number of studies is given within brackets.

Scope-related attributes			Applicability-related attributes	
Research objective	**Governing level**	**Studied policy and regulation implications**	**Region of study**	**Type of material**
Study the implications of governmental policy (2)	Macro (2)	Role of waste-based policies as drivers and barriers of valorisation (1) Influence of market-based policies to create competitiveness for secondary materials (1)	USA (1) EU (2) Belgium (1)	Shredder residue
Study the implications of institutional regulation (2)	Micro (2)	Potential users of recovered materials and gate requirements (2) Regulatory requirements for use of fines as a raw material in construction applications (2)	Sweden (1) California state (1)	Shredder fines

Studies focusing on regulatory conformance of valorisation measures: (Allen and Fisher, 2007) and (Johansson et al., 2017).
Studies focusing on policy implications of valorisation measures: (Nayak and Apelian, 2014) and (Dubois et al., 2015).

vantage of typical top-down macro-level policy studies. For example, the insights created on policy instruments such as disposal tax, tradeable recycling credits, and refunded disposal tax could be generally considered as a means for increasing the competitiveness of local recyclers (Dubois et al., 2015). Additionally, such studies can compare the driver- and barrier-effects of policy between regions, e.g. Europe and the USA (Nayak and Apelian, 2014).

In contrast, the particulars of micro-level studies can fluctuate substantially depending on the locality (e.g. state, county, or municipality) involved, and therefore, unlike its former counterpart, the applicability is geographically confined. Nevertheless, such knowledge is necessary to highlight the implications of different governing approaches for the handling of uncertainties and conflicts of interests between resource-efficiency and risk-mitigation objectives in different localities, and thereby, to develop locality-specific valorisation applications. For instance, country-specific standards have been implemented for the reutilisation of MSWI bottom ash in ground construction applications such as road base layer, parking lots, pavements, etc. Certain countries, e.g. Germany, Denmark, and Taiwan, have specified the minimum distance from drinking water sources and minimum elevation from groundwater sources, whereas other countries, e.g. Japan, Denmark, and the Netherlands, have specified the maximum allowed quantity and thickness (Dou et al., 2017; Van Gerven et al., 2005).

5. FUTURE RESEARCH GUIDANCE ON THE VALORISATION OF SHREDDER FINES

A prominent feature of the previous research on shredder fines is that the studies have often been performed in isolation within the individual topics. In such isolated topical research, not only the significance of the aspects of other topics is overlooked, but also the various inter-relations between them. Nonetheless, the valorisation of a heterogeneous material such as fines involves the interaction of the different topical aspects on different levels. Hence, a systemic approach that acknowledges such inter-relations (Figure 5) is deemed necessary to complement topical research (Andersson et al., 2019; Iacovidou et al., 2017) concerning fines management.

The valorisation of fines essentially involves process development to derive marketable fractions via exploiting the recovery potential and resolving the material constraints. Ragarding that, the key limitation of the previous research is that the process development has primarily taken place to address one specific resource opportunity (e.g. recovery of valuable metals) or material constraint (e.g. removal of heavy metals) at a time. Additionally, there, the investigation of material constraints is further confined to being used as a means for environmental risk reduction for (landfill) disposal, rather than for the identification of the need for upgrading for subsequent recovery. The aforementioned narrow approach to process development is also reflected in the approaches undertaken in material characterisation in previous studies. There, the established material properties are generally limited to a single type of resource (often base metals) or contaminants (often heavy metals). Such characterisation would not facilitate the process development to harness the full valorisation (the recovery of valuable resources as well as the utilisation of the whole of the material) potential. Furthermore, these studies have predominantly employed an ad-hoc collection of one or two samples, which would not reveal the over-time variation of composition that provides essential process design information. Consequently, the developed processes risk ending up redundant. It is also understood that research concerning shredder fines is still in an early phase where it is natural to investigate a particular topic in detail. Yet, the process development for the valorisation of such a heterogeneous residue needs to emanate from a comprehensive understanding of the material characteristics (a, Figure 5). That is, not only an array of chemical and physical properties is established, but also their distribution across the different size fractions and the over-time variation of the composition (Hernández Parrodi et al., 2018b). Such characterisation would facilitate the identification of the resource and contamination potential fully. Thereby, potential users of fines-derived resources (FDRs) could be identified for a variety of applications in order to not only avoid missing good valorisation opportunities but also aid in the integration of different types of recoveries to facilitate full valorisation (b, Figure 5). Then, their quality specifications (market requirements) could be established and

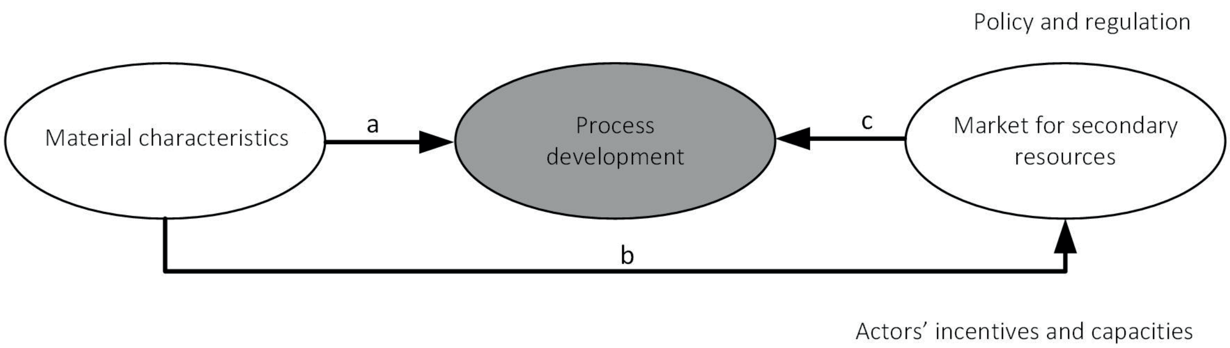

FIGURE 5: The systemic view of shredder fines valorisation. The arrows represent the inter-relations between the different research topics and other aspects concerning fines valorisation that have not been specifically addressed in previous research. a: Knowledge on material characteristics to enable process development; b: Knowledge on quality requirements of users of fines-derived resources, to enable process development; c: Knowledge on material characteristics to identify potential users of fines-derived resources.

the upgrading requirements to address the corresponding material constraints identified (c, Figure 5). Such a user requirement-based approach to process development, which is necessary to produce marketable outputs, is absent in the previous literature.

It is also quite natural to observe process development on a lab scale in an early phase of research. However, in order to ensure applicability, the results obtained it such studies need substantiation on a larger scale (e.g. pilot- or industrial-scale) (Mankins, 1995). Furthermore, associated sustainability concerns such as environmental impacts, costs of the involved processes, and income/savings from valorised products could simultaneously be considered as part of process development. Examples are the identification of particularly significant environmental impacts (hotspots) (Joyce et al., 2018), and key sources (material and energy inputs) of such impacts and potential substitutes (Joyce and Björklund, 2019).

Knowledge on drivers and barriers for initiating valorisation needs to be a key element as well in future research concerning fines management. There, the context of policy and regulation, and actors' (suppliers and users of FDRs) incentives and capacities play a vital role as governing conditions of fines valorisation. For instance, the implications of governmental waste policies on fines valorisation need to be established: that is, to what extent are the current policies reflected in legislation, and how are they implemented via institutional regulation practices? For example, in Sweden, significant uncertainties exist among the local institutions regarding what quality requirements actually rule in different situations and settings concerning the use of secondary materials, despite the EU and national policies for increased resource efficiency. Consequently, the high environmental and human health risk of contaminants in secondary resources is preconceived, and over-prudent limits on contamination levels are imposed (Johansson et al., 2017), which pose a major challenge in producing marketable FDRs. Providing insights on anticipated future policy changes is also important. In order to understand the real implications of different policy instruments (existing or anticipated) in practice, top-down approaches need to be complemented by bottom-up research, targeting how the shredding companies and potential users of FDRs would interpret different policy stimuli (micro-level), and in what capacities they would react. Such an in-depth understanding is crucial in the development of necessary policy interventions to both facilitate the technology development for fines valorisation and foster the marketability of FDRs. The significance of this perspective in accurately ascertaining the potential of residue valorisation has also been recognised in relation to the valorisation of MSWI bottom ash (Ribbing, 2007) and other types of industrial residues (Johansson et al., 2012; van Beers et al., 2009; Van Gerven et al., 2005). Some authors (Allen and Fisher, 2007; Johansson et al., 2017) have slightly touched upon these aspects concerning fines valorisation via establishing the gate requirements of potential users of FDRs; nevertheless, this area of knowledge, to a large extent, remains a gap in the literature. On the other hand, understanding the marketability of FDRs is incomplete without knowledge of the market forces at play and the dynamics of business development between shredding companies and users of FDRs. The former constitutes the structural elements of the market such as the current flows of shredder fines and their future prognosis, potential user demands on FDRs, and competition from other (often primary) resources (Porter, 1985, 1980). The latter consists of organisational aspects such as business strategies, value orientations, culture, and learning processes since it involves a change in current practices for both organisations under a new business contract (Peschl, 2007; Pettigrew, 2012). Without a thorough understanding of such market and organisational aspects, the true marketability of FDRs would not be assessed. Thus, endeavours for the valorisation of fines are unlikely to be realised and sustained.

6. CONCLUSIONS

The knowledge created in previous literature concerning shredder fines, in general, is scattered and incoherent. The main contribution of this article was to develop a coherent overview and an assessment of the previous research on shredder fines, and thereby, facilitate future research concerning its valorisation. The previous literature has collectively contributed to the development of various domains of knowledge, especially regarding specifics on the individual research topics, which is essential for the upgrading and recovery of shredder fines. Nevertheless, it also unveils a significant potential for improvement in the same regard.

Material characteristics studies are almost limited to a few common materials and elements, primarily revolving around the metal content. Comprehensive knowledge of the chemical and physical properties is required to establish the resource and contamination potential fully. There, undertaking strategic sampling procedures to capture the variation in composition is of paramount significance, as ad-hoc sampling is inadequate to do so. Process development studies have often taken place with narrowly conceived objectives, i.e. addressing one resource opportunity or contamination problem at a time. In order to facilitate full valorisation, it is crucial to identify a variety of valorisation alternatives and the potential for their integration. Furthermore, the substantiation of lab-scale results in larger-scale operations is essential to make the findings more applicable. Incorporating environmental impact and economic aspects already in the early phase of process development is also recommended to ensure sustainability, which is an aspect missing almost entirely in the previous research.

The main limitation of the policy and regulation studies is that the macro and micro levels have been investigated in insolation. Understanding the influence of governmental waste policies and institutional regulation on actors' incentives and capacities to produce and use FDRs serves a critical aspect of facilitating both technology development for fines valorisation and marketability of FDRs. There, the complementation of top-down research (macro-level) with actor-level implications (micro-level) is vital. Other aspects concerning marketability, such as the structural elements of the market for FDRs and the dynamics of business development between their suppliers and users, remain almost entirely absent in the previous literature.

Isolated topic-specific research is perhaps necessary at an early phase of knowledge development; nevertheless, the realisation of shredder fines valorisation is an outcome of various aspects, within the different topics, acting together influencing each other. Therefore, future research needs to undertake a systems perspective that shows this bigger picture, comprehending not only the significance of the individual topics but as importantly their inter-relations. It also facilitates the internalisation of the inter-relations of the different aspects into topical research.

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PYROLYSIS OF VARIOUS TYRE TYPES: CHARACTERISTICS AND KINETIC STUDIES USING THERMOGRAVIMETRIC ANALYSIS

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ABSTRACT

The present study entails investigating the fractional devolatilisation effect on the passenger car tyre (PCT) and truck tyre (TT). Thermograms and their first derivatives are used to profile devolatilisation of the PCT and TT rubbery fractions. Both tyre types rubbery fractions consist of mainly natural rubber (polyisoprene, NR) and synthetic rubber (polybutadiene, BR and styrene-butadiene, SBR). The NR is relatively significant compared to the BR and SBR in the TT. In the PCT, the rubbery fraction showed equal distributions between the NR and BR-SBR. Unlike conventional waste resources pyrolysis processes where the aim is towards energy recovery, novel pyrolysis is designed based on the characteristics of the materials in order to promote the production of the valuable products in addition to the energy recovery. Therefore, kinetic mechanisms pathways of the rubbery fraction devolatilisation from waste tyre pyrolysis are studied and models are developed. Model-free methods, such as Friedman and Kissinger are used to model the kinetics parameters.

1. INTRODUCTION

Solid waste management has become a global challenge mainly due to improper handling and processing of the end-of-life non-biodegradable materials. It is a common practice to directly dispose of the used materials in the landfills without consideration of the potential value of these materials since they consist of high amounts of carbon and hydrogen. This is mainly due to the lack of stringent environmental regulations and misunderstanding of the economic benefits from the recycling of these materials (Martínez et al., 2013). Both regulations and misinformation are one of the main reasons for the high rate of waste residues piles accumulation in the landfills. It is estimated that 1.5 billion new tyres are sold worldwide each year, while an estimated 4 billion waste tyres are currently reported to be in landfills and stockpiles. Therefore, if unattended to, the disposal of waste tyres will not only become an increasing environmental problem, but also a social and economic issue. It is therefore surprising, that there is the continuous dumping of waste tyres in landfills, which promotes stockpiling. This may result in the creation of breeding grounds for the pests and insects unless there is a substantial investigation of the valorisation of the waste materials, with the thermochemical conversion being one of them.

Among thermochemical processes for waste tyre conversion (Barlaz et al., 1993; Giugliano et al., 1999; Sienkiewicz et al., 2012), pyrolysis is one of the most promising process, since it results in products that have a variety of industrial and domestic applications (Quek & Balasubramanian, 2013; Williams, 2013). Pyrolysis is defined as the thermal treatment of waste tyres under inert conditions to yield pyro-gas, tyre derived oil (TDO) and pyro-char (Antoniou & Zabaniotou, 2013; Mui et al., 2004). The pyro-gas can be used as fuel for the endothermic pyrolysis process (Williams, 2013) or various other applications, e.g. when further processed to recover valuable chemicals, such as butadiene or isoprene. TDO's significantly high hydrogen and carbon element content and high concentration of valuable chemicals, e.g., DL-limonene, terpinolene and p-cymene characterise it as a potential alternative for liquid fuel, or as a feedstock for the production of valuable chemicals (Banar et al., 2012; Quek & Balasubramanian, 2013). The pyro-char mainly contains carbon and can be upgraded into the carbon black or activated carbon (Zabaniotou et al., 2004). However, to achieve these beneficial applications for the waste tyre pyrolysis products, intensive processing to improve products, such as product refining, upgrading, and purification, is required.

Therefore, prior characterisation of the pyrolysis feedstocks is critical since it can reduce or eliminate the processing requirements of the pyrolysis products. Materials characterisation is important in the design or modification

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of the pyrolysis process to target high yield and selectivity of certain chemicals or materials. Most significant tyre fraction which is the main source of volatile matter mainly consists of rubbers, such as natural rubber (polyisoprene, NR), butadiene rubber (BR) and styrene-butadiene rubber (SBR) (Williams, 2013). Moreover, the tyre consists of processing additives (fillers, vulcanisation accelerators, inhibitors, etc.), added during tyre compounding (Quek & Balasubramanian, 2013). It is, therefore, anticipated that kinetic mechanism study models of the pyrolysis of tyres are impossible. This has led to limited literature data on the rubber components characterisation of the waste tyres. Even worse, different rubbery components from different types of waste tyres have not been sufficiently investigated. Various materials characterisation investigations have been conducted using thermogravimetric analysis (TGA) in order to better understand the pyrolysis of a waste tyre (Kim et al., 1995; Conesa et al., 1997; Leung & Wang, 1999; Mui et al., 2004; Aylón et al., 2005; Mui et al., 2008; Lopez et al., 2009; Cheung et al., 2011; Quek & Balasubramanian, 2012; Lah et al., 2013; Williams, 2013).

To facilitate the investigation of the tyre rubbery components characterisation, waste tyre devolatilisation needs to be understood (Chen et al., 2001; Friedman, 1964; Kissinger, 1957; Senneca et al., 1999). Kinetics mechanism parameters, such as activation energies (Ea) and pre-exponential rate constants (k_0) in relation to the pyrolysis temperature and heating rate have been reported (Seidelt & Bockhorn, 2006; Lam et al., 2010; Cheung et al., 2011; Danon et al., 2015). From these studies, a trend illustrating that waste tyre devolatilisation kinetics mainly entails various parallel and serial devolatilisation reaction pathways, such as devolatilisation of the tyre processing oils followed by devolatilisation of the natural rubber and finally synthetic rubbers (SBR and BR). Additionally, waste tyre pyrolysis has been kinetically presented according to products volatilisation, such as gaseous products, variety of TDO sub-fractions (aromatics, aliphatics, polyaromatic hydrocarbons and heteroatom compounds) and char (Olazar et al., 2005).

A multi-stage pyrolysis process with 1) pyrolysis of tyre additives, 2) pyrolysis of polymerised rubbers and 3) pyrolysis of the cross-linked and/or cyclised rubbers (Lam et al., 2010) has been proposed. This study was supported by Cheung et al. (2011) with the following pyrolysis stages: 1) pyrolysis of the volatiles, 2) main chain scission (depolymerisation) to intermediates, 3) further depolymerisation of the intermediates (pyrolysis of cross-linked rubbers), and 4) cracking of the intermediates to shorter-chain compounds (degradation) (Cheung et al., 2011). Other authors, Danon et al. (2015), defined tyre rubbers devolatilisation kinetics as a three-stage process: 1) devolatilisation of additives, 2) crosslinking and depolymerisation of the rubbers into intermediates and volatiles, respectively, and 3) finally degradation of the intermediates to volatiles (Danon et al., 2015). Although there seem to be agreements on the observations, however, limited data have been reported on the comparison between the tyre types.

The present work focuses on the kinetic mechanism of two types of waste tyre (truck tyre and passenger car tyre) pyrolysis based on the rubbery fractions devolatilisation. The truck tyre compounding process is different from the passenger car tyre compounding process. Since there is a substantial difference in the tyre compounding process mainly of the rubbery component, characterisation by tracking of the devolatilisation profile of the rubbery fraction of each tyre will reveal this hypothesis. Kinetic mechanism parameters are not merely evaluated for the devolatilisation of the truck tyre rubbery fraction, through the application of thermogravimetric analysis (TGA), but the rubbery fractions are distinctively tracked to evaluate their kinetic mechanism parameters. Moreover, the precise graphical illustration of each rubber component fractions intensifies different stepwise devolatilisation for different types of waste tyres. The heating rate is varied between 10 and 50°C/min.

2. THEORETICAL BACKGROUND

Kinetic analysis requires tracking of the devolatilisation thermograms of rubbery fractions from waste truck tyres and passenger car tyres. The proposed model is based on the devolatilisation of the NR, BR and SBR. The conversion rate of each rubber fraction is given by the mass fraction devolatilisation equation, Eq. (1).

$$\frac{d\alpha}{dt} = k(T)f(\alpha) \tag{1}$$

where: α is a mass fraction, t is time, k(T) is reaction rate constant at temperature, T (K), and f(α) is the reaction model.

The time, t, and heating rate, β, can be related by Eq. (2).

$$\beta = \frac{dT}{dt} \tag{2}$$

where: β is the heating rate, and T is the absolute temperature.

The reaction rate constant, k, can be presented by Eq. (3).

$$k(T) = k_0 exp(\frac{-E_a}{RT}) \tag{3}$$

where: k is the reaction rate constant, k_0 is a pre-reaction rate constant, E_a is the activation energy, and R is the universal gas constant.

Therefore, by substituting Eq. (2) and Eq. (3) in Eq. (1):

$$\frac{d\alpha}{dT} = \frac{1}{\beta} k_0 exp(-\frac{E_a}{RT}) f(\alpha) \tag{4}$$

2.1 Determination of kinetic parameters

Kinetic parameters (E_a and k_0) are determined using various techniques classified according to i) pyrolysis conditions, i.e., isothermal or non-isothermal, or ii) mathematical analysis of the experimental results, i.e., linear or nonlinear. In the current study, non-isothermal pyrolysis conditions were used since the experiments were carried out at various heating rates (10, 20, 30, 40 and 50°C/min). The linear mathematical analysis methods for experimental results analysis were selected due to their conventional application in the analysis of thermal devolatilisation of the solids. These methods allow the determination of a linear relationship between the kinetic parameters using mass loss data generated at various devolatilisation rates. Linear regression methods using experimental results are applied to determine coefficients of the linear relation. Typical non-isothermal and linear methods were considered in the present study, in particular, the iso-conversional methods of Friedman (differential approach) and Kissinger (integration approach) (Friedman, 1964; Kissinger, 1957). The mathematical methods were used in the analysis of the experimental results.

2.2 Friedman method

Taking a natural logarithm of Eq. (4), the linear relationship is employed in the equation and resulted in Eq. (5).

$$ln\left(\beta\frac{d\alpha}{dT}\right) = lnk_0 + lnf(\alpha) - \frac{E_a}{RT} \tag{5}$$

Plotting $ln\left(\beta\frac{d\alpha}{dT}\right)$ versus $\frac{1}{T}$ at given reaction progress, α, for various heating rates yields a straight line with slope $\frac{-E_a}{R}$. Activation energy can be obtained from this slope without knowing the reaction function f(α). The pre-exponential reaction rate constant, k_0, is the y-intercept of a straight line. Friedman method is one of the model-free iso-conversional method (Friedman, 1964).

2.3 Kissinger method

The Kissinger method is based on the determination of the temperature at the maximum devolatilisation rate of the rubber fractions (NR, BR or SBR), at various heating rates (Kissinger, 1957). In the Kissinger method, Eq. 4 is modified to Eq. (6).

$$ln\left(\frac{\beta}{T^2}\right) = ln\left(\frac{k_0R}{f(\alpha)}\right) - \frac{E_a}{RT} \tag{6}$$

A plot of $\ln\left(\frac{\beta}{T_{max}^2}\right)$ versus $\frac{1}{T_{max}}$ at various heating rates yields a straight line that allows for the determination of the activation energy, E_a, from the gradient of the straight line, $\frac{-E_a}{R}$. The y-intercept can be used to estimate the pre-exponential reaction rate factor. Similar to the Friedman method, Kissinger method is another example of a model-free iso-conversional method.

3. EQUIPMENT AND METHOD

TT and PCT crumbs (steel- and fabric-free) samples with the particle size range of 0.6 to 0.8 mm were obtained from the local waste tyre recycler. A slightly adjusted version of ASTM E1131-08 (with X = 275°C) was used to determine proximate analysis. Additionally, the rubber composition of the crumb was determined using a procedure described by (Danon & Gorgens, 2015). The results of the proximate and elemental analysis are shown in Table 1.

The proximate analysing was done using the TGA which was carried out on both tyre types TT and PCT. To determine the mass loss as temperature increases, the maximum temperature was set to 800°C, nitrogen (N_2) and oxygen (O2) were used as a carrier gas to trace devolatilisation profile and to determine ash content, respectively. A TA 60WS model TGA (Shimadzu™, Kyoto, Japan) was used for all the experiments. The calculations were done using the Standard Test Method for Compositional Analysis by Thermogravimetry, designation: ASTM E 1131-08, (2008). All the experiments were conducted in duplicates to ensure the repeatability of the results.

4. RESULTS AND DISCUSSION

4.1 Thermogravimetric analysis

Shown in Figure 1 is the solid residual for truck tyre and passenger car tyre. The reported char values are the average estimated from all the heating rates (10 to 50°C/min). From the thermogravimetry (TG) profiles in Figure 1 it can be observed that, regardless of the heating rate, the profiles converge at the same value of solid residual at the end of the pyrolysis TG. This indicates that the heating rate effect on the mass and energy transfer is relatively insignificant and can be ignored in the analysis of the present study. Moreover, the observed char residuals, 36.2 wt. % for TT and similarly, the PCT at 36.2 wt. % (Table 2), is comparable to the previously reported work (Mkhize et al., 2016). This is not surprising since the fixed carbon mainly in the form of the carbon black added to the tyre compounding is likely to be the same for both the TT and the PCT. Therefore, the next step is the characterisation of the volatile matter, mainly rubber fraction of the two tyre types.

TT and PCT first derivative thermogravimetric (DTG) and reaction progress profiles on the pyrolysis temperature basis at various heating are illustrated in Figure 2. The shape of the curves of the TT is significantly different from that of the PCT. The observed distinction between the devolatilisation profiles for the two tyre types is critical, considering that the compounding process between the two tyre types is relatively different. Therefore, in order to relate the tyre compounding process, devolatilisation profile peaks from the two tyre types are analysed. Two distinctive peaks are observed in the PCT, mainly at higher heating rates, whereas, in the TT profile only one peak was observed and became firmly one peak at higher heating rates. This can, therefore, be attributed to the fraction composition difference of the rubber fractions from each tyre type. From previous investigations, it has been reported that TT rubber fraction mainly consists of natural rubber, while in the PCT, the rubber fraction is equally distributed between the NR and synthetic rubber (butadiene, BR, or styrene-butadiene, SBR). The equal distribution between the two rubber types in the PCT is supported by the identical peaks shown in Figure 2, particularly at a heating rate of 50°C/min. In the wake of new interests in the production of valuable products in addition to the energy recovery, the design of the pyrolysis system is depended on the characteristics of the feedstock. For example, the two distinguishable peaks, indicating maximum devolatilisation rate within a wide range of temperature (around 100°C different), is an indication of the potential of step-wise temperature devolatilisation of rubber fractions. As for the TT, the shape of the curves at a lower heating rate show two peaks, i.e. the main devolatilisation peak at low temperature and a small peak (shoulder) at the high temperature. The shoulder (relative small additional peak compared to the main large peak) on the high-temperature side of the peak disappears as the heating rate increases. In the earlier studies (Danon et al., 2015), this has been attributed to the elimination of the secondary reactions (primary products cracking) at higher heating rates, while promoting devolatilisation of the rubbers. Moreover, as observed in the previous studies, the effect of the heating rate on the devolatilisation peak temperature is characterised by an increase in the temperature associated with the maximum devolatilisation rate (in both the TT and PCT) as the heating rate increases.

4.2 Reaction Progress

The reaction progress of both the TT and PCT is illustrated in Figure 2. The reaction progress profile of the TT matches that of the PCT. However, at a refined view, the gradient of the profile is steeper for the TT as compared to the PCT. The low rate of the devolatilisation of the PCT (i.e., less steep gradient in the PCT is attributed to the two peaks as observed from the derivative thermogravimetric profiles. This also indicates not only the delay in the devolatilisation of the PCT compared to the TT but sufficient time for fractional devolatilisation of the rubber fractions. In order to show a better illustration of the fractional devolatilisation, the plots of the different tyre types activation energy, E_a as a function of time are critically analysed.

4.3 Friedman method

Figure 3 illustrates the estimated values of the TT and PCT linear plots, apparent activation energy, E_a, and pre-exponential factor k_0, as a function of reaction conversion with the differential-based, iso-conversional Friedman method. The linear plots of the TT have an identical gradient (from a plot of $\ln\left(\beta\frac{d\alpha}{dT}\right)$ versus $\frac{1}{T}$) as illustrated in Eq. 5. In other words, the activation energy (E_a) is independent of the reaction progress, therefore, a constant E_a with the progress of devolatilisation.

The linear plots for PCT differ from that of the TT in that there are two sets of the linear plots, i.e., the first set from reaction progress of 0.1 to 0.5 and the second set from reaction progress of 0.5 to 0.9. Each of these sets has a maximum peak at different reaction progress, i.e., maximum E_a at reaction progress of 0.4 and 0.8, respectively. Alternatively, the peak dependence on the reaction progress is expressed in E_a and/or k_0.

The TT activation energy (E_a) as a function of the reaction progress has a form of the peak with the maximum value at a reaction progress 0.7. For the rest of the reaction progress, the E_a remained relatively constant at approximately 166.3 kJ/mole. The E_a for PCT was estimated at 177.4 kJ/mole, slightly higher than that of the TT. The higher E_a for PCT compared to TT reflects a slight difference in the polymer composition. This can be attributed to the significant amount of NR in the PCT compared to the TT content of the BR and SBR. Therefore, the minimum energy required to devolatilise each NR, BR or SBR fraction differs. Such difference in the E_a may suggest and be supportive of the

TABLE 1: Proximate and elemental analysis of truck tyre and passenger car tyre.

Proximate analysis (wt. %)					
Tyre type	**Moisture**	**Oils**	**Volatile matter**	**Fixed carbon**	**Ash**
TT	0.6	5.6	56.0	30.0	7.8
PCT	0.6	5.4	55.9	29.9	8.2
Elemental analysis (wt. %)					
Tyre type	**Carbon**	**Hydrogen**	**Sulphur**	**Nitrogen & Oxygen[a]**	
TT	81.1	7.4	1.9	9.7	
PCT	79.3	7.5	1.2	12.0	

[a] *By difference*

TABLE 2: Kinetic parameters estimated using iso-conversional methods for PCT and TT.

Parameter		TT	PCT	PCT	
				PCT peak 1	**PCT peak 2**
Residue char (wt. %)		36.2	36.2		
E_a ($kJ.kmol^{-1}$)	Friedman	166.3	177.4	131.7	139.4
	Kissinger	128.5	177.7	136.1	165.0
k_0 (s^{-1})	Friedman	1.97×10^{12}	1.04×10^{13}	4.01×10^{8}	4.92×10^{7}
	Kissinger	4.20×10^{8}	8.91×10^{10}	1.48×10^{9}	1.22×10^{10}

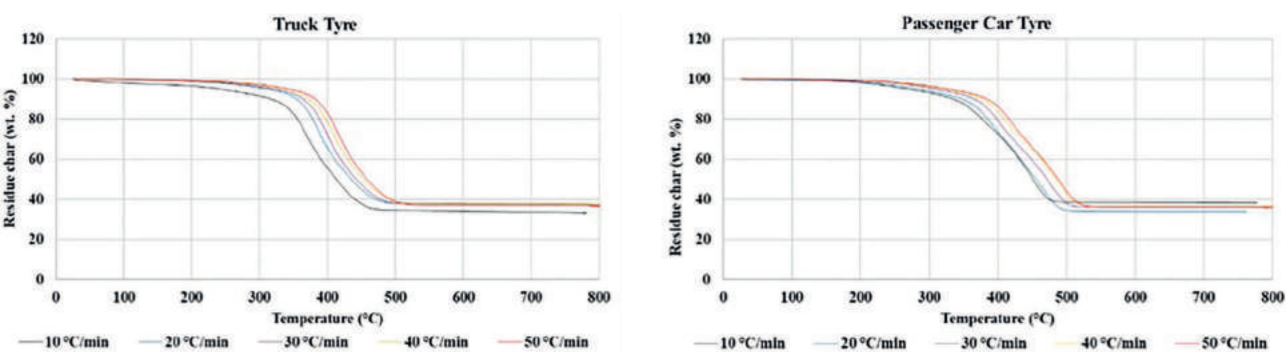

FIGURE 1: Percent devolatilisation of the various waste tyres showing residual char.

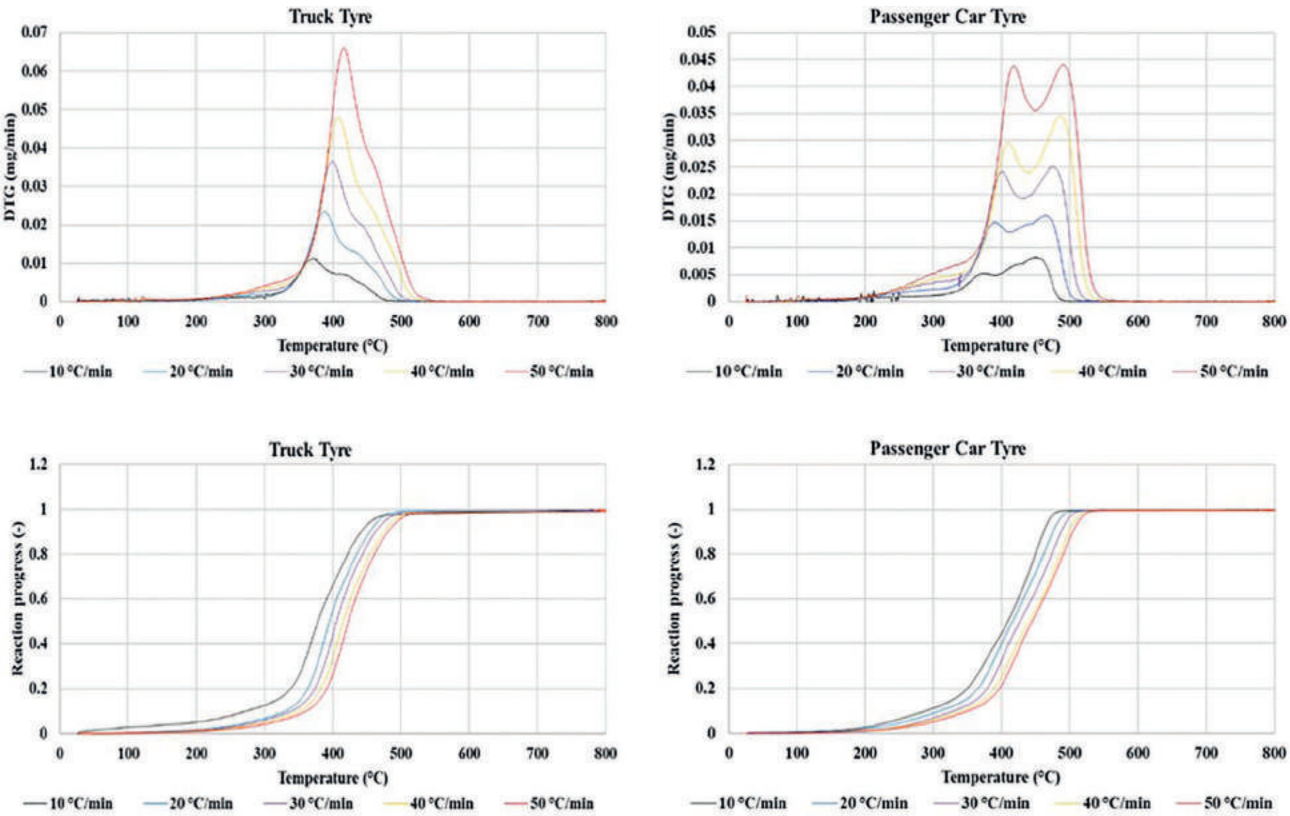

FIGURE 2: Profiles of the derivative thermogravimetric (DTG) and reaction progress of the waste truck tyre.

proposal of stepwise pyrolysis or selective devolatilisation of the rubbery components of the tyres. Typical ratios of NR to synthetic rubber (SBR and BR) are 2:1 and 4:3 for TT and PCT, respectively (Mui et al., 2004). Lopez et al. (2009) reported that the composition of a waste tyre was 60 wt. % rubber, of which NR and SBR were 29.95 and 29.95 wt. %, respectively (Lopez et al., 2009). Waste tyre crumb used in their investigation was steel and fabric free PCT. However, the rubber ratio was not reported. Another study conducted by the same group used TT waste tyre consisting of 58.89 wt. % NR representing the entire rubber content of the waste tyre (Lopez et al., 2009). The rubber composition of the waste tyre reported by Danon and Görgens (2015) was 64 wt. % NR and 36 wt. % synthetic rubber (SBR and BR) (Danon & Gorgens, 2015).

Finally, the pre-exponential factor, k_0 is reported in Figure 3. A similar trend as in the E_a on both the TT and PCT tyre types are observed with the k_0. The average k_0 for the TT was estimated at 1.97×10^{12}, while for PCT it was estimated at 4.20×10^{8}. If the definition of the frequency of molecules collision is adopted, it means that for the TT, the frequency of the molecules is significantly higher than the frequency of the PCT. Also, to be noted is the peaks at certain reaction progress, i.e., at 0.7 for TT and 0.4 and 0.8 for PCT, were observed. This is an indication that the kinetic parameters suggest that TT devolatilisation is one step, while PCT volatilization is a two-step process.

4.4 Kissinger method

The Kissinger method as an iso-conversional method provides an alternative to Friedman method. It is based on an integration approach and can be used to estimate E_a for both TT and PCT devolatilisation. The Kissinger plots for TT and PCT are shown in Figure 4. Using the Kissinger method plots, the E_a for TT was estimated at 128.5 kJ/mole while for PCT was at 177.7 kJ/mole. A significant difference between the E_a from the Friedman method and the Kissinger method has been observed in the TT at around 38 kJ/mole, whereas for PCT the difference was insignificant (around 0 kJ/mole). This was expected since the TT peak has the shoulder which disappears as the heating rate is increased, i.e., the shoulder is absorbed to the main peak which affects the estimation of the E_a. In the PCT profile, the two distinguishable peaks from the devolatilisation allowed separation of the different tyre rubber fractions as well as estimation of the E_a. This is also due to the fact that the two peaks from the PCT are almost identical and well separated.

An overlay of the TT and PCT derivative thermogravimetric profiles is illustrated in Figure 5. Since identical rubber components are likely to behave similarly under thermal devolatilisation regardless of the tyre type of origin, this constitutes an acceptable standard to associates kinetic parameters E_a and k_0 with the propose rubber component from the different tyre types. The PCT first peak corresponds to the TT main peak, while the second PCT peak corresponds to the shoulder of the TT main peak. As the heating rate is increased, both the TT main peak and PCT first peak's maximum devolatilisation temperature increases. Complete disappearance of the TT shoulder allows a

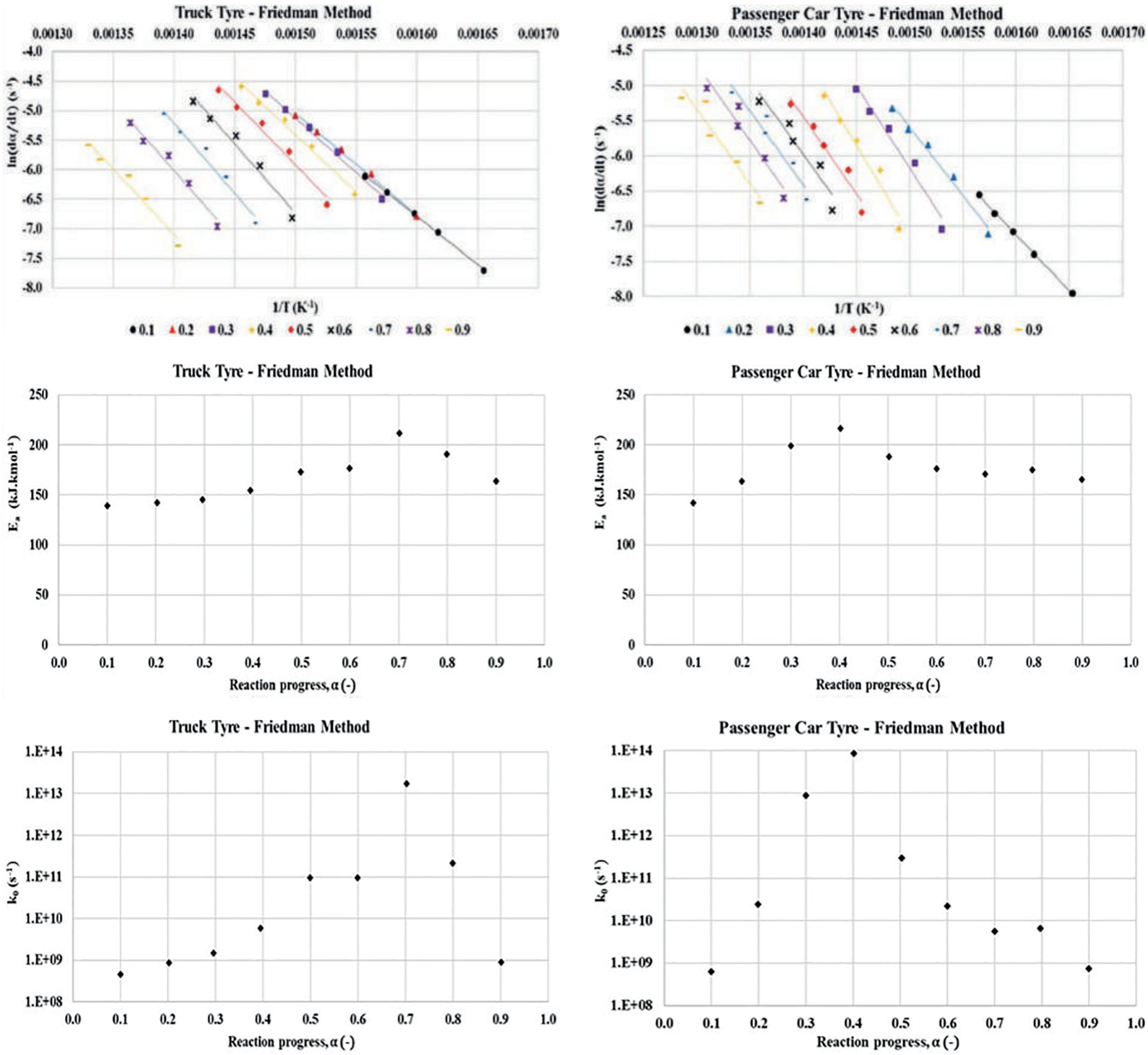

FIGURE 3: Iso-conversional Friedman method linear plots, activation energy, E_a, and pre-exponential reaction constant.

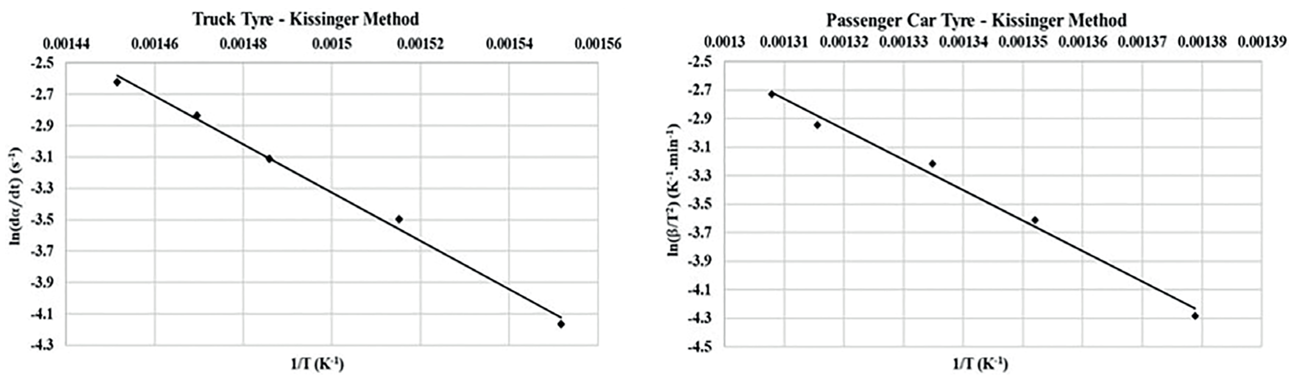

FIGURE 4: Iso-conversional Kissinger method plot for truck tyre and passenger car tyre at various reaction progress.

better illustration of the second peak from the PCT. Overall, the intensity of the TT peaks (except the shoulder) is higher than that of the PCT peaks. This can be attributed to the difference in the volatility of the rubber fractions of these types of tyres.

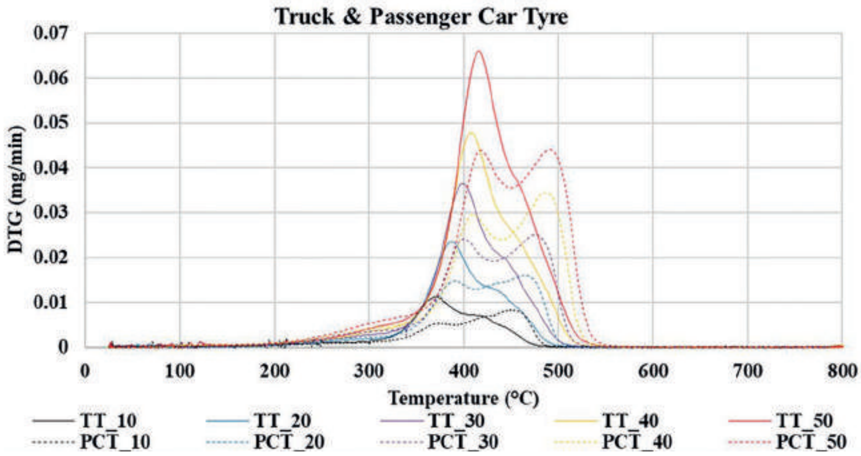

FIGURE 5: Overlay profiles of the derivative thermogravimetric (DTG) waste truck tyre and passenger car tyre.

4.5 Fractional devolatilisation of the PCT

An overlook of the potential of fractional devolatilisation of the PCT rubber is interesting since relatively two distinctive peaks were observed. Figure 6 shows the PCT fractional derivative thermograms and iso-conversional Friedman method analytical plots. It should be noted that in the ideal state, the two peaks have some effect on each other. However, for concept illustration purposes, the interaction effects of the peaks may be relatively less when compared to the potential benefits, such as product selective devolatilisation of heterogeneous materials.

The two distinctive peaks for PCT devolatilisation can easily be illustrated as two different plots. This implies that the two peaks can be model separately as was shown and illustrated by Danon and Görgens (2015). A significant improvement in the E_a as a function of α plots was observed when the PCT peaks were separately analysed or modelled, see Figures 3 and 6. The E_a versus α plots are better and consistent, each with a single peak shaped distribution. The average E_a was estimated at 131.7 kJ/mole on the peak 1 while the second peak 2 was estimated at 139 kJ/mole. The E_a for peak 2 is higher than that of peak 1 by 7.7 kJ/mole indicating a potential for the step-wise pyrolysis of the PCT. This can also be used as a basis for fractional recovery of the valuable products from waste tyre valorisation, most particularly as an indication that slightly more energy is required to devolatilize BR/SBR fraction. The parameter k_0 was at 4.01 x 10^8 and 4.92 x 10^7 for peak 1 and peak 2, respectively.

A similar trend on both the E_a and the k_0 was observed with the Kissinger method. The E_a for peak 1 and peak 2 were 136.1 and 165.0 kJ/mole, respectively. The difference between the E_a of peak 1 and that of peak 2 using the Kissinger method is substantial at 28.9 kJ/mole compared to the one from Friedman method. This may be a good support for the fractional pyrolysis of the PCT. However, the difference between the E_a from Friedman method and Kissinger method is quite significant at 25.5 kJ/mole. The k_0 is also comparable with the Friedman method generated value, i.e., for peak 1, k_0 of 4.01 x 10^8 and 1.48 x 10^9 were obtained from Friedman method and Kissinger method, respectively. Figure 6 shows the Kissinger method iso-conversional method for peaks 1 and 2 E_a estimation. The parameters for both the TT and PCT are illustrated in Table 2.

5. CONCLUSIONS

The increase in the heating rate up to 50°C/min increased the peak temperature of both the TT and PCT maximum devolatilisation temperature. The TT devolatilisation is characterised by one main fraction of the tyre rubber, which when the devolatilisation heating rate is increased, the peak shoulder disappears. The main peak from the TT profile was attributed to the natural rubber, the main rubbery fraction of the TT. Two distinctive peaks were observed in the devolatilisation of the PCT, peak 1 was sought to be the natural rubber (or polyisoprene), whereas peak 2 was associated with synthetic rubber (i.e., BR and SBR). The observation may suggest that there is a high potential for stepwise pyrolysis of the fractions from the tyre rubber, particularly PCT. The step-wise pyrolysis of the PCT is more favourable at a higher heating rate since the separation between two distinctive peaks is significant. Selective pyrolysis of the tyre rubbers may be beneficial to the recovery of the valuable chemicals from the waste tyre or other rubbery materials. The difference in the kinetic parameters of the components of the waste tyre rubbery fraction substantiates the difference in the reaction pathways. Therefore, waste tyre pyrolysis reactor design proposal should consider the distinctive behaviour of various rubbery fraction in the tyres during thermochemical devolatilisation of waste tyres. Moreover, since various rubber components devolatilisation was observed, the mirror various products can be observed and targeted for recovery.

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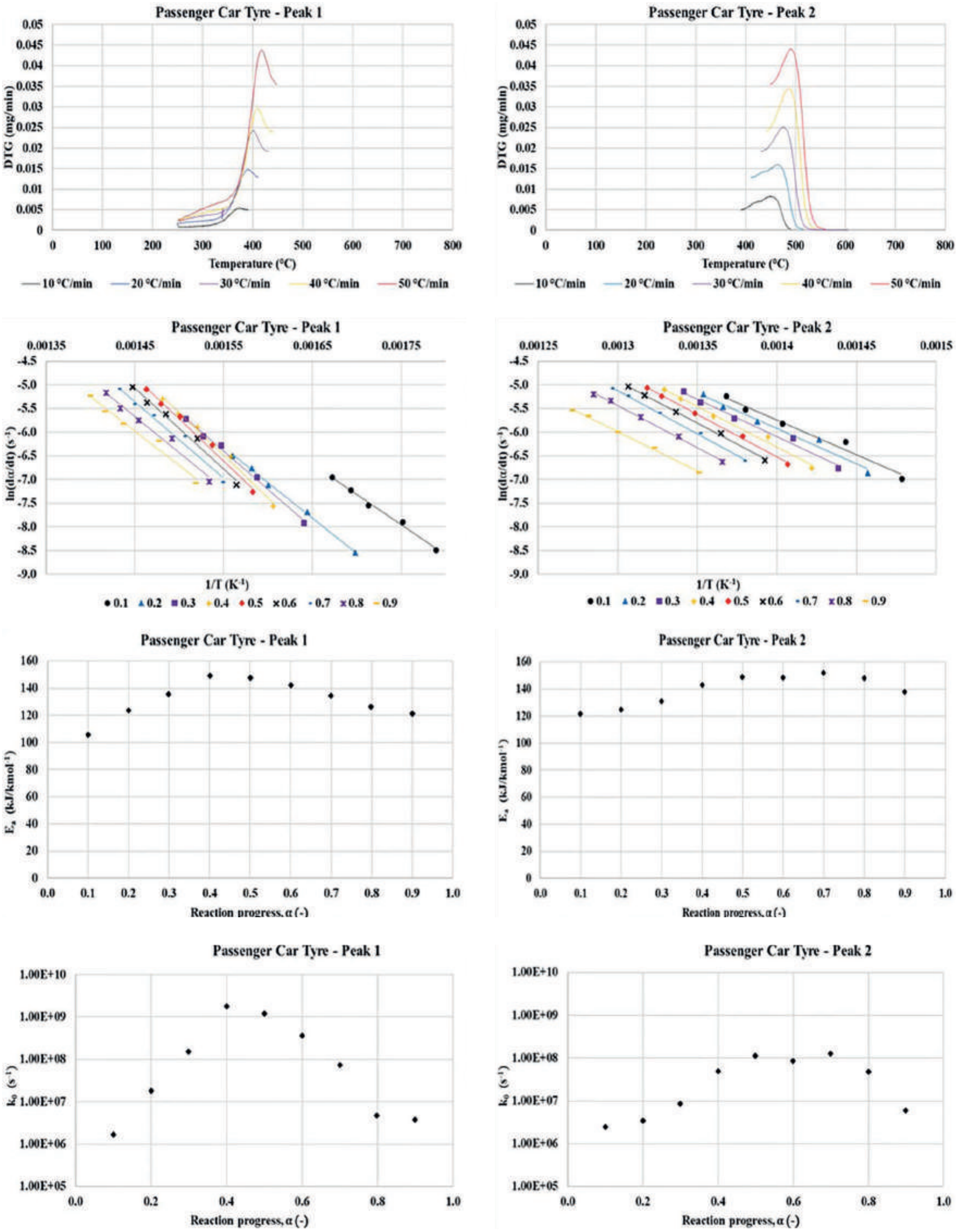

FIGURE 6: Fractional derivative thermograms and Iso-conversional Friedman analysis plots for the PCT.

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MINERAL WOOL WASTE IN AUSTRIA, ASSOCIATED HEALTH ASPECTS AND RECYCLING OPTIONS

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ABSTRACT

Mineral wool products are man-made vitreous fibres that are used as thermal and acoustic insulation materials and as substrates for horticulture. Mineral wool waste is generated from demolition activities by the building and construction industry. Unfavourable mechanical properties, such as low compressibility, elastic behaviour, high volume and low bulk density, cause problems in landfills when mineral wool waste is disposed of. Mineral wool waste with a certain content of carcinogenic fibres is classified as hazardous waste type 31437 g "Asbestos Waste, Asbestos Dust" in Austria, since some characteristics of such fibres are similar to those of asbestos fibres. An exception is those mineral wool materials that have been tested to be noncarcinogenic due to their characteristics of biological solubility or geometrical dimension. Such noncarcinogenic mineral wool waste is classified as non-hazardous waste type 31416 "Mineral fibres". Generally, it can be assumed that most of the industrial producers of mineral wool in the EU have not been producing carcinogenic material since 1998; however, carcinogenic mineral wool material has not yet been banned in Austria. Therefore, a segregation between so-called "old" and "new" mineral wool material is not necessarily possible. The medical aspects of mineral wool products are still controversial. The International Agency for Research on Cancer (IARC) evaluated mineral wool (glass wool and rock wool) as "possibly carcinogenic" in 1988 but revised this evaluation to "inadequate evidence in humans for the carcinogenicity" in 2002. Fibrous dusts that reach the alveolar region of the lungs undergo a congruent or incongruent chemical dissolution process. Alveolar macrophages ingest the intruded fibres and fulfil anti-infection and clearance functions. Biosolubility is a key property of this process. The recycling of mineral wool waste has not yet been performed in Austria due to economic inefficiency, technical problems and suspected health issues. However, some recycling and processing options already exist; other options are investigated in the project RecyMin, which compares different concepts with respect to environmental and economic criteria.

1. INTRODUCTION

Mineral wool is a man-made vitreous fibre that is primarily produced from glass, igneous rocks and slags. Mineral wool products are used as insulation material and in horticulture.

Mineral wool waste causes problems in waste management due to its high volume and low density, e.g., the lack of stability in the landfill body. Currently, most of the mineral wool waste in Austria is landfilled, and to date, no recycling is carried out. These problems have become increasingly urgent because of the higher quantities of this waste stream in recent years due to the separate collection of mineral wool waste and the higher amounts used for insulation.

Mineral wool waste is also under observation because of possible health aspects. In general, fibres of mineral wool can be released into the environment due to the production process, the usage of the product and the demolition and dismantling of buildings containing mineral wool products, which can cause health difficulties because of airborne respirable fibres with low biosolubility.

Mineral wool materials that have been produced with certain quality labels, such as the German RAL quality label for mineral wool products (approximately from 1998 and later), fulfil current requirements of biological solubility and may therefore not be listed as carcinogenic. All other mineral wool materials, at least those produced before 1998, might have a lower biosolubility and are classified by the

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European Union as “possibly carcinogenic to humans” (EU 2008). Therefore, it is very important to distinguish these two types of mineral wool materials due to hazardousness. The differentiation between glass wool and rock wool is even more important for the recycling process.

In this review, we provide an overview of the technical, health and legal aspects of mineral wool waste treatment.

2. MATERIALS AND METHODS

This article summarises the challenges of waste management, technical issues, and health aspects and discusses the legal aspects of mineral wool waste in Austria.

The literature research was performed by reading and summarising reviewed articles, books, legislations, guidelines and standards. Sources were selected based on their importance for the article in terms of waste- and health-related aspects as well as the concerning legislations.

A patent search has been performed in addition to the literature research to gather patent information regarding recycling and processing options for mineral wool waste and man-made vitreous fibres in general. Expert discussions with landfill engineers, waste collectors, waste processors and the mineral wool industry have been carried out to provide professional experience regarding the challenges of mineral wool waste in Austria.

3. RESULTS AND DISCUSSION

3.1 Technical aspects

Man-made vitreous fibres consist of fibrous inorganic substances. They are divided into glass fibres, glass wool, rock wool, slag wool and ceramic fibres. Mineral wool comprises glass wool, rock wool and slag wool (DGUV 2014). The term “wool” describes an omnidirectional accumulation of fibres with different lengths and diameters (DIN EN 1094-1 2008).

3.1.1 Production of mineral wool

Mineral wool is mainly produced from glass, slags and igneous rocks such as basalt and diabase. Waste glass is added as a secondary raw material (IARC 1988, BBSR 2011). The production of mineral wool products can be divided into the following steps: supplying the raw materials and energy sources, melting in a furnace, fiberization and collection, primary layer formation and finishing (Sirok et al. 2008).

Figure 1 shows the process of mineral wool production. The raw material is melted in a cupola furnace, and coal is mostly used as an energy source. The fiberization of the molten raw material is usually executed on spinner wheels (Sirok et al. 2008). The fibres are then collected in the wool chamber (Sirok et al. 2013). The primary layer is formed in the wool chamber and then folded by a pendulum. The stack of mineral wool is then brought to the required thickness and enters the curing chamber where the previously applied resin hardens (Sirok 2008). Following the preceding steps, the fibres are formed into different products, such as blankets, mats and other product types (IARC 1988).

3.1.2 Application of mineral wool products

Mineral wool is used for a wide range of applications, such as thermal and acoustic insulation material, fire prevention (DGUV 2014) and horticulture (TRGS 521 2002). Mineral wool products are primarily used at temperature ranges up to 300°C (TRGS 619 2013) but can also be applied at temperatures up to 600°C (DIN EN 1094-1 2008).

3.1.3 Waste-related aspects

Amount of mineral wool waste

A survey study on mineral wool waste in Europe (Väntsi et al. 2014) estimates that there are approximately 2.5 million tons of mineral wool waste produced in the European Union per year but the study notes a lack of data.

In Austria, an amount of 20,000-30,000 t/a of mineral wool waste was estimated by the Austrian Economic Chamber in 2018.

Challenges in practice

The management of mineral wool waste is technically challenging due to its high volume, low bulk density, high elasticity, poor compressibility and the consequential lack of stability in a landfill. Additionally, legal challenges arise from the distinction between old and new mineral wool waste. In contrast, during the collection of mineral wool, no distinction between glass wool and rock wool is made, which would be necessary for many recycling options.

The transport of mineral wool waste to a recycling plant or landfill is associated with high economic and ecological burdens due to its low bulk density.

The knowledge of mineral wool not being carcinogenic does not solve the problem from a waste management point of view. Mineral wool waste that occurs due to the dismantling of a building is primarily not relatable to a certain year of production or to a certain industrial producer. Because of the precautionary principle, this mineral wool waste must be classified as possibly carcinogenic or “old” mineral wool and therefore as hazardous waste (Wirtschaftskammer Österreich 2018), regardless of whether this is the case or not. For disposal, this mineral wool must be gathered in hermetically sealed packages, such as big bags, which results in an unstable landfill body.

No testing methods to analyse mineral wool waste at the construction site for its possible hazardous property (HP7, carcinogenic) have been developed yet concerning its biological solubility since the geometrical characteristics are unclear. Therefore, an unknown amount of mineral wool waste is wrongly assigned to the hazardous waste code. As a result of this situation, it is impossible to gather precise data on the specific amounts of hazardous and non-hazardous mineral wool waste in Austria.

3.1.4 Recycling options and patents

There has already been some research on the recycling options for mineral wool waste (Öhberg 1966) (Balkevicius et al. 2007) (Holbek 1987), but this only concerns mineral wool production waste (Väntsi et al. 2014) and not mineral wool waste from the demolition of buildings. Müller et al. (2009) developed a recycling method for slagging mi-

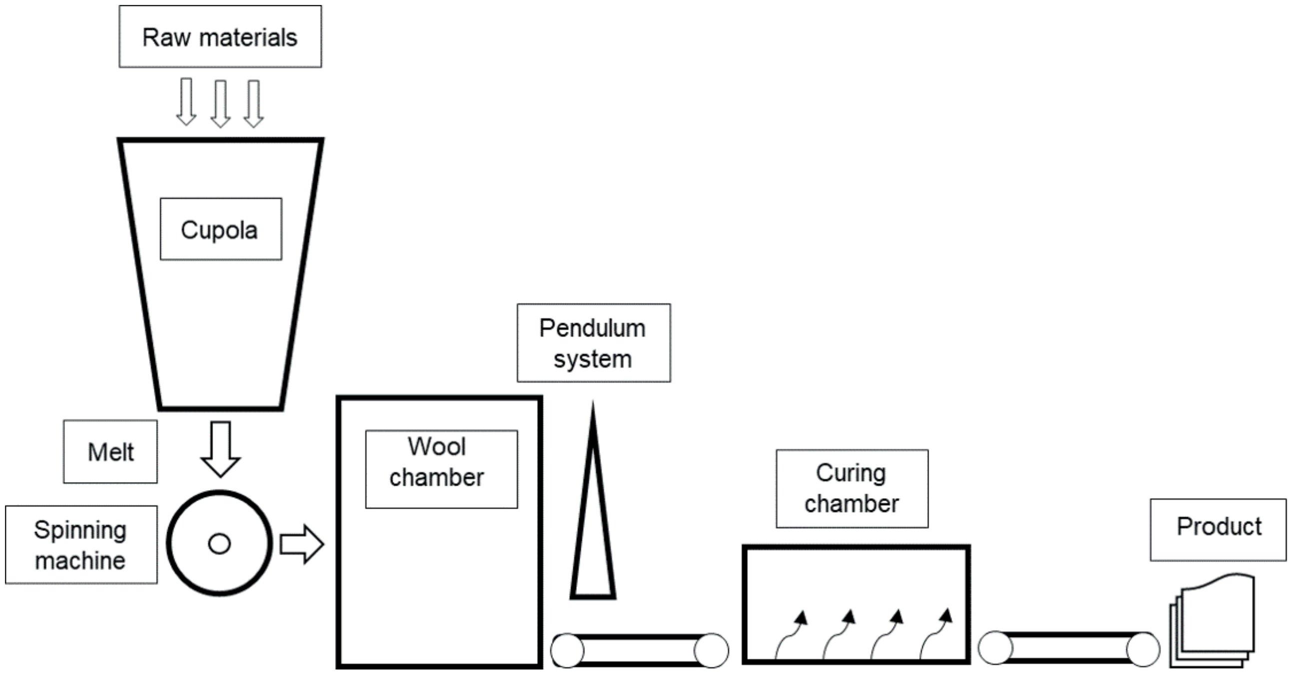

FIGURE 1: Schematic figure showing the production process of mineral wool (modified after Sirok et al. 2008, Institut Bauen und Umwelt e.V. 2012).

neral wool waste at the laboratory scale using a specific microwave technology. The slags created in this process might be used as products in the future as the hazardous property, i.e., the fibrous character in combination with low biosolubility, is destroyed.

The patented "re:cyKMF" process generates backfill material using mineral wool waste, binding agent suspension and water (Gröper & Lack 2016).

A mobile press to agglomerate waste of man-made vitreous fibres was patented by Wurzer Umwelt GmbH Eitting in 2016. The vehicle contains a compaction unit and is therefore able to execute the pressing of mineral wool waste where it accrues and to reduce the high volume of the mineral wool waste (Patent EP 3 168 037 A1). The agglomeration of mineral wool waste is an important preparation step prior to optimised disposal or recycling.

Project RecyMin

The project RecyMin focuses on mineral wool waste in Austria. It aims to develop innovative landfilling solutions and the recycling of mineral wool waste in backfilling and in the cement and glass/rock wool industries (Sattler et al. 2019).

The fundamental research approach is based on a waste management survey. Through a combination of waste management, process engineering and material science methods, a concept for the recycling of mineral wool waste will be developed. This concept, depending on logistical, economic and technical circumstances, includes an innovative disposal method, recycling through backfilling, in the cement industry and recycling in the mineral wool industry under consideration and the evaluation of ecological, economic and health aspects (Figure 2) (Sattler et al. 2019) (Vollprecht et al. 2019).

The challenge of varying and unknown potential for recycling and possible hazardousness of mineral wool waste should be solved by a combination of methods, including waste management life cycle assessment (LCA) and chemical, mineralogical and morphological material characterisation of waste in the laboratory. The processing of mineral wool waste aims to enhance the properties of mineral wool waste for landfill technology, recycling options and health characteristics. Mineral wool waste can be disposed of in the form of briquettes, as the low density and poor compressibility are improved by preceding processing by a briquetting press (Sattler et al. 2019) (Vollprecht et al. 2019). Using processed mineral wool waste as a backfilling material is another possible application (Höllen et al. 2015). RecyMin evaluates another application to recycle mineral wool waste in the cement industry. In the case of recycling in the mineral wool industry, the consequences on the procedural properties of the melt are investigated, and additives applied for the compensation of chemistry are used. The consequences of the proposed processes on waste management systems are investigated through an ecological-economic evaluation and summarised in a waste management context (Sattler et al. 2019) (Vollprecht et al. 2019).

3.2 Health aspects

3.2.1 Waste-related aspects

Health implications due to exposure to man-made vitreous fibres might be the irritation of the skin and mucous membranes, as well as health effects on the breathing organs (Valic 2012).

Due to their composition, synthetic vitreous fibres are degraded in the environment only under acidic or alkaline

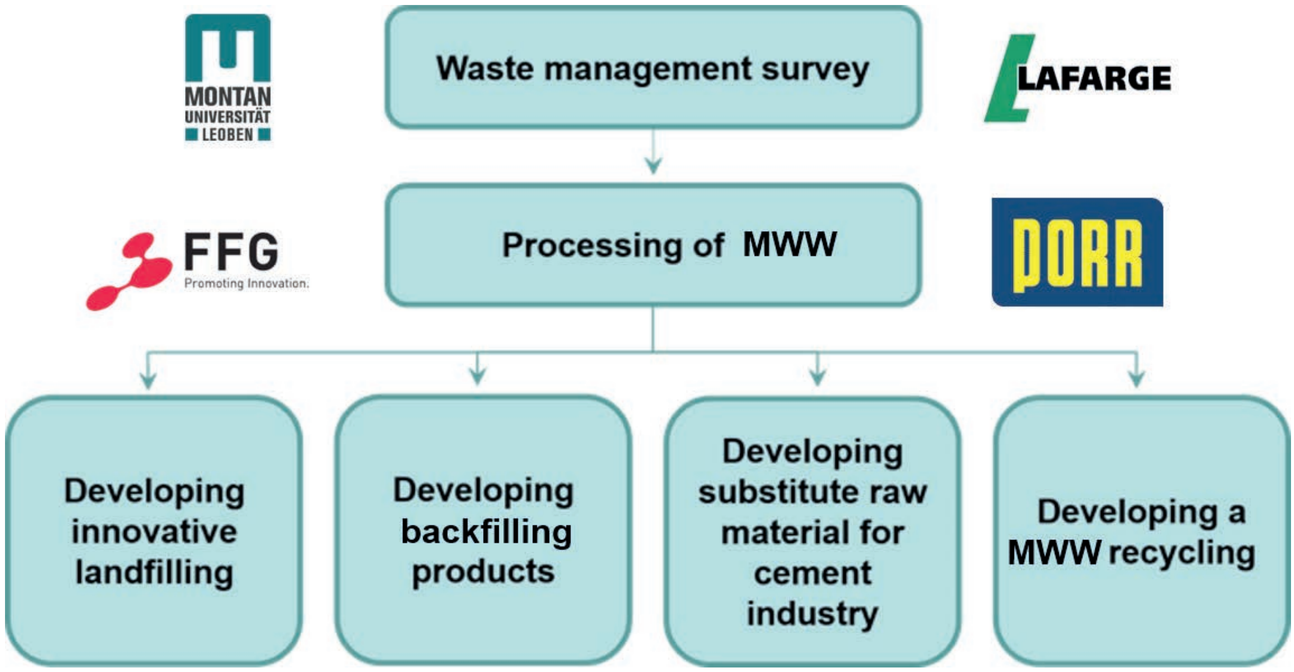

FIGURE 2: Schematic figure of the project RecyMin; MWW=mineral wool waste.

conditions by dissolution of the silicate network. Hence, the fibres can remain in soil and water over a long time. In particular, people working on construction sites (demolition, dismantling, building maintenance and repair) or in the fibre production industry can be exposed to synthetic vitreous fibres to a high degree (ATSDR 2004).

It is assumed that negative health effects are determined by certain fibre characteristics:

- Fibre length;
- Fibre diameter;
- In vivo durability and persistence (IARC 1988).

Fibres with dimensions of a diameter < 3 μm and length > 5 μm and a length to diameter ratio ≥ 3 : 1 can be deposited in the alveolar region of the lungs (IARC 1988); these fibres are called WHO fibres or critical fibres because the World Health Organisation (WHO) defined the respirable fibre dimensions in 1988.

Distinguishing between rodents and humans, in general, a larger amount of long respirable fibres are able to penetrate into human lungs compared to those of rats (Dai & Yu 1998).

3.2.2 Observations in humans

The inhalation of man-made vitreous fibres causes deposition of these fibres in the nasal, oral sections and upper lung airways at first. They are mostly transported to the stomach by a layer of mucous in the throat (ATSDR 2004). If respirable fibres are present, they can reach the alveolar region (Skinner et al. 1988) where they are exposed to the acidic intracellular environment with a presumed pH of 4.5-5 in the phagolysosome (ATSDR 2004) and undergo chemical dissolution or leaching processes due to the macrophages. The macrophages ingest the intruded fibres to fulfil anti-infection and clearance functions (phagocytosis). During phagocytosis, alveolar macrophages produce oxidising free radicals of many materials. They transport the absorbed materials through the lymphatic system to the lymph nodes (Skinner et al. 1988). Fibres are actively eliminated simultaneously by phagocytic cells. The fibre lengths are the decisive criterion for partial or complete removal (Lundborg et al. 1995).

The deposition of fibrous particles can cause inflammatory responses (Skinner et al. 1988), alveolitis, bronchitis and potentially fibrosis (Lippmann et al. 1971).

Pulmonary fibrosis is caused by man-made vitreous fibres of low biosolubility that stay in the lungs over longer periods of time (ATSDR 2004).

3.2.3 Observations in animals

Animal experiments showed that the lung reacts to the inhalation of foreign material such as fibres with a process called pulmonary inflammation, where macrophages increase and then remove the fibres. With increasing amounts of fibres, macrophages can also clump together (ATSDR 2004). This process may release reactive oxygen and nitrogen species, triggering potential DNA damage and therefore may foster tumour development (Coussens & Werb 2002). Further animal studies showed that repeated inhalation of certain types of synthetic vitreous fibres can cause scar-like tissues in the lungs and the surrounding membrane, making breathing more difficult, which is called pulmonary fibrosis. Such fibres stay in the lung over a long period of time and are therefore called durable or biopersistent. In addition to durability, the dose and duration of exposure and the fibre dimension are significant factors fostering pulmonary fibrosis, lung cancer and mesothelioma, respectively (ATSDR 2004).

3.2.4 Historical development

The discussion about the health aspects of mineral wool products started in the 1970s. The results of several studies (IARC 1988, Pott & Friedrichs 1972, Stanton et al. 1977) raised the suspicion of mineral wool being possibly carcinogenic (Draeger 2015). In 1988, the International Agency for Research on Cancer (IARC) published a monograph that reviews the carcinogenic risks to humans caused by man-made mineral fibres (IARC 1988). The IARC classified mineral wool (glass wool and rock wool) as "possibly carcinogenic" in this monograph based on epidemiological data and animal experiments (IARC 1988).

An IARC evaluation is executed as follows: the evaluations of the evidence of cancer in humans and the evidence of cancer in experimental animals are performed separately. The terms "sufficient evidence", "limited evidence", "inadequate evidence" and "evidence suggesting lack of carcinogenicity" are used. Other relevant data regarding the current evaluation are then considered. An overall evaluation is subsequently performed that implements the weight of evidence from the studies in humans and experimental animals as well as additional data (Baan & Grosse 2004).

Since 1988, there have been more long-term investigations, and these data were re-evaluated in the IARC monograph volume 81 of 2002 (IARC 2002).

In this monograph, it is evaluated that:

- There is inadequate evidence in humans for the carcinogenicity of glass wool;
- There is inadequate evidence in humans for the carcinogenicity of rock (stone) wool/slag wool;
- There is limited evidence in experimental animals for the carcinogenicity of insulation glass wool;
- There is limited evidence in experimental animals for the carcinogenicity of rock (stone) wool (IARC 2002).

The results of the evaluation in 2002 for carcinogenicity in humans are based on epidemiological information (Baan & Grosse 2004).

The manufacturers of mineral wool products responded to evaluations and founded the Joint European Medical Research Board (JERMB) in 1975. They started discussions about appropriate testing procedures and biosolubility with the World Health Organisation and founded umbrella organisations of mineral wool producers in Europe (EURIMA) and North America (NAIMA) (Draeger 2015).

Due to changes in industrial production from approximately 1996 onwards, mineral wool products with higher biosolubility have been produced (Kropiunik 2004), which would constitute the so-called "new mineral wool products". Biosolubility was not tested before that time. The differences between the chemistry of old rock wool and new rock wool products have been examined by Wohlleben et al. (2017). They found that most of the new rock wool products are high in alumina and low in silica content. Dissolution tests were conducted for 32 days at pH 4.5 and pH 7.4 with and without binder and at various flow rates. The removal of the binder accelerated the dissolution of the fibre. Size fractions of old mineral wool products and new mineral wool products were measured and showed that the respirable fraction of new mineral wool is low, but not less than that in old mineral wool (Wohlleben et al. 2017).

3.3 Legal aspects

3.3.1 Product regulations

The classification for carcinogenicity of a product is described in Regulation (EC) No 1272/2008 of the European Parliament and of the Council; therefore, several hazard categories for carcinogens are defined. Mineral wool with a content larger than 18% per weight of alkaline oxides and alkali earth oxides falls into the category of "suspected human carcinogens". To prove that this classification does not apply, the Note Q and Note R have to be fulfilled. The mineral wool product can be placed on the market if one of the four in vivo tests of Note Q on the one hand or Note R on the other hand is complied.

Note Q and Note R are defined as follows:

"Note Q:
The classification as a carcinogen need not apply if it can be shown that the substance fulfils one of the following conditions:
- *a short term biopersistence test by inhalation has shown that the fibres longer than 20 μm have a weighted half-life less than 10 days; or*
- *a short term biopersistence test by intratracheal instillation has shown that the fibres longer than 20 μm have a weighted half-life less than 40 days; or*
- *an appropriate intra-peritoneal test has shown no evidence of excess carcinogenicity; or*
- *absence of relevant pathogenicity or neoplastic changes in a suitable long term*
- *inhalation test.*

Note R:
The classification as a carcinogen need not apply to fibres with a length weighted geometric mean diameter less two standard geometric errors greater than 6 μm. (EU 2008)".

Another testing method besides in vivo and in vitro tests is the "carcinogenicity index" (CI). The CI is a test based on the calculation of a formula that implies certain oxide contents of a sample and is only applied in Germany. Calculations must be performed using the following formula: CI = $Na_2O + K_2O + B_2O_3 + CaO + MgO + BaO - 2\ Al_2O_3$. If the CI is larger than or equal to 40, according to "Technische Regeln für Gefahrstoffe" (TRGS 905), mineral wool produced prior to 1998 can be classified as "not carcinogenic". The test has the advantage of being an inexpensive, simple and fast method. The disadvantage is the frequent misclassification, especially in the case of rock wool waste, because of the high Al_2O_3 content. Mineral fibres with high alumina contents tend to be classified as cancerogenic, although they often show high solubility in in vivo and in vitro tests. Fibres that passed the in vivo test and are classified as "not carcinogenic" might have a CI lower than 40 and should be classified as "carcinogenic" after the CI (Ausschuss für Gefahrstoffe 2016).

In contrast to Germany (ChemVerbotsV 2000), it is not forbidden to place mineral wool products without exemption on the market in Austria after Regulation (EC) No 1272/2008. As a result, so-called old mineral wool products with lower biosolubility and, consequently, possibly carcinogenic impact can still be sold.

3.3.2 Waste regulations

Classification of waste in Europe

The European Waste Catalogue (2000/532/EC) regulates the assignment of waste types. Different waste types are described in the waste list. The types of waste are defined by a six-digit code for the waste and the corresponding two-digit and four-digit chapter headings (Table 1). A waste is considered hazardous when marked with an asterisk (EU 2000).

Classification of waste in Austria

The List of Wastes Ordinance regulates the assignment of a waste material to waste codes of the Austrian Waste Catalogue at a national level (BMLFUW 2003), with some exceptions to the waste codes of the European Waste Catalogue (2000/532/EC). This is necessary because of Austria's different waste classification system in contrast to the other EU member states.

Originally, a waste type has been classified by the OENORM 2100 in Austria (ÖNORM S 2100 2005); then, this classification has been taken over by the Austrian Waste List (AVV), which includes the five-digit code for the waste. Additionally, all hazardous waste codes are labelled with "g".

Waste codes for mineral wool waste

Hazardous mineral wool waste in Europe is assigned to "other insulation materials consisting of or containing hazardous substances" waste code 17 06 03*, and new mineral wool waste is assigned to "insulation materials other than those mentioned in 17 06 01 and 17 06 03" waste code 17 06 04.

Mineral wool waste in Austria must be assigned to the following three waste codes: 31416 "Mineral Fibres", 31430 "Contaminated Mineral Fibres" and 31437 g "Asbestos Waste, Asbestos Dust". Mineral wool waste consisting of mineral wool that has been produced with certain quality labels, such as the German RAL quality label (so-called new mineral wool), has to be classified as "Mineral Fibres", whereas mineral wool waste composed of mineral wool that was produced without any quality label (so-called old mineral wool) has to be assigned to "Asbestos Waste, Asbestos Dust" (Table 2). Old mineral wool waste is hazardous but may be disposed of in the asbestos compartment in a landfill for non-hazardous waste (DVO 2008).

TABLE 2: Chapters of the Austrian Waste List (AVV) and five-digit code for the waste.

Waste Code	Description
31416	Mineral fibres
31430	Contaminated mineral fibres
31437 g	Asbestos waste, asbestos dust

Asbestos is a naturally occurring fibrous inorganic material, and the hazards are attributed to the fibrous character (Skinner et al. 1988). These two parameters can be seen as similarities between old mineral wool waste and asbestos waste.

However, there are significant physical differences between mineral wool waste and asbestos waste, e.g., with respect to crystallinity, cleavage and biosolubility.

4. CONCLUSIONS

Mineral wool products of today with certain quality labels do show higher biosolubility due to the Regulation (EC) No 1272/2008 of the European Parliament and of the Council. Such new mineral wool products still contain respirable fibres with the fibre dimensions defined by the WHO. Mineral wool waste causes several problems in Austria. The impossible distinction between old and new, i.e., possibly carcinogenic and non-carcinogenic mineral wool waste, complex logistics and poor landfill stability are examples of occurring difficulties. An uncertain amount of mineral wool waste in Austria complicates an assessment of the recycling potential. To date, many studies have focused on the recycling potential of mineral wool waste from production. The technique of Müller et al. (2009) is realised at the laboratory scale. Compaction of man-made vitreous fibres by the patented press of Wurzer Umwelt GmbH Eitting might be a first step to improve landfill behaviour of the waste. The procedure of Gröper & Lack focuses on backfilling with products made from mineral wool waste. In contrast to these current recycling and processing options, project RecyMin is following a comprehensive aim and therefore addresses aspects such as landfilling, backfilling, recycling in the cement industry and recycling in the mineral wool industry. The comparison of the economic and ecological effects of the different examined aspects will allow an integrated evaluation.

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TABLE 1: Chapters of the list and waste codes for insulation materials (EU 2000).

<table>
<tr><th>Chapter</th><th>Description</th><th>Chapter</th><th>Description</th><th>Waste Code</th><th>Description</th></tr>
<tr><td rowspan="4">17</td><td rowspan="4">Construction and demolition wastes (including excavated soil from contaminated sites)</td><td rowspan="4">17 06</td><td rowspan="4">Insulation materials and asbestos-containing construction materials</td><td>17 06 01*</td><td>Insulation materials containing asbestos</td></tr>
<tr><td>17 06 03*</td><td>Other insulation materials consisting of or containing hazardous substances</td></tr>
<tr><td>17 06 04</td><td>Insulation materials other than those mentioned in 17 06 01 and 17 06 03</td></tr>
<tr><td>17 06 05*</td><td>Construction materials containing asbestos</td></tr>
</table>

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EVALUATION OF THE EFFICIENCY OF AUTOCLAVING HEALTHCARE WASTE USING BIOLOGICAL AND CHEMICAL INDICATORS

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ABSTRACT

Autoclaving is among the techniques used to treat health services waste belonging to the most applied group of biohazards and sharps. However, a shortcoming of the autoclaving treatment involves the operational parameters and appropriate frequency for monitoring the process because there is limited scientific evidence for using the optimal parameters. The objective of this study was to evaluate the efficiency of the autoclaving process, using biological and chemical indicators, at different temperatures and exposure times, in three enterprises that provide services of autoclaving health care waste. To this end, partnerships were signed with three private enterprises and an operational procedure was developed to conduct real-scale tests under three different scenarios, using temperatures of 150, 132, and 125°C at 15-min exposure times and reduced exposure times (4 or 10 min). The results showed that among the three projects, a statistically significant difference between exposure times of 15 min and 4 min of the healthcare waste under Scenario 2, at a temperature of 132°C, only occurred in one case. However, modification of the exposure time and/or operational temperature requires further studies as well as confirmation from an environmental agency.

1. INTRODUCTION

Significant progress and improvement in health services for risk elimination has resulted in an increase in the rate of healthcare waste generation, particularly in developing countries that have deficient normative instruments for the proper management of this waste (Windfeld, 2015).

Several authors have argued that there is no evidence that healthcare waste is, in fact, more contaminated than urban solid waste or that it can cause environmental diseases and contamination (Zanon, 2002; Cussiol, 2005; Costa e Silva et al., 2011). However, most authors establish an exception to sharps and biological waste, which require treatment before final disposal because they are capable of containing potentially pathogenic microorganisms and serve as a vehicle for disease dissemination, putting at risk the professionals who manage it (Nascimento et al., 2009).

According to Ciplak and Kaskun (2015), the technologies used in the treatment of healthcare waste are categorized into two groups: high-temperature technologies, which covers incineration, and low-temperature technologies comprising alternative technologies such as autoclaving and microwave. However, the authors noted that in developing countries there is inadequate means of targeting healthcare waste as it is an easy and low-cost method.

According to Windfeld (2015), in developed countries such as the United States and the European Union, incineration is the main form of healthcare waste treatment. However, these countries have been searching for centralized forms of incineration treatment to have better equipped facilities to control atmospheric emissions. However, because of the enactment of stricter atmospheric emission limits, there is a trend towards the closure of incineration plants in favor of alternative treatments.

Autoclaves, among the main alternative technologies to incineration, are equipment that use heat and temperature to inactivate microorganisms and were originally developed for sterilization of equipment and surgical and laboratory materials (Zhao et al., 2009).

The autoclave treatment of healthcare waste consists of the use of humidity, pressure, and heat, in a controlled manner, for inactivation of the microbial load in the waste mass. To conduct the treatment, the residues are disposed in the equipment and exposed to water vapor for a predetermined time and at a predetermined temperature (Pichtel, 2005). Steam waste treatment equipment is typically operated at minimum standards, between 121°C to 134°C,

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for 30 min, as a basis for the already consolidated application of autoclaving for sterilizing medical products (WHO, 2014).

However, there is a lack of normative instruments that regulate alternative treatments, such as autoclaving. Moreover, as highlighted by Taghipour et al. (2016) and Oliveira (2017), there is a lack of data in the literature that specify ideal parameters for autoclaving treatment, such as exposure time, temperature, and ideal pressures for microbial inactivation of healthcare waste as well as the establishment of appropriate frequencies for monitoring of the process by means of efficiency tests. Such a situation may pose risks to public health, particularly to workers in these treatment enterprises, as well as environmental impacts.

The efficiency of an autoclaving process depends on controlling the physical measurements of time, temperature, and pressure, as well as biological indicators (BIs) and chemical indicators (CIs) used (Sandle, 2016).

This efficiency evaluated using BIs is measured by a representative rate of the number of inactivated or killed microorganisms following treatment (Teng et al., 2015). The quantitative assessment of the reductions in the microbial populations considers the

resistance of different species present, and is conducted according to predefined scales (Rodrigues, 2008).

The State and Territorial Association on Alternative Treatment Technologies (STAATT, 1998) defined and quantified microbial inactivation as "Log10 Inactivation," characterizing it as the difference, before and after treatment, between the logarithm of the number of viable microorganisms. The common standard of microbial inactivation for healthcare waste treatment is Level III, based on the criteria established by STAATT.

The 1994 STAATT rated acceptable levels for ensuring the performance of health care waste treatment technologies were revised in 1998. Level III microbial inactivation refers to "inactivation of vegetative bacteria, fungi, hydrophilic / lipophilic viruses, parasites and mycobacteria in a log 6 reduction or greater, and inactivation of Geobacillus stearothermophilus spores and Bacillus atrophaeus spores with reduction equal to or greater than 4Log10" (STAATT, 1998).

Chemical indicators assess whether the temperature, pressure, and time parameters have been adequately achieved; they completely change in color if the vacuum system is functioning properly, helping to identify faults (WHO, 2016).

Some studies have evaluated the application of biological and chemical indicators to monitor autoclaves. These include Garibaldi et al. (2017), who evaluated the standard configuration of two autoclaves for the healthcare waste treatment of Ebola-infected patients, considering different exposure times, temperatures, and pressures. The authors found that 16 out of 19 autoclave cycles with the factory default setting were positive for the biological indicators inserted in the center of the load. The optimized parameters for dry waste autoclaving were: a time of 30 minutes, temperature of 134°C, and pressure of 20 psi.

Experiments with both real wastes and simulated loads were also performed using both different combinations of exposure time and pressure pulses as well as biological indicators to evaluate the process efficiency. The authors concluded that deeper pressure pulses are more effective than shallow pulses for treatment (Emmanuel; Kiama; Heekin, 2008).

Considering that the absence of a systematic standardization and evaluation of healthcare waste treatment technology via autoclaving may result in permanent damage to public health, professionals, and the environment, this work aimed to evaluate the efficiency of the autoclaving process, using BIs and CIs, at different temperatures and exposure times, in three enterprises that provide services of autoclaving of health services waste.

2. MATERIALS AND METHODS

Considering the lack of standardization of test frequency using BIs and CIs between enterprises, as well as the non-standardization of the locations of insertion of these indicators in the mass of waste, it was proposed that three enterprises, which provide services of autoclaving of healthcare waste, be tested to evaluate the efficiency of the process, using BIs and CIs at different temperatures and exposure times.

The three enterprises used pre-vacuum horizontal type autoclaves with the following capacities: company I - 400 kg / hour; company II - 600 kg / hour; company III - 600 kg / hour.

As highlighted by Garibaldi et al. (2017), "cycles that evaluate treatment efficiency should contain the biological indicators disposed within the waste load, considering that indicators outside the mass of waste may not reflect actual treatment conditions," a fact that demonstrates the importance of assessment of the mass of waste.

Thus, for the insertion of indicators in the mass of waste, it was proposed for the enterprises the support be manufactured (Figure 1), such that the indicators were inserted in the middle and at the bottom of the container, considering that some ventures conducted monitoring using indicators on the external side of the container; therefore, it is necessary to evaluate the treatment conditions inside the waste mass, where it would be more difficult to contact the vapor with the waste.

The indicators were arranged in the mass of the waste, both at the bottom and in the middle of the containers, approximately 10 cm and 50 cm from the base of the containers, such that the conditions in which the healthcare wastes were exposed during the treatment were evaluated.

Tests using chemical, biological, and class IV indicators, which are multiparametric indicators that are capable of reacting to two or more critical process units, were conducted for 10 consecutive days under three scenarios in the three enterprises. It should be emphasized that the enterprises were oriented to acquire the same indicators, of brands that are commonly used, considering that each BI lot can present a variation in its population, resistance, and time of inactivation of the microorganisms, as caused by genotypic and phenotypic variations in spore cultures (Sandle, 2016).

In two ventures an exposure time of 10 min was applied as the shortest exposure time and in another venture an exposure time of 4 min was applied. An exposure time of 15

FIGURE 1: Supports for insertion of indicators.

min was common for all the enterprises, considering that it is the standard time used during the treatment processes, according to Figure 2.

The exposure time of 4 min was adopted based on the arguments reported by McKeen (2018). The authors presented the most used parameters for two types of autoclaves (gravitational and pre-vacuum) for sterilization of medical-hospital products, in which pre-vacuum autoclaves at temperatures from 132 to 135°C require exposure times of 3 to 4 min. In this manner, it is interesting to evaluate if such parameters would also be suitable for waste decontamination. The exposure time of 10 min was adopted because it was an intermediate time between the times previously proposed.

Temperatures within a range between 145 and 150°C were adopted under Scenario 1 because they are temperatures commonly used by developments during the treatment process. The temperature of 132°C was adopted under Scenario 2 as the maximum temperature recommended by the manufacturers for use of the BIs, while the temperature of 125°C was used under Scenario 3 because it was within the temperature range (from 121 to 127°C) commonly recommended and used in sterilization processes for medical and hospital materials.

For the tests, the indicators commonly used by the companies were used. Two different brands of BIs (3M and Cristófoli) and CIs (class IV of the 3M and Cristófoli brands) were used, and the same lots were used in case of acquisition needs later. Each scenario was evaluated over 10 days to verify several possible situations of loads received, in relation to the variation in composition, by the enterprises.

The enterprise 1 uses BI 1262 of 3M, containing 105 spores of G. stearothermophilus, with a reading in 48 hours. The BI changes color to yellow if there are surviving spores in the ampoule, indicating positive results. The CI used by the enterprise is of class IV and also from the same manufacturer.

In enterprise 2 the BI and CI were fixed to the outside of the container and two stainless-steel supports, one of 70

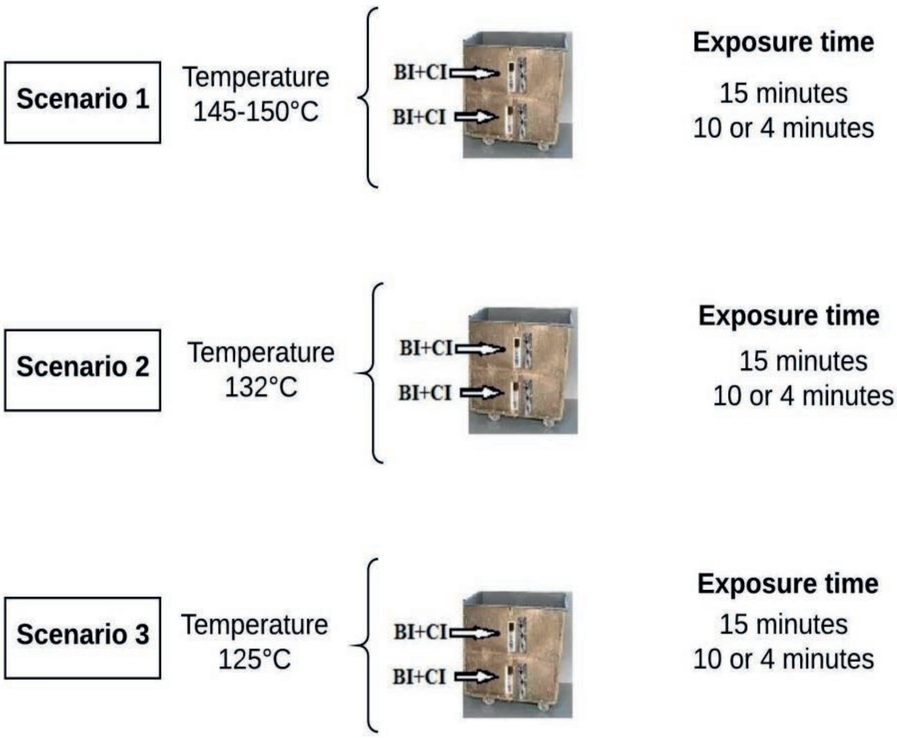

FIGURE 2: Tests plan.

cm and one of 1 m, were constructed for the insertion of the indicators in the mass of waste. The supports have a device for insertion of the BI, in the internal part, and another for the insertion of the CI, in the external part.

The enterprise 3 uses the 3M BI (Attest 1492 Super Fast Reading; American Type Culture Collection 7953 biological index) with a 1hour reading. The CI used by the enterprise is of class IV and also from the same manufacturer.

Following the treatment process, the indicators were removed from the containers. The CIs presented immediate results and the BIs were incubated, according to the manufacturers' instructions, for further reading of the results.

After all tests, the results were statistically analyzed using the Association or Independence Test (Chi-square test), which is used to evaluate the correlation between categorical variables. For this, two qualitative variables and the data were organized in a contingency table (Callegari-Jacques, 2003). The test statistic chi-square (χ^2), describes in a single number how the frequencies observed in each table cell differ from the frequencies that would be expected if there were no relationship between the treatments and results that define the lines and the table columns.

3. RESULTS AND DISCUSSION

3.1 Enterprise 1

The support, in wood, used by the enterprise to insert the indicators in the masses of waste presented some irregularities during the tests. The enterprise reported that the support did not withstand some tests and broke, requiring fabrication of new supports. It was observed that the material considerably deteriorated as it was subjected to the tests. However, the indicators were not lost amid the masses of waste, allowing their recovery and insertion in the incubator to analyze the results.

Table 1 shows the results of the tests applied during the project. It was possible to observe that some BIs were dry follows the incubation period, which made it impossible to read the results. We were informed that this situation has been frequently occurring in the enterprise. According to the manufacturer of the indicators, this may have occurred because of inadequate storage conditions.

From the analysis of Table 1, it is possible to observe that all the BIs evaluated under Scenario 1 at 150°C over 15 min of exposure, inserted at the bottom of the containers (10 cm from the base), showed inactivation of Geobacillus stearothermophilus spores. However, three BIs showed dryness, which impaired the reading of the results. The CIs, inserted under the same conditions to which the BIs were submitted, presented 90% satisfactory results.

Under this same Scenario 1, under the condition in which the BI was inserted within 50 cm of the container base, the results were similar to those of the BIs inserted at the bottom of the container. Five BIs showed dryness, rendering the reading unfeasible. As for the CIs, inserted under the same conditions, they presented 90% satisfactory results.

As for the 10-min exposure time under Scenario 1, none of the BIs showed spore growth for those inserted 10 cm from the base. However, under these conditions, five BIs were dry. The CIs, inserted under the same conditions to which the BIs were submitted, presented 100% satisfactory results.

For those BIs inserted in the middle of the containers (50 cm from the base), under the same aforementioned conditions, the problem of dryness in five BIs occurred; of the valid BIs, one showed spore growth. The CIs, inserted under the same conditions to which the BIs were submitted, presented 90% satisfactory results. Notably, the CI that presented an unsatisfactory result was not the one inserted in the same medium in which the BI showed spore growth.

Regarding Scenario 2, in which a temperature of 132°C was evaluated, considering an exposure time of 15 min,

TABLE 1: Results tests enterprise 1.

Tests	Enterprise1											
Scenario 1-150°C (Exposure time)	BI 10 cm			CI 10 cm			BI 50 cm			CI 50 cm		
	N*	NGS[1]	GS[2]	N*	S[3]	UNS[4]	N*	NGS[1]	GS[2]	N*	S[3]	UNS[4]
15 minutes	7	7	0	10	9	1	5	5	0	10	9	1
10 minutes	5	5	0	10	10	0	5	4	1	10	9	1
Scenario 2-132°C (Exposure time)	BI 10 cm			CI 10 cm			BI 50 cm			CI 50 cm		
	N*	NGS[1]	GS[2]	N*	S[3]	UNS[4]	N*	NGS[1]	GS[2]	N*	S[3]	UNS[4]
15 minutes	6	4	2	10	9	1	7	7	0	10	10	0
10 minutes	6	6	0	10	10	0	9	9	0	10	10	0
Scenario 3-125°C (Exposure time)	BI 10 cm			CI 10 cm			BI 50 cm			CI 50 cm		
	N*	NGS[1]	GS[2]	N*	S[3]	UNS[4]	N*	NGS[1]	GS[2]	N*	S[3]	UNS[4]
15 minutes	10	3	7	10	6	4	7	7	0	10	10	0
10 minutes	10	3	7	10	8	2	9	9	0	10	9	1

** N: Number of indicators (biological and chemical) evaluated*
[1] NGS: No growth of spores of Geobacillus stearothermophilus
[2] GS: Growth of spores of Geobacillus stearothermophilus
[3] S: Satisfactory result for chemical indicator
[4] UNS: Unsatisfactory result for chemical indicator

66.7% of the BIs showed satisfactory results and 33.3% showed unsatisfactory results. Under this scenario and at the time of exposure, four BIs showed dryness. The CIs, inserted under the same conditions to which the BIs were submitted, presented 90% satisfactory results. Notably, the CI that showed an unsatisfactory result was inserted in the same support as that BI that showed spore growth.

Under the condition in which the BI was inserted at 50 cm from the base of the container, none of the BIs showed growth; however, three BIs showed dryness, rendering the reading unfeasible. Regarding the CIs, all showed satisfactory results.

As for the results of the 10-min exposure time under Scenario 2, none of the BIs presented spore growth for those BIs inserted 10 cm from the base. However, under these conditions, four BIs were also dry. The CIs, inserted under the same conditions to which the BIs were submitted, showed 100% satisfactory results.

In the BIs inserted 50 cm from the base of the container, for the same aforementioned condition, none of the BIs showed growth and only one showed dryness. Regarding the CIs, all showed satisfactory results.

Scenario 3, in which a temperature of 125°C was evaluated, showed the most problems in terms of microbial inactivation. At an exposure time of 15 min, for the BIs inserted 10 cm from the base, 70% of the BIs evaluated showed spore growth. Under this condition, none of the BIs showed dryness which may have been related to the temperature at which the BIs were submitted, considering that the manufacturer establishes the maximum temperature as 132°C. Regarding the CIs, 40% showed unsatisfactory results, of which only two (i.e. one-half) were inserted together with BIs that also showed unsatisfactory results, demonstrating spore growth.

Under the condition in which the BI was inserted at 50 cm from the base of the container, none of the BIs that could be evaluated showed growth, but three BIs showed dryness, rendering the reading unfeasible. Regarding the CIs, all showed satisfactory results. Such a situation indicates that the bottom of the container (approximately 10 cm from the base) is a critical point for vapor penetration.

Regarding the results of the 10-minute exposure time under Scenario 3, the results of the BIs inserted at 10 cm from the container base were similar to those of the 15-min exposure time, in which 70% showed spore growth. However, only 20% of the CIs showed unsatisfactory results, both related to BIs that showed spore growth.

Under the condition in which the BI was inserted 50 cm from the base of the container, none of the BIs showed growth; however, a BI showed dryness, rendering the reading unfeasible. Regarding the CIs, 90% showed satisfactory results.

Notably, under all scenarios, the control BIs were also inserted in the incubator and all cases showed positive results with the presence of G. stearothermophilus in the ampoules.

From the statistical analysis of the BIs, it was possible to conclude that there was no significant difference in the inactivation of G. stearothermophilus spores between exposure times of 15 and 10 min., in relation to the different temperatures, in the three Scenarios evaluated. Thus, all the tests demonstrated that the inactivation of G. stearothermophilus is independent of the exposure times evaluated.

However, when there was a significant difference between the insertion positions of the indicators (10 and 50 cm from the container base), a statistically significant difference was observed between the position of the BIs of Scenario 3, at both exposure times. This result reveals that the BI position interfered with the inactivation of G. stearothermophilus spores, with the bottom of the container being a critical point for vapor penetration.

The enterprise ensured that, for those unsatisfactory BI results, the wastes were again submitted to the treatment process, until microbial inactivation efficiency was obtained, so that the healthcare waste were sent to final disposal.

3.2 Enterprise 2

During the validation phase of the supports, during the study, the results of the indicators were not satisfactory; thus, it was necessary to adapt and develop holes in all their extensions to enable steam insertion inside the supports.

Another necessary adaptation was the insertion of a BI in the support, considering that some BIs were showing water inside the ampoule, which made it impossible to read the results. In this sense, the BIs were repositioned on the supports with the lid facing upwards.

To perform the tests under Scenario 2, research of the new holes in the supports was conducted because of the sealing of the holes caused by the deformation of the plastic packages following the treatment process.

After adaptation, the indicators demonstrated good performance when they were in the treatment process, and did not deteriorate, which can cause the loss of indicators.

After the entire treatment cycle, the indicators were removed from the supports and evaluated and the BIs were inserted into the incubator to read the results. Quick use of the 3M mark, 1292 fast reading, containing 105 spores of G. stearothermophilus, with 3 h of reading, and CI class IV, from the same manufacturer, was completed. Table 2 shows the results of the proposed scenarios.

Notably, the enterprise was the first to conduct such tests in this manner; a trial period of 30 days was initially proposed under Scenario 1, to validate the supports and the test plan. However, because of operational issues, tests were performed for 24 and 25 days.

It is important to note that some BIs inserted in the support that evaluated the conditions at 10 cm from the base of the containers, under Scenario 1 for the exposure time of 15 min, showed water inside, which compromised reading of the results.

In this manner, the enterprise chose to perform most of the tests at the reduced time (4 min of exposure) under Scenario 1 with the indicators inserted in the middle of the waste mass (supported 50 cm from the base of the container) to avoid compromising the reading of the BIs.

From the analysis of Table 2, it can be observed that 76.2% of the BIs evaluated under Scenario 1, at 150°C over 15 min of exposure, inserted at the bottom of the containers (10 cm from the base), showed inactivation of Geoba-

TABLE 2: Results tests enterprise 2.

Tests	Enterprise 2											
Scenario 1-150°C (Exposure time)	BI 10 cm			CI 10 cm			BI 50 cm			CI 50 cm		
	N*	NGS[1]	GS[2]	N*	S[3]	UNS[4]	N*	NGS[1]	GS[2]	N*	S[3]	UNS[4]
15 minutes	21	16	5	24	24	0	24	23	1	24	24	0
4 minutes	1	1	0	1	1	0	25	21	4	25	25	0
Scenario 2-132°C (Exposure time)	BI 10 cm			CI 10 cm			BI 50 cm			CI 50 cm		
	N*	NGS[1]	GS[2]	N*	S[3]	UNS[4]	N*	NGS[1]	GS[2]	N*	S[3]	UNS[4]
15 minutes	20	12	8	20	20	0	20	12	8	20	20	0
4 minutes	20	2	18	20	20	0	20	2	18	20	20	0

** N: Number of indicators (biological and chemical) evaluated*
[1] NGS: No growth of spores of Geobacillus stearothermophilus
[2] GS: Growth of spores of Geobacillus stearothermophilus
[3] S: Satisfactory result for chemical indicator
[4] UNS: Unsatisfactory result for chemical indicator

cillus stearothermophilus spores and 23.8% of the tests evaluated did not show spore inactivation. However, the CIs, inserted under the same conditions to which the BIs were submitted, showed 100% satisfactory results. Under this same scenario, under the condition in which the BI was inserted 50 cm from the base of the container, 95.8% of the BIs showed satisfactory results of spore inactivation. In addition, the CIs evaluated under the same conditions also showed 100% satisfactory results. However, the results of the 4-min exposure time under Scenario 1 showed more spore growth (16% of the total BIs evaluated at a height of 50 cm from the container base), when compared to that of the 15-min exposure time. However, all CIs showed satisfactory results.

In relation to Scenario 2, in which the temperature of 132°C was evaluated at exposure times of 15 and 4 min, the enterprise chose to perform two tests per day, totaling 20 tests over 10 days. Under this scenario, it was possible to notice a greater divergence between the results. At an exposure time of 15 min, in both positions, 60% of the BIs did not show spore growth and 40% showed problems of microorganism inactivation, out of a total of 20 tests. However, the CIs showed all satisfactory results.

At an exposure time of 4 min, the situation was very unsatisfactory, considering that in both insertion positions the BIs showed 90% G. stearothermophilus spore growth, out of a total of 20 tests performed. Once again, the CIs showed 100% satisfactory results.

From the statistical analysis of the BIs, it was possible to conclude that for Scenario 1 there was no significant difference in inactivation of G. stearothermophilus spores between exposure times of 15 and 4 min under the two scenarios evaluated. However, under Scenario 2, there was a significant difference between exposure times, demonstrating that the inactivation of G. stearothermophilus spores is not independent of the exposure time employed. Thus, the exposure time of 4 min at a temperature of 132°C was not shown to be adequate for spore inactivation.

Notably, under all scenarios, the control BIs were also inserted in the incubator and all cases showed positive results, with the presence of G. stearothermophilus in the ampoules.

Considering that all the CIs showed satisfactory results, it was not possible to apply statistical tests for comparison. These findings demonstrate that using CIs only to assess the efficiency of the treatment process may not guarantee microbial inactivation in the mass of residues, although physical parameters are attained.

3.3 Enterprise 3

Enterprise 3 already used a support to perform BI tests. According to the managers of the project, BI loss was commonly recorded, both in the mass of the residue, because it became quite dense after treatment, and because of the dryness of the BI, which compromised the reading of the results. According to those responsible, after manufacturing of the supports, such problems were remedied.

However, the enterprise did not use CI to evaluate operational conditions. In this manner, it was proposed to acquire the indicator for conducting the tests. Considering that in one holder it was possible to insert only one indicator, two supports were required; one for insertion of the CI and another for insertion of the BI. Both were inserted in the mass of residues in the same position, being fixed to each other and in the container by a wire.

Following the treatment, the indicators were removed from the containers and evaluated, and the BIs were inserted into the incubator to read the results. The enterprise uses the BI of the mark 3M, super-fast reading Attest 1492, with readings of 1 h, containing 105 spores of G. stearothermophilus. The project opted to acquire the CI of the same manufacturer, in which the CI class IV was acquired. Table 3 shows the results obtained under the proposed scenarios.

Table 3 shows that, in general, the results of all the scenarios were satisfactory. It is possible to notice that all the BIs evaluated under Scenario 1, at 150°C over 15 min and 4 min of exposure, inserted at the bottom of the containers (10 cm from the base), showed inactivation of G. stearothermophilus spores. The CIs, inserted under the same conditions to which the BIs were submitted, showed 80% satisfactory results for both exposure times.

Under the condition in which the BI and CI were inserted within 50 cm of the container base, the results were also

TABLE 3: Results tests enterprise 3.

Tests	Enterprise 3											
Scenario 1-150°C (Exposure time)	BI 10 cm			CI 10 cm			BI 50 cm			CI 50 cm		
	N*	NGS[1]	GS[2]	N*	S[3]	UNS[4]	N*	NGS[1]	GS[2]	N*	S[3]	UNS[4]
15 minutes	10	10	0	10	8	2	10	10	0	10	10	0
4 minutes	10	10	0	10	8	2	10	10	0	10	10	0
Scenario 2-132°C (Exposure time)	BI 10 cm			CI 10 cm			BI 50 cm			CI 50 cm		
	N*	NGS[1]	GS[2]	N*	S[3]	UNS[4]	N*	NGS[1]	GS[2]	N*	S[3]	UNS[4]
15 minutes	10	10	0	10	10	0	10	10	0	10	10	0
4 minutes	10	10	0	10	10	0	10	9	1	10	10	0
Scenario 3-125°C (Exposure time)	BI 10 cm			CI 10 cm			BI 50 cm			CI 50 cm		
	N*	NGS[1]	GS[2]	N*	S[3]	UNS[4]	N*	NGS[1]	GS[2]	N*	S[3]	UNS[4]
15 minutes	10	10	0	10	9	1	10	10	0	10	10	0
4 minutes	10	10	0	10	8	2	10	10	0	10	10	0

* N: Number of indicators (biological and chemical) evaluated
[1] NGS: No growth of spores of Geobacillus stearothermophilus
[2] GS: Growth of spores of Geobacillus stearothermophilus
[3] S: Satisfactory result for chemical indicator
[4] UNS: Unsatisfactory result for chemical indicator

satisfactory for both exposure times, with no readability or unsatisfactory conditions occurring.

Regarding Scenario 2, in which the temperature of 132°C at an exposure time of 15 min was evaluated, all conditions also showed satisfactory results, showing only growth of G. stearothermophilus spores at an exposure time of 4 min, for the BI inserted in the middle of the container (50 cm from the base).

Scenario 3, in which a temperature of 125°C was evaluated, also showed satisfactory conditions similar to those of the other scenarios, presenting unsatisfactory results only in some CIs.

The developers stated that the manufacturer guarantees satisfactory treatment conditions at temperatures up to 121°C, which may justify the results of the three scenarios.

From the statistical analysis of the BIs, inactivation of G. stearothermophilus spores occurred between the exposure times of 15 and 4 under the three scenarios evaluated at the different temperatures evaluated.

In enterprises 1 and 3, it was observed that at the temperature and exposure time commonly used in the operation of the enterprises (145 or 150 °C, with healthcare waste exposure time of 15 minutes), both biological and chemical indicators presented satisfactory results.

From the results it was not possible to establish an ideal frequency for the use of the indicators in all scenarios, considering that in some enterprises there was a considerable variation between satisfactory and unsatisfactory indicators during the proposed test days.

However, in all the enterprises analyzed it was observed that at the temperature and exposure time commonly used in the operation of the projects (145 or 150 ° C, with exposure time of 15 minutes), the indicators, both biological and chemical, presented satisfactory results. This may indicate that the use of biological indicators at a frequency of use of twice a week would be appropriate for process monitoring.

4. CONCLUSIONS

The technique of autoclaving healthcare waste, although widely used for the treatment of healthcare waste from the biological and sharps group, presents some shortcomings in relation to the lack of specific legislation, to establish criteria and guidelines, mainly operational, to standardize the activity.

Although only one enterprise had a statistically significant difference between the exposure times for healthcare wastes of 15 min and 4 min, under Scenario 2 (at a temperature of 132°C), the results did not show that the ventures can, therefore, reduce the exposure time and / or the operating temperature, considering that the temperature of 125°C also showed the highest number of unsatisfactory results for enterprise 1.

The results also warned of the limitation of vapor penetration at the bottom of the containers, which must be observed and monitored by the enterprises, to evaluate, mainly, excess cargo in the containers.

In addition, another point to be evaluated by the enterprises is the use of operating temperatures above those suggested by the indicators' manufacturers, considering that in enterprise 1 the temperature of 150°C may have been a preponderant factor for the dryness of the indicators which impaired reading them.

Considering that some authors have reported on the lack of conclusive studies regarding the reliability of CIs, the results of the CIs obtained in this study indicate a limitation because they do not present similar responses to those of the BIs. Thus, it is clear that CIs should not act as a substitution for BIs but should be used for supportive monitoring to avoid false-negative or false-positive results.

It is necessary to conduct additional studies before any modification in the treatment process is carried out, mainly evaluating the insertion of indicators at different points and in higher quantities of containers per cycle, in addition to the evaluation and validation of an environmental agency, considering the licensing of the development.

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REMOVAL OF DISSOLVED METHANE AND OTHER CONTAMINANTS FROM LANDFILL LEACHATES IN ENGINEERED WETLANDS

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ABSTRACT

This paper will present detailed design and operational data for a full-scale horizontal flow reed bed system, specifically designed for removal of dissolved methane from leachates at a closed landfill in the UK Midlands, near to the city of Birmingham. Since commissioning during the summer of 2013, as reeds have grown, extensive operational data have demonstrated successful and complete removal of dissolved methane to below required concentrations, for safe disposal into the public sewer. The Shirley reed bed is very successful at removing high initial levels of dissolved methane (>95% removal), as per the intended requirements. This ensures that methane within the discharged effluent remains below the 0.14 mg/l consent concentration. Removal of methane has been achieved consistently for over 5 years from summer 2014 despite occasional increases in leachate flow rates above the design capacity for the reed bed. Additional removal of further contaminants is achieved by the reedbed, where the successful removal of high levels of iron and suspended solids from the leachate is observed. Monitoring has also noted significant seasonal nitrification of ammoniacal-N, although this was not part of the original purpose of the bed. Design, operational and monitoring data are presented and discussed, with discussion of how various practical issues were overcome during the six years of operation. The technology has wide application for many closed landfill sites throughout the world, and design loading data have been derived that will be valuable for operators to size reed beds appropriately at their sites.

1. INTRODUCTION

An alternative option to installing bespoke methane stripping systems at older landfills, where COD values and concentrations of ammoniacal-N may be starting to reduce, is to pass raw leachates through engineered wetlands in the form of reed beds. Dissolved methane is readily-degraded and oxidised by bacteria, preferentially before other COD components, because of the high energy available to the bacteria. Therefore, complete removal of the dissolved methane can potentially be achieved in a passive, and low maintenance system. This is important at closed and unmanned landfill sites.

The disposal of landfill leachates into public sewers, for combined treatment with domestic wastewaters, remains a common practice at many landfill sites. Leachates can typically contain concentrations of dissolved methane of up to 15 mg/l, and from first principles it can be determined that at concentrations of 1.4 mg/l or greater, explosive concentrations of methane gas can be generated in the headspace above such leachate, and potentially within the sewer itself (Robinson et.al, 1999). As a result, regulators in the United Kingdom routinely apply a factor of safety of ten, resulting in discharge limits of less than 0.14 mg/l for concentrations of dissolved methane in leachates being discharged.

This can therefore result in a need to reduce original concentrations of dissolved methane by more than 99 per cent, from 15.0 mg/l to less than 0.14 mg/l, on a reliable basis, requiring sound process designs and robust treatment systems. Even though rates of gas production may be much lower at relatively old and closed landfill sites, concentrations of methane gas can remain at 60% within the landfill mass, and so concentrations of dissolved methane in leachates often remain high for many decades.

Dissolved methane is readily degraded biologically in suitable aerobic conditions, and many relatively simple filter systems have been used successfully for this purpose (Hatamoto et.al, 2010). In landfill environments, the oxidation of methane in landfill gas, has regularly been

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observed to take place in the presence of naturally occurring methanotrophs (Stern et.al, 2007). Methane is readily oxidised biologically by these bacteria, in the presence of oxygen. Therefore, because oxygen enters the reed beds by passive diffusion, assisted to some extent by oxygen transfer via the reed plants, methane can be removed successfully. This removal has been demonstrated at Shirley reedbed, where methane levels must satisfy a 0.14 mg/l discharge consent (Robinson, H., 2017; Robinson, T., 2017; 2018).

Methane stripping systems can, and have been installed to remove dissolved methane, prior to the safe discharge of pre-treated leachates into the public sewer, for combined treatment with domestic wastewaters, but the stripping process requires power, frequent maintenance, and can result in precipitation of large amounts of scale as calcium, magnesium and iron are brought out of solution by the vigorous aeration involved (Robinson, 1999; Robinson et.al, 1999).

Reed bed systems have been used successfully both for the complete treatment of relatively weak leachates from old, closed landfills (Robinson, 1999; Robinson et.al, 1999), and also for the polishing of leachates that have been treated biologically, in order to enable effluents to be discharged safely into surface watercourses (Robinson, 1993, 1999; Robinson et.al, 2003; 2008; Robinson and Olufsen, 2007; Strachan et.al, 2007; Novella et.al, 2004). In almost all circumstances, greatest success has been achieved where concentrations of ammoniacal-N in liquids entering the reed bed do not exceed 10 mg/l, whether beds are operated as vertical or horizontal flow systems.

Reed beds have great potential to provide an environment in which effective biological oxidation and degradation of methane dissolved in leachates from closed landfill sites can reliably and efficiently achieve concentrations acceptable for discharge into public sewers, but few case studies have been reported.

This paper reports and describes in detail a full-scale project at a closed landfill in the UK Midlands, where a reed bed has been used successfully for this purpose since July 2013. Operational results are presented for a six-year period between 2013 and 2018.

2. SHIRLEY REEDBED BACKGROUND

Shirley Landfill Site is located to the South West of the city of Birmingham, in the UK Midlands, and is the responsibility of Worcestershire County Council. The site was originally quarried for sand and gravel during the 1970s, and was restored between 1981 and 1988 by filling with 1.2 Mm^3 of household wastes, over an area of 15 hectares. The average depth of the waste is about 8m with a maximum of 12 m and a minimum of 3 m.

A reed bed at Shirley was designed and constructed during 2013, primarily to reduce concentrations of methane in leachate draining by gravity from the landfill, where it was recognised that uncontrolled inflow of groundwater was a significant contributor to leachate generation rates. Concentrations of dissolved methane being discharged to sewer were routinely exceeding a recently imposed limit of 0.14 mg/l, and removal would take place by means of aerobic biological degradation, since methane is readily oxidised biologically by bacteria, in the presence of oxygen. Six years' data are available to demonstrate not only successful removal of methane (which is discussed in detail elsewhere; Robinson, H., 2017), but also provide valuable information on the limited and seasonal removal of ammoniacal-N being achieved by the bed.

3. REEDBED DESIGN AND OPERATIONS

3.1 Reedbed Design Types

Reedbeds are designed to pass flows of wastewater either horizontally (Figure 1), or vertically (Figure 2). Horizontal Flow Reed Beds (HFRBs) receive an inflow from an overflowing halfpipe structure at the inlet end of the bed,

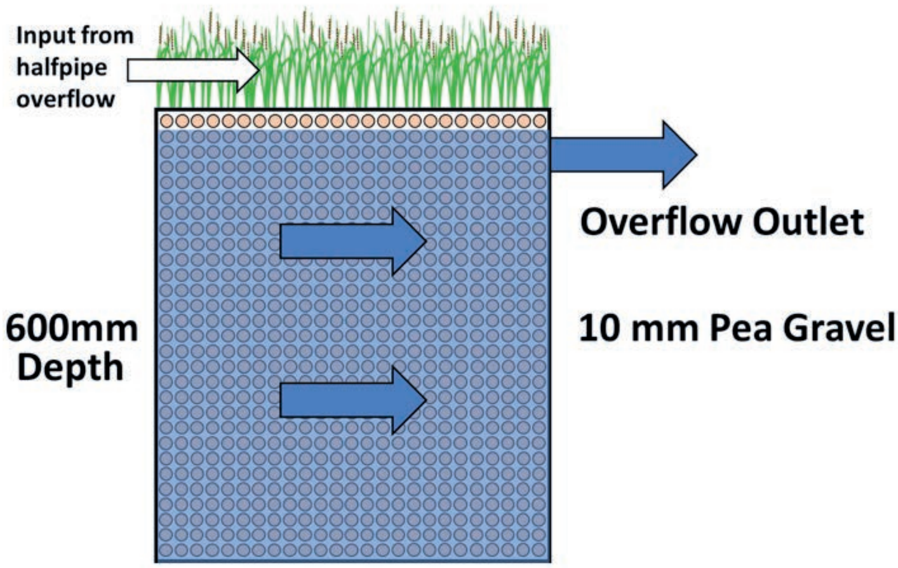

FIGURE 1: Cross-section of a horizontal flow reed bed (Robinson, T., 2018).

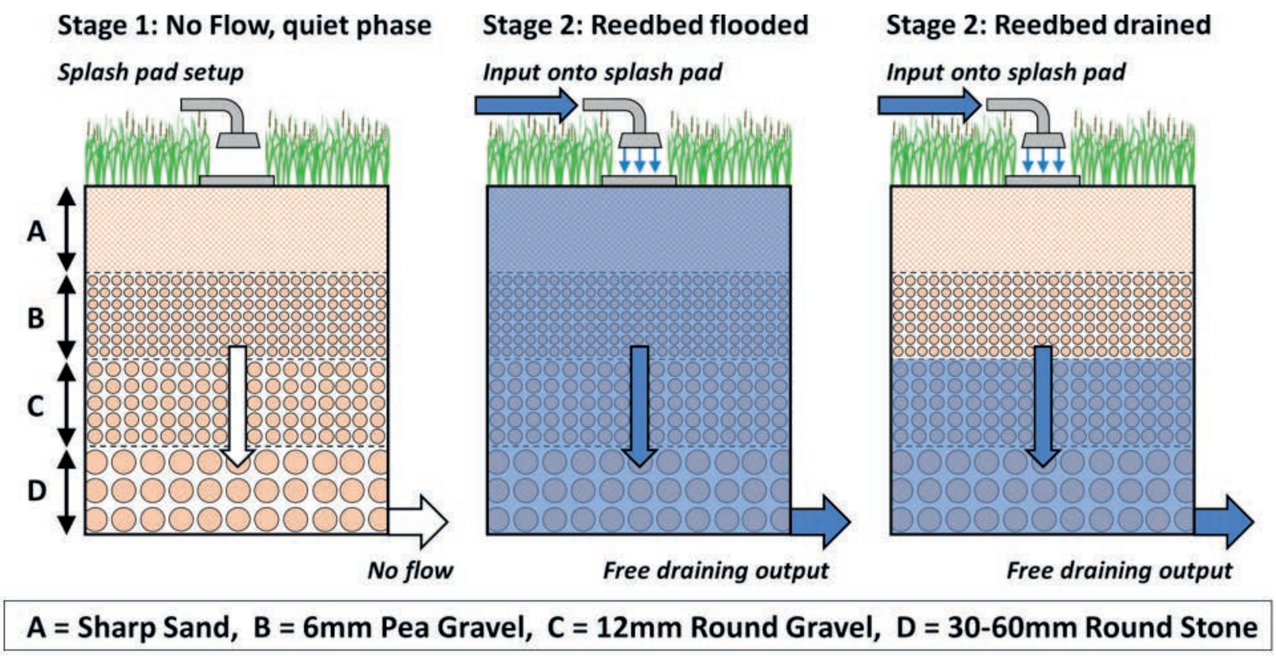

FIGURE 2: Cross-section of a vertical flow reed bed (Robinson, T., 2018).

before the water flows across the flooded bed, at a depth predetermined by the height of the overflowing outlet. Single-size gravel media (typically 10mm pea gravel) is generally flooded to just below the gravel surface, avoiding surface flows bypassing treatment, and allowing water to flow horizontally, at a steady rate.

In a Vertical Flow Reed Bed (VFRB), the media are a range of sizes, and water levels in the bed vary during treatment cycles. Incoming leachate, or pre-treated leachate, enters as occasional 'slug' doses, and floods the bed surface. The liquid gradually passes down through the bed, contacting oxygen in the spaces between the media. The bed becomes fully flooded, and effluent drains from the bottom of the bed. As the liquid drains out, fresh air, containing oxygen, is drawn down into the media of the bed. Eventually the bed drains completely, ready for another dose of feed. Vertical flow beds therefore have greater oxygen inputs, so can provide more treatment (e.g. nitrification of ammoniacal-N), but are usually not so good at solids removal (Robinson, T., 2018).

3.2 Mechanisms for removal of other contaminants within reedbeds

Although vertical flow reed beds have been reported to provide higher rates of removal of ammoniacal-N than horizontal flow beds, their reduced performance in achieving removal of solids, and the intrinsic simplicity of the horizontal bed, were key to the horizontal bed being selected at Shirley. Previous papers describe case studies of vertical flow reed bed systems (Robinson, T., 2017).

Iron (and suspended solids) are readily removed in a reed bed system, principally by oxidation and physical filtration processes. The rhizome system of the reeds within the gravel bed may contribute to improved performance, by enhancing the supply of oxygen, which is required to convert soluble iron (II) to insoluble iron hydroxide (III).

Although reed beds have a poor record for removal of ammoniacal nitrogen from effluents containing high levels of COD and BOD (for example, widely noted for direct treatment of domestic wastewaters), they are generally more successful in situations where concentrations of organic contaminants are much lower, (for example, biologically pre-treated leachates), and more oxygen is therefore available to nitrifying organisms, principally Nitrosomonas and Nitrobacter, which convert ammoniacal nitrogen to nitrite, and then to nitrate.

4. DESIGN AND CONSTRUCTION OF THE SHIRLEY REEDBED

The type of reed bed installed at Shirley is a lined, gravel-filled, horizontal flow bed. Reeds, Phragmites Australis, have been planted into the gravel at the site. Effluent enters at the inlet of the beds, travelling slowly through the bed following a horizontal flowpath, before flowing over a level control device within a chamber at the outlet end of the reed bed. Plate 1 below depicts the chamber containing the overflow point, at the end of the Shirley horizontal flow reed bed. This photograph shows the reed bed during its construction phase, when the bed was newly installed, and the reeds were freshly planted.

The reed bed at Shirley was constructed by Phoenix Engineering, during the first half of 2013. The design was based on flow information provided by the Council, which stated that mean flow rate would be about 50 m^3/d, and within a range from 24 m^3/d to a maximum flow of 78 m^3/d. Leachate draining from the site is captured by a series of French drains and a pipeline that runs to a chamber within the site, before being discharged into the public sewer. On several occasions previously, the limit set by the discharge

PLATE 1: The chamber at the outlet end of the Shirley Reed Bed during the construction phase of the reed bed

consent for dissolved methane was being exceeded, which had the potential to be hazardous.

It was recognised that uncontrolled inflow of groundwater into the landfilled wastes was a significant contributor to leachate generation rates. A reedbed was a far more sustainable and practical option for an unmanned, relatively remote, closed landfill site, than would have been provided by a mechanical methane stripping arrangement, and, although the development was in the green belt, it was recognised that the development was necessary to avoid pollution, and noted that the only alternative would have been to take leachate off-site in tankers, generating traffic and causing amenity impacts.

There was no means of buffering leachate flows from the landfill, as these arrived at the original manhole, from where flows to sewer were made, and the reed bed design did not seek to provide any additional leachate storage or flow buffering. Nevertheless, results indicated that although flow rates showed seasonal variation, they did not respond rapidly to rainfall events, as might be expected from a landfill where significant groundwater inflows were involved. Previous work had demonstrated a close link between general groundwater levels in the local aquifer, and leachate flow rates.

Leachate transfer arrangements required modification, with construction of a new deep chamber into which leachate would now drain from the site by gravity, and from where it would be pumped in a controlled way by duty/standby pumps into a new surface-mounted precast concrete header tank, having a diameter of 2.4 m and a depth of 1m (volume 5 m^3). This header tank was designed to encourage the quiescent settlement and retention of any silt or precipitated iron solids, with supernatant leachate overflowing to the reed bed inlet.

The reed bed is contained by an engineered earth embankment, and has a length of 50 m, a width of 7 m, a gravel depth of 0.6 m, and an estimated hydraulic volume of about 85 m^3, giving an estimated mean hydraulic retention time (HRT) of between 1 and 2 days at anticipated flow rates. Effluent from the bed drains into a discharge chamber at its remote end, flowing over a variable level control mechanism, which in most circumstances maintains water level within the bed just below the gravel surface. Plate 2 gives an overview of the entire reedbed treatment system.

5. ONGOING PERFORMANCE OF SHIRLEY REEDBED

The Shirley reed bed has always performed successfully, removing all methane from leachate entering it, includ-

PLATE 2: General view of Shirley Reedbed from the inlet end, showing the Leachate Header Tank in the foreground, September 2014.

ing when flows were more than double design rates during early 2014. Since the bed was commissioned in July 2013, routine sampling of raw and treated leachates has been carried out regularly, and all flow meters and recording instruments have performed accurately and reliably. The collection of such detailed data has enabled the production and interpretation of the following figures, which highlight the success of the reed bed as a treatment method for removing methane, suspended solids, and some ammoniacal-N from the leachate.

5.1 Operational data

The most significant impact on operation of the bed, since it was commissioned, has been the flows of leachate passing through it, which have sometimes exceeded the original design specification. In particular, there were extreme and record-breaking levels of rainfall during the early months of 2014, with more than double average rainfall amounts during January and February. This led to the reed bed receiving and treating leachate flows as high as 160m^3/d, with highest values recorded during late February/early March 2014 (see Figure 3).

During the full year from 1 October 2013 to 30 September 2014, mean leachate flow rate was just over 65 m^3/d – 30 per cent greater than predicted values, and the maximum flow rate of 163 m^3/d was more than double the anticipated maximum flow rate of 78 m^3/d.

During the first 3 months of 2014, more than 10,000 m^3

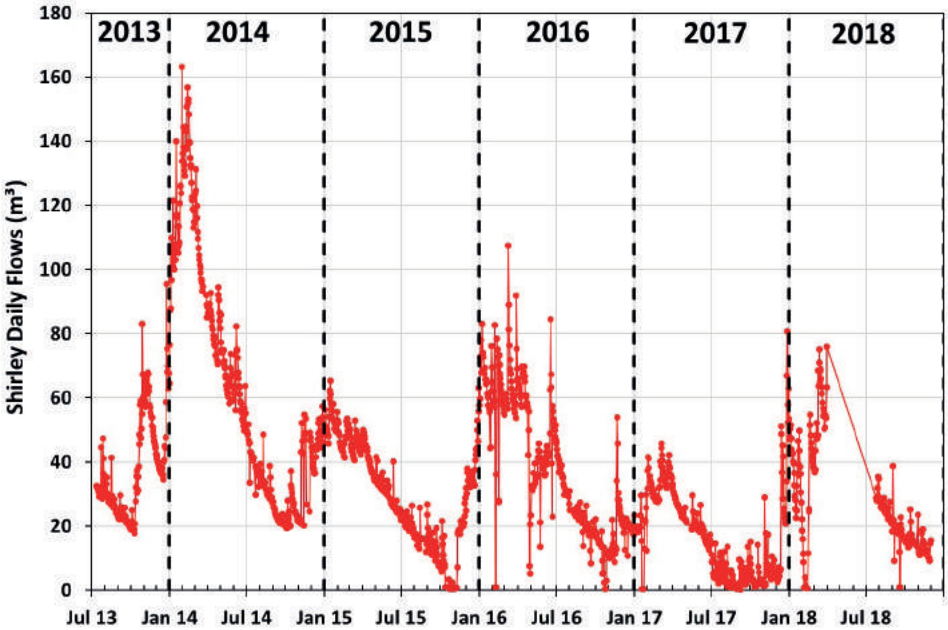

FIGURE 3: Daily volumes treated at Shirley, July 2013 to December 2018 (in m^3/d).

of leachate passed through the bed (10,348 m^3), at a mean flow rate of 115 m^3/d, with a maximum monthly flow of 3,766 m^3 during February 2014 (mean rate 134.5 m^3/d); 45 per cent greater than predicted maximum instantaneous flow rates, throughout the month.

Figure 3 highlights the very seasonal nature of the flow rates passing through the reed bed, whereby typical daily flows during winter months (excluding the winter of 2013/14) can be elevated above 60 m^3/day, whilst in summer and autumn months, the flows can reduce to below 20 m^3/day.

5.2 Treatment requirements and assessment of treatment efficiency

Table 1 presents the criteria for the discharge consent, as set by Severn Trent Water plc, for discharges of effluent from the Shirley Reed Bed. The maximum volume of effluent that was to be discharged to sewer, as agreed with Severn Trent Water, was set at 137 m^3 during any single 24-hour period.

Results comparing concentrations of various contaminants in incoming leachate flows are compared with values determined in treated leachate discharged to sewer, in Figures 4 to 11.

TABLE 1: Discharge conditions set by Severn Trent Water Limited on 14th August 2014, for wastewaters being discharged into the Upper Cole Valley Sewer.

Condition / Determinand	Units	Discharge consent
Maximum Discharge Rate	l/sec	2
Dissolved Methane	mg/l	<0.14
pH value	pH-Value	>6 and <10
COD	mg/l	300
Ammoniacal-N	mg/l	50
Phosphorus	mg/l	25
Suspended solids	mg/l	200

The effects of dilution during passage through the bed due to rainfall, or possible concentration from evapotranspiration losses, were determined by observation of concentrations of the two conservative ions sodium and chloride, which are not affected by any treatment processes of the reed bed. Results for sodium and chloride in raw and treated leachate are presented below in Figures 4 and 5 respectively.

Sodium and chloride results confirm that no significant dilution or concentration of contaminants took place during passage of leachate through the reed bed, which means that changes in concentrations of other contaminants can be entirely attributed to treatment being provided by biological and chemical changes taking place within the bed.

Also of interest is the fact that although flow rates of leachate from Shirley Landfill, and therefore rates of leachate flow through the bed, increased substantially during early 2014, this was not associated with equivalent dilution of the leachate being received for treatment. This is characteristic of landfills where high proportions of leachate being produced are derived from groundwater inflows.

5.3 Removal of methane concentrations

Figure 6 demonstrates that from the date that the reed bed began to receive leachate, on 12 July 2013, until the end of March 2014, all dissolved methane was removed completely, including during the periods of greatest flow. Even during those periods, concentrations of dissolved methane regularly exceeded 1mg/l in the incoming leachate flows.

From April 2014 to mid-June 2014, as flows reduced gradually from peak values of 160 m^3/d in early 2014 (dou-

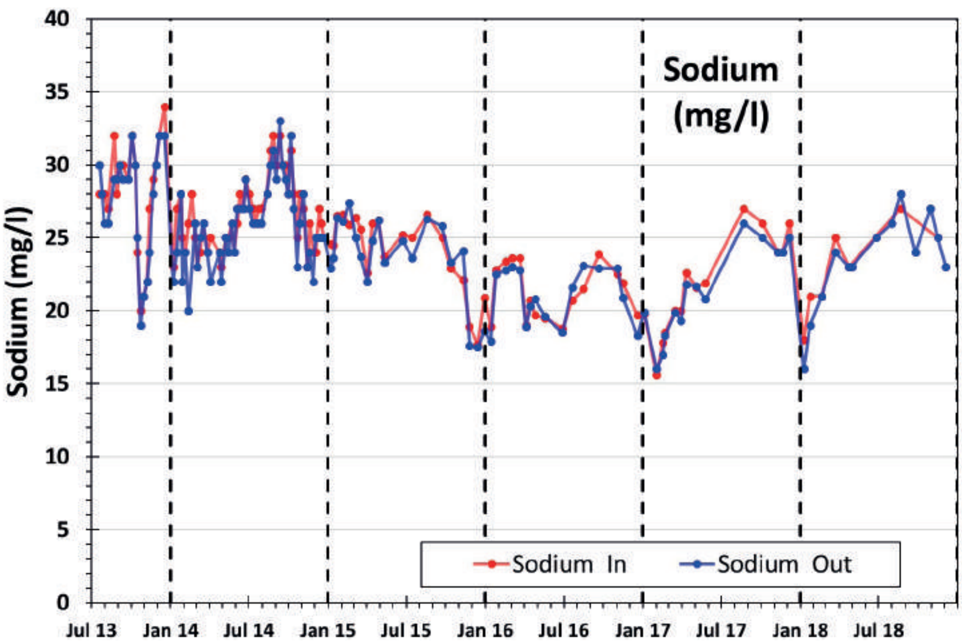

FIGURE 4: Variation in concentrations of sodium during passage through the reedbed, July 2013 to December 2018 (all results in mg/l as sodium).

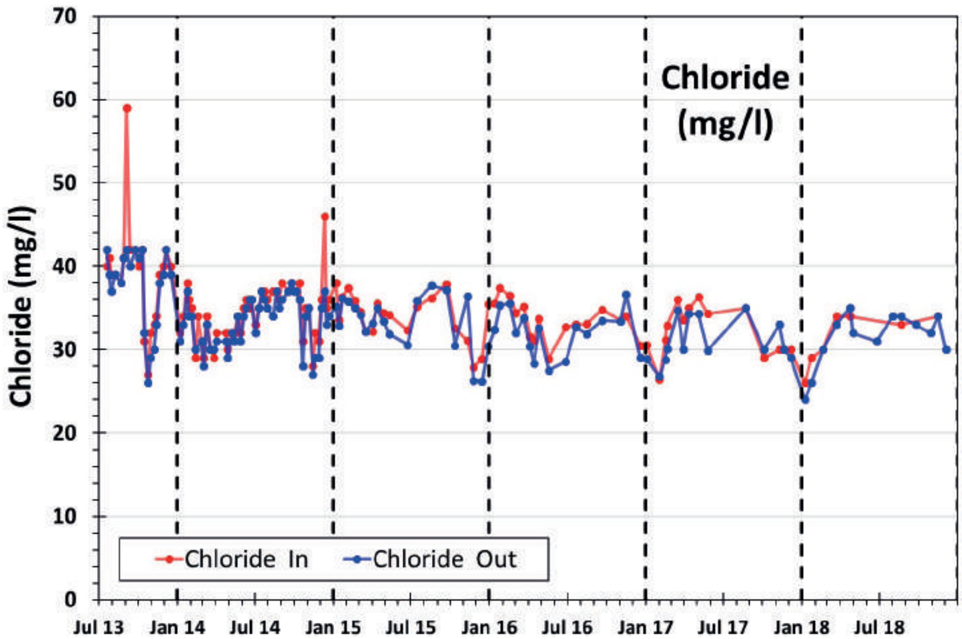

FIGURE 5: Variation in concentrations of chloride during passage through the reedbed, July 2013 to December 2018 (all results in mg/l as chloride).

ble the maximum design values), concentrations of dissolved methane in raw leachate rose significantly, reaching a peak value of 3.75 mg/l in mid-May 2014. During this period, low levels of dissolved methane were measured in treated leachate, which rose to a maximum value of 0.7 mg/l during July 2014, when raw leachate contained about 3.5 mg/l of dissolved methane (also nearly twice design values), at flow rates of 40 to 50 m^3/d.

The very high flow rates also had an effect of flushing out quantities of iron from within the landfill drainage system, some of which reached the surface of the bed near to the inlet, and had to be removed by works during July 2014. Although concentrations of dissolved methane in raw leachate remained above 2 mg/l, as leachate flow rates gradually declined towards 20 m^3/d by the end of September 2014, dissolved methane in effluent returned to below the consent limit of 0.14mg/l during August 2014.

Since August 2014, the reed bed has always achieved consistent removal of methane down to below the consent limit of 0.14 mg/l. Figure 6 shows that even after increased

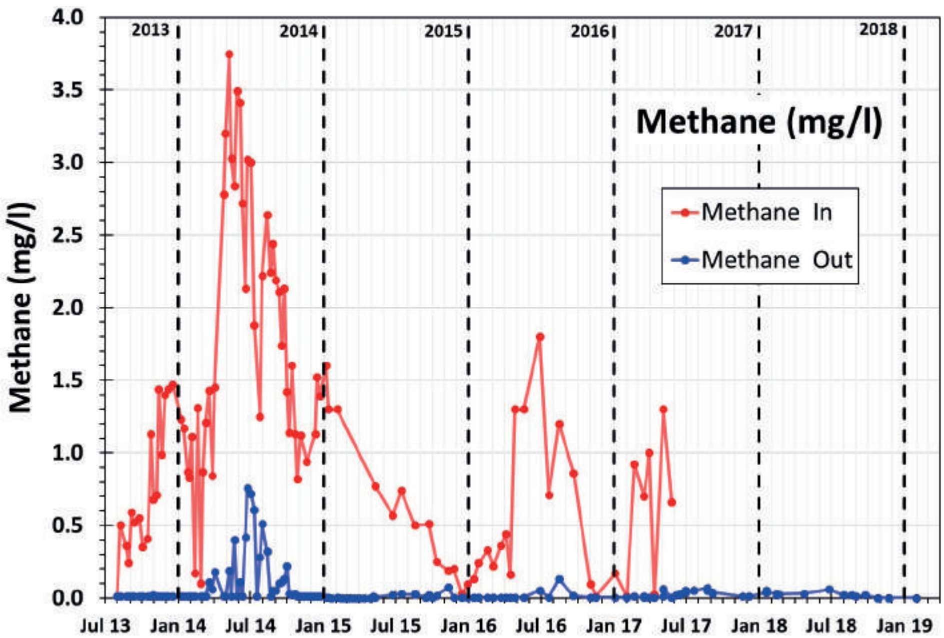

FIGURE 6: Concentrations of methane in raw leachate and following reedbed treatment, between July 2013 to January 2019 (all results in mg/l of methane).

flows following high rainfall during the winter of 2015/16, the methane concentrations within the effluent from the reed bed remained below the consented level.

Figure 7 presents the trends for methane and ammoniacal-N removal rates. This graph highlights that the reed bed is able to provide increased removal of methane when required, during periods of increased leachate flows passing through the reedbed.

Figure 8 displays data for both ammoniacal-N and methane loads that have been experienced by the reed bed between July 2013 and November 2018. It is evident from this figure that periods of increased ammoniacal-N and methane loading are experienced when higher volumes of leachate are being passed through the reed bed. Following the exceedingly high volumes of leachate that were experienced at the reed bed during winter 2013/2014 (mean monthly rates of up to 134.5 m^3/d), seasonal loading rates were much lower and more consistent during following years.

During summer 2017, it was decided that methane concentrations within the raw leachate would no longer be analysed. Instead, only the treated effluent methane con-

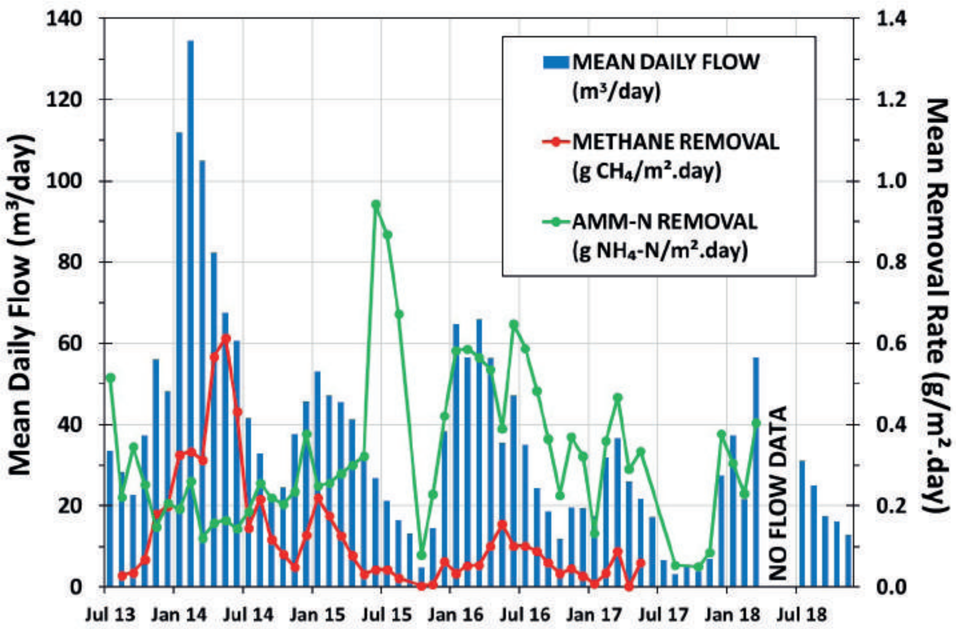

FIGURE 7: Removal rates for ammoniacal-N and methane, compared to the variations in flow rates through the Shirley reed bed system. July 2013 to November 2018.

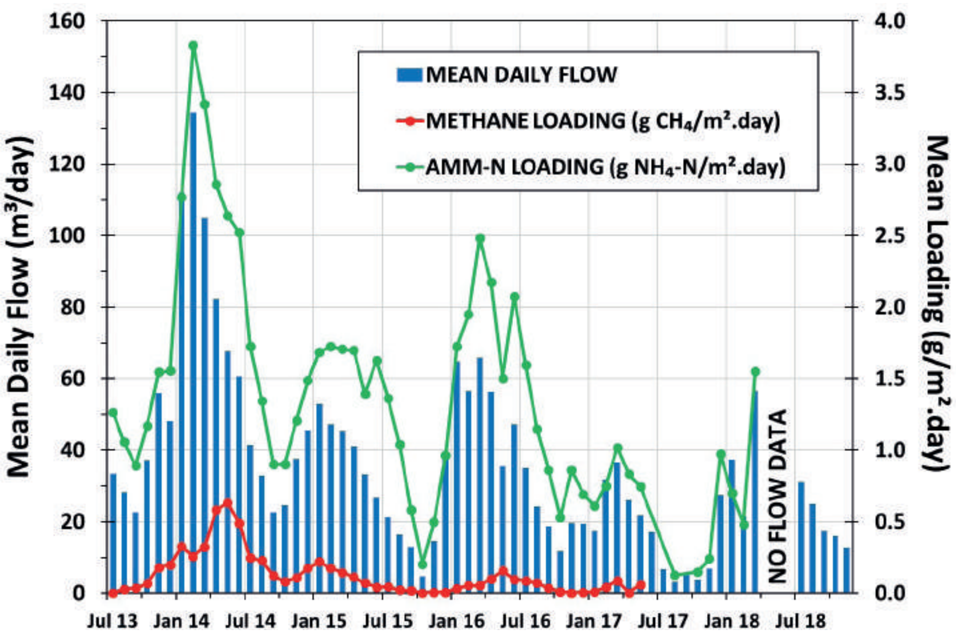

FIGURE 8: Ammoniacal-N and methane loading, compared to the variations in flow rates through the Shirley reed bed system. July 2013 to November 2018.

centrations would be observed. This highlights how successful the system has been, and the fact that the client at Worcestershire County Council is confident in the process reducing methane concentrations within the raw leachate, down to a level below the 0.14mg/l consent. Because of the decreasing concentrations of methane within the leachate, it was predicted that methane concentrations would not exceed 1.5mg/l, from May 2017 onwards.

Because methane samples have not been taken and analysed after May 2017, Figure 7 and Figure 8 do not display data for methane removal or methane loading rates after this date.

5.4 Treatment of other contaminants

Reed bed performance in terms of removal of other contaminants is discussed below. Figure 9 examines changes in COD values through the bed, which have been minimal. Figure 9 shows that influent COD is not impacted by changes in rainfall or seasonal flows, and highlights the overall mean leachate COD concentration of 22.8 mg/l is very similar to the effluent concentration of 20.3 mg/l.

Figure 10 presents results for ammoniacal-N in raw and treated leachates, which demonstrates that (i) there is significant removal of ammoniacal-N, and (ii) ammoniacal-N removal rates are greatest during warmer months, when loadings are reduced (Figure 8). Although concentrations of ammoniacal-N were lower during the period October 2013 to May 2014 (typically between 8mg/l and 11mg/l), removal rates were minimal (<10 per cent), due at least in part to the very high flow rates during this period.

However, during warmer months of each year, when flow rates were also reduced, although ammoniacal-N was typically present at between 12mg/l and 14mg/l, removal rates of up to 50 per cent were achieved during the period July to September 2013, and again during the summer periods of all following years. At slightly greater flow rates during summer 2014, ammoniacal-N removal rates of up to 25 or 30 per cent were still achieved.

Removal of ammoniacal-N was not any part of the specific design of the reed bed at Shirley, but is clearly being achieved to a significant extent during warmer summer months at the following rates:

- Summer: 0.6 to 1.0 gN/m^2.day
- Winter: 0.4 to 0.5 gN/m^2.day

5.5 Removal of iron concentrations

Figure 11 presents results for concentrations of iron during passage through the reed bed. These results suggest that very little iron is present in incoming raw leachate (<0.1mg/l) throughout the period from October 2013 to September 2014. We know that this is not the case, and observation demonstrates significant accumulation of iron oxides and hydroxides as rust-like deposits across the surface of the bed at the inlet end, and to a lesser extent across the entire bed (Robinson et.al, 2015).

The reason that very low iron concentrations were being determined and reported in raw leachates prior to September 2014, was that these samples were being filtered by the laboratory before being tested for iron. Following September 2014, all iron results have been determined on unfiltered samples, to provide more accurate data.

Although high levels of iron-rich sludge deposits accumulated at the inflowing end of the reed bed during the winter of 2014, the outlet end of the reed bed remained clear of these deposits (Plate 3). The high levels of sludge that did reside at the inlet end of the reed bed were easily removed by manual clearing of the surface of the reedbed in the affected areas. This clearing process of the reed bed is not expected to be a regular operation, and in this instance was only necessary following the extreme flow rates after the heavy rainfall experienced during the winter of 2013/14.

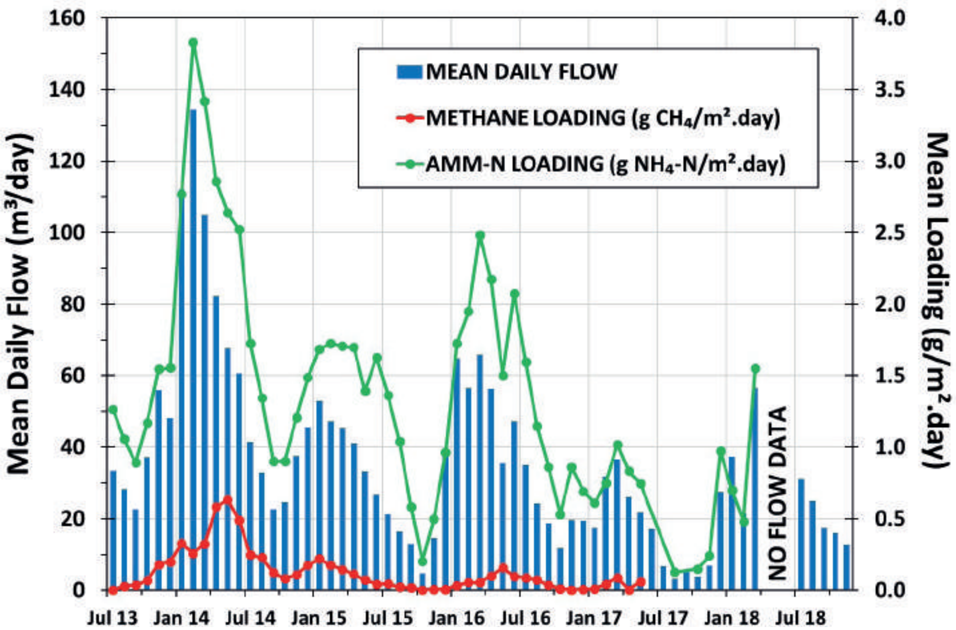

FIGURE 9: Variation in COD values during passage through the reedbed, July 2013 to December 2018 (all results in mg/l of COD).

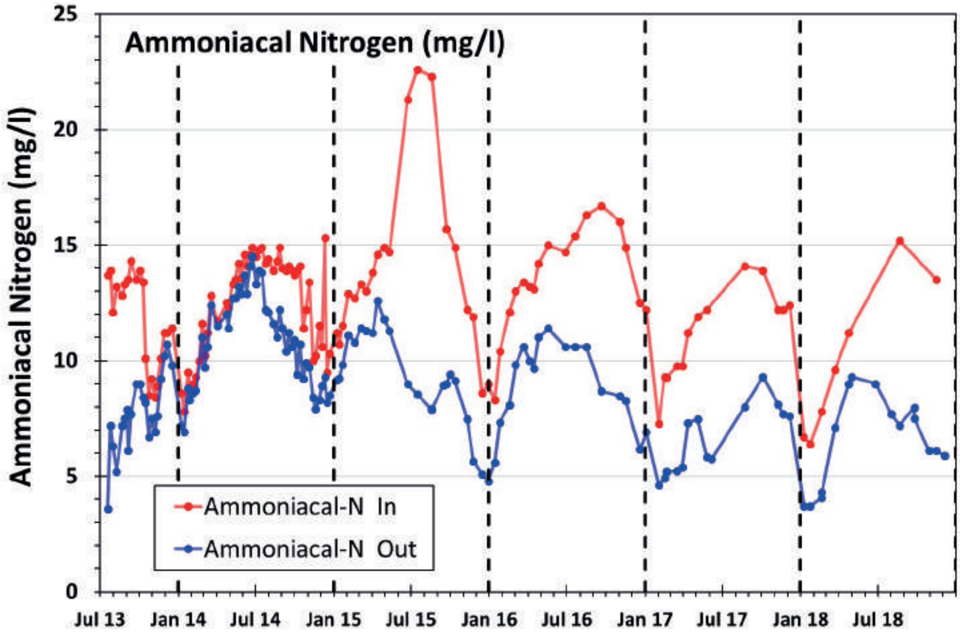

FIGURE 10: Variation in concentrations of ammoniacal-N during passage through the reedbed, July 2013 to December 2018 (all results in mg/l as N).

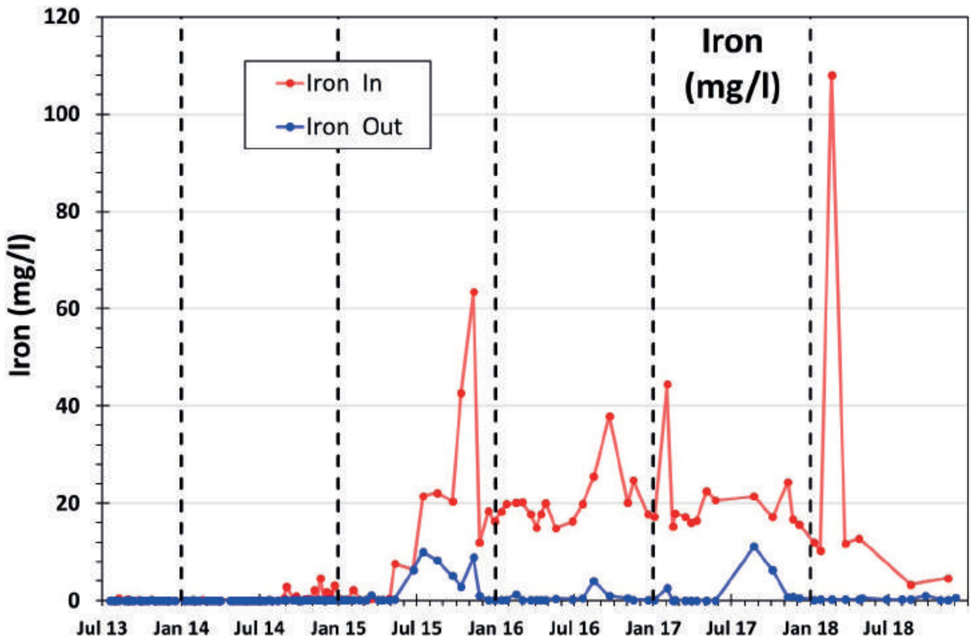

FIGURE 11: Variation in removal rates of iron following passage of leachate through the reed bed (all results in mg/l).

6. RESULTS AND DISCUSSION

The Shirley reed bed has performed well since commissioning during 2013, removing all methane from leachate entering it, achieving successful removal even when flows were more than double design rates during early 2014. The reed bed is very successful at removing high initial levels of dissolved methane (95% removal), as per the intended requirements. This ensures that methane remains well below the 0.14mg/l discharge consent.

As presented in Figure 8, the loading rate for methane into the reed bed has gradually reduced over time, allowing complete removal to be achieved. This is as a result of lower concentrations of methane within the incoming leachate from Shirley landfill in recent years. Figure 7 shows that when, in 2014 the methane concentration within the leachate was greater than 3 mg/l, and the daily flow of leachate was above 80 m^3/day, methane removal rates exceeded 0.6 g CH_4/m^2/d. During 2015 and 2016, maximum removal rates peaked at 0.2 g CH_4/m^2/d, when methane concentrations were between 1 mg/l and 2 mg/l, during periods when incoming leachate flow rates were between 40

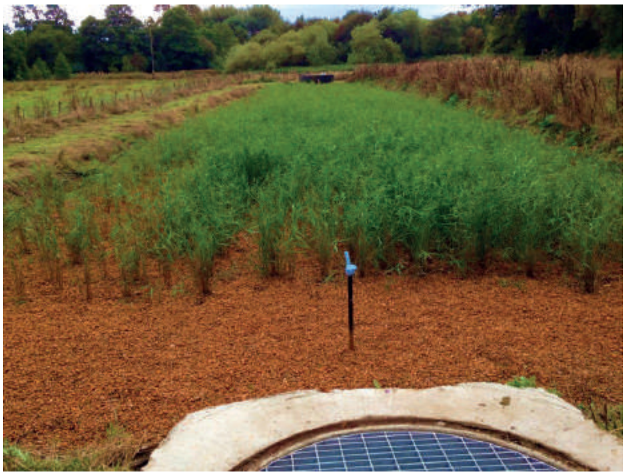

PLATE 3: Outlet end of the reed bed at Shirley during July 2014, showing very clean gravel and good reed growth.

PLATE 4: Outlet end of the reed bed at Shirley during Summer 2016, showing excellent reed growth.

and 70m^3/day. Because of the gradual reduction in methane production by the landfill, it has been determined that the methane concentrations within the incoming leachate have remained lower than 1.5 mg/l ever since 2017. Therefore, it has not been necessary for the methane removal rate to exceed 0.1 g CH_4/m^2/d, for complete removal to be maintained.

Because the reed bed has been removing methane so reliably for more than six years, Worcestershire County Council have decided that they no longer require analytical data for methane concentrations within the raw leachate. The council are satisfied to observe the methane concentrations within the outflowing effluent from the overflow discharge chamber at the end of the reed bed, which consistently removes methane to below the 0.14 mg/l discharge consent.

Seasonal removal of ammoniacal-N has taken place within the reed bed (up to 50 percent removal between 2013 and 2018), but this was not part of the design purpose of the bed. Nevertheless, as more data are obtained, it will be possible to obtain useful loading rate data for this removal.

Figure 7 demonstrates that removal rates of ammoniacal-N through the reed bed exceeded 0.9g NH_4-N/m^2/d during the summer of 2015, when raw leachate concentrations were between 20 mg/l and 25 mg/l. During following summer periods, maximum concentrations of ammoniacal-N within raw leachate were closer to 15mg/l, whilst removal rates peaked at 0.64 g NH_4-N/m^2/d during 2016, 0.47 g NH_4-N/m^2/d during 2017, and 0.4g NH_4-N/m^2/d during 2018. This gradual reduction in ammoniacal-N removal rate is due to the reduced flow rates of leachate through the red bed since 2016.

The reed bed has performed very well indeed, although extreme rainfall conditions during early 2014 caused flows of up to 160 m^3/d that were more than double maximum design values. Later in that year, concentrations of dissolved methane in the leachate reached 3.8 mg/l. This is nearly twice the reed bed design value of 2 mg/l, and breakthrough of methane within the effluent to levels of 0.7 mg/l was noted for a short period, until methane levels in leachate returned to 2 mg/l. Methane within the final effluent produced by the Shirley reed bed has consistently achieved the discharge consent set by the Environment Agency since this period in 2014, even when flow rates through the reedbed have exceeded the design parameters.

7. CONCLUSIONS

The Shirley reedbed treatment system demonstrates that a well-designed, constructed and operated plant is able to operate consistently, reliably, and cost-effectively, to meet stringent effluent discharge standards for dissolved methane at all times. The landfill owner has found the passive reed bed system to be cost-effective compared with other options such as tankering or methane stripping. The detailed operating data that this paper provides should give great confidence to both treatment plant operators, and to landfill regulators.

The reed bed continues to perform well, removing all methane from leachate entering it, following a brief period when flows were more than double design rates during early 2014. However, when concentrations of dissolved methane reached 3.5 mg/l, almost double design assumptions, later in the year, some dissolved methane was detected in treated leachate, with a maximum concentration of 0.7 mg/l recorded in early July 2014. As methane levels fell below 2.0 mg/l, essentially complete removal was again achieved.

This successful removal of methane has been achieved consistently for more than 5 years since summer 2014 and has continued through to the time of writing (summer 2019), despite the occasional increase in flow rates above the design capacity for the reed bed.

Some seasonal removal of ammoniacal-N continues to take place through the reed bed, however this was not part of the design purpose of the reed bed. Nevertheless, as more data are obtained, it will be possible to obtain further useful loading rate data for this removal.

Removal of iron, and potential accumulation of iron within the bed over the longer term, are being monitored. It is likely that a high proportion of that iron which has entered the bed to date comprises iron dislodged from the landfill drainage system during extreme rainfall events in early 2014. Some modifications to the raw leachate header tank arrangements are being considered, including introduction of compressed air into the pipeline transferring raw leachate into it, to encourage better oxidation and settlement of the iron within it. Prevention of iron accumulation in the bed by use of a header tank settlement arrangement has been implemented, and the reed bed design adopted at Shirley has great potential for adoption at many similar old and closed landfill sites.

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ASSESSMENT OF MICROPLASTICS IN THE ENVIRONMENT - FIBRES: THE DISREGARDED TWIN?

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ABSTRACT
The pollution of plastics into the environment is considered one of the major challenges of the current century. In particular, microplastic pollution is considered a significant threat to both human and animal life since removal is almost impossible once these plastic particles make their way into the environment. Unfortunately, when scholars evaluate microplastic contamination in the environment, synthetic fibres are often disregarded. This approach is flawed, considering that a large part of clothing consists purely of synthetic fibres, meaning they are omnipresent in every part of human activities along with emissions. This work takes a critical view of state-of-the-art analysis methods for microplastics in soil, water and air, with a special focus on their ability (or inability) to detect fibrous materials. A case study in the form of a critical evaluation was carried out to highlight common problems when detecting microplastic fibres, focusing primarily on the sampling of large volumes of water. Another case study explores microscopy as a means to analyse solid specimens regarding microplastic contamination. Furthermore, the sources of fibre pollution and the pathways they take in the environment before ending up in the maritime system are explored. Finally, this work aims to create and enforce standardised methods addressing microplastic pollution, which would potentially solve many of the current associated problems.

1. INTRODUCTION

One of the major challenges of this century is environmental pollution. In recent years, the scientific community has also focused on microplastics. While increasing efforts are taken in sampling and identification methods, microplastics seem to be found nearly everywhere – in soil, water, air, food and even purified drinking water (Pivokonsky et al., 2018). Every place humans have touched, we have left behind our presence in the form of these tiny microplastic particles and fibres. Even in remote places, such as glacier regions (Ambrosini et al., 2019), these particles have been found before. It is probably even possible that we left behind microplastics at the 1969 moon landing. Now that we have reached Mars, it is possible one could even say with some tongue-in-cheek humour that our microplastic problem has reached the interplanetary scale. However, apart from jokingly considering plastic waste in space, NASA has been considering using plastic waste to compress and immobilize garbage in spacecrafts (Fisher et al., 2008).

One of the first indications of terrestrial plastic pollution in sea water dates back to 1972 (Carpenter & Smith, 1972). The authors sampled particles in the size range between 2.5 and 5 mm from the Sargasso Sea surface. The term "microplastics" itself was mentioned in 2004 (Thompson et al., 2004) without, however, specifying a size range. On the one hand, the authors found granular material. On the other hand, the major fraction turned out to be fibrous and approximately 20 µm in diameter. Since then, a large number of studies have been published, but there is no common agreement about the size range of microplastics (Hidalgo-Ruz, Gutow, Thompson, & Thiel, 2012). Even if it has been suggested to define microplastics as particles below 5 µm (Arthur, Baker, & Bamford, 2009), an elaborate review revealed that the particle size of microplastics ranges between 1 µm and 20 mm for sediment particles and 0.5 to 29 mm for sea surface particles (Hidalgo-Ruz et al., 2012). The German Umweltbundesamt (Miklos, Obermaier, & Jekel, 2016) suggested using the terms listed in Table 1 to specify plastics in the environment in regard to their size, including nanoparticles (Comission, 2011). According to this definition, microplastics are smaller by a power of ten compared to the abovementioned 5 mm as an upper limit. However, the Austrian Umweltbundesamt distinguishes between large (1 to 5 mm) and small (1 µm to 1 mm) microplastic particles (Liebmann, 2015). Again, the sugges-

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TABLE 1: Definition of microplastics as suggested by the German Umweltbundesamt c.

Term for	size range
Macroplastics	> 5 mm
Mesoplastics	0.1 to 5 mm
Microplastics	0.1 to 100 µm
Nanoplastics*	< 100 nm*
PE	< 0.5
PS	< 0.5

According to the EU definition of nanoparticles (Comission, 2011).

tions differ by an order of magnitude.

The quantification of microplastics is handled just as inconsistently as the size range is handled. As shown in an elaborate review (Hidalgo-Ruz et al., 2012), microplastics are not only quantified as a number of items (-) or mass (g) but they can also be related to the area (m^2), volume (m^3), length of strandline (m) or mass of sediment (kg).

Even if a large number of studies are available, it is more or less impossible to compare results from different sources. A well-accepted standardization of sizes and units in the field of microplastics on an international level is a prerequisite for making any study results comparable.

In principle, one can distinguish between two types of microplastics, primary and secondary (Miklos et al., 2016). Primary microplastics are produced intentionally in small sizes for certain products, such as cosmetic products or detergents and cleaners. Secondary microplastics are generated from larger particles (macroplastics and mesoplastics) by physical, biological and chemical degradation.

There are many ways these microplastics may be emitted into the environment. For example, microplastics can end up in the maritime system through littering or by rain and flushing into rivers (GESAMP, 2015). These small particles are likely to be ingested by animals and will accumulate in the food chain, and then eventually in human food (Hantoro, Lohr, Van Belleghem, Widianarko, & Ragas, 2019). Most polymers in plastics are not a risk to human health directly and will likely just pass through the gastrointestinal system; however, the toxic additives in or adsorbed to the plastic materials may potentially leach out, be stored in tissue, accumulate and reach harmful levels (Carbery, O'Connor, & Palanisami, 2018). As these particles present a large surface area, they are also able to amass "persistent organic pollutants" (POPs), such as pesticides, industrial chemicals and burning residues from the environment. Therefore, while the materials themselves are harmless, additives and POPs may harm humans and the biosphere (Hantoro et al., 2019). This issue is well known, but the concrete impact on human health is not yet fully understood. However, the precautionary principle obligates us to search for solutions immediately.

Microplastics can enter the human body through the food chain; however, they can also enter the human body through the respiratory system. Fine particulate dust consists of a broad spectrum of materials. In particular, rubber particles, which are a large part of the dust fraction, can be considered microplastics (Liebmann, 2015). These rubber plastics, and other fibrous materials, are likely to be bio-persistent; however, most of them are suspected to be mucociliary clearable. Nevertheless, they may persist in the lungs and could cause long-term health effects (Gasperi et al., 2018).

There was major public disapproval when the issue of microplastics in cosmetic products, such as shampoos and toothpastes, was revealed for the first time, resulting in many companies committing to product changes (Park, 2016). On the other hand, the emission of plastic fibres through, for example, washing machines has yet to receive the same amount of attention, despite being even more relevant based on the emitted masses of microplastics. Based on a recent study in Germany, microplastics in cosmetic products account for 19 g/(cap*a), while fibre emissions from clothing accounts for 77 g/(cap*a) (Bertling, Hamann, & Bertling, 2018). Examining the instruction manual for a popular brand of washing machine, it is even suggested to just rinse the fibres out of the filter after it gets too clogged (Miele, 2019). Fibres are not officially disregarded in scientific research, but due to the ability of fibres to easily pass through sieves with wider meshes, they are often not detected, as described in the next paragraphs.

There is quite a broad spectrum of materials and morphologies applied when discussing fibres. Fibres used for textiles and clothing, which represent their major applications, are almost exclusively composed of polymers. In 2018, global fibre production reached 105.6·106 t (Chemiefaser, 2018). On the one hand, so-called natural fibres are derived from crops or animals. The most prominent representative natural fibre is cotton, with a volume of 26.9·106 t in 2018 (Council, 2018), based on the natural polymer cellulose. On the other hand, fibres can originate from technical processes, so-called man-made fibres, regardless of their material origin. Both natural (mainly cellulose) and synthetic polymers (e.g., polyester, polyamide, polypropylene, etc.) are used (BISFA, 2017). The volumes of fibres based on synthetic polymers are much higher (66.6·106 t in 2018 (Chemiefaser, 2019a) compared to fibres based on cellulose (6.8·106 t in 2018; (Chemiefaser, 2019a)). It should be further considered that among synthetic polymer fibres, PET plays a predominant role. In 2017, 53.7·106 t of PET fibres (Chemiefaser, 2019b) was produced, which means a share of 82.7% compared to a total of 64.9·106 t (Chemiefaser, 2019a) of synthetic polymer fibres.

Both fibre categories, man-made and natural, that are based on cellulose origins are biodegradable (Sular & Devrim, 2019). These fibres do not represent any specific threat to the environment and do not account for microplastics. However, considering synthetic polymer fibres only, the potential of these fibres being released into the environment is 66.6·106 t annually (Chemiefaser, 2019a). This represents a large volume, however, it is a smaller amount than the total amount of plastics, which reached 348·106 t in 2017 (PlasticsEurope, 2018). As mentioned above, it must also be considered that PET is by far the predominant polymer (53.7·106 t or 83 % (Chemiefaser, 2019b)), while 53 % (i.e., 184·106 t) of total plastics are polyolefins (Prata, 2018).

However, no generally valid statements can be made about fibres released into the environment, as fibres can be present in different forms. Table 2 shows different terms for fibrous structures, and it provides an estimation of their potential to release microplastics into the environment. As the chance for environmental pollution increases, the shorter and thinner the fibres are. The situation is, however, even more complex because fibres in textiles can be damaged due to abrasion (Textor, Derksen, Bahners, Gutmann, & Mayer-Gall, 2019) during their use and can then emit plastics into water (washing machines) or air.

2. ASSESSMENT METHODS

"The four basic factors which affect the quality of environmental data are sample collection, sample preservation, analyses and recording" (US Environmental Protection Agency, 1982). Although sampling guidelines have existed for more than 40 years, there are still improper actions in several areas of microplastics assessment , which may result in questionable conclusions derived from insufficient data. Hence, there is still a need for new methods and guidelines on sampling, preservation and analysis that address the specific characteristics of microplastics (GESAMP, 2015; Prata, da Costa, Duarte, & Rocha-Santos, 2019). Sample collection has the biggest impact on generating correct and reproducible results, however, no proper guidelines exist, and most works seemingly developed "their" methods through adaption to the specific sampling problem. Therefore, it is very hard to suggest a proper method that addresses every research case. However, the analysis of the samples can possibly be standardized, and that is the main topic of this work.

2.1 Solid samples

After sampling, it is important to separate the microplastic particles from other materials in the specimen. Mostly, methods such as optical-haptic separation, sink-float separation, flotation, sieving and chemical etching are used. Depending on the sample, the order of these procedures can vary from case to case. A review of the related literature shows that the analysis method of choice seems to be based on sink-float separation with additional chemical treatment beforehand or afterwards to eliminate organic components. Afterwards, the remaining plastic particles are usually identified with spectroscopic methods (D. He et al., 2018; Mai, Bao, Shi, Wong, & Zeng, 2018; Prata et al., 2019).

An exemplary process could consist of dispensing the sample in a heavy liquid, resulting in the separation of the floating plastic particles from sand and earth. Then, the floating particles are filtered from the liquid. Afterwards, the particles are weighed and analysed by a light microscope and/or near infrared spectroscopy. To assess heavy polymeric materials, such as polyesters (density ~1.4 g/cm³), solutions need to have a significantly higher density. Unfortunately, the liquids used are often very expensive and potentially toxic, or they have a density very close to 1.4 g/cm³. Furthermore, different publications use different separation fluids, which makes comparison problematic. Considering that polyesters are the most widely used polymers in fossil-based plastics or in synthetic fibres (Eyerer, Hirth, & Elsner, 2008), it is even more important to put a greater degree of effort into the development of new separation methods.

Agglomeration, biofouling and small particle sizes, in general, can be problematic. Air bubbles and/or agglomerates with plastic fibres may even float sand particles that are denser than the separation liquid or they may, vice versa, sink the plastic particles (see Figure 1). Thus, this step seems to be very critical. In addition to adding surfactants to limit these effects to a certain extent, ultrasonic treatment, for example, could be another option since its effectiveness for breaking agglomeration has been shown in other cases (Kusters, Pratsinis, Thoma, & Smith, 1993).

2.2 Liquid Samples

Analysis methods for liquid samples are mainly used to assess microplastics within samples from rivers, lakes, seas, sewage treatment effluents and washing machine wastewater outlets. The state-of-the-art techniques use sieving as the primary step with additional chemical treatment afterwards or even as the sole procedures for liquid samples (Napper & Thompson, 2016; Pirc, Vidmar, Mozer, & Krzan, 2016; Sun, Dai, Wang, van Loosdrecht, & Ni, 2019; Talvitie, Mikola, Koistinen, & Setala, 2017; Talvitie, Mikola, Setala, Heinonen, & Koistinen, 2017; Ziajahromi, Neale, Rintoul, & Leusch, 2017).

However, using sieving or filtration processes on particles and fibres is rather complex when a vast amount of water needs to be analysed because of low particle concentrations. Generally, there are two major challenges in using sieve analysis, namely, the choice of proper mesh sizes and the guarantee for a sufficient amount of accuracy of the mesh geometry. A multiple sieve stack will pro-

TABLE 2: Terms to specify different types of fibres according to their morphology (BISFA, 2017).

Term	Definition	Remarks in regard to relevance to microplastics
Filament	A fibre of very great length	Most likely not released during normal use or washing
Staple fibre	Textile fibre of limited but spinnable length	Release during normal use or washing possible; too large to be airborne
Flock	Very short fibres, intentionally produced for other purposes	Risk to be released during normal use or washing
Fibril	A subdivision of a fibre can be attached to the fibre or loose	High risk to be released during normal use or washing; loose fibrils most likely airborne
Fibre fly	Airborne fibres or parts of fibres (light enough to fly), visible as fibres to the human eye	Extreme high risk to be released during normal use or washing; airborne by definition

FIGURE 1: Partially floating quartz sand commingled with added polyester fibres and polypropylene granulate (model mixture) after sink-float separation in a heavy liquid; separation efficiency poor although accurate adjustment of density.

vide a more precise size analysis but proportionally lower material quantities at each mesh, leading to uncertainties during weighing. The second issue is particle deposition on the mesh surface, which can cause time-dependent changes in separation characteristics (see Figure 2). Often, even a small amount of material deposited on a sieve leads to blocking and a high pressure drop. This is particularly true for small mesh sizes. Thus, sieves featuring narrow meshes are often neglected. Consequently, a compromise must be found in sieve analysis between throughputs and measurement accuracy.

Microplastics in the aquatic environment have been studied extensively using these sieve analyses. Different sampling techniques and the use of sieves with different mesh sizes in particular result in data sets that are not comparable. Although many studies vary in sampling methods, some general statements can be made. The smaller the sieve mesh sizes are, the higher the amounts (or masses) of microplastics particles and fibres found (Murphy, Ewins, Carbonnier, & Quinn, 2016; Napper & Thompson, 2016; Talvitie, Mikola, Koistinen, et al., 2017; Talvitie, Mikola, Setala, et al., 2017; Ziajahromi et al., 2017). Regarding small fibre fragments (or flakes), such fragments were detected only where the authors used sieves of a mesh size of 25 µm or smaller for their experiments (Adam, Yang, & Nowack, 2019). High amounts of short fibres arise from textile handling, such as machine washing and tumble-drying. Here, the mass of fibres found on a 200-µm sieve was only a few milligrams (Pirc et al., 2016). Other studies on the washing of synthetic jackets, however, retrieved up to 2 g of fibres in a sieve stack with 333-µm and 20-µm screens (Hartline et al., 2016).

2.3 Air Samples

Be it in the office, outdoors next to a busy street, or in the countryside far from urban areas, microplastic fibres are steadily present in the air with varying concentrations. With every breath we take, we also inhale small amounts of these fibres into our respiratory system. We can ingest them even with the food we consume since these tiny particles can easily agglomerate on the food's surface (Gasperi et al., 2018).

Inside the human body, these fibres can penetrate deep into the lung, resulting in impaction and clearance problems. Fibres with lengths up to 250 µm were already found inside the deeper regions of the human lung (Prata, 2018). Due to their large surface areas, the deposited fibres are able to interact with organic material, which makes their removal by the human body complicated, and they may also release hazardous substances, such as the POPs addressed in the introduction (Rochman, 2015). Therefore, it is important to properly detect high-risk areas where the fibre and dust concentrations, or intake levels, become critical, e.g., in industrial environments, or when exposed to them for extended periods, such as in offices or even at home in bedrooms. For example, for those involved in the fibre production industry, the amount of lung disease is alarmingly high and it is significantly higher than for the rest of the population (Prata, 2018).

However, microplastic pollution from air has been discussed only in recent years. Air samples are usually obtained by standardised air samplers (PM10 high volume air samplers or small personal samplers) using absolute filter media to separate any particulate matter from the air flow throughput (Gasperi et al., 2018). To date, multiple studies have been able to detect fibrous materials in different environments. Microscopic and spectroscopic techniques were used afterwards to differentiate shape, e.g., fibres and particles, and the material, e.g., synthetic and natural. It was found that urban sites reach particle deposition rates up to 355 particles/m²*day, whereas suburban sites experience only approximately a third of that value (Rachid Dris, Gasperi, Saad, Mirande, & Tassin, 2016).

Additionally, assessment methods to determine indoor and outdoor fibre concentrations using vacuum cleaners are important. Settled dust was collected from the floor, and the captured dust specimens are then analysed by density separation techniques, as described in the previous chapter. Indoor fibre concentrations of approximately 1 to 60 fibres/m³ and outdoor fibre concentrations of approximately 0.3 to 1.5 fibre/m³ were found (R. Dris et al., 2017). An interesting result is that only approximately one-third of the indoor dusts were actually synthetic materials.

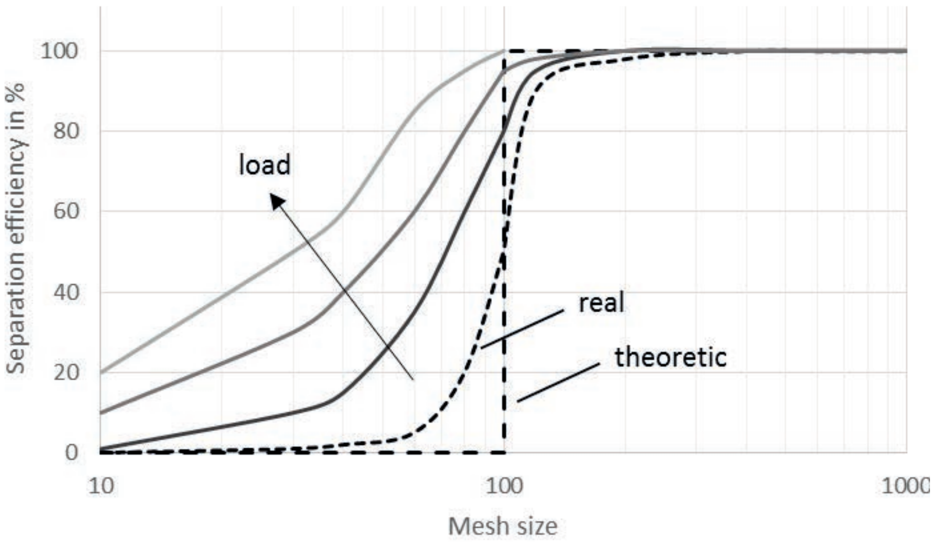


FIGURE 2: Separation efficiency curves of sieve meshes; low-dimension particle analysis is complex due to limited selectivity.

It should also be mentioned that these studies gather data from a few measurement points only and the contamination data are extrapolated to the scale of whole city areas. For example, it was calculated that 3 to 10 tons of fibres are deposited in the Parisian city region every day (R. Dris et al., 2017).

However, verification of these theoretical calculations is difficult. When dealing with measured values of the dust concentrations of airborne particles, the sampling methods are highly sensitive to differing environmental conditions. Wind, temperature, humidity, rain and even the location of the air sampler cause high fluctuation rates within the results. Furthermore, to measure very low particle concentrations (e.g., 0.3-1.5 fibres/m²), extended sampling times are needed to attain a number of particles that is clearly sufficient to make any kind of statistically sound assumptions. Understandably, this results in a high level of uncertainty in the measurement. For this reason, scholars suggest that the number of measurement points and repetitions should be higher to better extrapolate results to an entire city area (Almeida et al., 2007).

3. CRITICAL EVALUATION OF WATER SAMPLE ANALYSIS USING SIEVES

3.1 Evaluation of liquid sampling using sieves

To better illustrate the issue with these measurement techniques, the sieving operation for the samples was critically evaluated. As mentioned in the previous chapter, sieving is always the first step for analysis and it is regarded as the go-to method in the analyses of liquid samples, such as outflows of waste water treatment plants or the washing water of a commercial washing machine.

The main problem with sieving is not the method itself, but the use of inadequate mesh sizes, as already mentioned in the previous chapter, which results in a wide range of mesh sizes used, with most ranging between 20 µm and 300 µm. Considering that the width of a fibre is typically below 10 µm, it is very likely that even long fibres (>40 mm) can easily pass the wider meshes. To demonstrate this, the following tests were performed.

3.2 Materials and Methods

3.2.1 Materials

- Viscose Staple Fibres
- Viscose fibres were purchased with a titer of 1.7 dtex and a length of 40 mm.
- Lyocell Staple Fibres
- Lyocell staple fibres were purchased with a titer of 1.7 dtex and a length of 40 mm.
- Arbocel Fibre Dust

Arbocel FIC200 cellulose fibre dust was purchased, but titer and length distributions had to be determined with the MorFi optical fibre analyser due to a lack of exact specifications upon arrival. (See Figure 3)

3.2.2 Methods

Fibre Length Adjustment

Both the lyocell and the viscose specimen had a staple length of 40 mm and therefore needed to be shortened. This simulates short broken fibres as they might be found in a waste water stream. To shorten the fibres, a Hosokawa Alpine Rotoplex 20/12 RO cutting mill with 5.5 kW was used in conjunction with a 0.5 mm trapezoid cutting mill mesh and a 3 mm cutting mill mesh.

Three different fibre fractions were produced with this method:

- “0.5 mm” Lyocell
- “0.5 mm” Viscose
- “3 mm” Viscose

Fibre Length Analysis

To evaluate the length of the created fibre samples, an optical fibre analyser called MorFi from Techpap was used. For the analysis, a 0.2 g sample of the fibres to be analysed is suspended in 0.5 litre water and thoroughly mixed with a fibre mixer that is part of the MorFi measuring system. The

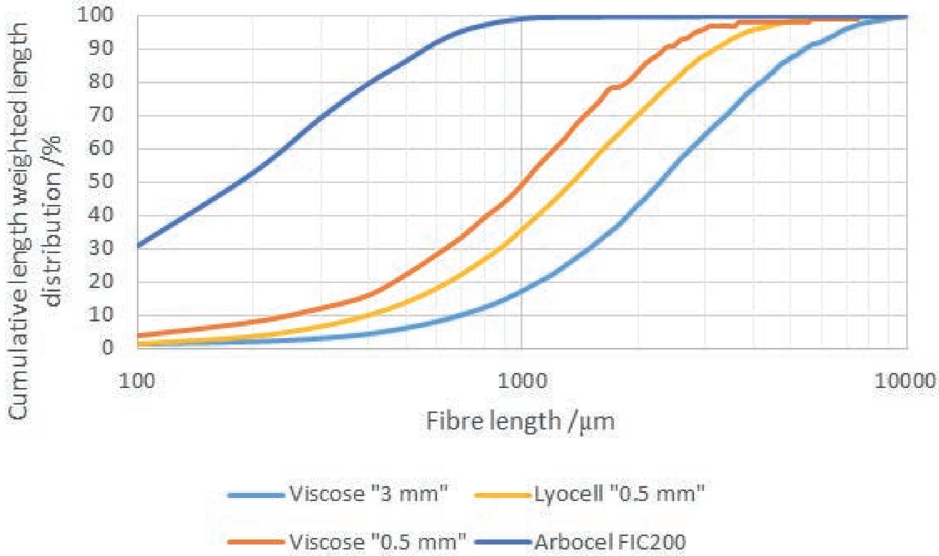


FIGURE 3: Length weighted fibre length distribution of the four samples

suspended sample is then poured into the sample reservoir of the MorFi, and the measuring device pumps the sample through a closed loop system from the reservoir back into the reservoir. While being pumped through that loop, the sample passes an optical analysis cell, where pictures of the fibres are taken. The pictures are analysed by a built-in algorithm to generate length-weighted fibre length distribution and a number-weighted fibre width distribution results. The measurements were repeated three times.

Weight Analysis

The weight of the fibre residues on the sieves was determined by first drying the whole sieve stack at 110°C for 24 hours and subsequently weighing each sieve with a Sartorius GP4102 laboratory scale with a maximum weight of 4100 g and a readability of d = 0.01. By subtracting the dry weight of the clean sieve before the measurements, the exact fibre residue weight per fraction can be determined.

3.3 Experiments, Results and Conclusion

For the evaluation of the sieving procedure, fibre samples were prepared from staple cellulosic fibres, more precisely, from viscose fibres and lyocell. Additionally, Arbocel fibre dust was used to simulate very short fibre fragments. While those materials are not plastic fibres, they do have the same shape and dimensions compared to their synthetic counterparts. They are also easier to handle in a laboratory setting since they are easier to disperse in water, which is essential for accurate measurements with the MorFi. The samples were prepared by grinding the staple fibres (length approximately 40 mm) in a cutting mill. By using different mesh screens within the mill, the three previously mentioned fibre samples were generated.

Before a sieving evaluation could be performed, the exact fibre length distribution of the four samples had to be determined. This was essential to being able to check which length fractions of fibres would be able to pass through a sieve with a certain mesh size. To evaluate the length of the created fibre samples, an optical fibre analyser MorFi from Techpap was used. The length-weighted fibre length distribution of the Arbocel, lyocell and the viscose samples are presented in Figure 3. The four fibre samples have very distinct distributions, with Arbocel on the shorter end with fibre lengths below 800 µm and on the longer end is viscose “3 mm” with fibres up to 10 mm.

To evaluate the retention capacity of different meshes, a sieve stack containing a 25 µm, a 100 µm and a 333 µm mesh sieve (Retsch test sieve according to ISO 3310-1) was chosen because sieves with these three mesh sizes are seemingly the most commonly used in the previously cited works. For the experiments, 10 g of fibres was suspended in 20 L of water and then slowly poured onto the sieve stack. The sieve stack was dried as a whole at 110°C in a climate cabinet for 24 hours until constant mass was achieved. The fibre mass deposited on every sieve was determined by weighing the corresponding sieve and subtracting the original sieve mass. The fibre lengths were measured for each fraction once again with the MorFi fibre analyser by analysing a 0.2 g sample of the sieve residues. This whole procedure was repeated for each of the four different fibre samples. The following distribution parameters (x values) were calculated from the data of each experiment and then combined as an average value.

In Figure 4, the average length-weighted fibre length (presented as quantiles $x_{10,1}$ $x_{50,1}$ and $x_{90,1}$) of each fraction is shown. The $x_{50,1}$ represents the median length-weighted fibre length of the residue on a certain sieve. As clearly shown in the figure, these median values for all used sieves are as high as the mesh size of the respective sieve with the next-highest mesh size within the stack. The $x_{90,1}$ represents the share of the fibres with the highest lengths on a sieve. It shows that fibres considerably (up to ten-fold) longer than the mesh size are able to pass through a certain sieve. The $x_{10,1}$ represents the smallest fibres within the deposited fibres on a sieve. Since these values are approximately equal to the mesh size of the respective sieves, no considerable amount of fibres with a shorter length were retained.

These results suggest that, for most cases, no fibres shorter than a certain mesh size are likely to be retained. Thus, if a collection of fibres below the evaluated mean lengths is necessary, sieves presenting mesh sizes in the respective range should be implemented.

However, already using a sieve with a mesh size of 25 µm, it was apparent during the experiments that the achievable flow rate is already very low; hence, a sampling of thousands of litres of water would be impossible with an even smaller mesh would be needed to sample highly diluted sources, such as waste water effluent (Murphy et al., 2016; Prata et al., 2019). Therefore, sieving with 25 µm mesh size could be a good compromise between quality and quantity.

4. HIGHLIGHTING DIFFICULTIES DURING OPTICAL ANALYSIS OF SAND SPECIMENS

Many approaches to assess contaminants in soil or water samples are rather complex. Quite often, many steps need to be taken, and the chance for errors is high despite the efforts to avoid this issue, as has been previously stated. It is therefore advisable to lead with a straightforward method. In this regard, one possible method to obtain a fundamental idea about whether a certain level of contamination is to be expected – especially suitable for solid samples – is optical imaging analysis. This method was performed in this work for various sand specimens to highlight the difficulties in such a procedure.

4.1 Materials

4.1.1 Beach sand

The samples were taken in Sardinia next to the Forte Village Resort, the venue of the Sardina Symposium in 2017. Tidal activity is known to distribute and/or accumulate items of various sizes across shorelines. The sampling position was defined at the mean sea water line as well as two positions with a 3 metre lateral offset to this baseline. In this fashion – and for the prevailing conditions at sampling (i.e., daylight, no rain, estimated Beaufort number 2 to 3) – one specimen normally without contact with the sea water, one with intermittent contact and one covered approximately full time were produced. Removing approximately 5-10 cm of the upper sand layer, samples of approximately 100 ml were taken at each position with a screw-on container that was closed immediately after sampling to avoid any additional contamination. It was ensured that the water line had retired beyond the respective positions before sampling. The procedures were repeated by two experimenters.

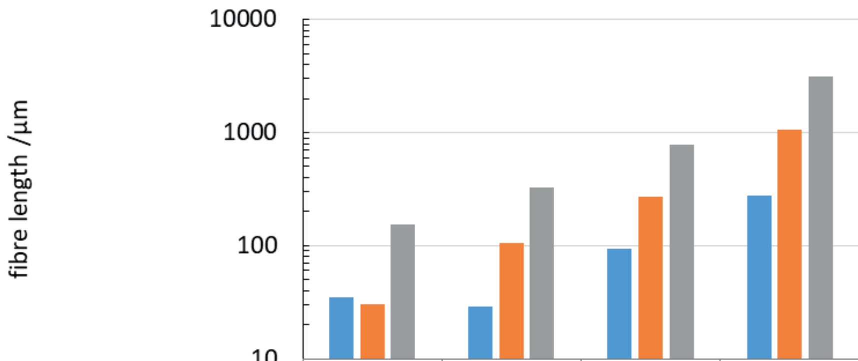
4.1.2 Synthetic sand mixture

To highlight the difficulties in optical analysis of sand specimens, two synthetic sand mixtures were produced. The first mixture consisted of 10 g of beach sand, which was intentionally polluted with 0.1 g PP powder (<100 µm). A second sand mixture was once again prepared with 10 g beach sand and 0.1 g of pink PET microfibres (<1 dtex). Each sample mixture was intensely mixed, and a small specimen of each sample mixture was taken for optical analysis.

4.2 Optical analysis

An optical image analysis with a digital light microscope VHX-7000 from Keyence (magnification between 2 and 2000) was carried out after spreading 1 g each of the sand samples on a microscope slide. Several measurement points on the slide were chosen and investigated in detail.

The images showed that the samples had varying morphologies and colours of the granules, with mean diameters ranging from approximately 0.2 – 5 mm, with a tendency to observe larger particles for the sampling positions closer to the waterline. The colour varied from transpar-



	<25	25-100	100-315	>315
x10,1 fibre length /µm	35	29	94	278
x50,1 fibre length /µm	30	105	273	1055
x90,1 fibre length /µm	156	326	779	3169

FIGURE 4: Average distribution quantiles (x10, x50, x90) of fibre length for different sieve fractions.

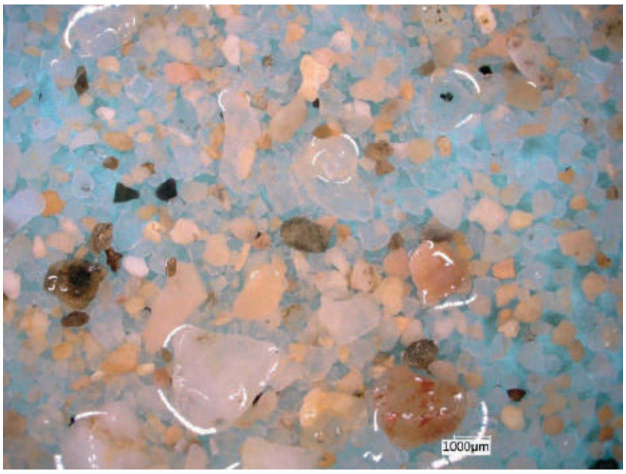


FIGURE 5: Light-microscope image of a beach sand sample drawn 3 m afar from mean sea water line.

ent-white over different shades of opaque-yellow to brown and a smaller portion of black grains (Figure 5).

At first glance, there seemed to be no contamination. However, this was only true for the samples in the millimetre size range. Upon raising magnification stepwise, every investigated sample was found to be polluted. For example, in Figure 6, two pictures show fibres surrounding small sand particles. Fibrous particles with lengths in the range of the sand particle diameters were found. These fibres had different colours and shapes, which made it possible to identify several as man-made, yet with other sample items this was ambiguous. For example, one suggestion that the fibres are of synthetic origin is the fact that the measured diameters ranged from approximately 10 – 30 µm, which is the classical order of magnitude that man-made fibres are produced to have (Sandip & Narsingh, 2007).

It was found that spherically shaped particles below a diameter of 50 µm occurred in high numbers, where colours and shapes suggested quartz particles; however, a clear differentiation was virtually impossible within this size range using optical methods only, for example, if the colours of the plastics were not easily distinguishable (pink particles, etc.). The IR method and Raman microspectroscopy (RM) could theoretically be used to identify the material; however, these methods are limited by the size of the particles (>10 µm) (Schwaferts, Niessner, Elsner, & Ivleva, 2019).

Figure 7 shows images of the synthetic sand mixtures. On the left, the mixture of PP powder with beach sand can be observed. Comparing the white PP powder with the initial sand samples, it may seem possible to distinguish the synthetic material from sand grains. However, this is only true because the size difference is known, and due to the high concentration of the PP powder, misshapen agglomerates are formed. Considering a real sample with a lower particle concentration, the distinction becomes impossible even with IR tools ("needle in a haystack").

On the right image in Figure 7, the synthetic microfibre beach sand mixture can be seen. The image shows that fibres are clearly distinguishable from sand particles. However, the distinction between plastic fibres and natural fibrous materials is not immediately obvious. For example, in the image, the distinction between the brown fibres (organic material) and the intentionally added pink fibres is easy; however, in the case of more naturally coloured synthetic fibres, it would be more difficult to make the distinction.

Identifying synthetic fibres is easier than distinguishing between microplastic and sand because most natural fibres can be degraded by certain agents (Mai et al., 2018).

Consequently, it can be concluded that microplastic particles – in the form of short fibre residues – were already found in a rather small sample from one restricted area. Clearly, this area is not directly contrastable to areas where humans have no access. Additionally, it can be concluded that optical analysis as the sole analysis procedure is not advisable since only pronounced differences in colour, shape or size are sufficient for the correct identification of microplastics.

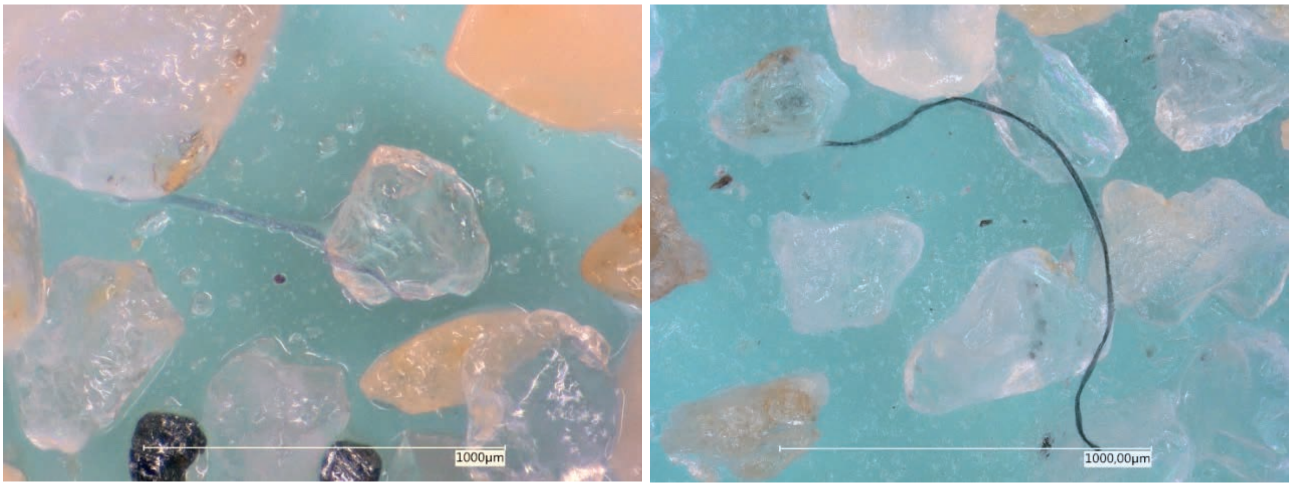


FIGURE 6: Examples of fibres found in beach sand specimen; left: coloured fibre or yarn (a twist is recognisable), right: another fibre type found.

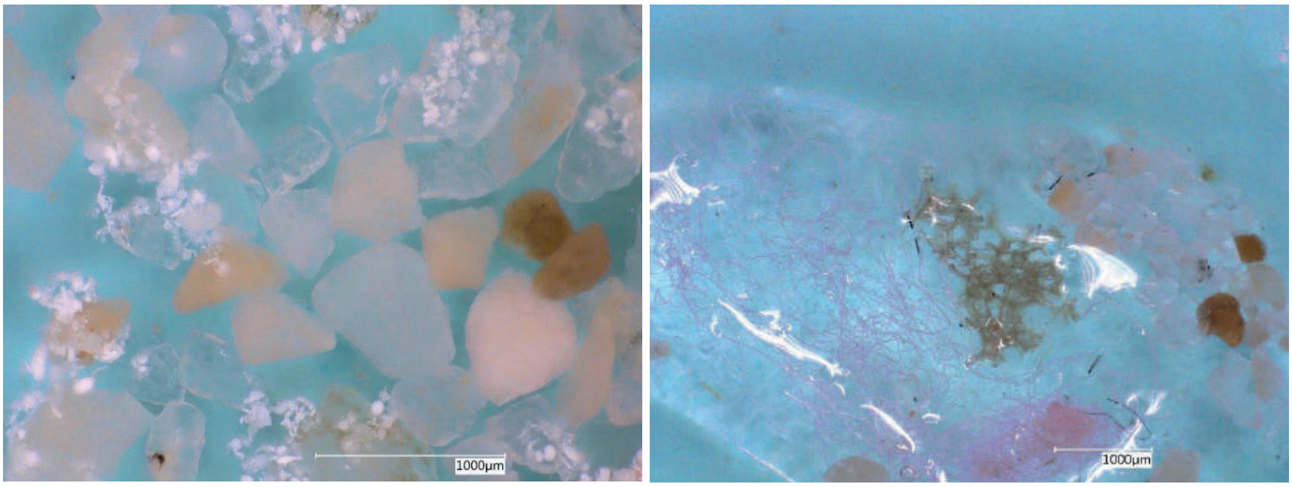


FIGURE 7: Examples of synthetic sand specimen; left: PP powder agglomerates encasing sand particles; right: PET microfibre agglomerates (pink filaments), a bundle of root or algae filaments (light brown) and unidentified black fibres.

5. EVALUATION OF FIBRE EMISSION SOURCES AND PATHWAYS

The problem of microplastics and, in particular, fibres released from textiles are currently intensively discussed. The laundering of textiles and apparel is believed to represent a major source for microfibres released into the environment (Napper & Thompson, 2016). In principle, three routes for the release of microplastics and microplastic fibres are possible, as sketched in Figure 8.

In the figure, "route 1" means the release of fibres during the washing of clothing. It is, however, important to consider that in many industrialised countries, waste water will be supplied to a waste water treatment plant (WWTP). Recent studies show that a WWTP will remove microplastics and fibres very efficiently, whereas sewage sludge represents a sink for microplastics and fibres (Schmiedgruber, Hufenus, & Mitrano, 2019; Sun et al., 2019). Based on this efficient cleaning of waste water, it makes little sense to use filtration systems to prevent the release of fibres from washing machines. This suggests that sewage sludge must not be applied to croplands, as microplastics and fibres will be dispersed into the environment (Corradini et al., 2019). It is further evident that the emissions of fibres from laundry into water are an issue in countries without comprehensive treatment of waste water by appropriate treatment plants. This route of fibre release into the environment will be relevant for low-income countries that cannot afford waste water treatment plants. In these countries, filtration systems for washing machines, as shown in Table 3, would be ideal, but most likely, people cannot afford the associated acquisition and operating costs. It can even be assumed that in many developing countries, laundry will be performed in watercourses (see Figure 9); thus, filtration systems are not applicable.

As with all products, textiles and clothing will also reach the end-of-life state at a given time. Despite a separate collection and a high rate for re-use that is well established in some developed countries, a large portion will end up as municipal solid waste named as "route 2" in Figure 8. It is evident that any uncontrolled dumping represents a potential source for microplastics and fibres. Even if waste disposal takes place in an engineered landfill, it is reported that microplastics can be found in the leachate (P. He, Chen, Shao, Zhang, & Lü, 2019). However, in several countries, a dispersion of microplastics into the environment via solid waste will not take place for the following reasons:

- Leachates from landfills must undergo waste water treatment, and thus, microplastics and fibres will end up in the sewage sludge.
- Landfilling of waste with more than 4 % total organic carbon is not allowed, which, also concerns textiles.

Finally, Figure 8 shows "route 3" by which microplastics and microfibres might enter the environment. During use, textiles will emit fibres or fibre dust. According to the BISFA(BISFA, 2017), the terms as specified in Table 4 are defined. Recent studies show that (plastics) fibres are present in indoor as well as in outdoor environments and that these fibres represent a source for microplastics (R. Dris et al., 2017; Rachid Dris et al., 2016). Furthermore, 20 years ago, it has been proven that cellulosic and plastic fibres can be found in human lung tissue (Pauly et al., 1998).

6. CONCLUSIONS AND OUTLOOK

In this work, evaluation methods for microplastics were examined and tested for their applicability to detect fibre emissions. There are some major challenges to overcome in the analysis, such as the issue regarding the publication of reproducible and, most importantly, comparable results. The issue of different mesh sizes used in the sampling of waste water illustrates the problem quite vividly. While the studies using small 20-µm-mesh sieves detect fibres in an amount hardly even countable, other researchers using sieves with mesh sizes above 300 µm conclude that their samples are free of microplastics since nothing was retained on the used sieves. A trade-off is needed in this regard. While using sieves with mesh sizes in the single-digit micrometre range provides rather accurate re-

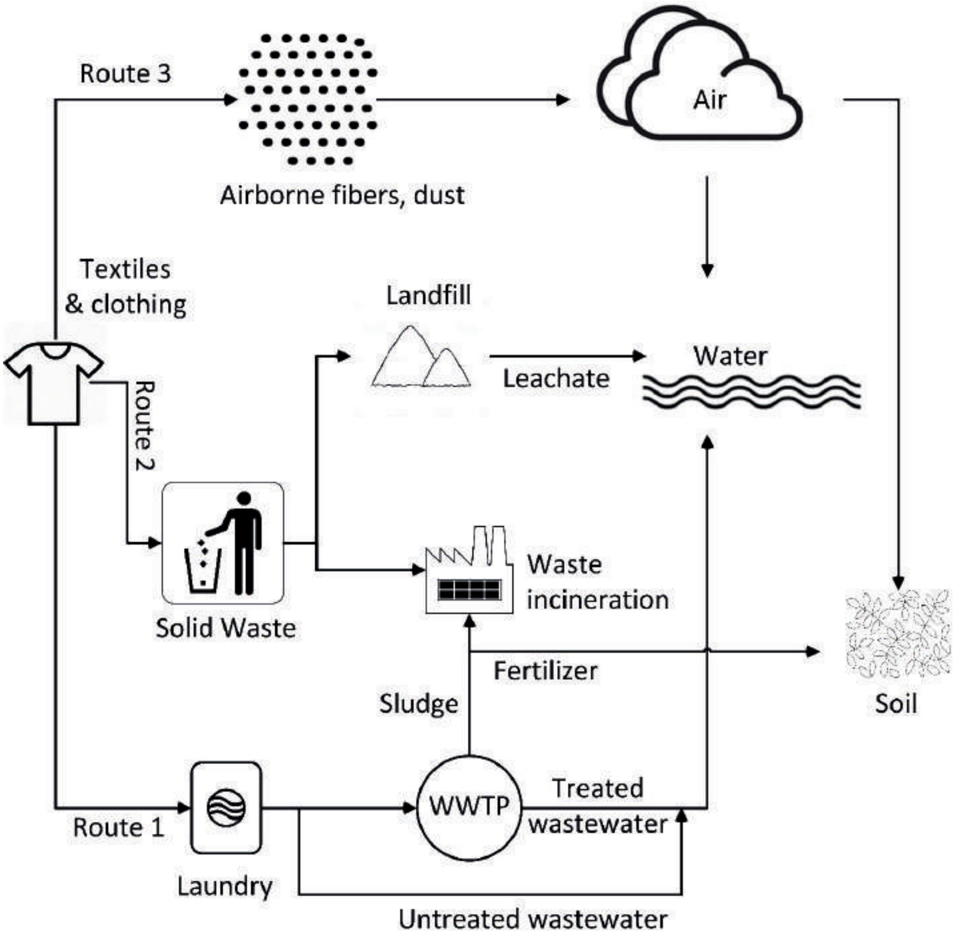


FIGURE 8: Schematic sketch of possible release of microplastics and microplastic fibers into the environment.

sults, it is impossible to analyse huge quantities of sample liquids. On the other hand, analysing vast volumes and using sieves with mesh sizes that allow most of the particles to pass through is also not very useful in most situations.

An issue that hitherto was not garnered much attention is the emission of microplastics into the air and the corresponding health risk for humans and animals. The currently available data are severely lacking, and the extrapolations from a few measurement points to whole city areas do not seem sufficiently reliable. Some studies identified fibres deep within human lung tissue, and others detected a correlation between diseases of the respiratory system and presence of high fibre concentrations in the work environment; however, no conclusive proof has been provided to correctly quantify the risks of fibre and dust.

Finally, it should be stated that while filters for washing machines and better waste water treatment plants are heavily discussed, many sources of microplastic fibres will most likely never be eliminated, for example, emissions from clothing or from the tires of cars into the air. Here, only biodegradable materials can truly stop permanent pollution of the environment with persistent substances. Many new innovative materials are being developed, but no such material has truly challenged the large competitor polyethylene terephthalate (PET), especially with respect to fibres. Highlighting this issue to the public could be the spark for a sustainable change in the sector.

TABLE 3: Selection of systems to prevent fibres to be released into the environment during laundry.

Principle	Short description	Name / company	Reference
Filter	Built-in or add-on filter for washing machine	Planet Care Limited	https://planetcare.org/en/
Washing bag	Washing bag collects released microfibers during washing	Guppyfriend	http://guppyfriend.com/
Laundry ball	Laundry ball catches released microfibers during washing	Cora Ball	https://coraball.com/
Filter	Filter mounted between washing machine and drain	Filtrol 160	https://www.septicsafe.com/
Filter	Filter mounted between washing machine and drain	Lint LUV-R	http://www.environmentalenhancements.com/
Filter	Filter mounted between washing machine and drain	Xeros' XFiltra	https://www.xerostech.com/

FIGURE 9: Laundry in the river Abidjan, Ivory Coast; Photo by Ferdinand Reus from Arnhem, Holland.

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TABLE 4: Terms for microplastics or microfibers which can be released from textile products according to BISFA 2016.

Term	Description
Fibre dust	Non-specific terms. Can cover many types of fibrous and non-fibrous species, including contaminants, usually present as mixtures of particulate matter. Recommended specific terms for airborne fibrous material are fibre fly, particulates from fibres, respirable fibre-shaped particulates
Fibre fly	airborne fibres or parts of fibres (light enough to fly), visible as fibres to the human eye
Fibril	A subdivision of a fibre can be attached to the fibre or loose
Particulates from fibres	Airborne particles, not visible as fibres to the naked eye. May or may not be of the polymer material of the fibre or have fibre shape under microscopic view
Respirable fibre-shaped particulates	Airborne particulates fulfilling the following dimensional conditions: length > 5 μm and diameter < 3 μm and length/diameter ratio of > 3:1

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CAUSE-ORIENTED INVESTIGATION OF THE FIRE INCIDENTS IN AUSTRIAN WASTE MANAGEMENT SYSTEMS

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ABSTRACT

In many European and North American countries, stakeholders and interest groups endeavour to improve fire prevention in the industry of waste management, disposal and recycling. Despite the industry's commitment to increasing safety and fire prevention measures, incidents have peaked in recent years, causing numerous problems: Environmental pollution, potential loss of resources, facility infrastructure and contractual partners, even higher insurance rates and deductibles. The most severe consequence which is recently spreading is the looming peril of a total loss of insurance cover.This paper discusses the comprehensive results of the survey and analysis of 285 fire incidents in the waste industry of Austria that have occurred within the recent decade. First, the historical development and reasons for the current increase in waste fires are stated, revealing waste-specific fire patterns such as self-ignition and the expanding range of potential ignition sources. Second, the statistical correlation between the probability of fire incidents and seasonal or climatic factors is shown. Third, the paper presents specific findings regarding the most commonly affected waste streams as well as the distribution patterns of ignition sources and causes. For example, most fire incidents occur in storage and transport areas (52.6% and 22.8%). Finally, probable driving forces are indicated and the potential development of risks and hazards from future waste fires are shown. The paper reveals a fundamental understanding of the conditions and incipiency of fires in waste management, disposal and recycling as well as gaps in our present knowledge that compellingly require further research.

1. INTRODUCTION

In many European and North American countries, stakeholders and interest groups endeavour to improve fire prevention in the industry of waste management, disposal and recycling. Despite the industry's commitment to increase safety and fire prevention measures, incidents have peaked in recent years (Nigl and Pomberger 2018, Messenger 2017, Fogelman 2018), causing numerous problems.

These problems are ranging from environmental pollution due to uncontrolled and hazardous emission, complaints by local residents and neighbours, via the potential loss of resources, facility infrastructure and contractual partners, through to the increase of insurance rates and deductibles. Even more, as a severe consequence the total loss of insurance cover is looming and recently spreading in Austria and Germany.

1.1 Background

The recently growing numbers of waste fires is concerning because their origins are little understood, nor is the topic of fire incidents and fire protection commonly discussed in scientific literature. There have been several studies of individual waste-specific causes of fire mainly from self-ignition (e.g. Holzer 2007; Ibrahim et al. 2013). But the problem of waste fires has rarely been addressed as a whole. Their causes and influence factors are often only assumed based on empirical values rather than on scientifically sound findings. General lack of knowledge about causes, circumstances and progress of waste fires has to be attested.

In recent years, batteries have presented a new hazard to waste management systems. Both increasing amounts of battery-powered devices and increased use of lithium-based high-energy batteries (both primary and secondary cells) have been observed. As a consequence, the number of accidents and claimed incidents related to portable batteries (especially lithium-based batteries) has steeply risen. Future volumes will continue to increase the potential risk of fire incidents across the whole end-of-life chain (from collection to storage and transport to treatment or recovery).

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1.2 Study objectives

This study was conducted in the BAT-SAFE interdisciplinary research project, aiming at assessing risks and hazards of portable batteries in waste management systems. The purpose of this study is to qualitatively and quantitatively examine previous incidents of fire to (1) identify potential risk factors or dangerous combinations of factors and (2) evaluate waste specific fire causes (e.g. the role of lithium batteries as sources of fire). Findings gained in quantitative research into fire causes are included.

The following questions were posed:

- What are the causes of waste fires, which are most relevant and how do they develop over time?
- When and where do these fires originate in waste management companies?
- What are relevant influence factors (e.g. season, daytime, weather, temperature) and in what way are they related?
  - Are fire incidents more frequent or longer on weekends?
  - How do weather patterns, seasonal or climatic conditions affect fire behavior?
- What are the respective drivers (influence factors)?

2. METHODOLOGY

The survey plan of this study followed two approaches including (1) collection and analysis of data via a stakeholder questionnaire and (2) collection and analysis of data from online press and media.

For the first approach, a detailed questionnaire was developed in order to gain data from personal, telephone and email-based questioning. The questionnaire covered about 30 different parameters, including specific data on fire conditions, waste streams involved, data of fire protection systems installed and of the waste management and treatment processes.

For the second approach, a detailed press and media research (including publicly available sources) was carried out to record as many incidents having occurred in waste management companies as possible (including recycling and waste disposal companies), drawing on the most relevant internet search engines (such as Google and Bing) as well as the search functions of relevant Austrian online press and media. A previously defined search term catalogue was used (see supplementary material, in German). In addition, the snowball principle was applied to obtain further incidents of fire. Moreover, the reports of (voluntary or professional) fire brigades and archives of online portals of Austrian fire services were searched for waste fires (for examples, see supplementary material).

In some cases attempts were made to obtain data on identified waste fires from the local voluntary fire brigade by phone-call. Unfortunately, this approach very often collided with matters of data privacy and protection.

Collection was followed by a revision of the recorded data, essentially concerning the elimination of double and false records. Where appropriate, missing data was completed with the help of other sources. Many parameters, however, had to be left blank or unknown because data for the defined parameters were often not available in public sources.

Unknown or not clearly assignable waste fractions have been designated as 'waste not defined'. The fire duration and amounts of damage were then plotted on an ordinal scale to account for any inaccuracies in the raw data collected. One reason for this approach was the relatively frequent and inaccurate indication of the amounts of damage, such as a 'six-digit damage', referring to financial damages somewhere between 100,000 and 999,999 Euros.

Subsequent data analysis, including descriptive statistics, linear regression and general linear modelling, was carried out using MS Office (Excel) and the statistical programmes STATISTICA and R.

After the assessment of long-term statistical data for the monthly mean temperature and precipitation in Austria, data for the city of Linz were applied to the model as average values.

To compare data collected in this project with reference values, the data sets were (1) standardised to the population size of the respective region, referencing one million capita, and (2) fire causes allocated to a defined scheme (see Table 4 for results).

The system boundaries of the study's research were set as follows:

- Spatially: Austria
- Temporally: 10 years (11/2007-11/2017)
- Availability: Data given by stakeholders (approach 1); publicly available data (approach 2)

The following data and parameters were collected for each fire incident:

- Date (derived: weekday, month, year, season)
- Time of fire alarm and time of fire extinction (derived: fire duration)
- Place, postal code (derived: federal state)
- Aggregates/facility/plant type
- Waste fraction
- Cause of fire
- Amount of damage
- Casualties or injured persons

In order to illustrate the spatial distribution of fire incidents, the coordinates of known facilities were obtained from providers of online topographic maps. When the facility or site of fire was not known, the central coordinates of the affected community were used.

3. RESULTS AND DISCUSSION

The first approach regarding the collection and analysis of data obtained by a stakeholder questionnaire yielded no quantitatively assessable result. Due to different reasons, such as (1) concerns about privacy and data protection, (2) worries about adverse consequences in dealing with authorities or insurance providers or (3) missing or incomplete internal fire reporting systems, the response rate by stakeholders was very low. Less than 20 of more than 150 stakeholders addressed in the questionnaire responded by providing data on specific incidents.

For this reason, only the results of the second approach regarding the collection and analysis of data from online press and media is discussed in the following.

3.1 Spatial and temporal distribution pattern

In total, 659 incidents of fire were collected during the media survey. Of these, 285 cases (43%) are accounted for by the waste industry (companies dealing with waste management, waste disposal and recycling). 65 other cases (10%) concerned waste fires in companies from other sectors (commerce and industry) and 309 cases (47%) involved waste fires in public and private sectors, mainly concerning fires at private collection sites and public waste bins. The focus of this study is on the 285 waste fires in the waste industry.

Figure 1 shows the geographical distribution of incidents collected from Austrian companies for the purposes of this project (waste management, disposal and recycling industry). The presented distribution pattern superficially matches the distribution pattern of the population in Austria and is also constrained by topology (e. g. sparsely populated alpine regions in western and central Austria).

The annual amount of fire incidents has increased over time during the surveyed period of ten years, both seasonally and annually (Figure 2). Apart from the high amount of unknown causes of fire, the figure shows that the diversity of known causes has increased over the years. This may be explained by the composition of different household waste fractions having more diversified in recent years (Nigl and Pomberger 2018).

The relative amount of self-ignition has risen since 2014, although self-ignition has been a serious problem in the mid and late 2010s already and was addressed both by scientific research and by suitable and applicable countermeasures (e.g. Pomberger et al. 2006, Schoßig et al. 2010, Held et al. 2011).

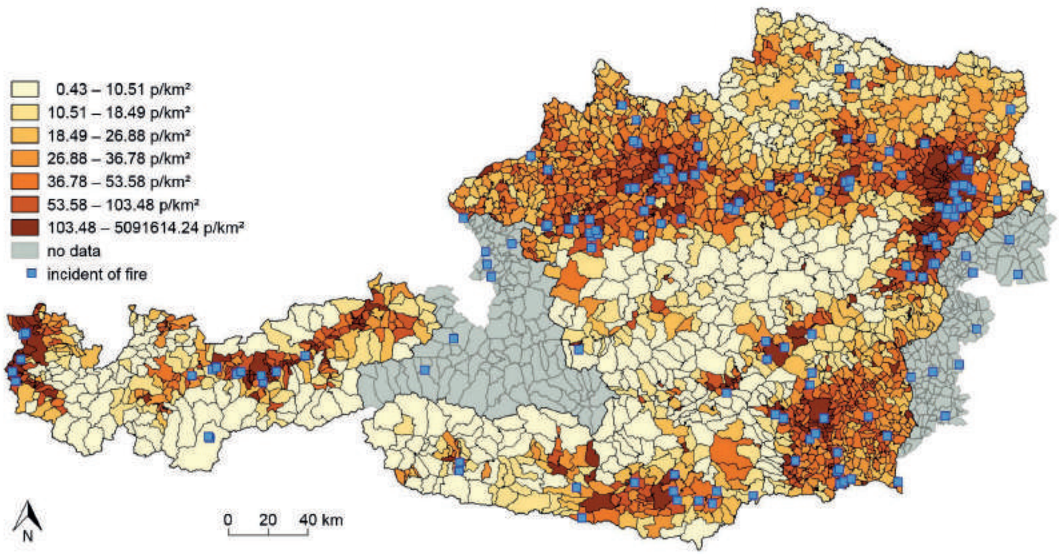

FIGURE 1: Geographical distribution of population density and incidents of fire in waste industry (n=285); chart of population density according to ODVIS-AT (2014).

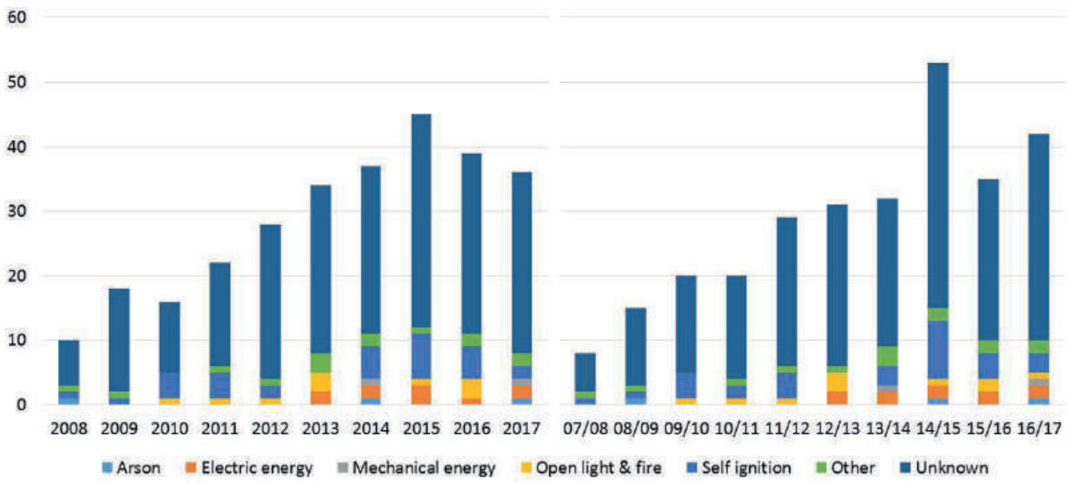

FIGURE 2: Annual and seasonal distribution of incidents of fire including causes of fire (n=285).

There are two plausible reasons for this phenomenon: Unlike arson or fire originating from technical or electrical faults, waste-specific causes of fire are often hard to identify (Nigl and Pomberger 2018). Fire investigators may incline to conclude on self-ignition as the cause of fire, if there is no evidence for any other origin. On the other hand, waste fires involving or caused by lithium-ion batteries are increasing in recent years and (like other causes of fire) are often difficult to identify, leading to a mistaken conclusion on self-ignition.

Comparable data for waste fires can be found in references from the United Kingdom, Sweden, Austria and the German federal states of Saxony and North-Rhine Westphalia.

The Environmental Agency investigated waste fires in the United Kingdom from 2001 to 2013 (Oliver and Brown 2014). Ibrahim et al. (2013) considered cases of fire occurring in permanent and temporary waste storage facilities (including waste for incineration, digestion and composting) in Sweden between 2000 and 2010, recorded by means of stakeholder surveys. In addition, data on cases of fire from two parliamentary inquiries in the German federal state of Saxony are available (Sächsisches Staatsministerium für Umwelt und Wirtschaft 2007, Sächsisches Staatsministerium für Umwelt und Wirtschaft 2015). LANUV (2016) surveyed waste fires in North-Rhine Westphalia in the period of 2011 to 2015. Comparative data on fire incidents in Austria date from the period of 2004 to 2007 (Holzer 2007).

Except for the United Kingdom, whose numbers are already at a very high level, the rates of fire incidents are rising over the respective period (Figure 3).

From a methodical point of view, it should be noted that these data sets are based on different survey methods. The location parameters of the data sets (presented in Table 1), however, are distributed in a similar way despite different statistical populations and survey methods.

The month-by-month graph of incidents in Austria is shown in Figure 4. The months of March to August are well above the theoretical average of 23.75 fire incidents per month while September to February are significantly below.

TABLE 1: Location parameter for annual amounts of fire incidents per million capita.

Region	Minimum	Maximum	Median	Mean	SD
United Kingdom	4,16	6,72	5,50	5,41	0,83
Sweden	0,67	3,10	1,66	1,76	0,93
Austria	1,20	5,21	2,39	2,90	1,30
Saxony	1,86	4,65	3,06	3,07	0,92
North-Rhine Westphalia	1,25	2,44	1,76	1,84	0,50

An analysis of the influence of seasonal or climatic factors (monthly mean temperature or monthly average precipitation) shows a statistically significant correlation to the number of fire incidents. The number of fires per month correlates with the monthly mean temperature and monthly mean precipitation (data for Linz adopted for Austria), indicating that fires are more common in warm and rainy summer months than in winter months. The correlation is modest with an R^2 of 0.62 (for precipitation) and R^2 of 0.54 (for temperature) but statistically highly significant ($p=0.002$ and $p=0.006$). Since these two variables have a high degree of combined explanatory value (autocorrelation and high correlation to each other, see Figure 4), their individual influence on the dependent variable in a combined consideration (general linear model) is not significant.

The statistical correlation can be explained both by higher microbial activity and by a higher probability of exothermic chemical-physical processes triggered in hotter environments. The months of March and September are not perfectly in accord with the trend. An effect of their role as transitional periods between winter and summer may be assumed. A higher sample size in the course of further surveys or the observation of an extended survey period could help clarify this issue.

The distribution of fire incidents across days of the week shows that fires scatter around an average value of 40.7 from Monday to Saturday, only Sunday has to be considered a low outlier (Figure 5). In addition, long-burning incidents (speaking of long fire durations) are distributed

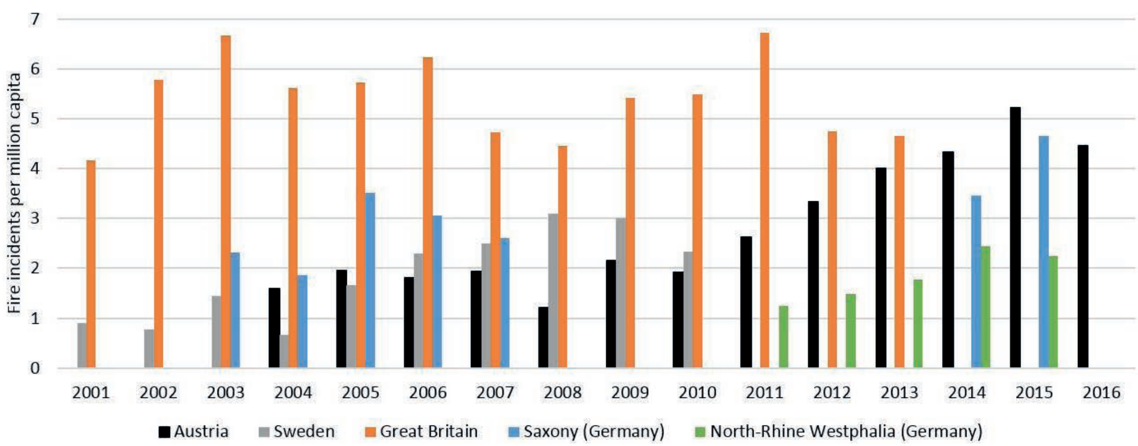

FIGURE 3: Comparison of the annual amount of fire incidents per million capita for Austria (data from this study and literature) and different European regions (data from literature).

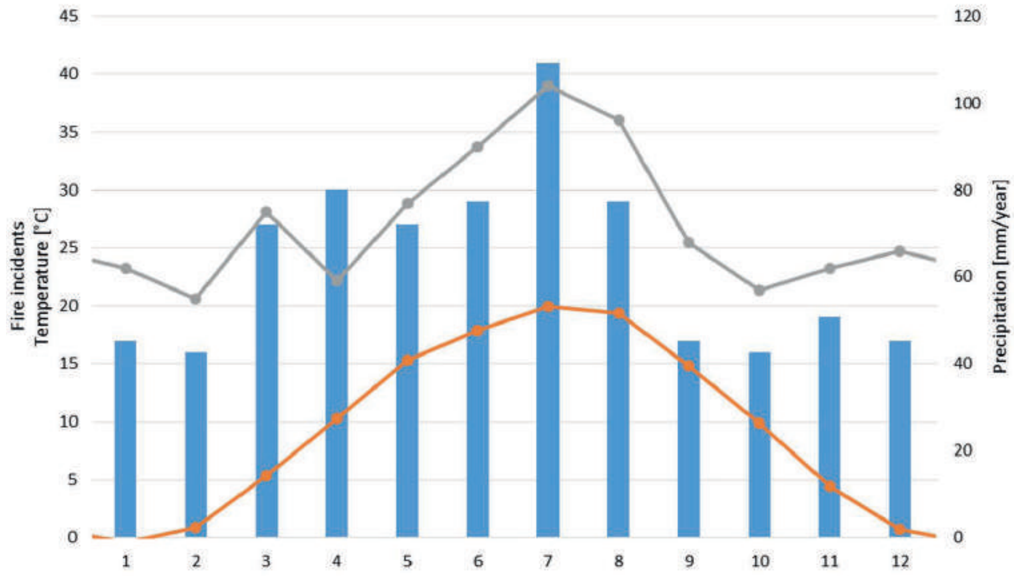

FIGURE 4: Monthly distribution of fire incidents compared to mean temperature and mean precipitation per month (n=285).

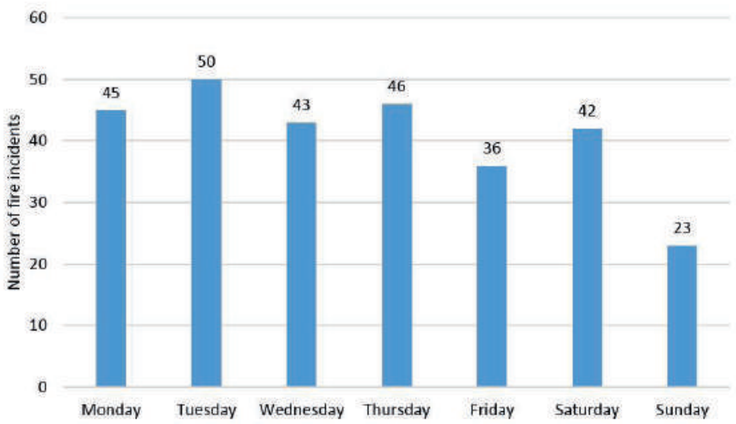

FIGURE 5: Distribution of fire incidents across days of the week (n=285).

equally across the days of the week. Hence, the common assumption that (long-burning) incidents are more common on weekends when staff does not act as an early detection unit could not be confirmed. On the contrary, the results suggest that the probability of fire incidents is related to the operating activities of the facilities, meaning that fires are more frequent on workdays.

3.2 Local distribution pattern

In 227 cases, rather detailed information about the location of the fire ignition was available whereas in 58 cases, the locations was unknown or only available at plant or facility level (see Table 2). Especially with regard to the distinction between input and output storage areas, a precise allocation of cases was not possible because required information was not available.

Note that the terms for aggregates and plant/facility areas are based on the respective sources of information. Correctness of the given information was not always verifiable and therefore the data is not fully consistent (e.g. waste processing vs. recycling facility or waste treatment vs. sorting plant).

From an investment point of view, there is a concentration of fire incidents on shredders (a total of 30 cases) and storage areas (about 150 cases). According to the experience of individual plant operators, the relative amount of fires associated with crushing processes is significantly higher (even up to 50% of the cases may occur during crushing processes).

Another 65 cases of fire relate to the transport areas and can be assigned to collection or transport vehicles (18 cases) and waste containers (47 cases). In the latter case, there is no evidence for whether these incidents should rather be assigned to the transport or to the storage area.

3.3 Distribution of waste fractions and causes of fire

The distribution of the cases of fire across waste types or waste fractions involved is shown in Table 3.

Slightly more than 40% of the incidents could not be assigned to a specific waste type or fraction (waste not defined). Municipal solid waste accounts for 69 of the 285 cases (24%), including 41 cases of residual waste (house-

TABLE 2: Distribution of fire incidents across waste facilities and their areas/sections (n=285).

Specific locations of fire origin	**227**
Aggregates / facility areas	**54**
Waste bunkers	6
Delivery halls	2
Feeding/intake hoppers	1
Exhaust extraction and filtration systems	3
Conveyors	9
Shredders, shredding facilities	30
Waste turner, waste mixer	3
Storage areas	108
Outdoor storage areas	31
Storage (boxes)	31
Storehouses	29
Bales, bale storages	3
Waste piles	6
Baler, press containers	8
Transport areas	65
Collection and transport vehicles	18
Waste containers	47
Unspecific locations of fire origin	**43**
Recycling facility	10
Waste treatment facility	8
Waste sorting plant	8
Waste recovery facility	2
Composting plant	1
Municipal waste collection center	6
Landfill	8
Unknown location of fire origin	**15**
Total sum	**285**

hold waste), 17 cases of bulky waste and 11 cases of commercial waste. In 7 cases (separately collected), spent batteries could be identified as the burning waste fraction. Another 15 times there was no certain waste involved in the fire because these cases were caused by technical defects.

Table 4 shows a comparison between identified causes of fire and (1) waste-specific causes of fire according to Holzer 2007, Sächsisches Staatsministerium für Umwelt und Wirtschaft 2007, Sächsisches Staatsministerium für Umwelt und Wirtschaft 2015 and LANUV 2016 and (2) the distribution of fire causes in other industrial and commercial sectors in Upper Austria (BVS OOE 2018). It is observed that causes of fire in waste industry are often unclear or unknown (26-78%) and more specific than in other sectors. Fires caused by lightning strokes, heat energy and devices or tank explosions, on the other hand, are no issue in waste industry while self-ignition is much more frequent.

Observing the average fire duration is relevant as it is directly related to the potential amount of damage, as shown in Figure 6. The size of the bubbles in the graph corresponds to the number of incidents.

TABLE 3: Distribution of fire incidents across involved waste fractions (n=285).

Waste fractions		
Waste (not defined)	117	41.1%
Residual household waste	41	14.4%
Bulky waste	17	6.0%
Commercial waste	11	3.9%
Organic waste	16	5.6%
Waste metals	18	6.3%
Waste plastics	13	4.6%
Waste paper	10	3.5%
Other waste material (not defined)	6	2.1%
Batteries	7	2.5%
Waste electrical and electronic equipment	3	1.1%
Other hazardous waste (not defined)	3	1.1%
End-of-life vehicles	2	0.7%
End-of-life tires	1	0.4%
Grinding sludge or dust	2	0.7%
Refuse derived fuel	1	0.4%
Shredder light fraction	1	0.4%
Workshop waste	1	0.4%
No waste	15	5.3%

The correlation shows that waste fires lead to a higher level of damage the longer the fire burns. The graph shows that two scenarios will not plausibly occur: Long-burning incidents causing a small amount of damage and quickly extinguished incidents causing a high amount of damage. This result also corresponds to the model presented by Richter (2007) on the correlation between amount of damage and progress of the fire.

Finally, when examining the size of incidents, the survey showed that waste fires have a typical pyramidal-shaped distribution with a lot of incipient fires, a medium amount of small fires and a few large fires. However, the 285 fires included in this study have to be considered the tip of the iceberg because incipient and small fires frequently occur in waste companies and are often not recorded in (online) press and media.

4. CONCLUSIONS

The investigation of the recent ten years of fire incidents in the Austrian waste industry was conducted in this study by two different approaches. Due to privacy and data protection concerns, worries about adverse consequences and missing or incomplete internal fire reporting systems, the first approach yielded no assessable result. The results of the second approach showed a clear correlation between the number of fires per month with the average temperature and the average precipitation per month. The influence of rather short-term weather phenomena on the occurrence and incipiency of waste fires, however, requires further investigation.

TABLE 4: Comparison of causes of fire in waste industry and other industry sectors.

Region	**Austria**	**Austria**	**Saxony (DE)**		**NRW (DE)**	**Upper Austria**	
Period	**2007-17**	**2003-07**	**2003-07**	**2014-15**	**2011-14**	**2012-16**	
Branch	Waste management, disposal and recycling					Industry	Commerce
Source (data adopted)	this study	Holzer 2007	SMUL 2007 & 2015		LANUV 2016	BVS OOE 2018	
Number of fire incidents	n=285	n=58	n=57	n=33	n=94	n=163	n=1018
1. Lightning stroke						7%	19%
2. Self-ignition	11%	17%	23%	36%	33%	7%	4%
3. Heat energy & devices			2%			6%	17%
4. Mechanical energy	1%		2%	6%		22%	12%
5. Electrical energy	4%		2%	3%		25%	17%
6 .Open light and fire	4%	2%	5%		4%	9%	15%
7. Tank explosion						1%	
8. Arson	1%		33%	6%	9%	3%	10%
9. Other	5%	3%	7%	3%	29%	2%	1%
10. Unclear, unknown	75%	78%	26%	45%	26%	18%	6%
Total	**100%**	**100%**	**100%**	**100%**	**100%**	**100%**	**100%**

With regard to locations most affected by fires, a distinct focus on storage areas was found. In treatment facilities, fire incidents will likely take place in shredders.

The widespread assumption that fires increasingly occur on weekends (when staff members do not act as early detection units) could not be confirmed since long-burning fires are distributed equally across the days of the week.

There is also an evident correlation between the duration of the fire and the amount of damage. Therefore, the industry resp. the companies should aim to reduce not only the number of incidents but also the average fire duration to minimise the extent of damage. Recommendable measures include early fire detection methods or other fire prevention (e.g. automatic extinguishing systems).

Comparing the results of this study with data from the United Kingdom, Sweden, Saxony and North-Rhine Westphalia shows that high levels of incidents in waste management companies are not limited to nations. Rather, the waste industry as a whole is prone to fires (Nigl and Pomberger 2018), requiring more attention paid to industry-specific measures in order to control this hazards in the future.

The rising relevance of batteries as sources of ignition (fire cause) did not reflect in this study, although, experiences of individual plant operators suggest exactly the opposite. On the one hand, there were only a few incidents publicly reported where separately collected batteries were involved in fires. On the other hand, misplaced lithium batteries were rarely identified as source of ignition in other waste streams. Waste-specific fire causes are often difficult to identify and fires caused by batteries often falsely allocated to self-ignition.

Finally, this study shows the ceiling of specific knowledge about fires in the waste industry. Further research is imminent and inevitable to highlight the high relative amount of unknown parameters such as causes of fire, involved waste fractions or pinpointing the exact locations of fires.

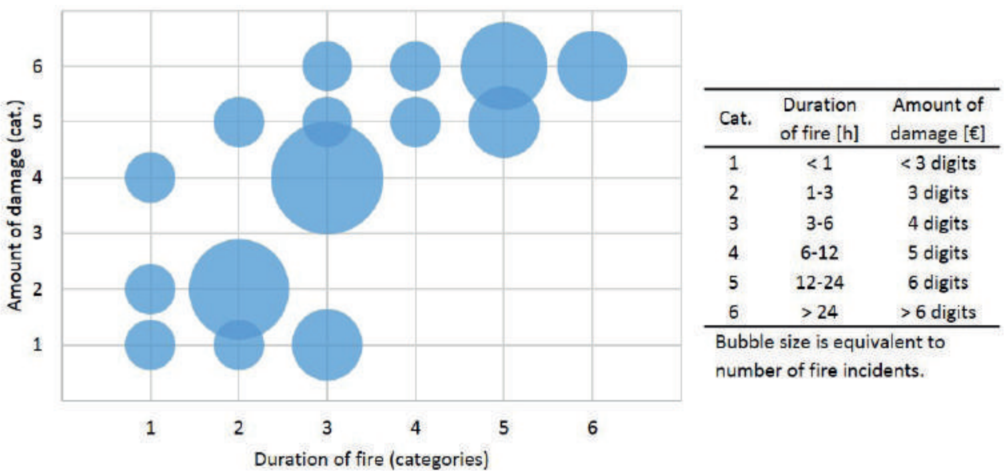

FIGURE 6: Bubble graph showing the amount of damage depending on the fire duration (n=30).

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CISA

ENTERING ROCINHA: A GIS APPROACH FOR THE IMPROVEMENT OF SOLID WASTE MANAGEMENT IN A SLUM IN RIO DE JANEIRO (BRAZIL)

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ABSTRACT

Solid waste management (SWM) is recognized worldwide as an important issue to be addressed with the aim of enhancing liveability. The Rocinha favela (slum) depicts a synthesis of the challenges to be faced in this field: high population density, lack of space and narrow streets, residents struggling with low incomes. In Rocinha, service coverage is lacking and inadequate in numerous sectors, including SWM. The present study investigates the reasons underlying this inadequacy. As an initial step, waste streams were analysed and the SWM system and its criticalities delineated. All accessible information was used, with Geographical Information Systems (GIS) playing an important role in data processing. In the final discussion, a small-scale and decentralized waste management network to be implemented in collaboration with the centralized collection system is suggested. This study was conducted as part of the Polimipararocinha project focussed on the overall urban re-qualification of Rocinha.

1. INTRODUCTION

Waste management in low-income countries is frequently called upon to deal with a series of challenges, as described by UN-HABITAT (2010a). From an economic point of view, the costs of waste management may account for up to 20-40% of the total costs sustained by municipalities, with coverage of these costs through fees proving difficult, particularly in poor areas. Inequalities in service provision may ensue as a consequence both of the above and discrimination.

Although Brazil is not formally considered a low-income country, but rather as a major emerging national economy (together with Russia, India, China and South Africa, they are referred to as the "BRICS" countries), inequality affects the distribution of wealth, with favelas representing one of the most evident signs. At the same time, this huge country has a vast experience in inclusion of the informal sector, particularly in the field of waste management (the case of waste-pickers or "catadores do lixo", organized in cooperatives) (Dias, 2011).

The settlement studied in this paper, Rocinha, recognised as a neighbourhood of Rio de Janeiro (RJ), is served with a series of public services (health, education, waste management, etc.) (EMOP, 2012). Urban plans and public policies for Rocinha have been developed, although they have either not always been adopted, or only partially and inefficiently adopted (Silva, 2015), and service coverage appears as lacking and inadequate. Moreover, Rocinha is characterized by numerous social conflicts, including "criminalization of poverty", presence of narco-traffickers, and generalized scarce trust in the institutions (Ceppi, 2017). Institutional lack of coordination may represent an obstacle to implementation of interventions (Rekow, 2016), potentially to an even greater degree than poor regulation. In this situation, a distinction should be made between sanitation (infrastructural and managerial choices) and hygiene (behavioural change and good practices) rather than placing the burden of an improper waste management on individual stakeholders.

The study was conducted as part of the Polimipararocin-

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ha project. A team made up of researchers and technicians with expertise in different areas (urban studies, architecture, energy efficiency, waste management, ecosystem service analysis, water management), worked together using the Integrated Modification Methodology (IMM) to define a comprehensive framework of interventions within Rocinha, aimed at transforming the favela into a more sustainable environment, whilst pursuing an improved quality of life for its inhabitants. To this regard, waste management is deemed fundamental in view of its acknowledged impacts on public health and the environment (Wilson et al., 2015).

2. CONTEXT

Rocinha is a neighbourhood of Rio de Janeiro, located in the southern part of the municipality (Figure 1). Rocinha is a favela or “comunidade” and was considered an informal settlement (slum) until 1993, when it was officially recognized by the Municipality of Rio de Janeiro. In 2012, a police pacification unit (UPP - Unidades de Polícia Pacificadora) was established as a strategy to maintain control of the territory against drug trafficking. Currently, several public services are located within Rocinha, including health centres (Centro de Saúde) and the national water and sanitation company (CEDAE - Companhia Estadual de Águas e Esgotos).

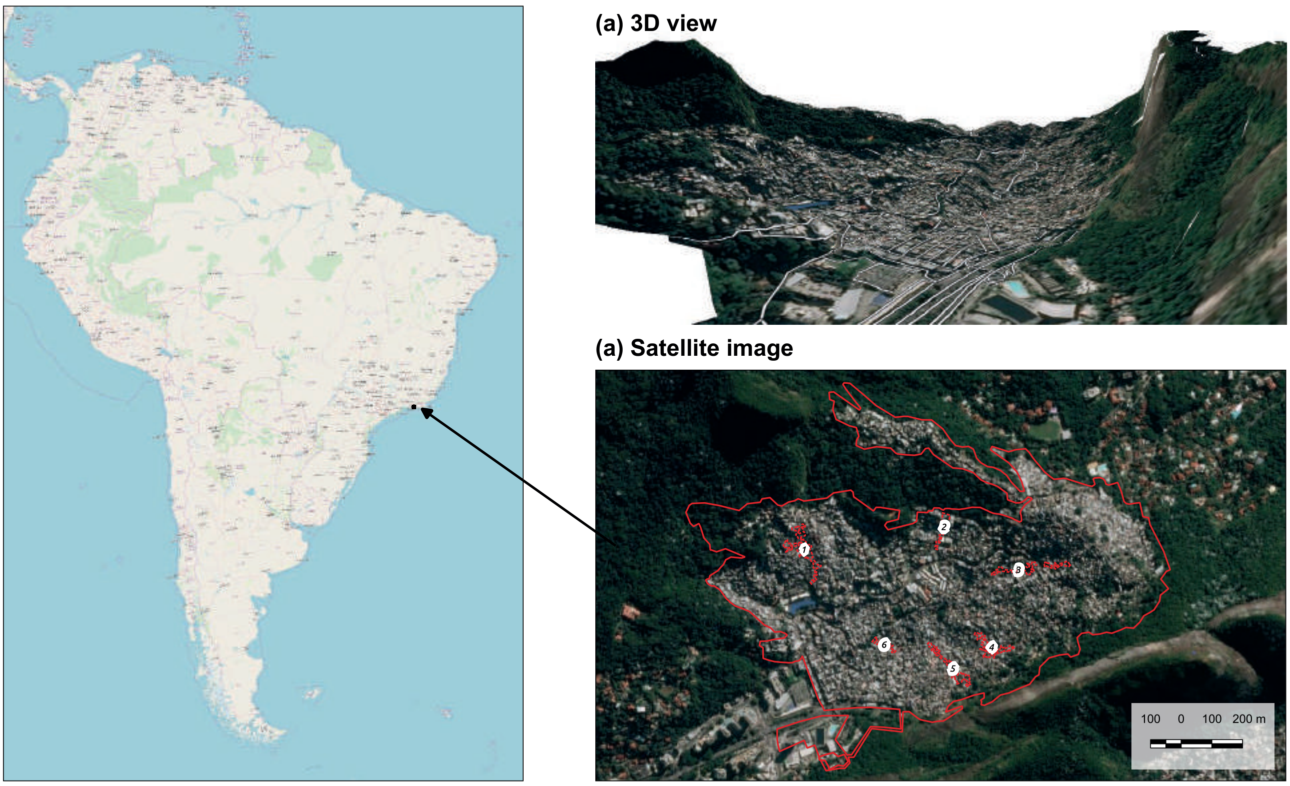
Located between two of the richest neighbourhoods of Rio de Janeiro, Rocinha is one of the poorest, with a Human Development Index (HDI) of 0.735 (IPP, 2017) and an economy mostly based on the informal sector (EMOP, 2010b). It has a high population density, which is difficult to evaluate due to a series of varying estimates. Considering an area of 144 ha, the density for the year 2010 was approx. 500 inh./ha (IPP, 2017). The area rises between two granitic mountains with a morphology characterized by steep slopes, narrow alleys and lack of space.

These characteristics pose serious challenges to the establishment of an effective solid waste management (SWM) system. The location of containers is difficult throughout all areas, with only a few streets accessible to vehicles. The capacity of containers is apparently not sufficient to collect all the waste conferred, and the frequency of collection is inadequate. On moving away from the main road, waste is left on sidewalks and empty terrains or thrown inside drains and sewers. A concerted collection effort is therefore required for citizens living in poorly connected areas where waste dumping may derive from a traditional culture (Carvalho, 2016).

The consequences of improper waste management are well-known: water contamination; sanitation problems due to foraging of disease vectors such as rodents and insects; flooding caused by obstruction of drainage networks; collapse of waste masses onto surrounding buildings. All these issues, at times enhanced by the urban characteristics of the area, are present in Rocinha (EMOP, 2012).

2.1 Functioning of the SWM system

A description of the SWM system for the whole Municipality of Rio de Janeiro should first be provided to set the scene and clarify the current situation (PMGIRS, 2016;



Contains map data: © OpenStreetMap contributors (ODbL, www.openstreetmap.org); © 2019 Microsoft (www.bing.com/maps)

FIGURE 1: Geographic location and context for Rocinha.

EMOP, 2012; UFRJ, 2015). Both the Authorities responsible for the environment, sanitation and waste management and the respective legal frameworks are organised across three different administrative levels (federal, state, municipal) (EMOP, 2012). At a municipal level, the main focus is on legislation regulating Integrated Solid Waste Management. Other laws relate to separate collection, the downstream market for waste (with measures on the mandatory use of recycled waste in the works of the municipal public administration), and the informal sector. Laws related to other issues (sustainable development and climate change, water and sanitation, urban planning) should also be considered. Finally, several plans capable of impinging on waste management are present at a national and local level (e.g. Plano de Desenvolvimento Sustentável; Programa Alternativo de Coleta Seletiva; Lixo zero) (PMGIRS, 2016).

In 2011, a new logistic waste management system was developed for Rio de Janeiro, taking into account three steps (collection, temporary storage and final disposal). Comlurb is the public company responsible for waste collections, which are subsequently transferred to Waste Transfer Stations (Estações de Transferência de Resíduos - ETRs). ETRs receive MSW from Rio de Janeiro and other cities. Out of seven ETRs, the ETR Caju is dedicated to receiving waste from the AP2 area, which includes Rocinha. Waste from the ETRs is then forwarded to the Waste Treatment Centre CTR-Rio at Seropédica, opened in 2011.

In 2014, Comlurb managed the collection of 8370 t/d of waste on average for the city of Rio de Janeiro, corresponding to 91% of an overall 9227 t/d received by Transfer Stations (ETRs). This waste stream was sent to CTR-Rio Seropédica (93.2%); separate collection accounted for 0.5% (PMGIRS, 2016).

The collection system in Rocinha has been described by several authors (Toledo, 2009; Azevedo, 2010; EMOP, 2012). Comlurb operates inside the favela and is responsible for waste collection, cleaning of public areas and maintenance of green spaces and street furniture. Street cleaning and waste collection may also be performed by the “gari comunitario”, a street-sweeper identified by residents’ associations, when streets and alleys are too narrow to allow the passage of vehicles. Tractors and small compactor trucks are used within Rocinha, together with several types of containers (compactors of about 15 m^3, skips of 1-2 m^3, dumpsters of 1.5 m^3) and concrete slabs for street collection. Approximately one hundred people are purported to be involved in waste collection within Rocinha.

3. POLIMIPARAROCINHA PROJECT

The project builds on the collaboration with a Non-Governmental Organization (NGO), “Il sorriso dei miei bimbi”, located in Rocinha. The NGO invited the researcher to implement a scientific approach in an attempt to identify strategies aimed at improving the overall liveability of the favela.

The theoretical background of the project is underpinned by the Integrated Modification Methodology (IMM), aligned with the UN Sustainable Development Goals (SDGs) 2030. One of the 17 goals identified within this framework, SDG 11, relates specifically to cities and human settlements and how to render these inclusive, resilient and sustainable. According to the IMM approach, cities are regarded as Complex Adaptive Systems (CAS), with the main focus being on simulating their operative mechanisms, involving both the internal subsystems and external systems that the cities are part of. The result is a methodological interpretation of the SDG Number 11 (Sustainable cities and communities), suggesting locally-based actions.

3.1 The Integrated Modification Methodology (IMM)

IMM methodology is based on a multi-stage process composed of five integrated phases, respectively: Investigation, Formulation, Modification, Retrofitting and Optimization. IMM intends to assist city-planners and decision-makers by providing them with a fully-integrated design process to transform an existing urban context into a more sustainable one. Starting out from SDG 11, IMM systematically provides a holistic correlation with all other SDGs (Tadi and Vahabzadeh Manesh, 2014).

IMM attempts to modify the behaviour of the entire complex system by means of local intervention scenarios. Based on morphological and functional arrangements, IMM devises a local system mapping used both to investigate the base and predict future scenarios with the aim of improving an existing system. This process allows the identification of areas in which the potential of the context is unexpressed and where major resources exist. Following application of the IMM, Rocinha was considered a specific urban CAS and analysed to assess performance. In particular, the investigation phase highlighted how Rocinha is affected by a lack of open spaces and an existing street network unsuited to providing adequate links and ready access.

Accordingly, the transformation concept is driven by an aim to provide additional open spaces and boost the integrity of street networks by relocating a small number of low-rise buildings where possible and beneficial. The research project comprised an objective evaluation of the morphological organization and related environmental performances and qualities, focusing on different sectors of intervention.

3.2 Waste and other sectors of intervention

As previously explained, one of the pillars of the methodology lies in assessing the link between the different sectors of intervention and their relationship with urban morphology, pursuing a multidisciplinary and integrated approach. In addition to waste management, the main focus of the present research, other related sectors include energy, ecosystem services, wastewater, mobility and food. These will be described briefly in the following paragraphs.

3.2.1 Energy

The energy project aims to fulfil the basic electrical needs of the low-income households of the favela by exploiting solar energy through photovoltaic panels installed on the rooftop of buildings where incident solar radiation is abundant. By doing so, solar energy exploitation will also be able to reduce electricity theft and increase grid reliability inside the favela. Simple access to electricity should be considered while exploring the waste management system, with particular focus on household-level solutions. Furthermore, the exploitation of waste as a source of energy may be evaluated.

3.2.2 Ecosystem Services

An Ecosystem Services (ES) based approach was applied to estimate environmental conditions in view of the fact that urban liveability is influenced by natural resources (e.g. soil, air and water). Compromised and environmentally-degraded areas, more subject to potential disaster events such as landslides or flooding, were identified. The ES-based approach is pivotal in setting nature-based solutions (NBS) to contrast risks and improve both urban ES and local quality of life. NBS use natural processes and ES for functional purposes, addressing ecological, social and economic challenges. In this case, NBS mitigate flood, drought, erosion and landslide. Improper waste management influences flooding, largely caused by the clogging of drains.

3.2.3 Mobility

The current mobility system of Rocinha presents a series of critical issues. Many recent studies carried out to investigate mobility in the favelas highlighted the need to improve the general accessibility of these communities through interventions focused on supporting the non-motorized sector, in particular by implementing bicycle-based systems. The "Rio Conecta" plan for mobility also enhances this aspect. The lack of an integrated mobility system capable of supporting the existing subway station means that the underground is able to cover only limited areas alongside the main street of Rocinha. Following analysis of both the road network and mobility index value, an affordable and integrated electric bike-sharing system has been identified as a priority. The bikes will be charged using locally produced energy, and the whole system linked to an Urban Management System (UMS), which will collect mobility data for further transportation strategies. Limited accessibility is a major obstacle for waste collection, strongly influencing waste management. The proposed solution for Rocinha will consequently take into account conclusions arising from this analysis.

3.2.4 Water and wastewater

The collection of wastewater and rainwaters is fundamental to protect public health and provide for an appropriate management of the territory. A "pilot" area has been identified on which to implement the proposed procedures. Sewer systems were sized based on a water supply of 150 L/day per capita, envisaging collection of 80% of the supplied water in the sewage. Resulting overall discharges are very low, which could lead to settling and therefore septic problems. However, the diameter of network ducts was set at 150 mm to avoid problems of occlusion. The possibility of conveying a small percentage of rainwater to "clean" the pipes when necessary was considered. The designed pipelines are easy to lay and inspect, and, during construction, must be protected against possible mechanical and climate-induced damage. Water should undergo treatment prior to release into the environment.

Although not strictly related to hydraulic aspects, the need for direct community involvement should be highlighted – both to guarantee a correct use of the network (for example, no solid waste should be discharged into the pipes) and provide for improved land management. Community involvement will allow contribute towards raising awareness and fostering interest in the work; a skills workshop will be set up to train the users in correct construction and sewage network management techniques, which would help to generate an economic activity, albeit small.

3.2.5 Food

Food quality and nutritional awareness represents a crucial issue for the majority of the Rocinha children and their parents, who have scarce access to fresh food and receive no information on how to eat healthily. Rocinha is characterized by a high population density and almost total absence of voids, with the steepness of the territory rendering any kind of traditional cultivation impossible. The project therefore envisaged the implementation of a series of innovative sustainable urban agriculture strategies, e.g. green roofs, vertical farming and aquaponics. Besides providing fresh fruit and vegetables, this intervention aims to boost the local economies. Food concerns are also closely linked to water issues (gardens increase drainage of impermeable surfaces that otherwise generate critical runoff) and to the waste sector that promotes the local use of compost derived from organic waste.

4. MATERIALS AND METHODS

The purpose of the study was to suggest a series of potential improvements to be applied to the SWM system in Rocinha. The first step was to create a comprehensive picture of the situation, identifying existing waste flows and estimating the amount and composition of the same. Consequently, efforts were made to collect and analyse all available data to define a baseline.

Subsequently, a geographical analysis was performed in view of the fact that organization of a solid waste management system is interlinked to the topology of the target area, including the distribution of buildings and street network. Data related to waste generation were used in this stage to clarify the distribution of solid waste within Rocinha.

Finally, this framework was used to add consistency to a new proposal for waste management, scaled on the six pilot zones identified by the project.

Details on data sources and analytical methods are provided below.

4.1 Data analysis

All available data from public sources was collected to evaluate the amount of waste generated in Rocinha and its composition. Existing data was largely based on information provided by Comlurb, although other sources were also consulted. When possible, complete data series were used for analysis (IPP, 2017; IBGE, 2017; SNIS, 2017) and aggregated data were used to correct or validate the estimate (PMGIRS, 2016; EMOP, 2012; UFRJ, 2015). The analysis was performed using the open-source software R-Stat and related packages (R Core Team, 2015; Wickham, 2007; 2009; 2011).

Data on the composition of waste was only available for Household waste. For the whole Municipality of Rio de Janeiro, data was first collected in 1981 (Lima and Sorliuga, 2000; Comlurb, 2012), while data for Rocinha were only available for one single year (Comlurb, 2012). Numerical data for both Rio de Janeiro and Rocinha is listed in Table 1. Data display a coherent picture, which was taken as the basis for the assessment.

Finally, the number of inhabitants of Rocinha was estimated. Evaluating the population of an informal settlement is not an easy task due to the lack of reliable information. Official data range from approx.70,000 (IBGE, 2010) to 100,000 (EMOP, 2010) inhabitants. The electricity provider Light S.A. estimates 165,000 inhabitants, while citizens' associations suggest a range between 180,000 and 220,000 people. Calculations presented in this paper assume 211,000 inhabitants, based on the Polimipararocinha project best guess, although further statistical and demographic analysis is required.

4.2 Geographical analysis

The analysis was performed using the open-source software QGIS and related plugins (QGIS, 2017).

The base map for the analysis, a shapefile containing every single building with an estimate of the number of floors, was prepared and is being kept updated by the Universidade Federal do Rio de Janeiro (UFRJ). Other geographical information, i.e. data relating to administrative boundaries and Census areas, are available online on the website of the Municipality of Rio de Janeiro (IPP, 2017).

The first step was to define the number of inhabitants per building. As a first estimate, all buildings were assumed to be residential, and the total floor area calculated by multiplying the number of floors of each building by its ground floor area. The population of Rocinha was divided by the total floor area, yielding a value of 0.17 inh./m², which was used to assign a defined number of inhabitants to each building. The number of inhabitants multiplied by the daily per capita generation of waste provided a map of the distribution of waste produced in Rocinha, useful for a general overview but also in calibrating intervention on the pilot zones.

A second step was carried out to ascertain the situation of collection points within the favela. Maps describing the location of waste containers in Rocinha were found in previous studies (Azevedo, 2010; EMOP, 2012) and partially updated by means of a manual inspection using GoogleStreetView. To estimate the amount of waste reaching each collection point, Voronoi polygons were used. The Voronoi region for each point represents the set of points in the plane for which that point is the closest (Fortune, 1987); consequently, the sum of the amount of waste calculated for each building included in the Voronoi region will represent the amount of waste reaching the related collection points.

TABLE 1: Characterization of Households waste in Rio de Janeiro (RJ) and Rocinha for the year 2012, weight percentage (GPA, 2012).

Fraction	Rio de Janeiro (%)	Rocinha (%)
Paper / Cardboard	15.99	11
Plastic	19.14	21.82
Glass	3.28	0.5
Organic matter	53.28	60.67
Metal	1.57	1.69
Inert waste	1.81	1.11
Leaves	1.35	0.52
Wood	0.34	0
Rubber & tyres	0.22	0.44
Tissues	1.76	1.11
Leather	0.21	0
Bones	0.01	0
Coconut	0.82	0
Paraffin	0.05	0
E-waste	0.2	1.04
Total	**100**	**100**
Specific weight (kg/m³)	**133.02**	**111.17**
Moisture content (%)	**36.57**	**39.46**

5. RESULTS

5.1 Solid waste flows for Rio de Janeiro and Rocinha

All available quantitative information relating to waste in Rocinha was gathered. A database was prepared, considering the following entries: year; numeric value; unit of measure; location (boundaries of the area); phase (production, collection or disposal); source (whether the waste originated from households, street sweeping, etc.); who performed the collection; final destination of the waste; reference for data; material fraction (food waste, metals, plastic, etc.); type of data (whether estimated, measured, or derived from a survey). Information relating to each parameter was not always available.

The database represented the starting point for analysis; several waste streams (depending on the source of the waste), collected by Comlurb or by private enterprises and forwarded to a series of final destinations, were identified. In the collection phase, the total waste quantities may be divided into different streams:

- Public waste, from street cleaning and sweeping;
- PublicGreen, green public waste from pruning and cutting;
- FreeRemoval, bulky waste removed for free by a specific service provided by Comlurb;
- Household waste, also referred to as "domestic" waste;
- RCC, Construction and Demolition waste;
- RSS, Healthcare waste;
- Separate collection.

The role of waste pickers ("catadores") was not considered in this calculation as no available data accounted for their involvement.

Information was available only for Rio de Janeiro, but not for Rocinha. Consequently, the data available for Rocinha was corrected based on the results obtained from analysis of solid waste flows in the main city, as described below.

The only data available for Rocinha related to the collection of Public and Household waste by Comlurb . Moreover, public waste (Public2) was an aggregated data comprehensive of Public, Public Green and FreeRemoval waste, as previously described.

As a first step, both Public2 and Household waste were increased to take into account the contribution of the Private sector, accounting for 11% of the total amount of collected Public, PublicGreen and FreeRemoval waste, and 16% of the total amount of collected Household waste in Rio de Janeiro.

Public2 waste was then sub-divided into Public, PublicGreen and FreeRemoval waste, and percentages related to other waste streams (RCC, RSS, Separate collection) were estimated. These results, representing the collection phase, are described in Figure 2.

Data presented in Figure 2 represent solely the percentage of waste collected by public or private enterprises; however, the percentage of waste which is not appropriately collected should also be evaluated in order to estimate overall waste generation. In Rio de Janeiro, approximately 59% of Household waste is estimated as being appropriately collected, while the remaining 41% is not collected and is improperly disposed of (burned, buried, dumped, etc.) (UFRJ, 2015). The amount of collected waste in Rocinha was consequently increased by calculating the amount of improperly collected Household waste (18,870 t per year) to estimate the amount of total produced waste (60,411 t per year in 2014). The final daily solid waste production of approx.166 t is in line with the estimate of 152 t/d by EMOP (2012).

Considering a population of 211,000 inhabitants, the daily per capita generation of total solid waste is estimated at 0.78 kg/d/inh., while the daily per capita production of Household waste is estimated at 0.59 kg/d/inh.

5.2 Distribution of solid waste within Rocinha

The daily per capita production of Household waste was used to calculate the daily production of Household waste for each building (Figure 3, (a) Base map). Voronoi regions were subsequently calculated for each formal ("Collection point") or informal ("Dumping point") collection point. Figure 3 (b) Elaboration, illustrates Collection points and Dumping points using different sizes depending on the amount of waste conferred to each point, ranging from less than 100 to more than 7000 kg per each day.

When applied to the real world, the choice of Voronoi regions represents an approximation which fails to take into account the presence or absence of streets or alleys, or the differences in altitude that characterise Rocinha. Nonetheless, the choice made was based on the lack of a detailed map of streets and alleys.

The map presented in Figure 3 represents an initial tool for use in the identification of critical points, yielding a first estimate of the equipment (such as containers) needed. Moreover, the amount of waste disposed of at each collection point may be used as a proxy of the impact of the intervention described in the next paragraph.

5.3 A new proposal

The proposal intended to improve the solid waste management system in Rocinha was based largely on a detailed analysis of the current state-of-the-art. Previous studies, which had identified three major stakeholders (companies,

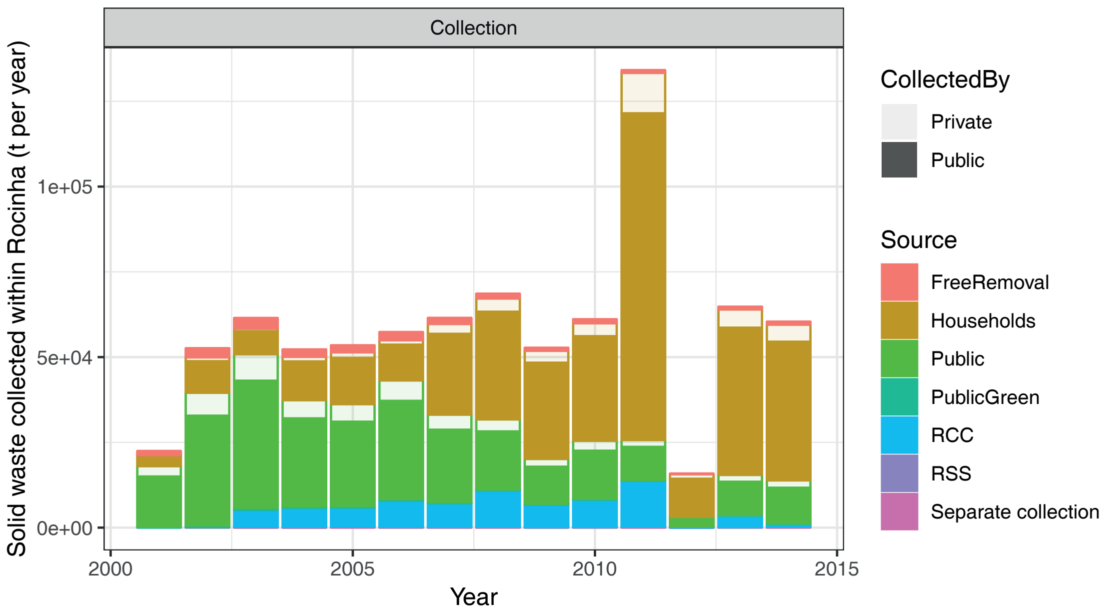


FIGURE 2: Rocinha: solid waste streams in the collection phase from 2001 to 2014.

local communities, public authorities), and detailed actions or solutions (Azevedo, 2010), such as underground containers, to be implemented with an aim to tackling the problem of space (EMOP, 2012), were also considered. It should however be underlined that none of these proposals has yet been implemented due to conflicts between the State and the municipal management (Silva, 2014).

The current situation may be summarized as follows: waste collection is not guaranteed throughout the favela due to its morphological characteristics, logistic concerns and lack of resources. Together with a scarce environmental awareness, this leads to improper disposal of waste, exacerbating other issues which may over time result in a situation of emergency (floods or disease outbreaks). The need to deal with emergencies drains the already scarce resources from the overall system. In an attempt to overcome this vicious circle, a collection system operating parallel to the existing system managed by Comlurb has

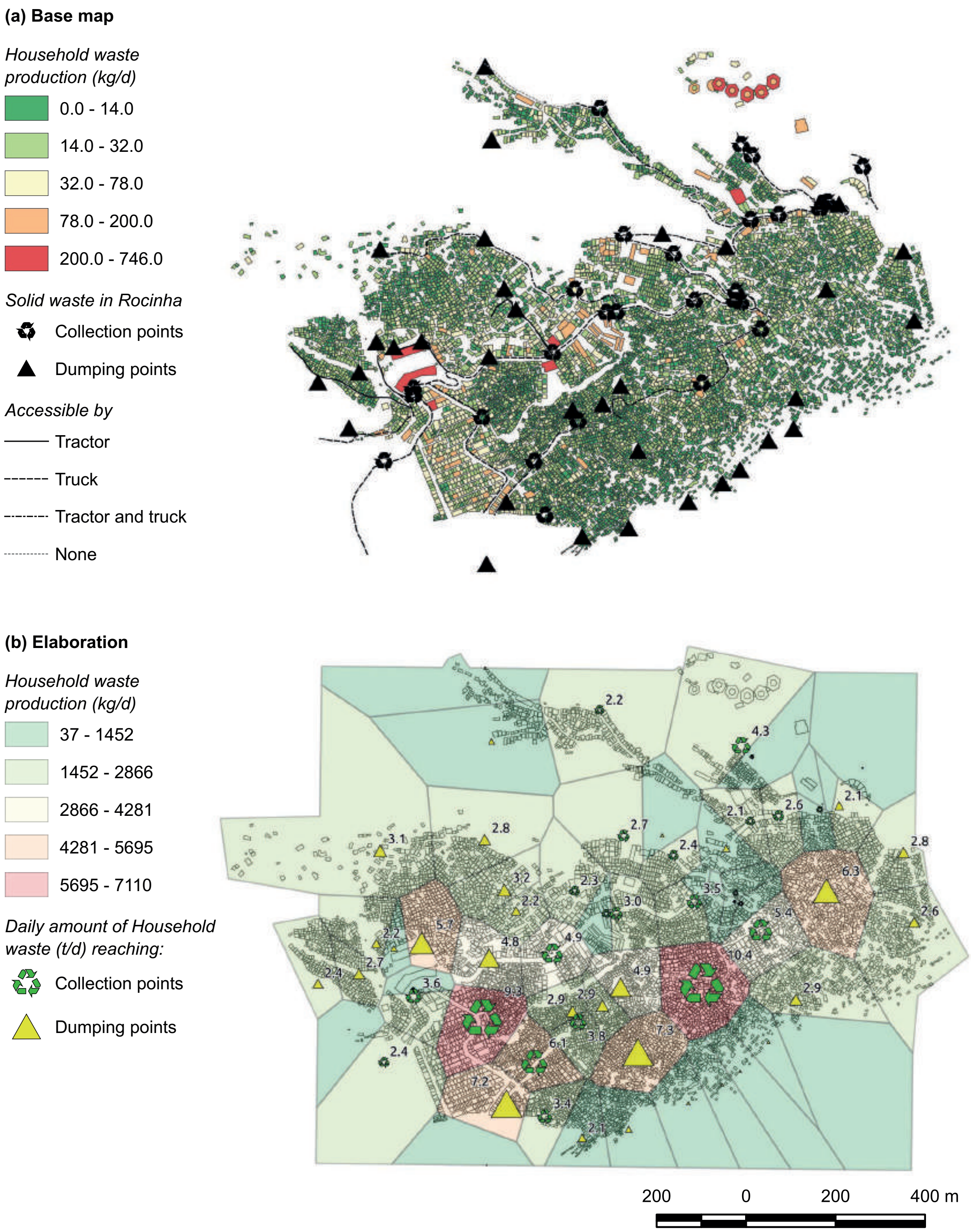


FIGURE 3: Household waste production in Rocinha: (a) Base map - daily production for each building, location of Collection and Dumping points, collection routes; (b) Elaboration – Household waste production per area (kg/d) and daily amount conferred at each point of disposal (t/d).

been devised to divert valuable materials from the general waste collection. This collection system will be organized in small-scale nodes, each representing the focal point for the whole collection system in a specific area. In each node, a group of workers operating either independently or in cooperatives, will assume responsibility for waste management. All nodes will be connected in a network. Separate collection of Household waste will be implemented, targeting both the organic fraction and recyclables, with sale of recyclables representing a source of income for people involved in the system.

The reasons for this proposal can be summarized as follows.

First, the choice to proceed with small scale interventions requires little investment. This will consequently allow for a series of different strategies: the establishment of nodes may be promoted and coordinated by institutions, although may even arise as a grass-root initiative moving from local communities and stakeholders with a high environmental concern (e.g. community-based organizations) or an economic interest (e.g. waste pickers).

An important example is provided by the cooperative Rocinha Recicla, established in 2016 in the context of the project "De Olho No Lixo" (promoted by the State of Rio de Janeiro) (INEA, 2016). Rocinha Recicla employed up to 30 workers, all resident in Rocinha, and is currently focused on the cleaning of roads and water channels, together with a series of other awareness-raising activities: Ecomoda, a dressmaking laboratory using recovered waste fabrics, and FunkVerde, a workshop for the production of musical instruments using discarded materials. Until November 2018, Rocinha Recicla managed a warehouse in which recovered materials (plastic, metals, paper and cardboard) collected by its workers or by autonomous waste pickers were prepared for sale, and soap was produced from exhaust kitchen oil. The warehouse was subsequently demolished by the Municipality of Rio de Janeiro without notice (INEA, 2018), and the cooperative is currently awaiting the assignment of a new location, as promised by the authorities.

When activities resume, Rocinha Recicla, together with its laboratories Ecomoda and FunkVerde, should be considered an important node of the entire network. Waste pickers, many of whom already collaborate with the cooperative, should also be involved in this system. Brazil has a long tradition of self-organization of this informal economy sector, with previous experiences maintaining how waste pickers benefit from a higher level of organization: the existence of a network will allow scale economies to be implemented, including shared use of equipment (such as presses, shredders, etc.) and improved access to the secondary raw material market.

Secondly, the promotion of separate collection at a household level is fundamental to obtain high quality secondary raw materials and provide a safer working environment for collection workers. Separate collection will target recyclable waste (plastic, metals and paper/cardboard) with a high (potential) economic value. Moreover, since recyclables from Rocinha are not currently forwarded to any sorting plant, this diversion will produce no negative effect on the cooperatives involved in waste separation in other nodes of the waste collection system of Rio de Janeiro.

The abundant organic fraction present in Rocinha will also be targeted. An important goal of the project is to reduce the amount of waste improperly disposed of, with the aim of improving public health and safety. This fraction has been estimated as corresponding to approx. 41% of the Household waste generated, and this value accordingly adopted as the target amount to be diverted by means of separate collection. Analysis has revealed how this value is achievable solely if both recyclables and the organic waste fraction are targeted. Threshold values for the efficiency of separate collection to be achieved over the medium term are in the range of approx. 45% for the organic fraction and 73% for recyclables (plastic, metals and paper/cardboard).

Finally, the strategy is focused especially on collection, as only a small portion of the waste will be treated inside Rocinha. This amount will vary depending on land availability, considering that the on-site recycling of waste requires space for storage and treatment, which represents a considerable constraint for Rocinha. Waste not managed within the proposed intervention will be forwarded to the official collection point managed by Comlurb and will enter into the system described in Paragraph 2.1.

5.3.1 Organization of the new system

In the context of the general framework of this strategy, an organizational proposal has been elaborated. For each node, a series of actions relating to the individual stages will be evaluated: separate collection, primary storage, local treatment and awareness. These actions may be described as follows:

- Separate collection is based on the distribution of street containers for conferment of the organic fraction and recyclables. However, innovative solutions aimed at addressing the lack of space within Rocinha should be identified. Alternative solutions (door-to-door collection, location in schools or other facilities) may be explored.
- "Eco-Centres" are designed for the primary storage of recyclables. The storage period is linked to space availability (less space, shorter period) and also on the size of the vehicles accessing the Eco-Centre (smaller vehicles, a shorter period of storage OR a higher number of trips).
- A multi-purpose hexagonal structure, known as the "Ecoponto Kiosk", will serve as: information point to explain the functioning of the SWM system and raise awareness; distribution point for items to enhance separate collection, recycling and re-use of materials (paper bags for organic collection, "take-back" glass bottles, locally produced compost) and Personal Protective Equipment (PPE) for waste workers; collection of paper, metals and plastic.

The feasibility of local treatment of the organic fraction will be assessed for both composting and anaerobic digestion, depending on the local situation (e.g. land availability, distance from buildings, demand for compost in the surrounding areas…). Composting may indeed be divided into sub-modules and managed on a community basis, with little labour cost. The compost can be used in synergy with other actions proposed within the project: land restoration for the mitigation of hydraulic risk (see 3.1 Ecosystem services) or vertical garden for the local production of food (see 3.5 Food). The compost may also be used in the maintenance of green areas, and, if high-quality levels are achieved, eventually sold on a local market. The anaerobic digester will be designed according to a local project implemented in Rio de Janeiro (Mattos and Farias, 2011). Both safety issues, such as the need for a buffer zone around the digester, and environmental issues, will be taken into consideration: for example, the potential for local use of the produced biogas should be evaluated to avoid any leakage (methane poses a high risk of fire and explosion).

5.3.2 Analysing the impact of the intervention

The study presented in this paper aims to provide technical support for the establishment of a node. Indeed, following the initial identification of an area, the study will allow the amount of waste produced within the specific area, together with the quantities of organic and valuable fractions such as paper, plastic and metals, to be calculated. Subsequently, the space required for street collection, storage and local treatments can be assessed and appropriate locations identified.

An analysis performed on the six zones identified within the Polimipararocinha project is presented as an example of the procedure.

Design choices are summarized hereafter:

- Separate collection. The size of the containers was calculated considering typical specific bulk weights for each fraction (organic: 0.3 t/m^3; paper: 0.13 t/m^3; plastic: 0.072 t/m^3; metals: 0.12 t/m^3) and a maximum weight of 80 kg for the full container. Collection is performed daily.
- "Eco-centres" feature a maximum height of 2 m. Maximum storage period was set at 7 days as a first choice.
- The "Ecoponto Kiosk" has a side of 2.5 m, a total surface area of approx. 16 m^2, and can accommodate a volume of up to 1.83 m^3 of plastic, 1.1 m^3 of paper and 0.3 m^3 of metal. The "Ecoponto Kiosk" will be located in areas currently devoid of structures.
- Composting slabs will have a maximum height of 1.5 m, and a width of 2 m at the bottom and 1.5 m at the top; the depth of the smaller module will reach approx. 2 m; a reduction of circa 30% of the volume is hypothesised. The anaerobic digester is circular shaped with 2 m radius and requires a circular buffer zone with 10 m radius. Retention time is assumed to be 30 days.

The final results are compiled in Table 2. For each zone, the area required for the implementation of each strategy is presented, together with the impact on the available empty spaces existing in each specific area. The number of containers required for each zone has also been calculated.

The impact produced on empty spaces is minor, with the exception of the anaerobic digester, which requires a safety zone and can only be realized in Zone 2. The inter-

TABLE 2: Design of the new system for waste management in the six pilot zones of the project.

	Zone 1	Zone 2	Zone 3	Zone 4	Zone 5	Zone 6	Total
Number of inhabitants	**4058**	**4166**	**4774**	**4404**	**4227**	**2234**	**23863**
Amount of diverted waste Weight (t/d), of which	1.26	1.30	1.49	1.37	1.32	0.70	7.44
• Organic fraction	0.66	0.67	0.77	0.71	0.68	0.36	3.85
• Plastic	0.38	0.39	0.45	0.42	0.4	0.21	2.25
• Paper	0.19	0.2	0.23	0.21	0.2	0.11	1.14
• Metal	0.03	0.03	0.03	0.03	0.03	0.02	0.16
• Volume (m^3/d)	9.24	9.49	10.87	10.03	9.63	5.09	54.4
Total area required (m^2):	**98**	**486**	**115**	**108**	**88**	**62**	
Primary storage (m^2)	4	7	7	7	7	3	
Ecoponto Kiosk (m^2)	16	No	16	16	No	16	
Secondary storage (m^2)	25	26	30	28	26	14	
Composting (m^2)	53	-	62	57	55	29	
Anaerobic digester (m^2)	-	452	-	-	-	-	
Impact on empty spaces	5%	36%	6%	6%	7%	14%	
Equipment required:							
Metals - Bins 0.12 m^3	0	2	1	0	2	0	
Paper - Bins 0.24 m^3	2	6	4	4	6	0	
Organic - Bins 0.24 m^3	10	12	12	12	12	6	
Plastic - Cages 2 m^3	2	4	4	4	4	2	

vention on six zones will affect approx. 24,000 inhabitants (11% of the population in Rocinha).

Zones are shown in Figure 4, together with collection or dumping points which will be affected by the implementation of the project. The reduction of Household waste disposed of at each point is shown as a percentage: it is noteworthy to observe the impact on dumping sites surrounding Zone 1 (20% and 22%) and Zone 5 (25%), while percentages of approx. 50% are perceived in the vicinity of Zone 4. An approximately 46% reduction in official collection points will result in an increased collection capacity.

6. CONCLUSIONS AND PERSPECTIVES

The analysis conducted, yielding a preliminary estimation of the distribution of solid waste generation within Rocinha, may prove valuable in the decision-making process for solid waste management.

The method proposed for use in evaluating impact of the intervention can be rapidly applied to different areas, facilitating a preliminary estimation in terms of equipment and spaces needed; this estimation may then be used in support of funding applications. The application of a method based on information gathered and analysed remotely may contribute towards reducing the typically time-expensive planning stage, saving resources which can be consequently be assigned to field data collection and feasibility studies.

Should the proposed intervention be implemented, it is likely to produce a positive impact on the existing situation. A small investment would foster a general improvement of the SWM system. Moreover, in terms of space required, the solution would appear to be compatible with the high population density in Rocinha. Finally, this small-scale approach based on involvement of the informal sector is a good fit with the relative socio-cultural context.

Successful implementation of this proposal would however need to take into account a series of different aspects, as briefly listed below.

A comprehensive economic analysis would be needed, to understand whether the system would prove to be self-sustainable or may be contingent upon public investments.

The participating stakeholders (public authorities, the academy, civil society organizations and NGOs, companies, and waste pickers) should subsequently be identified; the involvement of these stakeholders if of the utmost importance, as a high level of community engagement is mandatory in achieving an effective waste management system. To this regard, the lively civil society of Rocinha would undoubtedly play a key role in the successful implementation of the system and raising of awareness.

The potential integration of the proposal with other institutional programs should also be evaluated, bearing in mind the numerous initiatives undertaken in the past. An agreement should be reached with Comlurb to diversify management of the collection, cover a larger number of areas, and to obtain institutional or economic support of other projects. Every effort should ultimately be taken to avoid conflicts that might negatively affect the successful outcome of the project.

ACKNOWLEDGEMENTS

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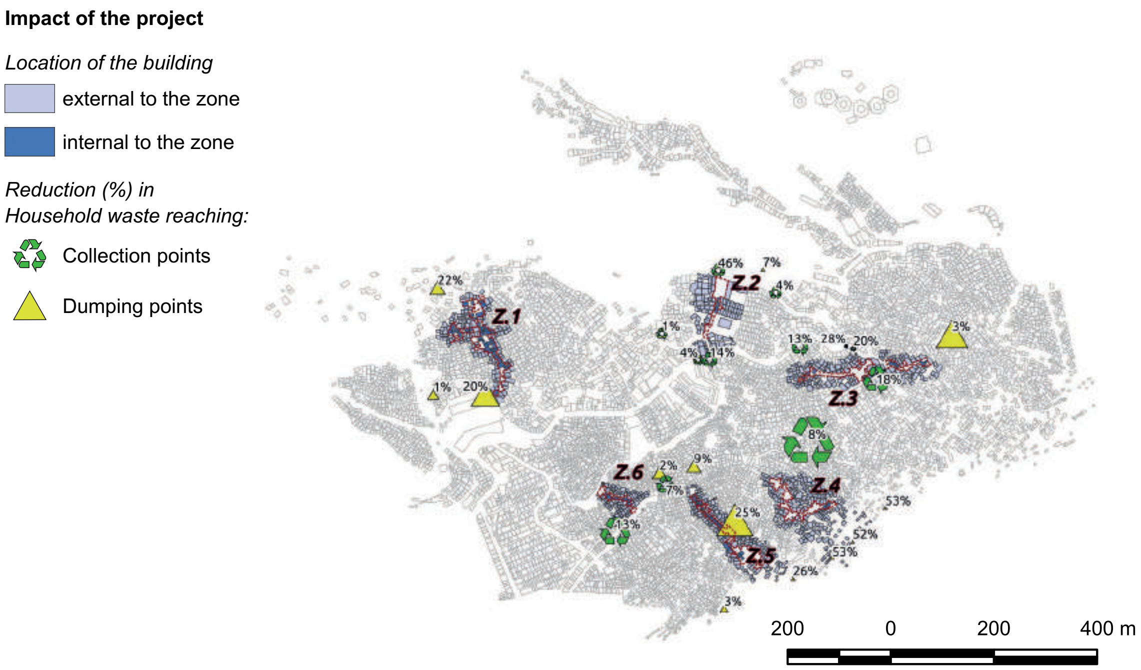

FIGURE 4: Impact of the implementation of the project on the pilot zones: reduction of Household waste reaching collection and dumping points (t/d).

Gisele Silva Barbosa and Roberto Machado Correa; Marco Contardi and Pietro Ceppi from the Foundation Getulio Vargas (FGV); Maria Izabel de Carvalho for sharing her social-based perspective on waste management issues in Rocinha, her hometown; Professor Stefano Mambretti and Professor Gianfranco Becciu; Silvia Ronchi, Hadi Mohammad Zadeh and Anita Tatti, the operational team of Polimipararocinha project, for their efforts.

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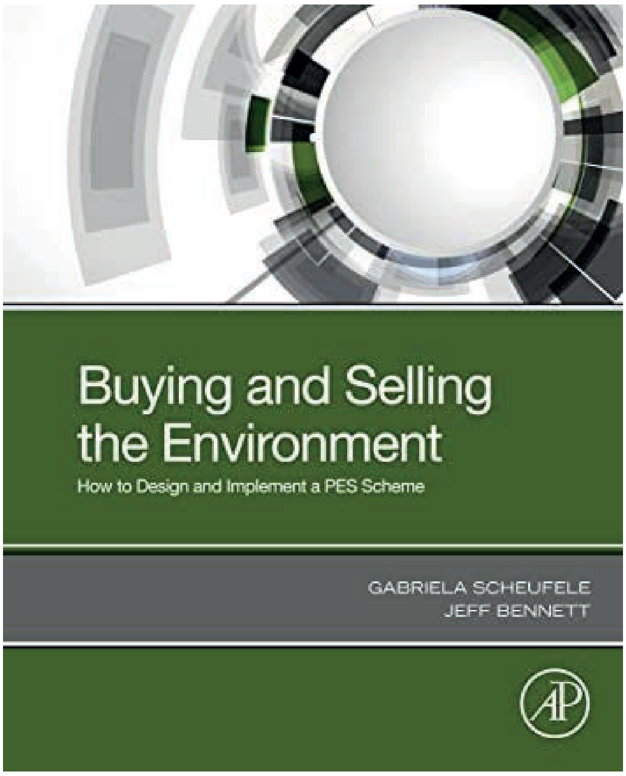
BOOKS REVIEW



BUYING AND SELLING THE ENVIRONMENT - HOW TO DESIGN AND IMPLEMENT A PES SCHEME

Edited by Gabriela Scheufele and Jeff Bennett

Payments for environmental services (PES) are attracting growing interest worldwide as the natural capital becomes increasingly depleted under mounting, often conflicting, land-use pressures, proving definitely greater than socially optimal due to different sources of market failure (e.g., presence of externalities, public recognition of the majority of natural assets, imperfect definition of property rights and asymmetry of information).

PES are widely recognized as cost-effective policy instruments of use in improving environmental protection and management, and rewarding people for their efforts in providing environmental and ecosystem services, such as biodiversity conservation, soil stabilization, carbon storage and sequestration, etc.

PES are voluntary transactions between the consumer and the provider, conditional to the provider securing service provision (Wunder, 2005). They explicitly recognize the need to match the interests of landowners and users both to societal and external interests and internalize what would otherwise be an externality (Engel et al., 2008). The conversion of natural ecosystem areas for anthropogenic activities has progressively reduced the wilderness and natural habitats, thus constituting an increasing threat to environmental services. This ever-emerging scarcity renders environmental services potentially subject to trade, as external beneficiaries of environmental services might be willing to pay landowners to implement the best practises to ensure environmental and ecosystem conservation and restoration. In this respect, PES schemes can be viewed as an attempt to implement the Coase Theorem (Coase, 1960), or alternatively as a subsidy to environmental service providers, eventually combined with a user fee for consumers of environmental services. In this context, PES schemes are an alternative to the direct governmental provision of environmental services and a means of correcting market failures or policy distortions. Nonetheless, PES are unable to address any environmental issues and their scope for application is limited to problems in which externalities play a major role and voluntary approaches are unlikely to be effective and efficient. The successful implementation of PES schemes is indeed affected by the nature of environmental services as public or club goods. PES programs differ in design characteristics and reflect the social, economic and political context in which they are implemented.

Buying and Selling the Environment - How to Design and Implement a PES Scheme, edited by Gabriela Scheufele and Jeff Bennett and recently published by Elsevier, provides an overview on the creation of operational and efficient PES schemes. The stimulus for the book derives from a research project commissioned by the Australian Center for International Agricultural Research (ACIAR) to explore the feasibility of adopting PES schemes for the management of natural resources in the Lao People's Democratic Republic (Lao PES scheme). The book consists of nine chapters and seven annexes, presents a step-by-step approach to the design of PES schemes with a core focussed on applied economics. Chapter 1 provides a general introduction and a road map to readers, as well as the economic foundations and basic principles underpinning PES and PES schemes. In Chapter 2, the characteristics of PES scheme contexts are described, the rationale for developing and implementing a PES scheme is discussed and alternative schemes are investigated. Chapter 3 analyzes the biological and physical background of any PES scheme and the relationship between environmental management actions and their potential outcomes, with the aim of identifying actions to be performed by sellers and benefits gained by buyers. Chapter 4 investigates market

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demand and buyers' willingness to pay for environmental services and describes the most widely implemented techniques to elicit individual preferences and individuals' willingness to pay, whereas Chapter 5 is centred on market supply and sellers' willingness to accept payments to provide environmental services. Chapter 6 describes the PES markets and the equilibrium price is identified, whilst Chapter 7 analyses contractual agreements and the role of intermediaries and brokers. In Chapter 8, performances of PES scheme are assessed and discussed and potential improvements focussed on increasing the cost-effectiveness of PES schemes are investigated. Finally, Chapter 9 provides a review of the main concepts presented in previous chapters, analyzes advantages and pitfalls and presents an overview of the contexts in which PES schemes are most likely to be successfully implemented. In each Chapter, a reference to the application of concepts discussed in the specific context of the Lao People's Democratic Republic is made, and materials produced in the Lao PES scheme project are provided in annexes.

The book is mainly intended as an instruction manual for practitioners, policy makers and advisors in governmental and non-governmental organizations, providing an insight into practical issues and barriers to the design and implementation of efficient PES schemes in real world situations. The volume provides a user-friendly step-by-step guidance of PES schemes, reviews the underpinning basic economic principles and enhances practical knowledge through a case study of natural resource management.

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RESEARCH TO INDUSTRY AND INDUSTRY TO RESEARCH

REMOVING POLLUTANTS, HEAVY METALS AND PFAS FROM LEACHATE USING MAGNETIC FORCE

Leachate is a consequence of landfill processes and operations, which, if not properly managed, monitored and treated, can be harmful to the surrounding environment. Landfill leachates consist of soluble organic and inorganic compounds as well as suspended particles. Depending on weather leachate flow can increase (during rainy season) or decrease (during dry/summer season).

The composition of waste changes dramatically over the life of the landfill due to chemical degradation and biological decay of organic matter present.

In general, landfill leachate pollutants can be categorized into four groups as dissolved organic matter, inorganic macro components, heavy metals, and organic compounds.

The pH of the leachate does not vary much and is often considered as alkaline as the pH is 7.0 to 8.5.

The current paradigm for wastewater and leachate treatment technologies are based on one or a combination of the following solutions:

- Dissolved Air Flotation (DAF)
- Electro-coagulation;
- Aerated lagoon;
- Filtration;
- Biological treatment.

Some of the challenges a landfill owner is facing when choosing a well-suited technology is finding a solution that fits the existing space and constructions, the complexity of the treatment process, the operating expenses and how to handle the sludge from the treatment system in terms of both large volumes and the degree of dry matter content.

Magnetic particle separation: a new hybrid system

The treatment system developed by Mivanor AS is called MivaMag ™, a combination of chemical and mechanical wastewater treatment (Figure 1).

Chemical wastewater treatment

Chemical treatment used in combination with magnetic particle separation is often based on coagulation and flocculation.

Suspended particles vary in source, charge, particle size,

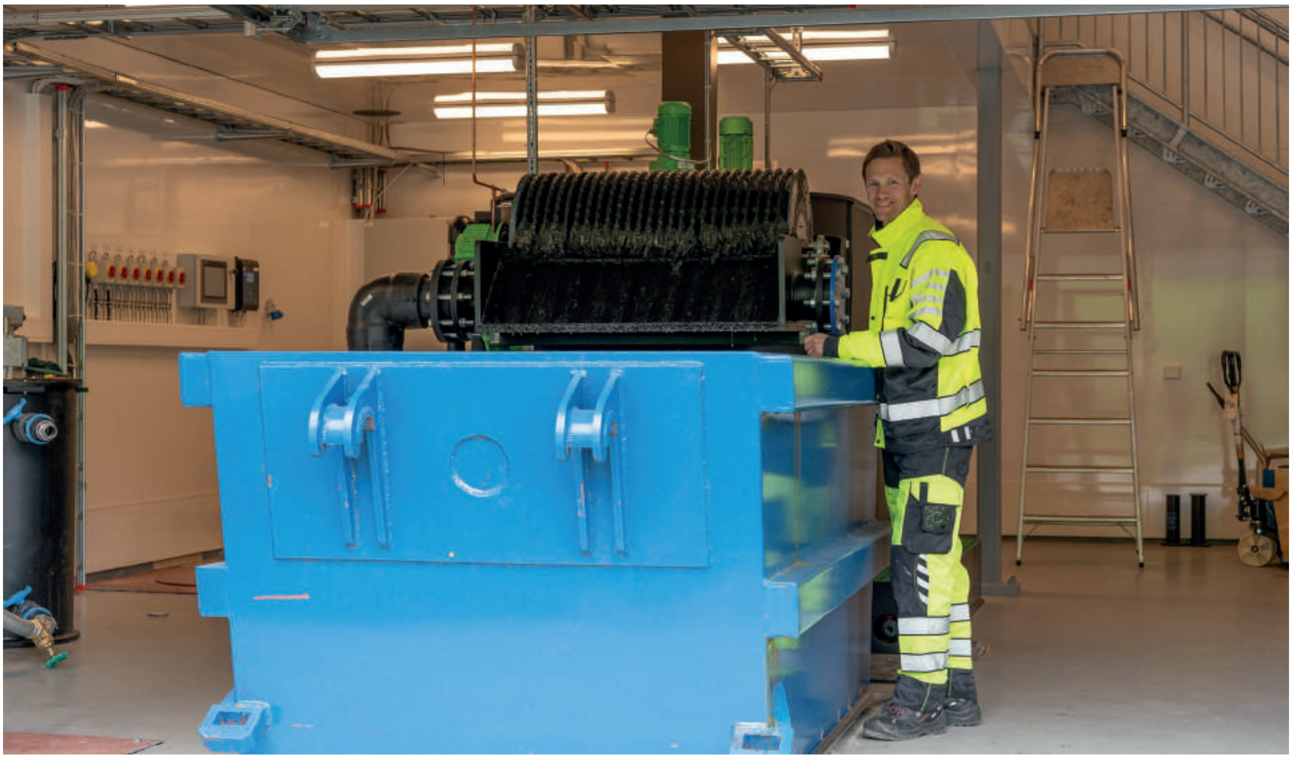

FIGURE 1: The tratment system “MivaMag ™”.

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Detritus / Volume 09 - 2020 / pages III-V
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shape, and density. Correct application of coagulation and flocculation depends upon these factors. Suspended solids in water are the main target and have a negative charge. Since they have the same type of surface charge, they repel each other when they come close together. Therefore, suspended solids will remain in suspension and will not clump together and settle out of the water, unless proper coagulation and flocculation is used.

Coagulant chemicals with positive charges are added to the water to neutralize the negative charges on non-settleable solids (such as clay and color-producing organic substances).

Inorganic coagulants such as aluminum and iron salts are the most commonly used.

Once the charge is neutralized, the small suspended particles can stick together and form so called "micro-floc".

Flocculation increases the particle size from submicroscopic micro-floc to visible suspended particles. Micro-floc particles collide, causing them to bond to produce larger, visible macro-flocs.

The main mechanism of building macro-flocs is based on bridge bonding using organic polymers.

Magnetic Particle separation

Untreated water is pumped into the system, and through mixers where a coagulant and flocculant agent are added. Further the water is led into basins where an adjusted circulation ensures an optimal capture and flocculation.

During flocculation an iron-based powder is added to the process. The powder consists mainly of Fe_3O_4 and it is called magnetite. Magnetite is a rock mineral and has a ferrimagnetic character.

Adding magnetite to the treatment process enables the treatment system to extract the pollutants using magnets as the floc mass floats up to the surface of the basin, and is separated and pressured through a rotating, magnetic drum.

The pollutants are removed from the leachate at the same time as the sludge is dewatered. Dewatering happens due to magnetic forces where small amounts of water the pressed out of each floc.

The technology does not require use of a filter or an extra de-watering unit, something that makes it compact and cost-effective.

Treatment results

Each treatment unit has a capacity of 40 cubic meters of water per hour.

The results recorded from after treating leachate so far show that the magnetic particle separation reduces:

- 50 – 70 % of the organic pollutants
- 25 – 95 % of the of heavy metals
- 80 – 90 % of the phosphorus
- 70 – 90 % of the suspended particles

The sludge has a dry matter content of 30 % to 40 %. The sludge is therefore easy to collect and transport for deposit.

Expanding the leachate treatment to include pfas removal

The treatment process is also possible to expand as Mivanor has developed a demonstrative treatment module for removing PFAS from leachate and wastewater (Figure 2). This research is on-going and Mivanor is exploring the treatment results from a couple of sites dealing with PFAS in water.

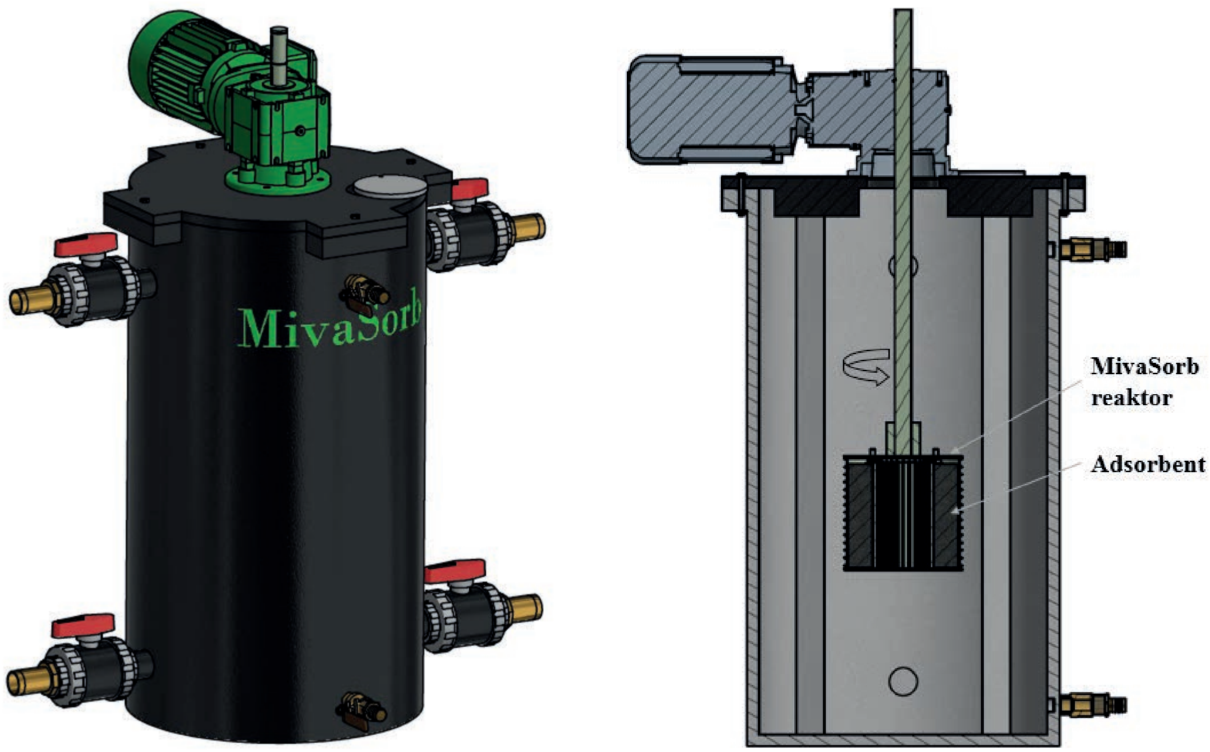

FIGURE 2: Demonstrative treatment module for the removal of PFAS.

Per- and polyfluoroalkyl substances (PFAS) are a group of man-made chemicals that includes PFOA, PFOS and many other chemicals.

PFAS are used in many ordinary consumer goods, as well as industrial products because of the ability to make an object water and oil repellent. Some examples are frying pans, paints, outdoor clothing and also firefighting foam.

These chemicals are persistent in the environment and in the human body, meaning they don't break down and they can accumulate over time – leading to several negative health effects like cancer (for PFOA), thyroid hormone disruption (for PFOS) and general effects on the immune system.

For these reasons, PFAS has risen to be of great importance for the EU and in December 2019 EU launched a strategy for acting and minimize the exposure of PFAS in the environment and to human beings [1].

Most landfill leachates that Mivanor has tested contain about 200-1000 ppt of PFAS mainly PFOS and PFOA.

To enable landfill owners to preserve masses containing PFAS in secured landfill areas and ensure that the leachate does not contaminate the surrounding nature with PFAS, Mivanor has developed a treatment solution that can be added to the magnetic particle separation process as a polishing step.

The product is called MivaSorb, and is designed to remove PFAS from polluted waste- or groundwater through adsorption in a rotating packed bed. MivaSorb consists of a cylindrical rotary container inside a vessel.

Due to rotary movement of the reactor, adsorption medium will enter the reactor from the bottom of the MivaSorb vessel and water is circulated through the reactor and adsorption material. In this way the pollutants encounter the adsorbent many times over. Besides this, loading and re-loading of the reactor with adsorption medium can be automated.

The MivaSorb treatment method enable the treatment to be quite flexible in terms of the possibility to combine of several units and increase the flow capacity. The treatment system is designed so that the loading and reloading is automated, and there is no backwashing required.

Typical treatment results for the PFAS removal is:

- 90-99% reduction of PFAS compounds
- 90% reduction of other ions in water

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Mivanor is a total supplier of treatment solutions for industrial wastewater. In order to ensure that our delivery will fulfil the customer's wishes and requirements, we supply each individual customer with a solution based on small-scale experiments with the relevant water and a carefully considered design.
This provides customers with a good estimate of the treatment results and operation costs prior to a system being ordered. Mivanor is ISO-certified in accordance with quality, health, safety and environment standards.
Today, the company has delivered 16 MivaMag ™ treatment plants.

(1) https://www.documentcloud.org/documents/6586418-EU-Strategy-for-PFASs-FINAL-VERSION-December-19.html

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

by Rainer Stegmann

Artists seldom provide an interpretation of their own work; they leave this to the observer. Each of us will have his/her own individual view of a specific piece of art, seeing different contents and experiencing a range of own feelings and emotions. Bearing this in mind, I created this page where you will find regularly selected masterpieces from different epochs and I express my thoughts on what the work conveys to me personally. My interpretation will refer specifically to the theme "Environment". Any comments or suggestions regarding this column should be addressed to stegmann@tuhh.de

CASPAR DAVID FRIEDRICH / The Wanderer above the clouds - 1817, Kunsthalle (Art Hall) Hamburg, Germany.

What does a well- dressed man with a walking stick alone on a ledge above a sea of fog? The way he stands there gives me the impression of a self-confident, well situated may be successful person. He is looking into the distance. We see nature around with rocks passing through the fog and in the distance gentle mountains. The sky above is lightly clouded and the sun slightly illuminates the scene discretely from the side.

Firstly I brought the person in connection with the anthropocene, the age of humans dominating nature. The way he positioned himself may give the impression that he looks over "his" territory, all mine in the sense "I am lord of nature".

But thinking more about this masterpiece perhaps may be things that we do not see on this piece of art are of importance: around 1800 cities grew quite rapidly, people lived closely together with many animals around, there was a lot of noise and dust. Since there was no or only poor waste and sewage collection and no adequate treatment, most of the excrements and rubbish ended up on the streets or areas near the houses. Many of the streets were not paved which - dependent on the weather - created mud and dust. In some cities once a week people - often prisoners and prostitutes – collected waste and swept the dirt. But even then cities were very dirty and stinky and diseases arose inside the city walls.

May be this elegant person escaped from this unpleasant atmosphere for a while and hiked in the late morning to this beautiful place with a magnificent view in order to enjoy nature, silence and clean air. The fog made his unhealthy city invisible. Perhaps he wanted to gain new power for his life down in the city. He is not sitting down, which means he is only on a short visit to this place, but he took a long way for enjoying this moment.

We have to a certain extent a similar situation today: in big cities air is often highly polluted, traffic produces a lot of noise and hectic, contaminated sites cover significant areas, lot of trees had to give way for new buildings. For restoration often an "artificial nature " is created.

This situation encourages many people to enjoy their holidays in nature. But increasing amounts of people visit these natural areas and we have to pay attention not loosing them...

Next issue: I will present another masterpiece from the Hamburg Art Hall: a painting by Karl Kluth "Coast in North Schleswig", which is located in Northern Germany.
He was a German artist who lived from 1898 until 1972 mainly in Hamburg. He was member of the "Hamburger Session" a group of artists engaging for new impulses and worked for a while together with Edvard Munch in Norway. Later he became Professor at the Hamburg Art School.

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A PHOTO, A FACT, AN EMOTION

"Massive pollution from all waste of the city, leather industry and other factories surrounded Dhaka city makes the major river Buriganga so much polluted that no living creature is in the water anymore."

"TOXIC BATH"

Dhaka, Bangladesh

Mahfuzul Hasan Bhuiyan, Bangladesh

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This photo was selected to participate in the third edition of Waste to Photo in 2019, the photo contest connected to the Sardinia Symposium, International Waste Management and Landfill Symposium organised by IWWG.

Waste to Photo is conceived with the specific aim of recreating a scenario representing the global situation with regard to waste and landfills, ranging from the developing countries to the more industrialised nations.

The 3rd edition of the contest officially closed on 31st May, receiving over a hundred entries. During the Symposium, a photography exhibition was set up using a selection of the most significant shots and a jury consisting of members of the IWWG Managing Board and professional photographers voted for the best photo.

Mr. Mahfuzul Hasan Bhuiyan won the first Prize of the 2019 Edition with the photo “Toxic Bath” taken in Dhaka, Bangladesh, where leather industries make the Buriganga river almost unlivable.

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Mahfuzul Hasan Bhuiyan

He is one of the key Architectural Photographers as well as Documentary Photographer in Bangladesh. He graduated with a Bachelor of Architecture degree from Bangladesh University of Engineering & Technology.

His passion has brought him to photographic practice professionally. His extended interest in different photographic genres inspired him to capture life around the globe.

He is a co-founder of Absurd Photos Ltd and currently represented by ZUMA Press inc.

He has achieved hundreds of international awards in the field of photography and exhibited in several countries as well as his works have been published worldwide. He is one of the co-founders of Bangladesh Society of Photographic Art (BSPA) and Chairman of Patronage & Promotion in Image Colleague Society (ICS), USA.

He has been a fellow of several international photographic clubs worldwide. Mahfuzul also severed as a jury member for many national and international photographic competitions.

(Contents continued from outside back cover)

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CONTENTS

(Contents continued on inside back cover)

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