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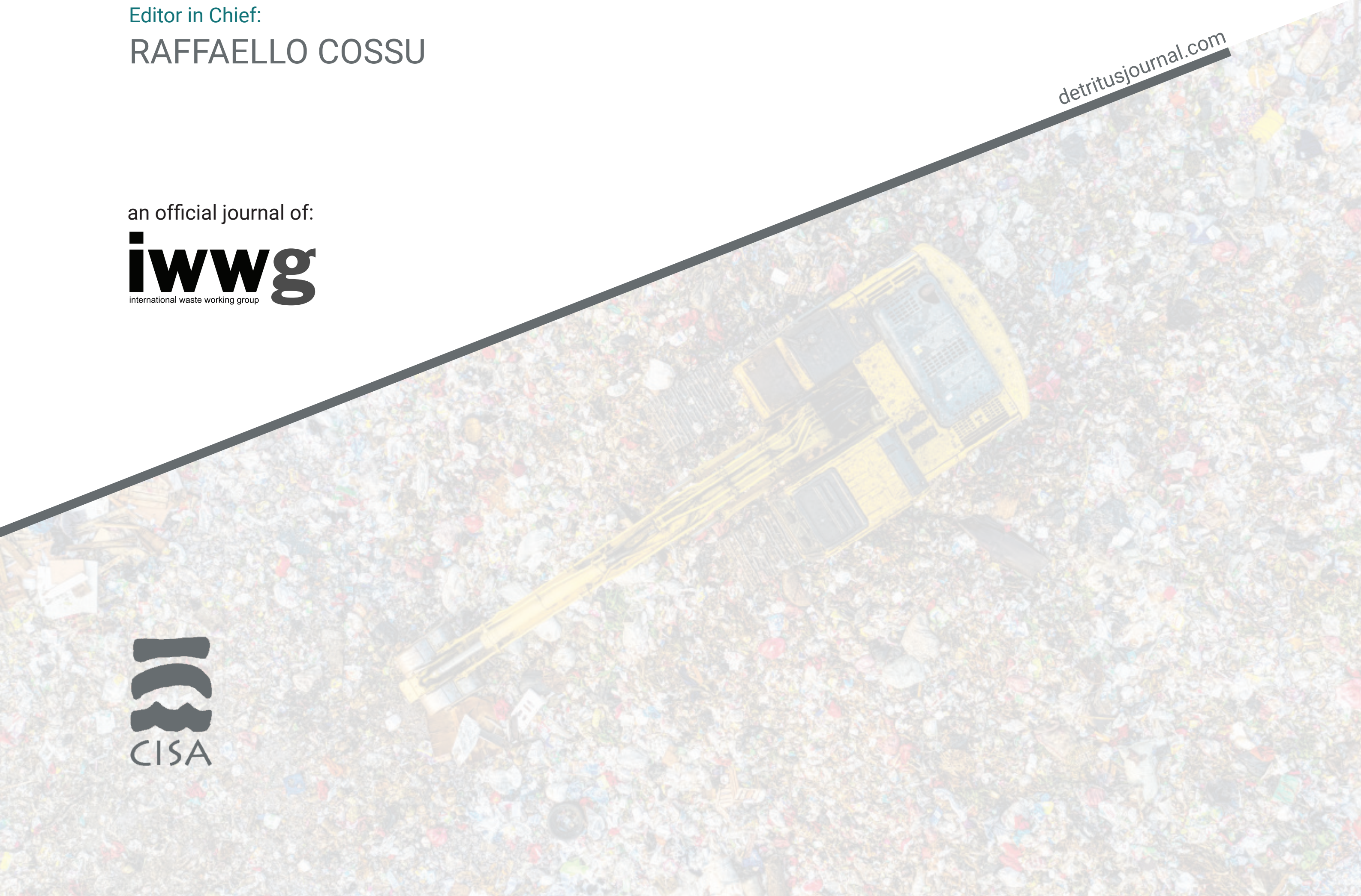

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Editorial

ENHANCED LANDFILL MINING, THE MISSING LINK TO A CIRCULAR ECONOMY 2.0?

PREFACE TO THE SPECIAL ISSUE ON RESOURCE RECOVERY THROUGH ENHANCED LANDFILL MINING

The transition towards a resource-efficient, climate neutral and circular economy is one of the Grand Societal Challenges of today, as recently endorsed by the European Commission in its European Green Deal (EC 2019). As part of this Green Deal the Commission will also launch a new Circular Economy Action Plan (foreseen in March 2020), which will present a "sustainable products" policy that prioritises reducing and reusing materials before actually recycling them, moving up the Waste Hierarchy. Undoubtedly this is a step in the right direction.

Nevertheless, this shift towards a more pro-active Circular Economy vision does not yet address the question of what Europe and other countries in the world will do with the vast amounts of industrial and consumer waste that have been disposed of in waste dumps and landfills over the past 100 years. In this context Enhanced landfill mining (ELFM) has been proposed as an "out of the box" approach to address the dark side of the Circular Economy vision: i.e. how can we deal with the waste of the past, irrespective of the urgent need to avoid new waste creation and disposal in the future?

As regards Municipal Solid Waste landfills, ELFM has been defined as "the safe conditioning, excavation and integrated valorisation of landfilled waste streams as materials and energy, using innovative transformation technologies and respecting the most stringent social and ecological criteria" (Jones et al. 2013). In this paradigm landfills are considered as resource stocks awaiting future mining in order to recover valuable resources in an integrated way. As has been estimated by the European Enhanced Landfill Mining Consortium (EURELCO), Europe comprises most likely more than 500,000 landfills (Jones 2018) (which is an upgrade of the previous estimation of 150,000 – 500,000 landfills). More importantly, approximately 90% of these landfills are to be considered as "non-sanitary" landfills, predating the EU Landfill Directive of 1999. In order to avoid future environmental and health problems, many of these landfills will at some point require expensive remediation measures. For these landfills, ELFM can be seen as a combined resource recovery and remediation strategy, which can drastically reduce future remediation costs, reclaim valuable land, while at the same time unlocking valuable resources.

Special Issue

The present Detritus special issue "Resource recovery through Enhanced Landfill mining" presents the results of the Horizon 2020 MSCA Innovative Training Network NEW-MINE, a project which aims to develop innovative concepts, technologies and methods for integrated resource recovery and remediation of landfills containing Municipal Solid Waste (MSW). More specifically, the project investigates the full value chain of landfill exploration, excavation, material separation, recovery and upcycling of landfilled materials and energy resources as well as the reclamation of land. In addition, an integrated environmental and economic assessment framework for ELFM is developed and the stakeholders perspectives on ELFM, i.e. the Social License to Operate (SLO), is studied.

NEW-MINE flowsheet

The NEW-MINE project is based on the flowsheet presented in Figure 1, which is the outcome of concerted research performed at the Remo landfill site in Houthalen-Helchteren, Belgium (Jones et al. 2013). This flowsheet, which is to be considered as just one of the possible pathways for performing ELFM, consists of several key operations: waste excavation, mechanical processing to produce recycled materials (e.g. ferrous and non-ferrous metals, aggregates) and to generate a fraction concentrating the components with high calorific value (wood, plastics, textile, etc.). This high calorific fraction is subsequently fed to a thermochemical conversion process, in which a synthetic gas is produced that can be further processed to produce hydrogen, methane or biofuels (Bosmans et al. 2013). As a by-product of the thermochemical conversion, and depending on the applied process, slags or ashes are formed, which subsequently can be upcycled to produce building materials, such as inorganic polymer binders or glass ceramics (Machiels et al. 2017).

Overcoming technological barriers towards ELFM

The ELFM concept is based on treating landfills as anthropogenic deposits (Krook and Leenard 2013). Consequently, the whole value chain known from traditional mining – i.e. exploration, excavation and processing – can be adapted to assess the resource potential of the landfills. Efforts have been made to use geophysical data for the

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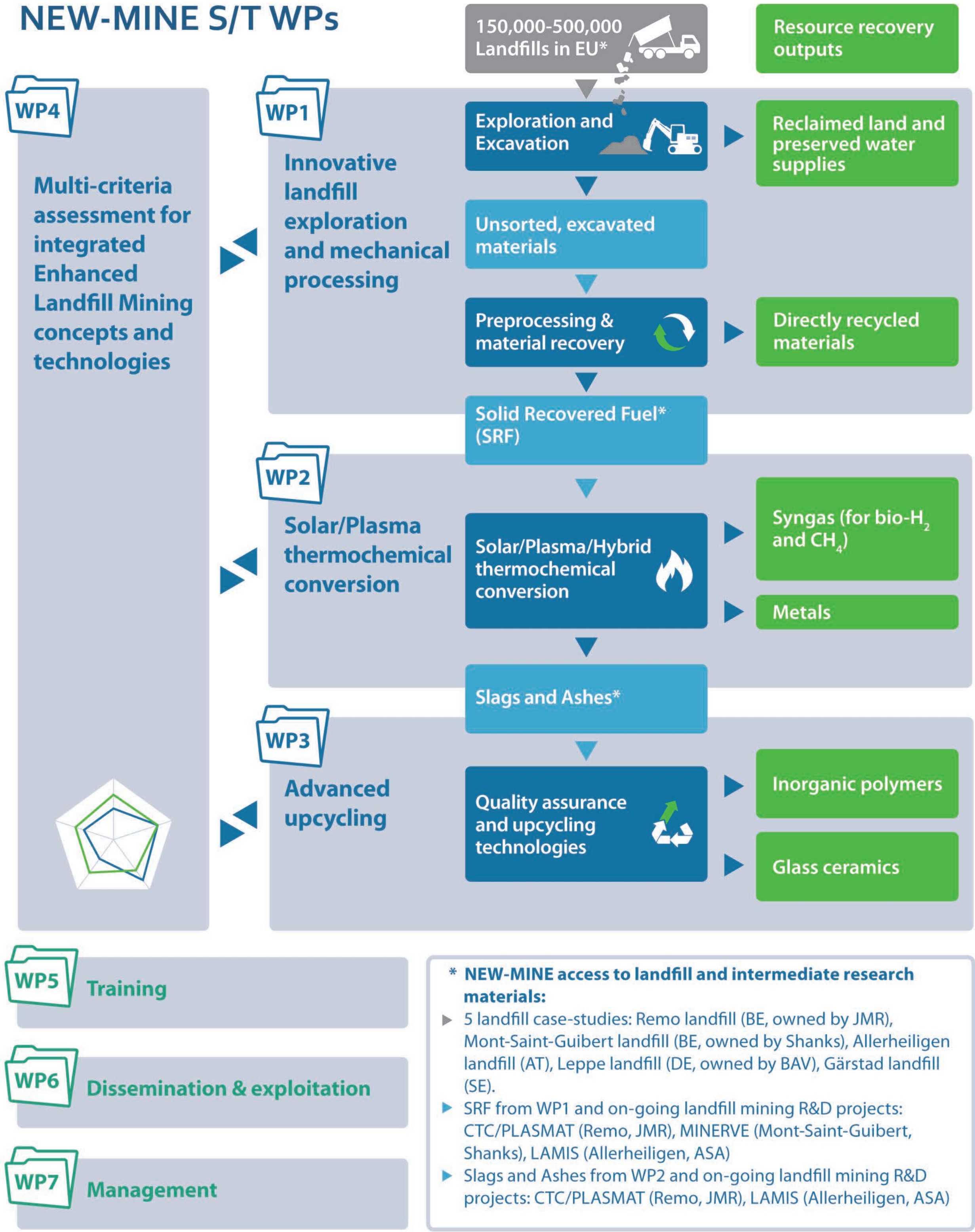

FIGURE 1: Overview of the NEW-MINE work packages and value chain approach

delineation of landfill bodies and to distinguish zones within a landfill containing different waste types (e.g. Municipal Solid Waste versus Industrial Waste). In the NEW-MINE project geophysical data have been correlated with drilling data, from which the different components in the waste can be quantified. A key question is whether geophysical measurements can be used to determine the presence of certain components (e.g. metals), or even better, to quantify their amounts. With respect to this challenge, Vollprecht et al. (this volume) determine whether the metal content of MSW landfills can be estimated through measurement of the magnetic properties.

Once excavated, the waste from the landfills is passed through a series of mechanical separation processes. Within NEW-MINE, state-of-the art processing schemes as well as novel methods have been explored (Figure 2. The key aspect is to understand how excavated, surface-defiled, aged waste from landfills behaves in these processing schemes with respect to fresh MSW. In this Special Issue the results of the tests with both ballistic separators (Garcia Lopez, this volume) and sensor-based sorting (Küppers, this volume) are reported. As non-ferrous metals can form an important part of the revenues of ELFM, the non-ferrous metal output produced in the mechanical separation has been studied in detail (Lucas et al., this volume).

Particular attention is paid to the “fines” fraction, i.e. the size fraction < 90 mm. This fraction often represents the Achilles heel for an ELFM operation as it typically comprises more than 50% of the waste volume and hardly any valorisation solutions have been identified for it (Quaghebeur et al., 2013). A detailed characterisation of the fines fraction of the Mont-Saint-Guibert landfill in Belgium provides more insight in these materials (Hernández Parrodi, this volume). In a corresponding paper, Hernández Parrodi (this volume) presents a detailed fines processing scheme involving particle size classification, ferrous- and non-ferrous metal extraction, density separation and sensor-based sorting, all with the aim to produce different output fractions that can be valorised, including a soil-like fraction, building aggregates as well as ferrous and non-ferrous fractions.

Another key fraction that needs to be valorised in an ELFM operation is the high calorific fraction, which is one of the outputs from the mechanical processing of the excavated waste. This fraction can be thermochemically converted to produce a synthetic gas, metals and, depending on the conversion technology used, slags or ashes. To improve the profitability of an ELFM project, the outputs of the thermochemical conversion are further upcycled to products with a high added value. Firstly, syngas can be upcycled to produce H_2, CH_4 or liquid fuels. Secondly, slags and ashes can be upcycled to produce binders and low-carbon building materials. Within NEW-MINE, two routes have been studied for valorisation of slags/ashes, i.e. the production of inorganic polymers (Ascensão et al., this volume) and glass ceramics (Rabelo Monich et al., this volume).

Within NEW-MINE, an integrated systems analysis framework has also been developed, in order to specify key economic, environmental, technological, social, market and policy conditions and measures for facilitating the implementation of ELFM projects. Hernández Parrodi et al. (this volume) aim to embed landfill mining as a strategy in current waste management systems, taking into account (i) reduction of the landfill volume, (ii) reduction of risks and impact and (iii) increase in resource recovery and overall profitability. In the work of Esguerra et al. (this volume) the economic assessments performed in the framework of landfill mining are reviewed. Finally, while assessments of ELFM have until now mainly focused on environmental and private economic issues, societal impacts have rarely been analysed. Einhäupl et al. (this volume) therefore developed a method for integrating stakeholder archetypes in the assessment of ELFM projects.

FIGURE 2: Overview of key aspects of the NEW-MINE value chain. Upper part from left to right: Geophysical exploration, excavation and ballistic separation at the Mont-Saint-Guibert landfill in Belgium. Lower part from left to right: sensor based sorting, tapping of slag derived from gasification of the calorific fraction derived from EFFM, pavers produces using an inorganic polymer derived from ELFM SLAG. *Pictures from C. Bobe, C. Garcia Lopez and J.C. Hernàndez Parrodi.*

FIGURE 3: Public acceptance as a key aspect in the implementation of an ELFM project. *Source: https://www.geograph.org.uk/photo/372466*

To mine or not to mine, that's the question

This brings us to the question: what is the status of ELFM in terms of real-world implementation? Despite progress in the technical aspects of ELFM, at this point, the first, full-scale industrial, resource recovery-driven ELFM-project is still to be developed. Multiple barriers seem to persist (see Figure 3), as confirmed also by experiences outside New-MINE, in Italy (Cappa, this volume).

First of all, market barriers for ELFM remain: ELFM-derived (recycled) products need to compete with too cheap primary resources, as external environmental and health costs are typically not internalised in their price. As a result, only when land reclamation can provide substantial additional revenues will the economics of ELFM become positive in the present market context. Furthermore, several industries are still reluctant to absorb ELFM-derived materials.

Secondly, local communities may take some convincing about ELFM projects in their backyard, as experienced in the Remo landfill case. The resistance of only a handful people who (metaphorically) take up arms and initiate time-consuming court cases can be enough to block an ELFM project for years. Obtaining the Social License to Operate for ELFM projects is, therefore, not straightforward, even when multi-actor facilitation processes and "citizen science" concepts are employed.

Finally, legislation for ELFM on the EU level has not yet come to terms with the dynamics of the ELFM concept. The fact that the ELFM Amendment that was agreed by the European Parliament in 2017 was later blocked by the European Council highlights that there is still a long way to go before ELFM is accepted as the new standard by policy makers. This represents a major delay for getting ELFM implemented at the EU-level.

In reality this implies that Europe basically still considers landfills as "end stations" for obsolete waste, rather than as "dynamic resource stocks" that can be re-injected into the economy when the time and the economics are right. The importance of the required paradigm shift with respect to the definition of a "landfill" – from a static (linear) view towards a dynamic (circular) perspective – will need to be put on the agenda again in the coming years so that we can clean up our historic waste legacy. Only then can we truly speak of a Circular Economy 2.0 version in which "climate and resource frontrunners" (EC 2019) are facilitated rather than blocked.

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CHARACTERIZATION OF LANDFILL MINING MATERIAL AFTER BALLISTIC SEPARATION TO EVALUATE MATERIAL AND ENERGY RECOVERY POTENTIAL

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ABSTRACT

For decades, ballistic separators have been used in Europe as a means of sorting waste to separate mixed waste material streams at material recovery facilities and municipal solid waste treatment plants. Currently, with the growing need to remediate landfill sites, ballistic separators can be employed to recover calorific fractions from excavated landfill material within the framework of enhanced landfill mining. Ballistic separators provide multiple separation steps in one machine: they sort flat two-dimensional materials from rigid three-dimensional materials, while the material is screened to a selected particle size at the same time. The present study shows the results obtained during an investigation performed at the landfill in Mont-Saint-Guibert, Belgium. The main objectives were to acquire first-hand information regarding the efficiency of the ballistic separator in relation to processing old and untreated landfilled material and to study the potential of the landfill as a reservoir of secondary resources. The excavated material was processed through a pre-treatment chain of steps, including material classification and separation, as well as particle size reduction. As a first step, the material was processed with a ballistic separator using two different mesh sizes, 200 mm and 90 mm. Subsequently, the performance of the separator in question was evaluated, especially regarding its effectiveness in the production of refuse derived fuel. The two-dimensional flow was characterized by combustible materials from municipal solid waste and the three-dimensional by construction and demolition waste. As a result, 46% (dry basis) of the input material were fines particles <20 mm, 3% had a calorific value of 22.4 MJ kg^{-1}, 1% 16.0 MJ kg^{-1} and approximately 1% were magnetic metals that could be recovered by mechanical processes. The results of processing and valorising the potential resources disposed in landfills are essential for the implementation of enhanced landfill mining since revenues from material and energy recovery could contribute to the economic feasibility of the project.

1. INTRODUCTION

As the global demand for raw materials rises, while the availability of natural resources remains limited, alternative sources of materials demand investigation and, therefore, new techniques need to be developed in order to maximize the use of existing materials. A political framework to tackle these challenges was set by the 2020 Strategy of the European Union (EU). This strategy serves as the basis for the 7th Environment Action Programme, which seeks to develop a sustainable, low-carbon and resource-efficient economy where waste is turned into a resource (European Parliament, 2013). Regarding historic waste, it is of major interest to identify the potential and examine which secondary resources could be recovered from the existing 150,000 to 500,000 landfills that are estimated to be located in the EU (Hogland et al., 2010). This subject is investigated in enhanced landfill mining (ELFM) projects, which aim for "the safe conditioning, excavation and integrated valorisation of (historic and/or future) landfilled waste streams, both as materials (Waste-to-Material, WtM) and energy recovery (Waste-to-Energy, WtE), using innovative transformation technologies and respecting the most stringent social and ecological criteria" (Jones et al., 2013). ELFM contributes to create a circular economy and to reduce the EU's dependency on imports of raw materials, driven by the goals of reclaiming land, re-

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gaining landfill capacity and protecting groundwater, as has previously been achieved by landfill mining (LFM) projects (Hermann et al., 2014; Hernández Parrodi et al., 2018a).

This study focuses on the characterization of landfilled material that was directly processed with a ballistic separator after excavation. Figure 1 shows the working principle of a ballistic separator, which is a standard processing equipment, normally installed in packaging waste treatment plants before sensor-based sorting or after in-feed with subsequent drum screen. Due to the inclination and upward movement of the paddles, three-dimensional (3D) materials (heavy, hard and round particles) move downwards and are separated from the two-dimensional (2D) materials (lighter and soft particles), such as plastics, paper and textiles, which are collected at the top end of the paddles. A third output stream, the under-sieve fraction, is produced due to the screening property of the paddles. The latter can be varied by adjusting the screen of the paddles according to a certain particle size. Material characteristics, such as weight, form, size and elasticity, can influence the movement of the particles and can, therefore, affect the sorting efficiency of the equipment.

A bottom-up approach, in contrast to a top-down approach analysing historical data (Bhatnagar et al., 2017), was chosen in this study. Especially since most older landfill sites did not register the type and amount of material deposited during their active phase (Jones et al., 2013). However, landfilled materials are heterogeneous waste streams that need to be separated and treated before they can be recovered (Quaghebeur et al., 2013). Previous studies provide information on the applicability of full-scale and state-of-the-art technology used currently in waste-treatment plants (García López et al., 2018; Kaartinen et al., 2013; Maul & Pretz, 2016).

The novelty of this case study is the use of a ballistic separator as the first step of the mechanical process prior to shredding. This method allows the recuperation of fractions that are suitable for the production of refuse derived fuel (RDF) with a high heating value in its original size, in addition to other valuable materials, such as inert, glass and metallic materials.

Landfilled waste from which RDF might be produced includes plastics, paper, wood and textiles (calorific fractions). In addition, metals, glass and perhaps mineral and organic materials could be recovered (Kaartinen et al., 2013; Quaghebeur et al., 2013). The material composition of landfill sites varies according to different factors, such as the time of deposition, the type of stored waste, the meteorological and hydrological conditions of the site, and the collection area, since waste composition is influenced by population density, consumer behaviour and waste-sorting habits (Quaghebeur et al., 2013; Wolfsberger et al., 2015a). Therefore, a holistic characterization of the landfilled waste contributes to assessing the suitability of a site for ELFM, which is determined by the share of usable recyclables from the excavated waste, among others, and to predict the revenue and costs of an ELFM project (Hermann et al., 2014).

The purpose of this study is to evaluate the performance of the ballistic separator "STT 6000"-STADLER® Anlagenbau GmbH" with landfilled material. This evaluation was achieved by characterizing each output stream of the ballistic separator. The materials composing these streams were identified by manual sorting and were subjected to laboratory analyses, such as moisture content, particle-size distribution (PSD), calorific value and ash content, to obtain qualitative and quantitative information and determine the suitability of the calorific fractions for the production of RDF.

2. MATERIALS AND METHODS

2.1 Site description

The examined landfill site is located in the municipality of Mont-Saint-Guibert (MSG) in the province of Walloon Bravant, Belgium. This site was established on a former sand

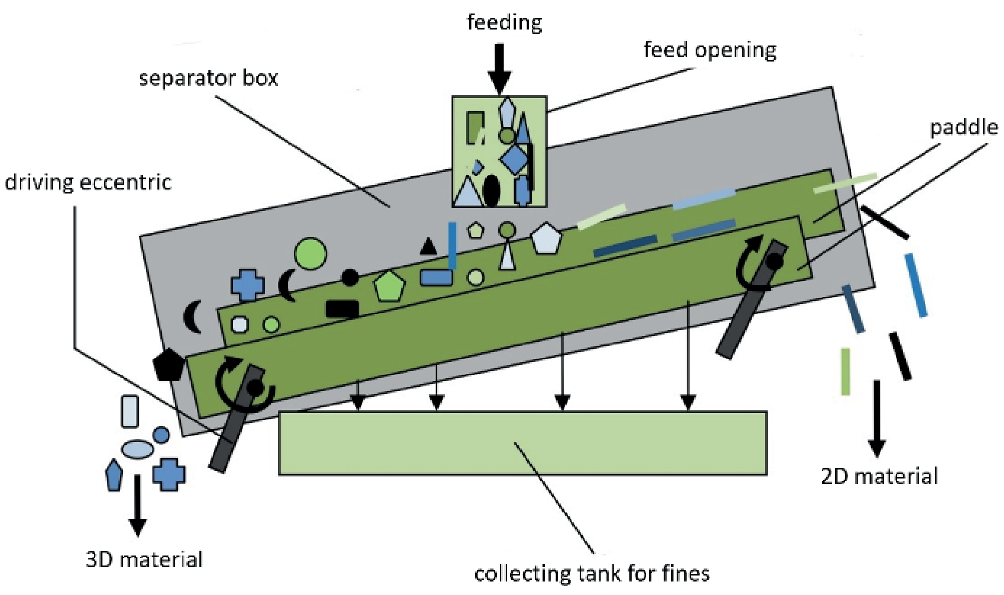


FIGURE 1: Schematic diagram of a ballistic separator (adopted from Martens & Goldmann, 2016).

quarry and has served as one of the main disposal sites of municipal solid waste (MSW), non-hazardous industrial waste and construction and demolition waste (CDW) since 1958 (Bureau d´études greisch, 2002). The site covers an area of approximately 44 ha, of which 26 ha belong to the most recent part and 2 ha to the oldest part. The oldest part of the site has no bottom liner and, nowadays, the biogas collection system has been removed, while the leachate collection system is still in place and operational. The present investigation was carried out in the old part of the landfill, which has an estimated depth between 30 m and 60 m and where at least 5.7 million m^3 of waste were deposited between 1958 and 1985 (Hernández Parrodi et al., 2018b).

In September 2017, before the excavation, a geophysical exploration was performed in an area of approximately 2150 m^2. Using electromagnetic induction, the depth of the cover layer and the soil properties were estimated. Based on the results of the electrical conductivity, the area was then divided into four batches. This paper focusses only on batch 1. A future publication will show the results of all four batches.

As shown in Figure 2, the waste was covered by a clay layer with a thickness of about 4 m. This layer was removed in order to keep it separated from the landfill waste. The processed pit was 5 m long, 5 m wide and 4 m deep; a total volume of 130 m^3 was excavated and treated. During the excavation, a layering of different types of materials was observed. From top to bottom: 4 m cover layer, 2 m CDW and 2 m MSW. For the extraction of the buried material, an excavator with a toothed digging-type bucket was employed, while for the manipulation of the excavated material a wheel loader was used. The weight of the material was measured with a weighing bridge (resolution of 50 kg).

2.2 Mechanical processing and sampling campaign

After the excavation, the material was directly fed into the ballistic separator "STT 6000", the specifications of which are given in the Appendix. The motivation for the choice of this specific equipment was the following: i) the conglomerates of the input material would be loosened up due to the agitation (crankshaft eccentricity) on the screen deck, ii) its availability for treating CDW (Sigmund, 2018), iii) no pre-shredding of the input material is needed (saving the wear and energy on the shredders for the infeed material), iv) sorting large items increases sorting quality (large parts can be removed in one piece), v) saving of space on the site due to the separation of 3 fractions in one step (vs. a drum type, by which only particle size can be sorted), and vi) its robust design. Thus, these characteristics could lead to better sorting processes (enhanced mechanical treatment) with respect to effectiveness, wear and energy consumption. Even if previous studies recommend low moisture content (<15%) for an effective sorting process (Giani et al., 2016; Martens & Goldmann, 2016), this study was performed without drying the material prior to the processing.

FIGURE 2: Excavation at the Mont Saint Guibert Landfill, Belgium.

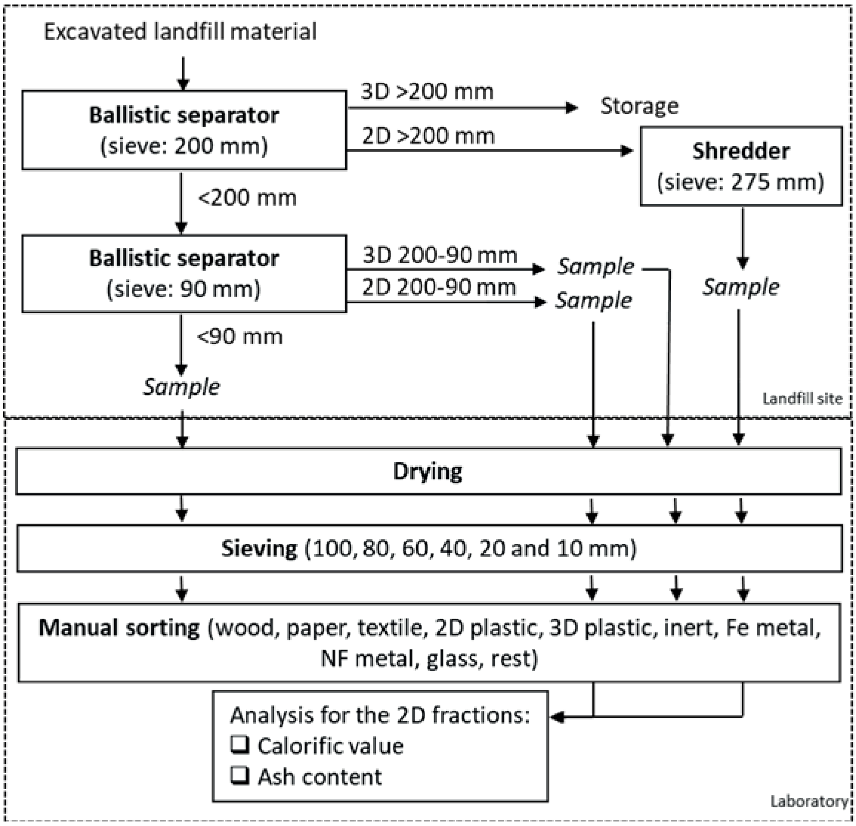


FIGURE 3: Scheme of the mechanical pretreatment, sampling campaign and methodology used in the laboratory.

The landfilled material was first sieved with a mesh size of 200 mm and subsequently with a mesh size of 90 mm, as can be seen in Figure 3. The output "2D >200 mm" was fed into a shredder (sieve: 275 mm) to reduce the particle size down to <275 mm; hereafter, this output stream is referred to as "2D <275 mm".

The sampling campaign followed the same methodology as in the case study in Halbenrain (García López et al., 2018), based on the German Directives (LAGA PN 78; LAGA PN 98). The "3D >200 mm" output stream was not sampled; instead, all the output material was sorted in situ into different categories (Table 1). Moreover, 60 m^3 of the obtained fraction <200 mm were subsequently processed with the ballistic separator with a mesh size of 90 mm, from which 3 additional output streams were obtained: 2D 200-90 mm, 3D 200-90 mm and <90 mm.

The following numbers of samples were taken exclusively during the mechanical treatment and were adapted to the time of the process to achieve the maximum level of representation: i) 2D <275 mm, 8 samples (n=8), summing to 132 kg, ii) 2D 200-90 mm, 9 samples (n=9), summing to 63 kg, iii) 3D 200-90 mm, 7 samples (n=7),

TABLE 1: Classification of the landfilled material by categories.

Category	Material	Age of site (as of 2018)
I	Wood	All types of wood
II	Paper	Paper, cardboard, composite carton
III	Textile	All types of textiles
IV	Plastic 2D	Bags, foils
V	Plastic 3D	PP, PET, HDPE, LDPE, PVC, PS, others
VI	Fe metals	Iron
VII	NFe metals	Non ferrous metals: copper, aluminium, lead, others
VIII	Inert	Mineral fraction (stones), ceramic
IX	Glass	Colorless glass, green glass, brown glass, others
X	Rest	Rubber, foam, EPS, silicone, melted plastics, sandpaper, hazardous waste (e.g. sanitary material), unidentified, composites
XI	Fines	Particles < 20 mm

summing to 154 kg and iv) <90 mm, 12 samples (n=12), summing to 116 kg. All the figures with fluctuations given in this study are based on the corresponding number of samples. The samples were further characterized in the Department of Processing and Recycling at the RWTH Aachen University.

2.3 Characterization of landfill mining material

2.3.1 Moisture content and particle size distribution

Based on the DIN CEN/TS 15414-1 all samples were dried but at a reduced temperature of 75°C to prevent plastics from melting, which can happen at higher temperatures.

After drying, the samples from the output streams 3D 200-90 mm, 2D 200-90 mm, <90 mm were sieved with a drum sieve and a box sieve at the Department of Processing and Recycling (RWTH Aachen University), except for the ones from 2D <275 mm, which did not provide realistic information regarding the size of the original material. As a result, seven particle size fractions were generated: 200-100 mm, 100-80 mm, 80-60 mm, 60-40 mm, 40-20 mm, 20-10mm and <10 mm.

2.3.2 Material composition by output stream

All the particle size fractions >20 mm from 2D <275 mm, 2D 200-90 mm, 3D 200-90 mm and <90 mm were sorted manually into eleven categories, listed in Table 1. There is no category for organic waste (food scraps, green waste, etc.) because they were not distinguishable after at least 15 years inside the landfill. It is likely that the organic material was degraded to soil-like material (Quaghebeur et al., 2013). At this point, it must be mentioned that no pure and clean materials can be obtained by manual sorting without washing or using other pretreatment due to surface defilement and material agglomeration.

2.3.3 Calorific value and ash content

Such characteristics as calorific value, amount of organic carbon, total carbon, ash content, and hydrogen and nitrogen contents are needed to be measured to assess the efficiency for WtE applications (Quaghebeur et al., 2013). In this case, only the gross calorific value (GCV) and the ash content were determined. These values give information about the recoverable energy and amount of residue produced in the combustion process (Kuchta, Hobohm, & Flamme, 2017). Before conducting the analysis, several mills (hammer mill, disc mill and cutting mill) were used to reduce the size of the particles down to 1 mm. After the particle size reduction of each category in each sample, the GCV was determined according to the DIN 51900-1 and the ash content based on DIN EN 15403. The measurements were only conducted for the 2D output fractions (2D 200-90 mm and 2D <275 mm), which were estimated to have a high heating value. Metals and glass were assumed to have a GCV of 0 J/kg and an ash content of 100%.

3. RESULTS AND DISCUSSION

3.1 Moisture content and particle size distribution

The moisture content landfilled material plays an important role when considering the material processing (Hull et al., 2005). Previous experiences include that moisture contained in excavated waste did not impede its processability, but it might have an impact on the processing efficiency (Kaartinen et al., 2013). Drying could (i) reduce the amount of surface defilement; increase both the quality of the recyclable materials and the efficiency of sorting processes, (ii) enable a more efficient and precise particle size classification in the screening and sieving processes, (iii) decrease the total amount of material to be processed and perhaps transported and (iv) raise the calorific value (Hernández Parrodi et al., 2018a).

The fluctuations in the moisture content within the samples of every output stream are represented in Figure 4. The similar moisture contents of 2D <275 mm and 2D 200-90 mm, with averages of 31% and 32% respectively, allow a first estimation about their composition, which is described in detail in the section "3.2. Material composition". The output "<90 mm" is characterized by a slightly lower average moisture content of 28%, while "3D 200-90" mm contains considerably less water, showing an average of 12%. The latter correlates with the low capacity to hold water of materials usually found in the 3D output stream of ballistic separators, such as stones, metals, glass, rubber and wood.

It has been reported that the moisture content and amount of organic matter are interrelated and decrease with the age of the waste due to microbiological activity (Quaghebeur et al., 2013). Dating back to the 1960s and 1970s, the investigated waste from MSG can be considered as old. Nevertheless, the amount of water is still notable, with a range of 9-41%, which might be explained by the thick layer of clay used as cover material and the type of waste landfilled. More permeable material, such as compost, leads to higher degradation rates and thus faster water reduction.

Due to this moisture, fine particles likely adhere to surfaces and larger particles which may lead to an increased share of the fine fraction after drying (Kaartinen et al., 2013) .Combined with the subsequent sieving, this may contribute to the reduction of surface defilement and as a consequence, compliance with the requirements for other waste treatments, e.g., sensor-based sorting if considered for further processing.

Drying the material is an operational cost but can reduce the total mass being transported and fed into the mechanical processing; hence, the throughput might be increased and transportation costs reduced. Other effects of drying are an increase in the CV and more effective sieving results.

Figure 5 illustrates the total mass (dm%) by particle size (mm) in each sieved output stream from which it can be deduced that a dried sample of landfilled material contains more fine particles than it did under humid conditions. After drying, 51% of the output stream "2D 200-90 mm" is smaller than 80 mm, even though it was first classified as >90 mm by the sieve of the ballistic separator. Moreover, fines <10 mm in this output stream make up 30%, which is almost as high as the share of particles >100 mm (35%).

For the "3D 200-90 mm", 28% is <80 mm. This statement can only be made with some reservations, since

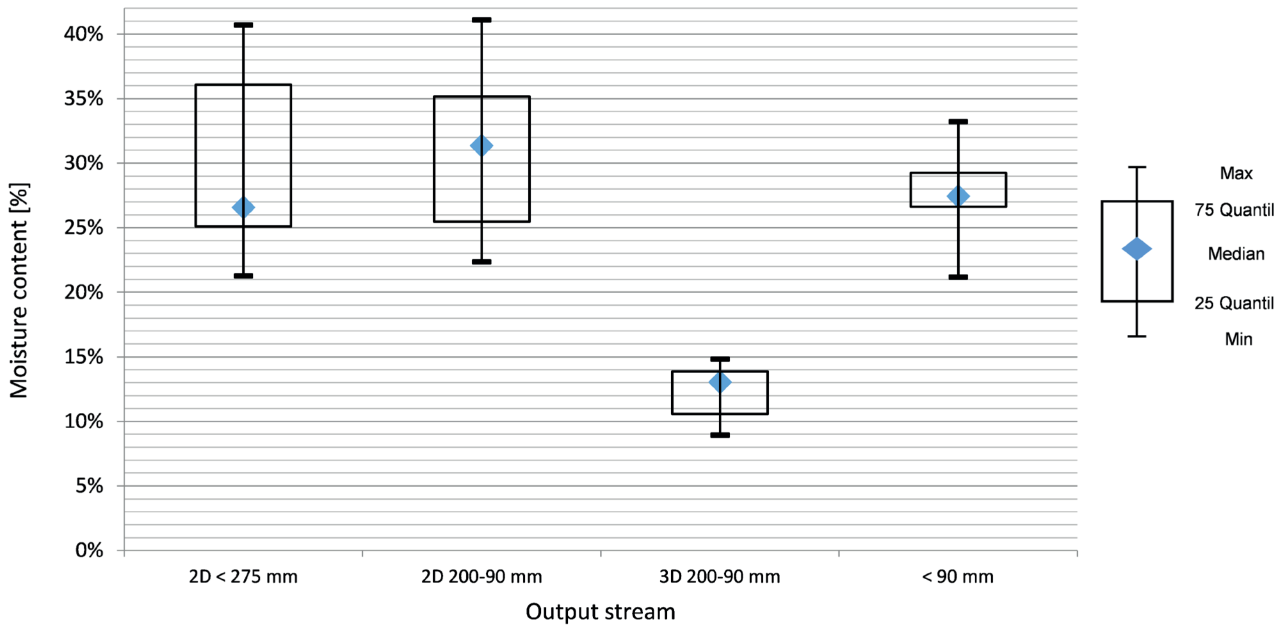
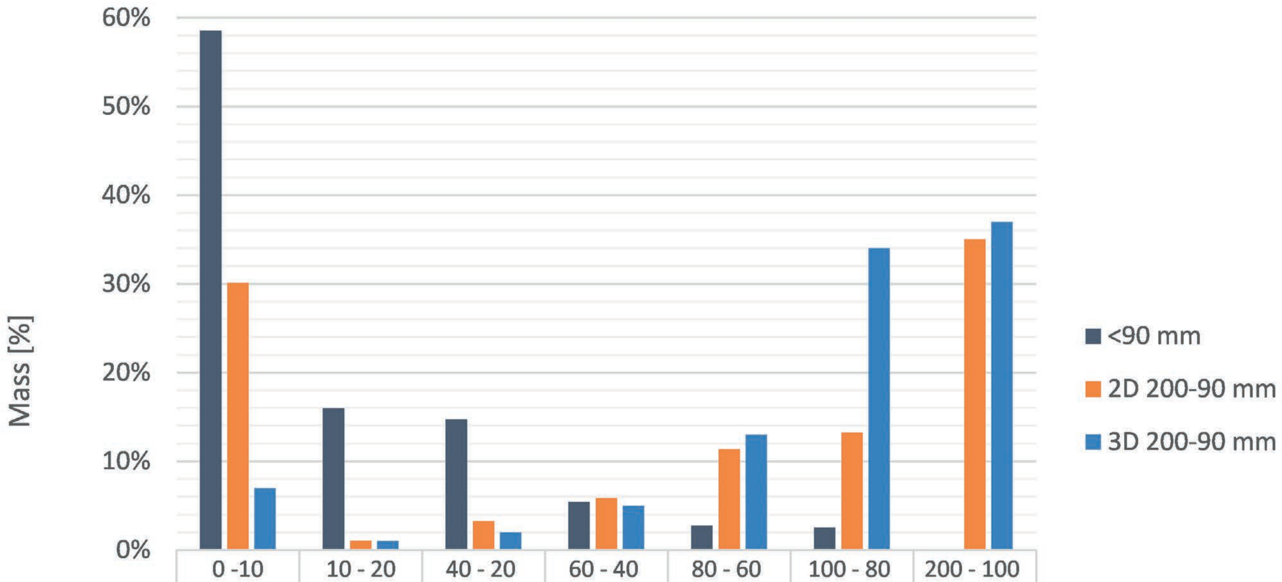


FIGURE 4: Fluctuations in the moisture content by the output stream of the ballistic separator. Number of samples shown, n=8, n=9, n=7, n=12.



	0 -10	10 - 20	40 - 20	60 - 40	80 - 60	100 - 80	200 - 100
<90 mm	58,5%	16,0%	14,7%	5,4%	2,8%	2,6%	0,0%
2D 200-90 mm	30,1%	1,1%	3,3%	5,8%	11,4%	13,3%	35,0%
3D 200-90 mm	7,0%	1,0%	2,0%	5,0%	13,0%	34,0%	37,0%

Particle size [mm]

FIGURE 5: Distribution of the total mass (dm%) by particle size in each sieved output stream of the ballistic separator STT 6000 using landfilled waste.

some bricks broke during the sieving in the drum sieve, especially in the first run with a mesh size of 100 mm. This means that the actual throughput at particle size 100 mm is lower than indicated in Figure 6 and that the particle-size fraction 200-100 mm composes more than one third of the whole output stream “3D 200-90 mm”. Besides, more fines than those originally contained in the samples are generated due to abrasion processes while sieving the material.

In addition, there is a concentration of material in the smaller particle size fractions: 34% of “2D 200-90 mm” is < 40 mm and 30% is <10 mm. Regarding the output stream <90 mm this observation is more significant, with values of 90% and 59%, respectively.

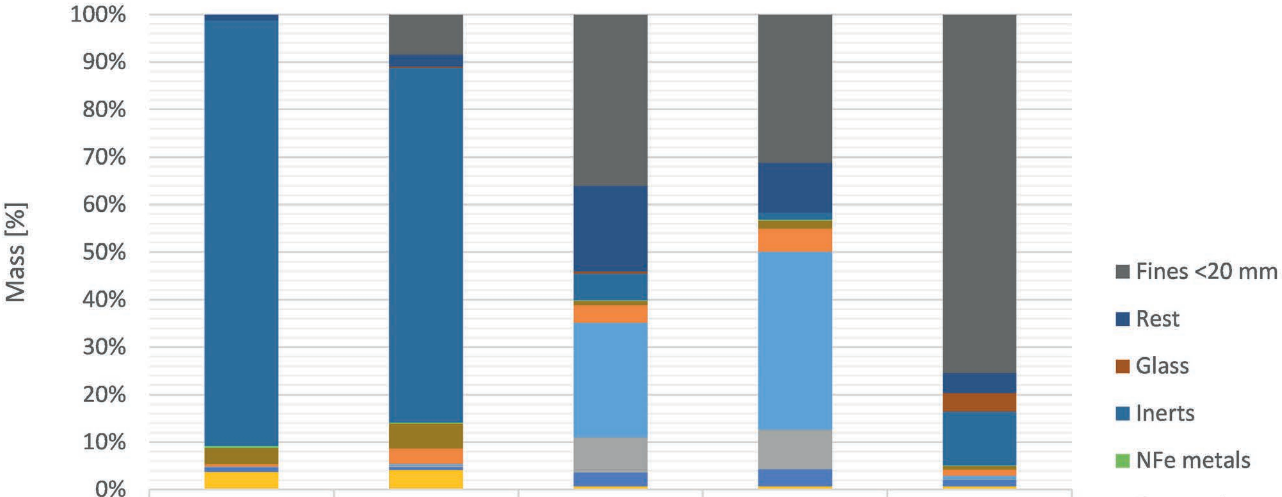
In line with the above-described findings, the moisture content of all samples varies between 9 and 41% and on the average makes up for almost a third of the RDF potential fractions. The PSD reveals a high percentage of fine material, not only in the output stream <90 mm but in all outputs due to the large amount of impurities found between the large particles, which are separated by the drying process.

3.2 Material composition

The excavated waste from MSG (batch 1) consisted of three main categories: inert, fines and plastics. In this study, all particles with a particle size <20 mm are defined as fines. Figure 6 gives an overview of the material distribution in all the output streams generated by the ballistic separator. As expected, most fines are found in the output stream <90 mm, additionally both, 2D >200 mm and 2D 200-90 mm, show considerable amounts of fines. Inert material is concentrated in the 3D output streams (3D >200 mm and 3D 200-90 mm). The ferrous metals (Fe metals) are mainly found in the 3D output streams: 3% in the >200 mm and 5% in the 200-90 mm output, as well as wood with 4% in the both 3D output streams. It must be noted that the mass percentages of "3D >200 mm" are given on a wet matter (wm) basis, while the other streams flows refer to dry matter (dm). Detailed figures (Fig. A.1-3) with the average values of the masses by particle size in each output can be found in the Appendix.

3.2.1 Output stream 3D 200-90 mm

The box whisker diagram in Figure 7 illustrates that the inert fraction is by far the largest with a median of 75%, while the other fractions all range below 10% with tendencies toward 0%; the shares of the categories paper, textile, 2D plastics, Non Ferrous (NFe) metals and glass are less than 1%. The same explanation as above may be considered for the high amount of inert material: the upper layer of the pit consisted of CDW. The second largest fraction is that of fines (8%), followed by Fe metals and wood, 5% and 4% respectively. Regarding Fe metals and wood, it was observed that half of the samples contained compounds of both materials in the fractions 60-40 and 40-20 mm. As the magnetic forces of a magnetic separator attract ferrous materials, wood parts containing nails were categorized as Fe metals. On the other hand, a large piece of wood (100-80 mm) with nails was identified and classified as wood. Another compound found was wood with 3D plastic.



	3D > 200 mm (wm)	3D 200-90 mm (dm)	2D < 275 mm (dm)	2D 200-90 mm (dm)	< 90 mm (dm)
Fines <20 mm	0,0%	8,4%	36,0%	31,2%	75,5%
Rest	1,2%	2,5%	18,0%	10,5%	4,2%
Glass	0,0%	0,2%	0,4%	0,0%	3,8%
Inerts	89,6%	74,7%	5,7%	1,5%	11,4%
NFe metals	0,5%	0,2%	0,1%	0,2%	0,2%
Fe metals	3,4%	5,2%	0,9%	1,7%	0,8%
3D Plastics	0,5%	3,3%	3,7%	4,8%	1,2%
2D Plastics	0,1%	0,3%	24,1%	37,5%	0,8%
Textile	0,0%	0,2%	7,4%	8,4%	0,1%
Paper	0,9%	0,7%	2,9%	3,5%	1,3%
Wood	3,8%	4,1%	0,7%	0,7%	0,7%

Output stream

FIGURE 6: Composition of all output streams by categories.

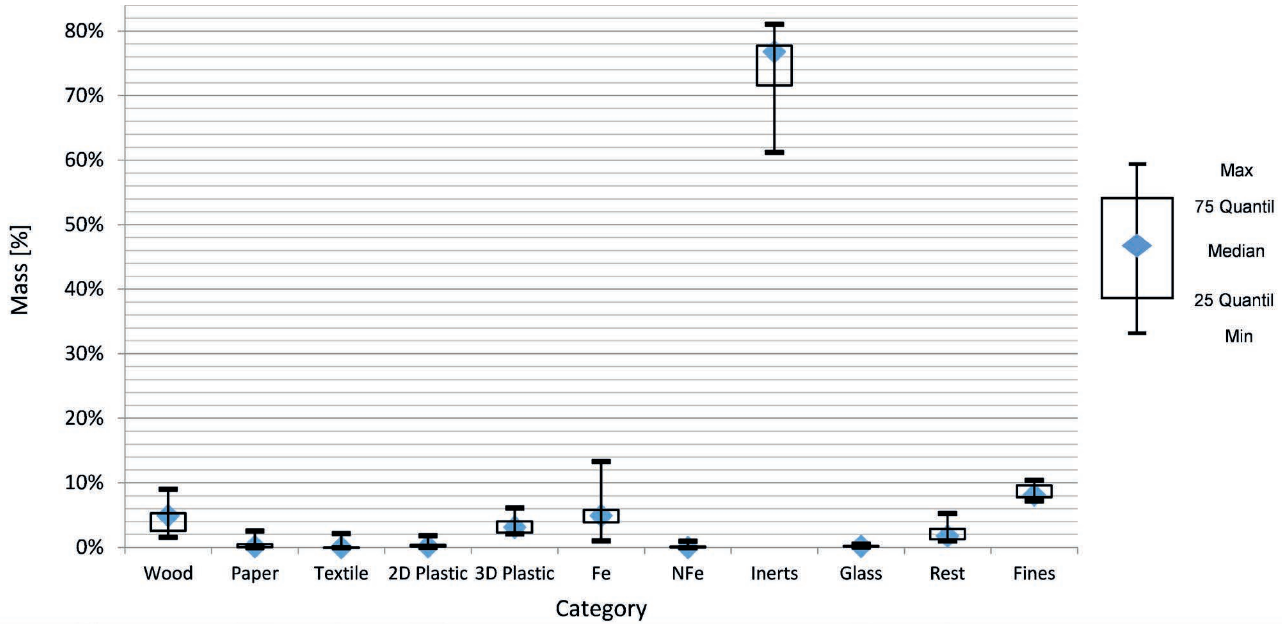


FIGURE 7: Output "3D 200-90 mm": fluctuations in the material composition, dm % (n=7).

The fluctuations indicate that the samples do not differ strongly from each other. The largest variations are observed in the categories wood, Fe metals and inert material, which also correspond to those obtained in previous LFM investigations.

Figure 8 represents the composition of each particle size fraction indicating its weight as it relates to the total weight of the output "3D 200-90 mm". The two larger coarse fractions make up most of the material, 37% and 34%, but breaking and abrasion in the drum sieve have affected the PSD distribution. Wood presented in the output "3D 200-90 mm" is distributed equally between the three larger particle size fractions, whereas most iron particles are counted in the 100-80 mm fraction (2%). The low quantity of glass (0.2%) might be explained by the fact that glass is likely to break and pass the sieve, both when previously transported and landfilled and during the excavation and mechanical stress in the ballistic separator, finally ending in the output stream <90 mm.

Looking to further processing steps, the enrichment of one material by sieving could result in difficulty due to similar distribution curves. Therefore, different sorting treatments such as magnetic separation, sensor-based separation (Beel, 2017; Martens & Goldmann, 2016), or float-sink separation (Bauer et al., 2018; Kranzinger et al., 2017) could be considered. Generally, the composition resembles the output "3D >200 mm" although it shows an increased number of fines generated in the PSD process (particle breakage and the drying process). Hence, the ballistic separator meets the expectations for enrichment of 3D materials.

3.2.2 Output stream 2D <275 mm

As shown in Figure 9, the output stream 2D <275 mm has a heterogeneous composition in comparison with 3D 200-90 mm, where the fine fraction (<20 mm) holds the largest share with 36% of the total. These fines are mainly composed of a mix of 2D plastics and soil-like material that may have been generated during the shredding process and due to the high detention time inside the shredder (Figure 10). To prevent losing this part of the combustibles in the fine fraction, sieving with a mesh size of 10 or 5 mm is suggested. Moreover, a considerable number of particles (particle size approximately <3 mm and smaller) were attracted by the magnet, being magnetic soil-like. Comparable findings are also described by Quaghebeur et al., 2013, where the amount of metallic iron in the magnetic fraction was between 8 and 9%.

The second-largest fraction is made up by 2D plastics with an average share of 24%, followed by Rest with 18%. Many particles categorized as Rest during the sorting process are compounds, such as carpets and nappies. They are mainly made of plastic, textile and cellulose, thus this category may be considered as a highly calorific fraction. Another compound that was mainly present in the output stream "2D <275 mm" consists of cables. If the 2D and 3D plastics, Rest, wood, paper and textile are considered highly calorific materials, 57% of the total mass has RDF potential. However, wood is insignificant with less than 1%. The same applies to glass and both Fe and NFe metals. Metals could not be separated completely due to attached pieces of plastic, mostly two-dimensional, textile or other materials.

3.2.3 Output stream 2D 200-90 mm

As it can be seen in Figure 11, the grain-size fractions >60 mm in the 2D 200-90 mm flow are especially rich in plastic foils (>45%). As expected, the main component of this flow are 2D plastics with an average content of 41% (Figure 12). In contrast, 32% were fines (<20 mm), which may reduce the RDF potential drastically. However, the categories Rest and wood could contribute to increase the

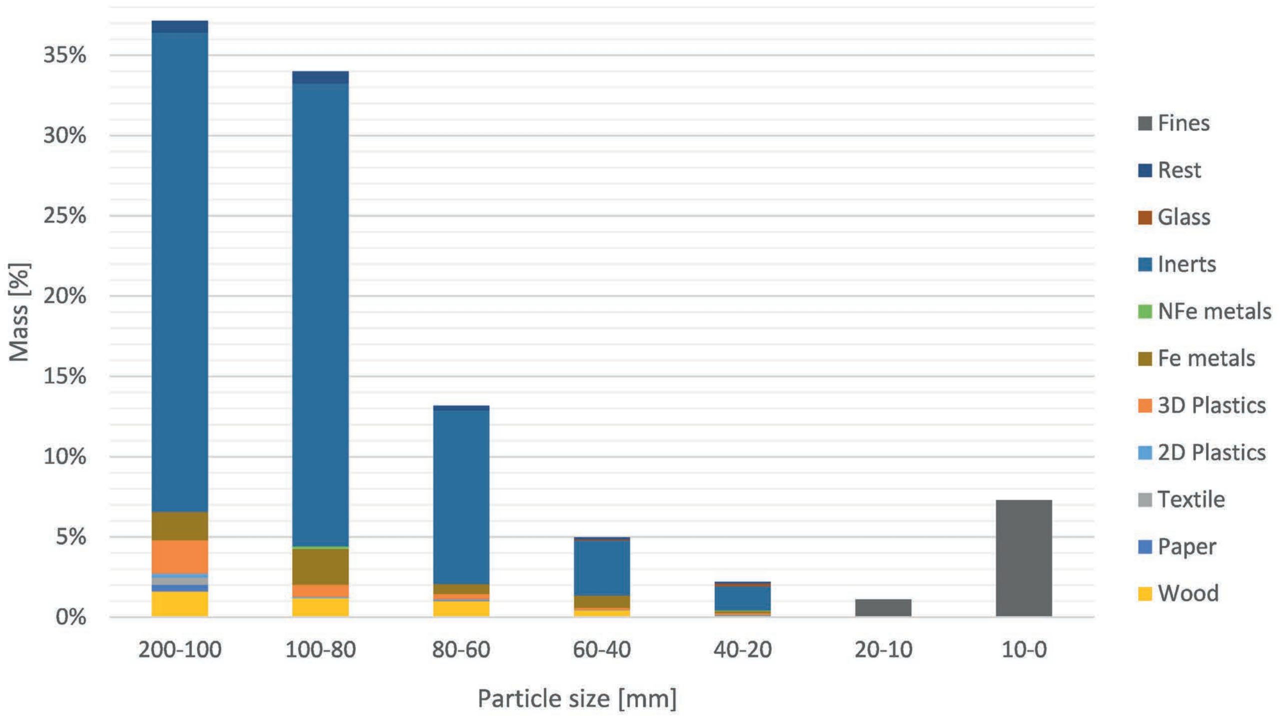


FIGURE 8: Material composition of the output stream "3D 200-90 mm" by particle size.

calorific value with a share of 7% and 6% respectively, since they are considered combustible materials.

Furthermore, it can be said that fraction 40-20 mm has a very heterogeneous composition consisting of all materials except glass, while the other sorted particle sizes are characterized by 2D plastics as the main component.

All inert material can be discharged by sieving with a mesh size of 80 mm, as shown in Figure 15. Apart from wood and fines, all other categories show similar distribution curves, so sieving alone will not be suitable for the enrichment of single categories as it is also the case for 3D 200-90 mm. Regarding the heterogeneous composition of both 2D streams compared to the 3D stream characterized by CDW, it can be deduced that the composition of this stream consists mainly of MSW with a high content of potential combustibles.

3.2.4 Output stream <90 mm

Most of the output stream (75%) consisted of fines <20 mm, Figures 13 and 14. These were not sorted manually, but as a general observation the particle sizes between 20 and 10 mm consisted mainly of glass and those <10 mm of soil-like material, comparable to the Fines of 2D <275 mm and 2D 200-90 mm. Within the categories (>20 mm), inert material makes up the largest part ranging between 5-23% of the total, with an average of 14% Also worth mentioning is the amount of glass in fraction 40-20 mm (4%). Together with the glass present in fraction 20-10 mm this confirms the previous assumption that glass is likely to break and pass to smaller grain-size fractions.

Looking at the PSD of the materials in Figure 13, a conclusion similar to the other outputs can be made: sieving with a mesh size of 40 mm may yield a material stream poor in glass but further enrichment does not seem realizable with the mesh sizes used in the analysis.

3.2.5 Overview of all output stream

Normally the 3D output streams of ballistic separators used to separate fresh waste contain stones, metals, glass, rubber and wood (H. Martens, 2016). In comparison, the results of the present paper show that the same applies for landfilled CDW and MSW with one exception: glass is only present in the finer fraction of the output stream <90 mm. Thus, considering the enrichment of highly calorific materials in the 2D streams, mostly 2D plastics, the ballistic separator seems to meet the expectations.

Nevertheless, a mass balance of the whole process must be considered to estimate a reliable potential for the whole landfill. This means that the composition of every output stream must be related to its share in the total flow when extrapolating the absolute mass of all materials stored in the landfill. Figure 15 presents the mass balance considering wet and dried material. However, the total water content must be higher than the indicated 25% because the moisture content of the output "3D >200 mm" was not analysed in this study and the water is evaporated during separation. Nevertheless, it points out the high share of particles in the output stream <90 mm, 58% of the input material, compared to 4% (dry) with RDF potential from which the fines separated by sieving must still be subtracted. Two main reasons are considered for the predominant share of fines: the longer the waste is stored, the more organic material can be degraded; thus, the amount of fines increases with time (Maul et al., 2016). Besides, the cover layer is a source of fines because usually soil is used as a daily cover (Kaartinen et al., 2013).

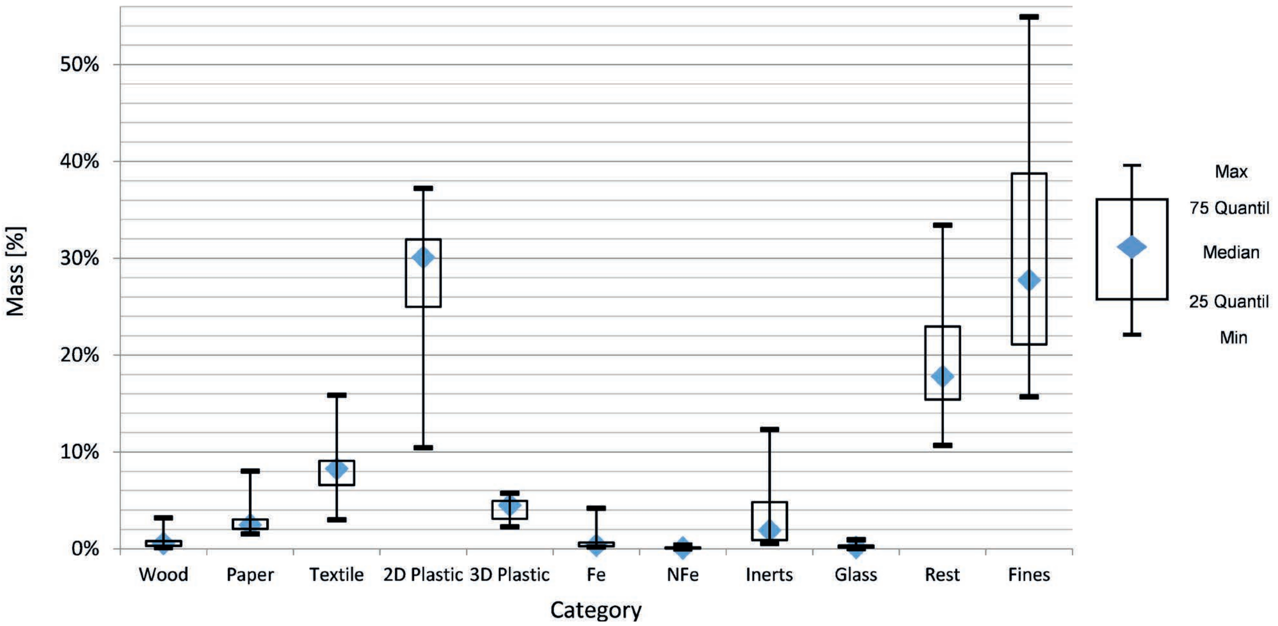


FIGURE 9: Output “2D <275 mm: fluctuations in the material composition, dm % (n=8).

FIGURE 10: Fines (<20 mm) from the output stream “2D <275 mm”.

3.3 Calorific value and ash content

In order to assess the quality of the potential RDF produced by the ballistic separator, Table 2 shows the distribution of the calorific values, the ash content and the mass percentages by material categories in both 2D output stream, 2D <275 mm and 2D 200-90 mm.

As expected, 2D plastic, with a considerable share of the output streams, is the material with the highest calorific value, 35.1 and 40.9 MJ $kg^{-1,}$ followed by the 3D plastics with an average of 32.4 and 30.6 MJ kg 1. In the case of the Rest, the values are 18.5 and 23.3 MJ kg-1, which reinforce the previous assumption that this category can be regarded as a highly calorific fraction. Compared to the mean net calorific value of 21.9 MJ kg-1 (dry and ash free) for RDF (Phyllis database, 2018), the results of this study range within the same magnitude at 16.0 and 22.4 MJ kg^{-1}. Nevertheless, the influence of the fines is significant since they make up a third of the mass. This means that by separating fines from the stream before combustion, the amount of energy produced could be increased and the ash content could be reduced. In addition, only materials with ash contents <60% can burn autonomously (Seifert & Vehlow, 2017). In this regard, the 2D output streams have an ash content average of 50.5 and 40.4, thereby meet-

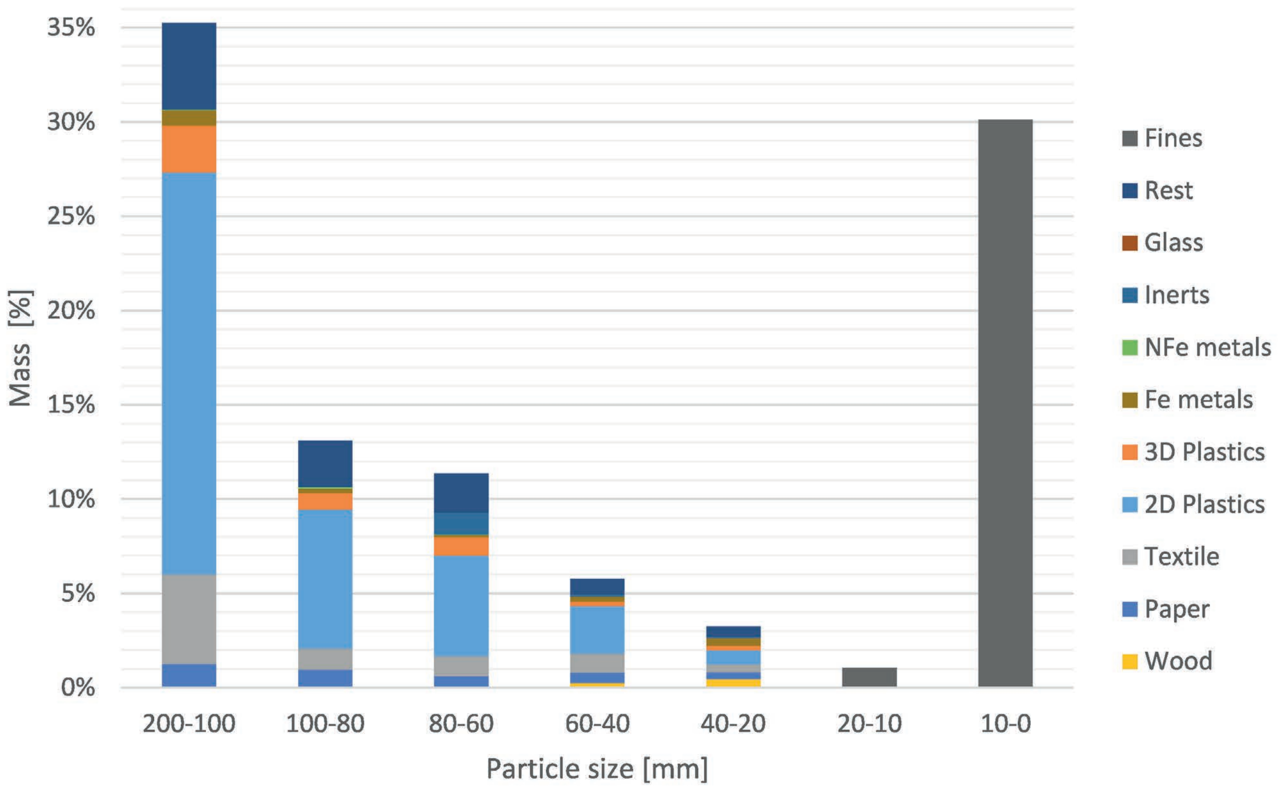


FIGURE 11: Material composition of the output stream ”2D 200-90 mm” by particle size.

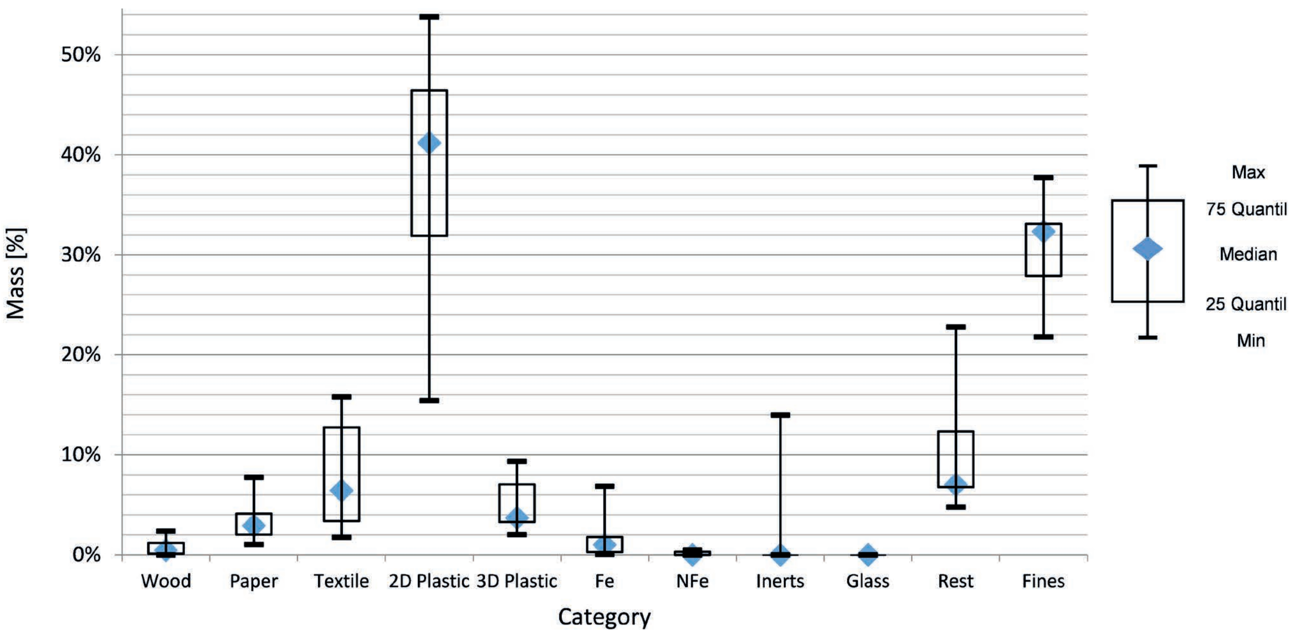


FIGURE 12: Output “2D 200-90 mm”: fluctuations in the material composition, dm % (n=9).

ing this requirement. Moreover, low ash contents are also desirable to reduce the dust loading. It can be deduced from the low amount of materials that are not suitable for RDF (metals, inert and glass) and thus degrade the burning parameters, that the ballistic separator enriches RDF potential materials in the 2D stream as expected since the downgrading fines come from the surface of the RDF particles and were generated in the subsequent drying process.

3.4 Comparison with previous LFM investigations

The total range of moisture content was between 9 and 41%, comparable with data in the literature of 18 to 40% for 17 to 40 year old waste (Hernández Parrodi, 2018a),

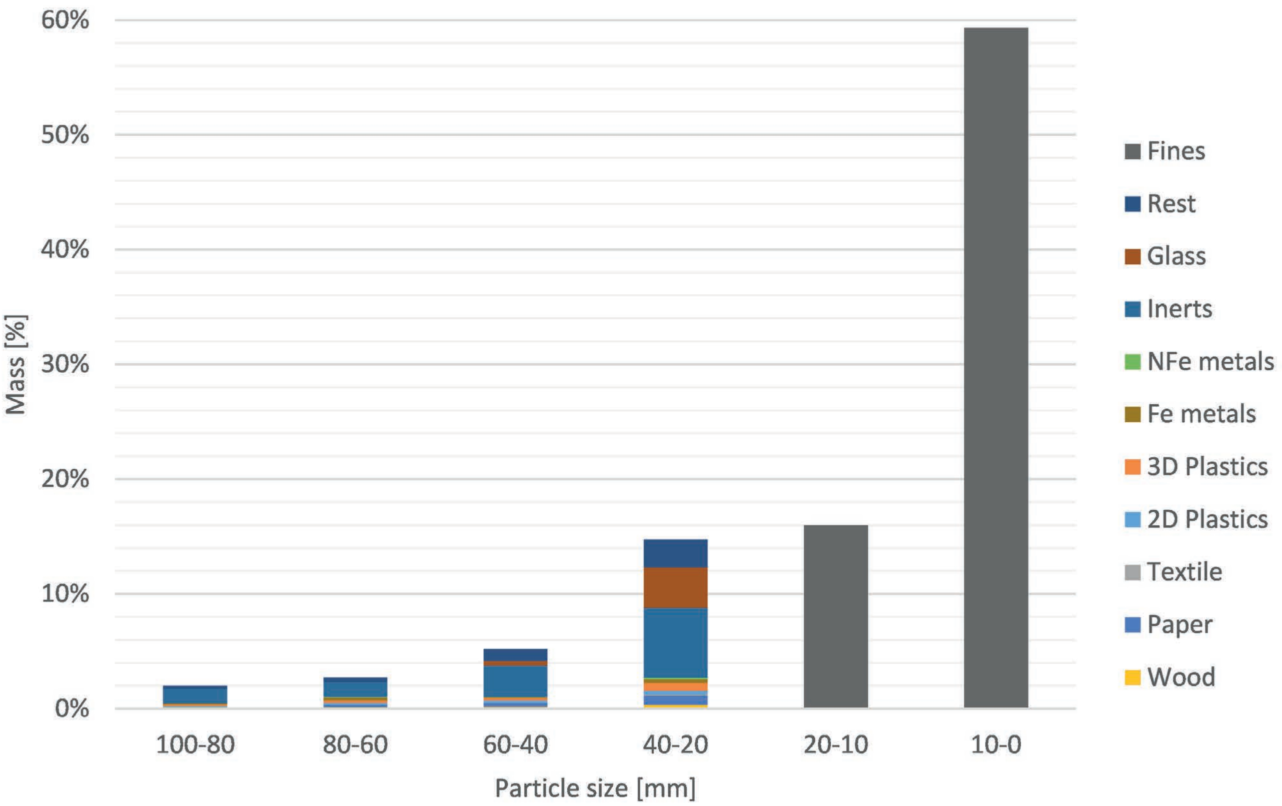


FIGURE 13: Material composition of the output stream ”<90 mm” by particle size.

even if data from “3D >200 mm” is missing for a complete comparison. Apart from the age, other important parameters can influence the water content of the material, e.g., geographical location, landfill layering (water permeability), particle size and the type of waste landfilled.

The characteristics of the mined material depend strongly on the type of waste that was initially landfilled. This becomes evident when looking at the material composition indicated on a wet and mass basis from previous studies (Bhatnagar et al., 2017; Jani et al., 2016; Kaartinen et al., 2013; Quaghebeur et al., 2013; Van Vossen & Prent, 2011; Wolfsberger et al., 2015b). In Table 3, the inert fraction (17% in this study) makes up more of the total when CDW was deposited. Municipal landfills, on the other hand, are characterized by 10% of inorganic substance (concrete, stones, and glass), 20-30% highly calorific fraction and 27-54% soil-like materials. Thus, the share of plastics in MSW is considerably higher than the 3% found for mixed waste in this paper. The same holds for household waste materials, such as paper and textiles: the percentages counted are below the values reported in the literature (0.4% and 2-7%, respectively). With a share of approximately 46%, the fines (<20 mm) are in accordance with MSW but different particle sizes are used to define the fines or soil-like materials. The metal concentrations in most LFM projects range below 5%, nevertheless the 1% found seems low because more metals are expected from CDW, especially structural steel.

Similar to the difficulty encountered when comparing PSD, materials are classified into different categories by different researchers, especially overlapping categories (e.g., for plastics and metals) making comparison more challenging. Moreover, the efficiency of sorting, and therefore the results of the composition, depends on the applied technique: manual, mechanical or sensor-based sorting.

Findings on the GCV and ash content are compared to results presented by Quaghebeur et. al., 2013 for the REMO landfill site in Belgium. It stands out that the calorific value of the main combustible fraction (2D plastics, 35.1 and 40.9 MJ kg$^{-1)}$ is considerably higher than for plastics analysed by Quaghebeur et. al., 2013 (19.0-28.0 MJ kg^{-1}) with an ash content of 20-35% compared to 21,7% and 12,8% (12,8% and 11,8% for 3D plastics), depending on the 2D output stream in this study. Similar differences are noted for the ash content of paper and cardboard, which amounts to 25-61% at the REMO site but is only estimated to be 12% in this study. The CV of paper and cardboard is more conformable in both studies: 6.7-12.0 MJ kg^{-1} and 15 MJ kg^{-1}, respectively. The results for fines are in accordance with the compared data in which a CV of 1.3-4.8 MJ kg-1and ash contents of 64.4-87.5% are reported.

Summarizing the above comparison, it can be said that the mixture of CDW in the examined material with MSW influences the RDF potential negatively because MSW landfills are found to consist of more combustible materials. The high amount of fines described in most LFM projects is similar, whereas the CV and ash content are restricted in their resemblance. Standardized or widely agreed-on

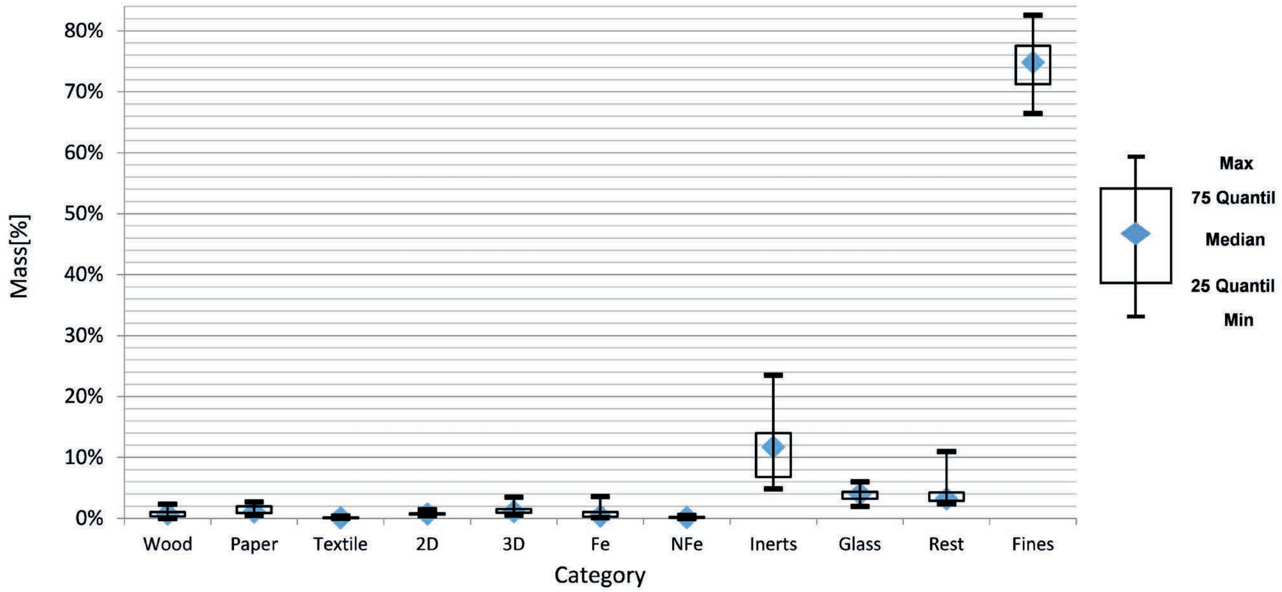


FIGURE 14: Output "<90 mm": fluctuations in the material composition, dm % (n=12).

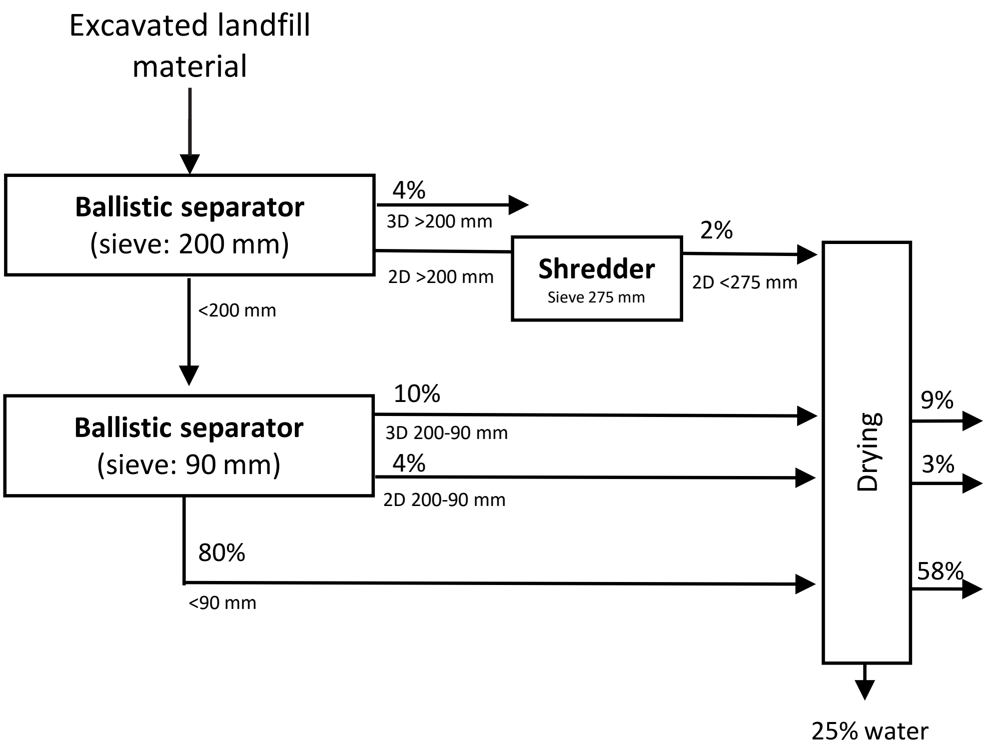


FIGURE 15: Mass balance of the ballistic separator with the two sieving steps of 200 mm and 90 mm (all indications refer to the input 100%).

material categories and particle size fractions to increase comparability are desirable.

3.5 Recovery options

Generally, two different recovery processes can be distinguished: WtM, which creates new materials according to the properties of the recovered materials and WtE, which can substitute for the use of fossil fuels. For fresh plastic waste both recovery options are practised. WtM (for plastics) is subdivided into primary recycling without modifications of the polymers, secondary recycling with downgrading and tertiary recycling with depolymerization reactions. Primary and secondary recycling requires high-quality incoming waste. Moreover, the possibility of meeting the limits on heavy metal content must be questioned, taking into account the diverse additives that were used in the past for plastic production. Tertiary recycling seems suitable for landfill-mined plastics but only a few industrial

TABLE 2: Average composition, GCV and ash content of the 2D <275 mm and 2D 200-90 mm output streams (n=8, n=9, respectively). Data from the REMO landfill (Quaghebeur et al., 2013) were used to compare the results.

Categories	2D < 275 mm			2D 200-90 mm			REMO, Belgium (2013)	
	Mass	GCV	Ash content	Mass	GCV	Ash content	GCV	Ash content
	[% of total]	[MJ/kg]	[%]	[% of total]	[MJ/kg]	[%]	[MJ/kg]	[%]
Wood	0,7	15,2	10,3	0,7	16,9	10,9	-	-
Paper	2,9	16,0	22,0	3,5	15,2	12	7.3 - 13.0	25.0 - 61.0
Textile	7,4	22,5	32,0	8,4	22,7	19,6	-	-
2D Plastics	24,1	35,1	21,7	37,5	40,9	13,5	19.0 - 28.0	20.0 - 35.0
3D Plastics	3,7	32,4	12,8	4,8	30,6	11,8		
Fe metals	0,9	-	100,0	1,7	-	100	-	-
NFe metals	0,1	-	100,0	0,2	-	-	-	-
Inert	5,7	0,1	98,4	1,5	-	99,6	-	-
Glass	0,4	-	100,0	0,0	-	-	-	-
Rest	18,0	18,5	34,7	10,5	23,3	16,8	-	-
Fines	36,1	2,0	79,0	31,2	1,9	88	2.1 - 5.7	64.4 - 87.5
Total	100,0	16,0	50,5	100,00	22,4	40,4	-	-

plants applying this process exist in Europe (Quaghebeur et al., 2013). Likewise, the heterogeneity and high level of contamination of paper, textiles and wood would require extensive and expensive treatment for WtM valorisation of those materials. As mentioned above, the Rest of the "2D 200-90 and 2D <275 mm" also appears suitable for WtE. Hence, for most excavated materials WtE valorisation as "Solid recovered fuels (SRF) is the most realistic marketable material" (Bhatnagar et al., 2017). RDF differs from SRF in that the latter guarantees a certain quality of the fuel, since SRF must be produced from non-hazardous waste and fulfil certain fuel qualities (Rotter et al., 2011). Therefore, more parameters, apart from the characteristics presented in this study, are needed to evaluate the efficiency and quality of the RDF generated from landfilled materials and the emissions of the combustion process. Those parameters are the amount of organic carbon, total carbon, hydrogen, nitrogen, sulfur, chlorine, fluorine, bromine and heavy metals (Quaghebeur et al., 2013). It is obligatory to indicate the concentration of chlorine according to the specification DIN EN 15359 due to its corrosive impact and the property of mobilizing some of the metals into the flue gas (Kaartinen et al., 2013). Heavy metals are of special concern for SRF made from pretreated waste (Wolfsberger et al., 2015a). Therefore, information should be gathered about the history and origin of the landfilled waste. The requirements that the above parameters must meet depend on the combustion system used and on applicable laws and authorization procedures (Kuchta et al., 2017).

If metals are not separated during the SRF/RDF production process and are directly valorised as WtM, they can be recovered from the ash after incineration (Seifert & Vehlow, 2017) .As glass is an inert material, WtM is considered for its valorisation providing an efficient separation (Quaghebeur et al., 2013).

Of major importance is the recoverability of the fine fractions, as they make up around half of the total material excavated and have been found to be challenging in previous investigations (Hernández Parrodi et al., 2018; Jones et al., 2013). A WtE application of the fine fractions, as described by Quaghebeur et. al., 2013, is not considered to be applicable for the present material because of the age of the waste, the resulting high degree of degradation, the low CV and the high ash content. Thus, WtM options are to be considered.

One possibility is the use of the fine fractions as a cover layer in operating landfills. At the Kudjape landfill, Estonia, the fine fraction was used as a methane degradation layer. This valorisation requires a low degree of contamination (Bhatnagar et al., 2017). The use of fines and conditioned inert materials as filler and construction material generally can be considered if they comply with the limit values for such activities (Quaghebeur et al., 2013).

Heavy metals are especially expected to be found in old landfills and can contribute to decrease the costs of ELFM if recovered (Garcia Lopez et al., 2018). From these findings, the recovery of the major and trace metals from the fine fractions could be an option to meet the globally increasing demand. Conducting leaching tests and XRF analysis can give a more precise composition determination and allow an estimation of the marketable potential. Apart from the potential economic benefit of heavy metals recovery from the fine fractions, another positive effect can be the resulting reduction of their uncontrolled leaching out of the landfill (Hernández Parrodi et al., 2018; Bhatnagar et al., 2017; Quaghebeur et al., 2013; Kaartinen et al., 2013).

A third option would be to re-landfill the fine fractions, but this procedure bears costs instead of revenues and

TABLE 3: Comparison of the material composition of this study with previous LFM investigations, adapted from Hernández Parrodi, 2018a.

Type of information	This study, 2018 (MSG, Belgium)	Various countries (Van Vossen and Prent, 2011)	Högbytorp, Sweden (Jani et al., 2016)	Kuopio, Finland (Kaartinen et al., 2013)	Kudjape, Estonia (Bhatnagar et al., 2017)	Lower Austria, Austria (Wolfsberger et al., 2015)	Houthalen, Belgium (Quaghebeur et al., 2013)
Type of waste	MSW + C&D	Various	MSW + C&D	MSW	MSW	MSW	MSW
Age of waste [y]	40 - 50	Various	5	5 - 10	10	13 -20	14 - 29
Average moisture content	25%	-	-	-	-	29 - 55%	48 - 66%
Fines/ Soil-like material	46%	55%	27%	50-54%	29%	47%	44% (12)
Stones	-	3%	28%	-	18%	-	-
Inert/minerals	17%	6%	-	-	-	6%	10% (6)
C&D	-	9%	-	-	-	-	-
Limestone	-	-	5%	-	-	-	-
Asphalt	-	-	3%	-	-	-	-
Glass/ceramics	2%	1%	6%	-	5%	1%	1.3% (0.8)
Plastics (3D/2D)	3%	5%	-	23%	22%	18%	17% (10)
Soft plastics	-	-	1%	-	-	-	-
2D plastics	2%	-	-	-	-	-	-
3D plastics	1%	-	-	-	-	-	-
Other plastic/ Composites	-	-	7%	-	-	4%	-
Organic/kitchen waste	-	5%	-	-	-	-	-
Paper/cardboard/ PPC	1%	5%	-	4 - 8%	5%	3%	7.5% (6)
Paper	<1%	-	4%	-	-	-	-
Wood	1%	4%	15%	6 - 7%	5%	-	6.7% (5)
Textile	<1%	2%	3%	7%	-	6%	6.8% (6)
Leather	-	2%	-	-	-	-	-
Rubber	-	-	0%	-	-	-	-
Wood/leather/rubber	-	-	-	-	-	9%	-
Total metals	1%	2%	-	3 - 4%	3%	5%	2.8% (1)
Fe metals	1%	-	0%	-	-	-	-
NFe metals	<1%	-	0%	-	-	-	-
Other/ Rest	3%	3%	-	2%	13%	1%	-
Non-MSW	-	0%	-	-	-	-	-

is not in line with the introduced goals of ELFM projects (Kaartinen et al., 2013).

Independently of the recovery option chosen for the fine fractions, their separation from other excavated materials by sieving can be essential for further processing, since it raises the purity and CV of the RDF stream and improves the possible efficiency of sensor-based sorting techniques (Hernández Parrodi et al., 2018; Maul et al., 2016). Moreover, the bulk density of the material is a very relevant parameter for the design of the mechanical treatment. Although no quantitative statement was done in this analysis, it can be said that landfilled material presents higher bulk densities than fresh MSW, since the landfill-mined material showed more fine material than those typical of MSW.

4. CONCLUSIONS

This study found that the 3D-output streams (>200 mm and 200-90 mm) of the ballistic separator consisted mainly of coarse CDW, whereas more heterogeneous MSW was yielded in the 2D-output streams (>200 mm and 200-90 mm). By comparing 57% of the combustible materials (plastics, paper, textiles, leather and wood) in the output "2D <275 mm" and 65% in "2D 200-90 mm" to 75% of inert material in "3D 200-90 mm", the efficiency of the separating process was suitable for the landfilled material. The 2D-output streams were characterized by a higher moisture content (average, 30%) than the 3D streams (12%), which caused elevated amounts of fines (<20 mm) to adhere to larger particles.

The laboratory analysis showed an average CV of 22

MJ kg^{-1} and an ash content of 40% for the "2D 200-90 mm". To fully determine the quality of the produced RDF, further tests that characterize the generated flue gas and remaining ash should be conducted.

A considerable share of the total excavated waste was made up of the output stream <90 mm, which was the greatest output stream of the ballistic separator with a share of 80% (wm). Its amount is expected to increase if the material is dried beforehand. As a further processing step, separating the fines by sieving is advisable to remove inert materials and enrich the WtE-stream with high CV materials, since fines contain a significant amount of inert materials, which are not suitable for RDF production. WtM valorisation of the recovered materials, such as inert materials as construction sand, soil-like materials as cover material for operational landfills or metals for recycling, could be considered for the fine fractions. If the separation of metals, glass and stones is possible, the amount of material that has to be re-landfilled would be decreased significantly.

The results of this study cannot be directly transferred to other landfills because compositions depend on different factors such as the origin of the waste, time of storage and physical conditions of the site. In this study, the ballistic separator shows an enhanced mechanical processing due to the share of CDW, which mostly consists of heavy 3D-materials. Further investigation is required in order to state that different types of waste (industrial waste, MSW and CDW, independently) have the same rate of enrichment of high calorific materials as in this study.

Although this study assesses the technical aspects by characterizing the different output streams generated by the ballistic separator, cost efficiency also needs to be taken into account when considering the feasibility of a full-scale ELFM project. Additionally, a remaining challenge is the assessment of costs and revenues of a recovery process by estimating the total amount of deposited waste and relating it to the market prices of the recoverable materials.

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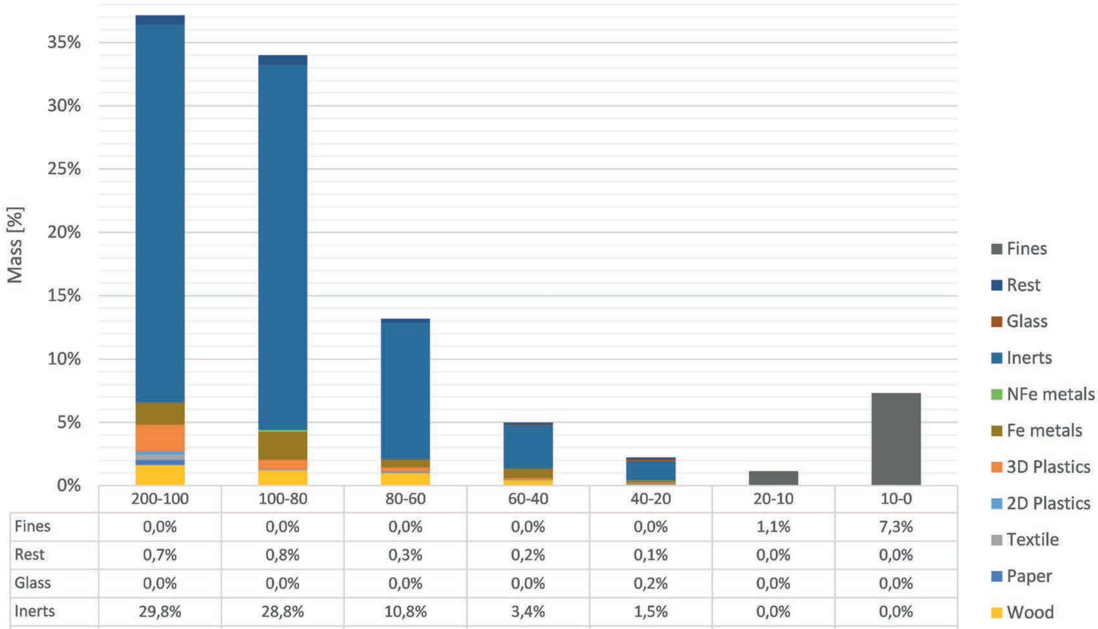
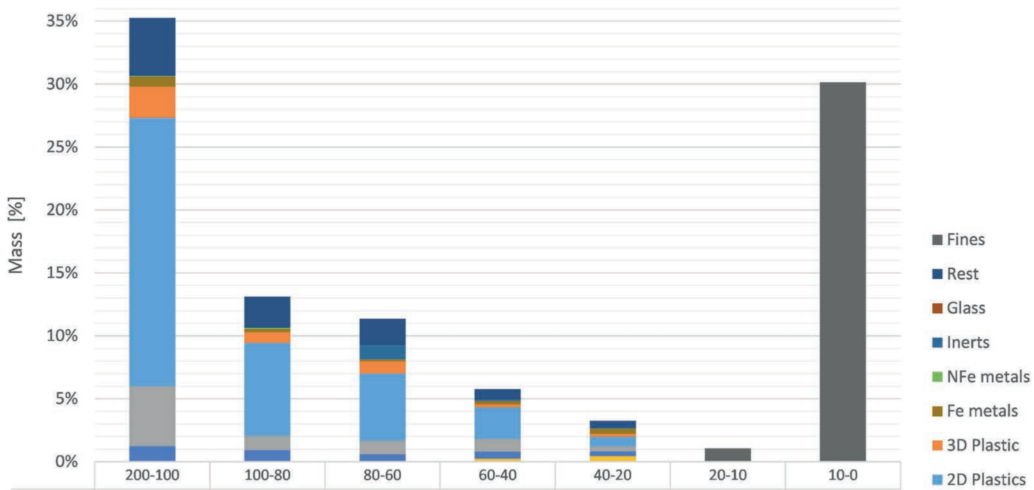
APPENDIX A



	200-100	100-80	80-60	60-40	40-20	20-10	10-0
Fines	0,0%	0,0%	0,0%	0,0%	0,0%	1,1%	7,3%
Rest	0,7%	0,8%	0,3%	0,2%	0,1%	0,0%	0,0%
Glass	0,0%	0,0%	0,0%	0,0%	0,2%	0,0%	0,0%
Inerts	29,8%	28,8%	10,8%	3,4%	1,5%	0,0%	0,0%
NFe metals	0,0%	0,1%	0,0%	0,0%	0,0%	0,0%	0,0%
Fe metals	1,8%	2,2%	0,6%	0,8%	0,1%	0,0%	0,0%
3D Plastics	2,1%	0,7%	0,3%	0,1%	0,1%	0,0%	0,0%
2D Plastics	0,3%	0,1%	0,1%	0,0%	0,0%	0,0%	0,0%
Textile	0,4%	0,0%	0,0%	0,0%	0,0%	0,0%	0,0%
Paper	0,4%	0,0%	0,0%	0,0%	0,1%	0,0%	0,0%
Wood	1,6%	1,2%	1,0%	0,4%	0,1%	0,0%	0,0%

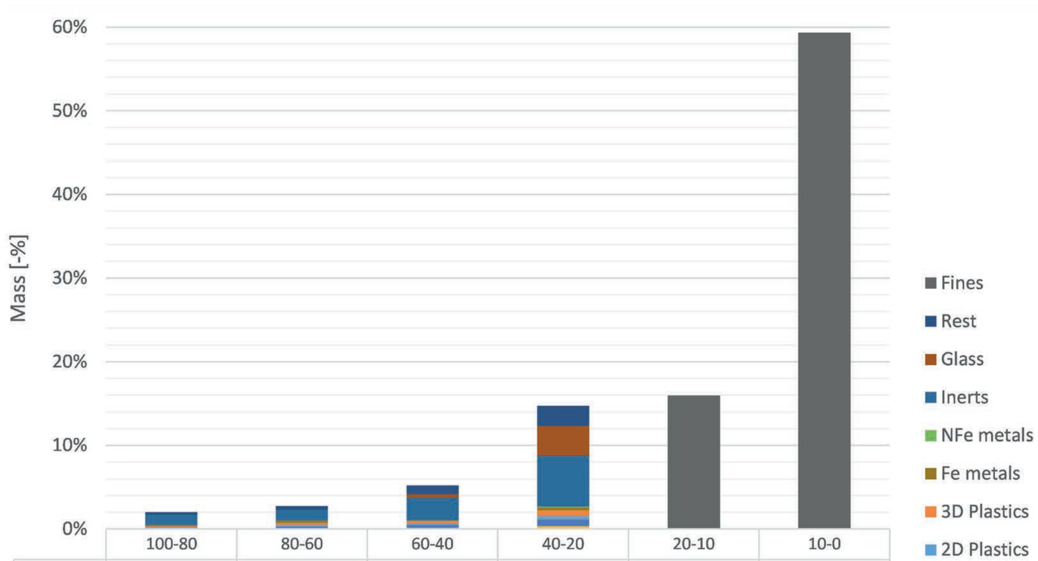
FIGURE A1: Total mass composition, dm %, by particle size of the output “3D 200-90 mm”.



	200-100	100-80	80-60	60-40	40-20	20-10	10-0
Fines	0,0%	0,0%	0,0%	0,0%	0,0%	1,1%	30,1%
Rest	4,6%	2,5%	2,1%	0,9%	0,6%	0,0%	0,0%
Glass	0,0%	0,0%	0,0%	0,0%	0,0%	0,0%	0,0%
Inerts	0,0%	0,0%	1,2%	0,1%	0,1%	0,0%	0,0%
NFe metals	0,1%	0,1%	0,0%	0,0%	0,0%	0,0%	0,0%
Fe metals	0,8%	0,2%	0,1%	0,2%	0,4%	0,0%	0,0%
3D Plastic	2,5%	0,9%	1,0%	0,2%	0,2%	0,0%	0,0%
2D Plastics	21,3%	7,4%	5,3%	2,5%	0,8%	0,0%	0,0%
Textile	4,7%	1,1%	1,1%	1,0%	0,4%	0,0%	0,0%
Paper	1,2%	0,9%	0,6%	0,6%	0,4%	0,0%	0,0%
Wood	0,1%	0,0%	0,0%	0,2%	0,5%	0,0%	0,0%

Particle size [mm]

FIGURE A2: Total mass composition, dm %, by particle size of the output “2D 200-90 mr



	100-80	80-60	60-40	40-20	20-10	10-0
Fines	0,0%	0,0%	0,0%	0,0%	16,0%	59,3%
Rest	0,3%	0,5%	1,0%	2,4%	0,0%	0,0%
Glass	0,0%	0,0%	0,4%	3,5%	0,0%	0,0%
Inerts	1,3%	1,3%	2,7%	6,1%	0,0%	0,0%
NFe metals	0,0%	0,1%	0,0%	0,1%	0,0%	0,0%
Fe metals	0,1%	0,3%	0,1%	0,3%	0,0%	0,0%
3D Plastics	0,1%	0,2%	0,2%	0,7%	0,0%	0,0%
2D Plastics	0,1%	0,2%	0,2%	0,3%	0,0%	0,0%
Textile	0,0%	0,0%	0,0%	0,1%	0,0%	0,0%
Paper	0,0%	0,2%	0,4%	0,8%	0,0%	0,0%
Wood	0,1%	0,1%	0,1%	0,3%	0,0%	0,0%

Particle size [mm]

FIGURE A3: Total mass composition, dm %, by particle size of the output “<90 mm”.

APPENDIX B

TABLE B1: Technical specifications of the ballistic separator.

Manufacturer	**STADLER®**
Modell	STT 6000
Dimensions (L x W x H) (mm)	6030x2970x6040
Weight (kg)	25000
Engine output (kW)	30
Adjustable Angle	15°; 17.5°; 20°
Surface (m^2)	14.3
Maximum output (m^3/h)	200
N° of paddles	5
N° of fans	No fans available

TABLE B2: Technical specifications of the sieves.

	Drum sieve	**Box sieve**
Manufacturer	Self-made by the Department of Processing and Recycling (RWTH Aachen University)	Siebtechnik
Type of movement	Rotating	Circular Vibratory Screen
Input power (kW)	2.2	0.75
Diameter of drum screen (mm)	1500	-
Installation	Polygonal screen with 8 screen linings	-
Dimensions of screen linings (mm)	950x560	500x500
Revolution	Adjustable	1400 RPM
Mesh sizes (mm)	10; **20; 40; 60; 80; 100**; 120; 140; 160; 200; 240; 300	1; 2; 4; 6.3; **10; 20**; 31.5; **40**; 50; **60**; **80**; 100
**The mesh sizes used during this study are marked in bold.*		

TABLE B3: Technical specifications of the sieves.

Type of cutting mill	**Large cutting mill**	**Small cutting mill**	**Disc mill**
Manufacturer	Reto	Dreher	Retsch
Input power (kW)	37	2.2	1.75
Rotor peripheral speed (m/s)	9	10	10
Rotor cutting diameter (mm)	350	160	
Rotor length (mm)	450	200	
Mesh sizes (mm)	4; **5**; 6; 8; **10**; 12; 15; 20; 30; 40; 50; 60; 70; 80	**1**; **2**; 3; 4; 5; 6; 8	**0.5**; 0.75; **1**; 1.5; **2**; 4; 6; 8; 10
**The mesh sizes used during this study are marked in bold.*			

TABLE B4: Technical specifications of the hammer mills.

	Large hammer mill	**Small large mill**
Manufacturer	Hazemag	Condux
Type of crusher	High speed	High speed
Main types of loads	-	Impact, Shearing
Weight (kg)	-	30
Input power (kW)	18	3
Revolution	adjustable up to approx.19 m/s	2850 RPM
Mesh sizes (mm)		
Feed grain size (mm, max.)	*approx. 360 x 280*	*100 x 60*
Rotor width (mm)	-	*150*

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POTENTIAL OF SENSOR-BASED SORTING IN ENHANCED LANDFILL MINING

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ABSTRACT

In landfill mining, simple technologies and processing chains are frequently applied to excavated material in order to extract recyclable metals and high-calorific fractions used in energy recovery. Sensor-based sorting is one way to extract more and better material from a landfill. Two testing series have been performed using state-of-the-art technology to assess the technical feasibility of classifying and sorting landfill material with the aid of near-infrared spectroscopy. Fractions were classified as inert and combustible and sorted by particle sizes ranging from 90-30 mm, from 30-10 mm and from 10-4.5 mm for water content levels of 0 wt% and of 15 wt%, respectively. Additional tests were applied different landfill mining materials. Polypropylene (PP), polyethylene (PE) and polyvinyl chloride (PVC) products were produced, using sensor-based sorting, from a mixed fraction of particle sizes ranging from 60-200 mm. Both test series applied air-classified heavy fractions gained from two distinct processing schemes of landfill mining projects in Belgium and in Austria. Results show that the separation and classification of inerts and combustibles is feasible, enriching inert fractions with purities of 97.7 wt% to 99.6 wt% derived from inputs whose inert contents achieved 85.6 to 98.8 wt%. Efficient sorting is a function of the level of pre-processing, water content, relative amounts of adhesive fines, input composition and particle size ranges of the input material. Results from the second test series show that PP, PE, PVC and other materials can be successfully distinguished, achieving correct classification and ejection into respective product fractions of 91.8-99.7 wt%.

1. INTRODUCTION

In the past, landfills were considered cost-effective and final means of waste disposal (Krook et al., 2012). Nowadays such landfills pose both a problem and a chance. Spatial constraints, landfill-based hazards like leachate and methane emissions (Danthurebandara, M. et al., 2015) and shortage of landfill volume (Wörrle, J., 2018) can be arguments in favour of landfill mining (LFM) activities (Quaghebeur et al., 2013, Mor et al., 2006, Sormunen et al., 2008).

LFM is usually expensive and not economically feasible. Economic feasibility could be achieved, however, by using mechanical processing to recover marketable valuable materials. Since the 1950s, LFM projects have mostly applied simplified process chains using a screening stage and optional subsequent air classification and magnetic separation, among other processes, to render mechanical processing as cost-effective as possible (Krook, J. et al., 2012). As a consequence, only limited amounts of landfill resources (metals and refuse-derived fuel (RDF) could be recovered in the past (Krook et al., 2012). This limitation may be due to the fact that the reduction of environmental impacts and landfill remediation commonly has been given preference to the recovery of land or of landfill volume (Danthurebandara et al., 2015).

By contrast, the design of enhanced landfill mining (ELFM) is targeting the extraction of valuable materials for recycling (waste-to-material, WtM) and for energy production (waste-to-energy, WtE). The TönsLM project, for instance, has developed and examined scenarios based on rather complex process chains and innovative technologies. In addition to comminution, ballistic separators, screening and magnetic separation, also eddy current separators and near-infrared (NIR) sorters were considered in processing (Breitenstein et al. 2016), enabling higher and

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purer yields of recyclables. This resulted in better market prices, facilitating economically improved mining.

Studies have shown that ELFM is mainly inspired by the recovery of landfill volume and by the extraction of metals and of high-quality combustible fractions (Danthurebandara et al., 2015, Jones et al., 2013, Kieckhäfer et al. 2017). Most published ELFM projects so far are still at the planning stage, however. One main caveat of any practical implementation of (E)LFM projects is the risk that expected costs may exceed the achievable profit (Kieckhäfer et al. 2017). If framework conditions for ELFM should change, the mining of landfills will become more attractive to landfill operators (Kieckhäfer et al., 2017). This paper intends to shed more light on the potential recovery of recyclable and energetically valuable materials.

Utilization of the high-calorific fraction derived from (E)LFM has been practically tested in the recent past (Rotheut and Quicker, 2017; Wolfsberger et al., 2015). The relative amount of heavy metals contained in a high-calorific fraction limits its use in Austrian co-incineration plants (Wolfsberger et al., 2015). Since the distribution of heavy metals in the individual groups of substances and particle size ranges may vary significantly, suitable pre-treatment (separation of fractions contaminated with heavy metals) can prevent exceeding the limit values (Wolfsberger, T. et al, 2015, García López, C. et al., 2018).

Studies of Rotheut, M. and Quicker, P. (2017) concerning the energetic utilisation of RDF from LFM have shown that also the RDF's properties may vary a lot, affecting in particular calorific value, water and ash contents. Material from the 'Pohlsche Heide' landfill was excavated, for example, and processed in a state-of-the-art mechanical-biological waste treatment plant. Once the metals had been removed, the light fraction recovered using air classification was thermally converted into RDF. Calorific values between 9.2 and 23.9 MJ/kg, ash contents of up to 49.6% and water contents of 9.1% to 30% were observed. These findings as well as the sometimes high chlorine content caused the fuel properties of the examined material to be ranked as troubling.

Regarding mono-combustion of high-calorific fractions from LFM, the relative amounts of HCl and SO_2 included in the raw gas have been observed to exceed customary process values. In addition, the generation of steam has varied strongly while the bottom ash has shown finer particle size distributions and increased contents of Cl in the eluate. In clean gas, the only elevated readings related to the relative amount of HCl. Based on these and further experiments the authors conclude that co-combustion of RDF from LFM with RDF from municipal solid waste (MSW) in a 1:1 ratio seems feasible while mono-combustion of non-pre-treated material remains troubling (Rotheut and Quicker, 2017).

Mechanical recycling and other options, such as gasification, pyrolysis and hydrogenation, can only be pursued if more elaborate pre-treatment (cleaning, drying, comminution and thorough sorting) takes place (Zhou et al., 2014). The waste-to-energy (WtE) route is expected to be a plausible method of utilising high-calorific fractions from (E)LFM (García López et al., 2018; Quaghebeur et al., 2013) while a waste-to-material (WtM) route is not considered promising due to the increased levels of contaminants in recyclables (Quaghebeur et al., 2013). Generating potential RDF (pRDF) of a quality sufficient for (co-)combustion from (E)LFM requires the separation of material classes containing contaminants like PVC (chlorine), minerals (increased ash content) and metals.

Compared to the calorific fractions, recycling inert constituents such as metals, glass, ceramics and stone is estimated to be more promising (Quaghebeur et al., 2013). Mechanical processing, e.g. the separation of wood, paper and plastics from inerts, is required in this case, either to reduce the TOC or the Cl content of fractions intended for construction aggregates.

Sensor-based sorting (SBS) can be utilised to separate impurities from the respective fractions (pRDF and inert) in order to comply with the boundaries of both the high-calorific fraction and the inert fraction. One promising approach is the treatment of 3D or heavy fractions derived from air classification or ballistic separation. Generating suitable products from an ELFM process may require carrying out not only single but also multi-stage sorting steps (cascading application of SBS technology). In this way, several types of plastics can be separated, resulting in pure product fractions. Near-infrared technology was found helpful both for distinguishing inert from pRDF and for differentiating types of plastic that then could be separated using compressed air blasts (Kieckbehäfer et al., 2017; Beel, 2017).

Flawless functioning of SBS units is achieved by pre-treating LFM material. An important step of preconditioning is drying the landfill material, hopefully to improve subsequent mechanical processing (García López et al., 2018). Contaminants such as fine adhesions and coarse particles should be separated and fractions of processable particle size ranges produced using pre-classification. The resulting particle size ranges have to match machine requirements of subsequent processing units. In addition, this classification can be used to subdivide the material flow into volume and mass flows suitable for further treatment, preventing overloading of the downstream processing equipment. Pre-processing can also be used to enrich the valuable substances or to separate contaminants into specific grain size ranges and material flows (Pretz and Julius, 2008).

If the treatment is carried out on a landfill site, SBS technology must be adapted to adverse environmental conditions. The treatment can be affected by weather conditions or by dust, raising the risk of e. g. soiling on light bulbs that would cause malfunction or impairment of the SBS equipment (Gundupalli et al., 2017).

This paper discusses near-infrared-(NIR)-based sorting since this technology provides a wide range of possible applications. However, NIR sensors can only analyse the surface of particles and are therefore particularly susceptible to external contamination and adhesions (Pomberger and Küppers, 2017). That is why research regarding the application of this technology in LFM is of particular interest. Two applications for NIR SBS technology are examined:

- Enrichment of the inert fraction using the separation of

combustibles (plastics, paper, cardboard, wood, etc.);
- Distinction and separation of various types of plastic from a LFM process to create unpolluted fractions.

To verify the technical feasibility of these applications, the distinctness of combustibles, inerts and impurities is examined and quantified.

2. MATERIALS AND METHODS

ELFM projects were conducted at Mont-Saint-Guibert (MSG), Belgium, and on the Halbenrain landfill, Austria. Excavations and mechanical processing in MSG were carried out to verify the suitability of a ballistic separator as the first processing unit in the mechanical processing of LFM material. In Halbenrain, the purpose of on-site examinations was to test whether the available mechanical biological treatment plant was able to process LFM material.

At the MSG landfill, municipal solid waste (MSW) and construction and demolition (C&D) waste were disposed of between 1958 and 1985. For the purposes of this examination, the top clay layer was removed first. Next, a total of about 425 m³ of landfill material was excavated in four batches and treated with the ballistic separator (STADLER STT 6000). The LFM material was processed in two stages, using mesh sizes of 200 mm and 90 mm to produce five output streams (cf. Figure 1). Detailed information concerning landfill site, processing, sampling and landfill composition is provided by García López et al. (2019) and Hernández Parrodi et al. (2019).

The fine fractions samples (<90 mm), gained during ballistic separation, were divided into two fractions and treated at two different water content levels (0 wt% and 15 wt%) or using screening into the grain size ranges of 90-30 mm, 30-10 mm, 10-4.5 mm and <4.5 mm, followed by air classification. Heavy fractions from air classification (90-30 mm, 30-10 mm and 10-4.5 mm) yielded input material for SBS experiments and were therefore drawn on during the examination.

At the Austrian Halbenrain landfill, about 500 t of LFM material were excavated and processed in an on-site mechanical and biological treatment plant in 2016 to study options for RDF and metal recovery. The treatment included

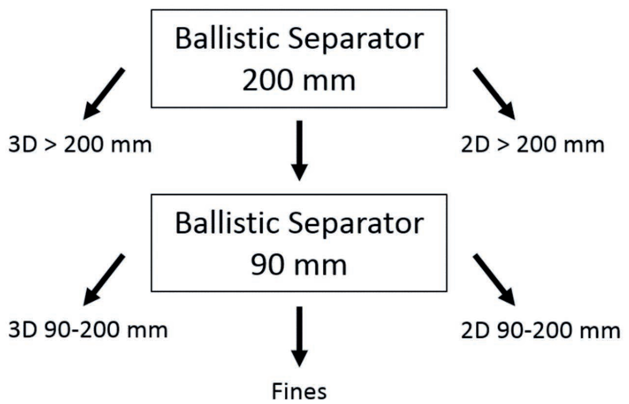

FIGURE 1: Procedure for mechanical treatment of landfill material in MSG.

three to four weeks of biological drying in rotting boxes, followed by shredding, multiple screening stages (screen cuts: 200 mm, 60 mm, 14 mm), separation of ferrous and non-ferrous metals using magnetic and eddy current separation and, finally, air classification. More detailed information is provided by García et al., 2018.

For SBS examinations, samples were taken from the heavy fraction of 200-60 mm. These samples were handpicked and all particles (1377 pieces) categorised by visual inspection and Fourier Transformed Infrared Spectroscopy (FTIR, Agilent Technologies, Cary 630) according to material types: polypropylene (PP), polyethylene (PE), polystyrene (PS), polyvinyl chloride (PVC), polyethylene terephthalate (PET), and residuals.

A hyperspectral imaging (HIS) near-infrared chute sorter (sensor: EVK HELIOS NIR G2 320, spectral range 990-1700 nm) was used for performing the SBS experiments. The spectral resolution of the sensor was 3.18 nm, its spatial pixel width being 1.60 nm. The frame rate of the line sensor was 476 Hz for an exposure time of 1800 µs. During the experiments, the side of a spatial pixel in the direction of movement was always less than 1.6 mm long. Sorting recipes were created by recording sample objects of each material class. Two recipes emerged:

- Recipe 1 – for treating fractions of 90-30 mm, 30-10 mm and 10-4.5 mm from MSG (objective: separation of combustible from inert materials of the heavy fractions);
- Recipe 2 – for treating air-classified heavy fractions from the Halbenrain landfill (objective: distinction of plastic types PP, PE, and PVC from PS, PET and residuals).

Recordings of the recipes contained spectra allocated to pixels on all sample objects. Spectra from several pixels were collected and an average spectrum for each material was created. These spectra were used as references for classifying object pixels. The classification of different materials was improved by including only such wavelength ranges that displayed significant differences (cf. Figures 2 and 3). As a result, a pseudo-colour was allocated to each object pixel. All objects were then assigned to the material class represented by the predominant pseudo-colour. Particles were separated using air blasts from a compressed air nozzle bar to validate the sorting efficiency of the recipes. Low throughputs were applied to avoid overlapping objects, allowing to quantify correct classification rates unaffected by variations of the throughput.

The spectra used in recipe 1 are given in Figure 2. In total, 16 spectra of pulp-based materials (wood, paper and cardboard) and bone (red), plastics (green) and inerts (blue) were used for this recipe. Wavelength ranges included in the classification cover 1120-1273 nm, 1342-1527 nm and 1618-1674 nm (areas marked red in Figure 2). Soot-blackened plastics cannot be classified by spectral data reflected in the NIR range since they absorb much of the irradiation. Their classification as plastics was facilitated by using the low intensity of reflected radiation.

For evaluating the results of trials based on recipe 1, an

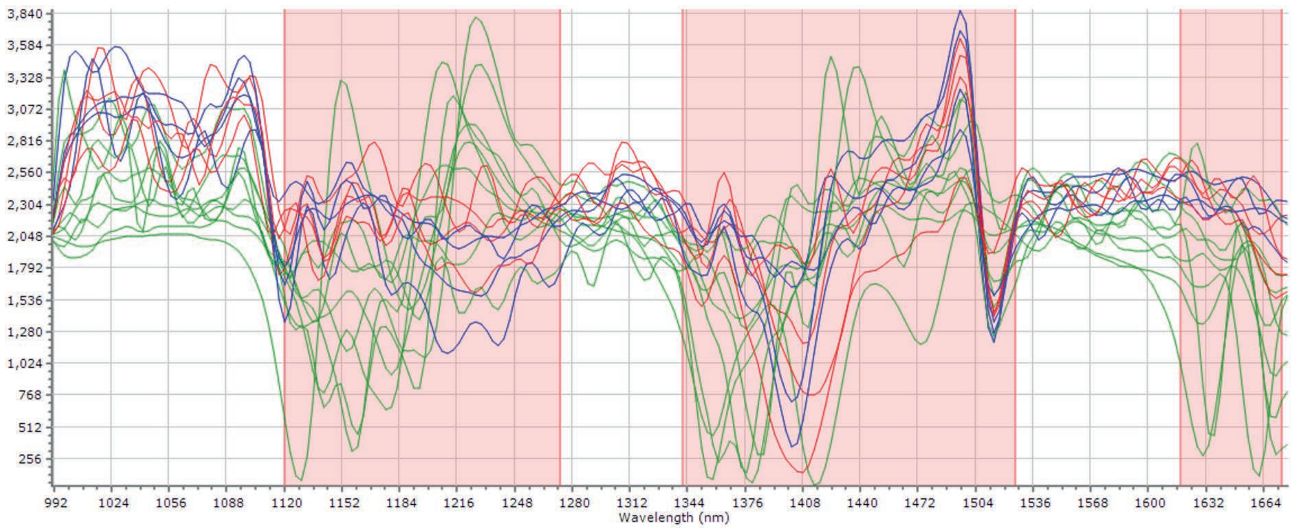

FIGURE 2: Spectral data used to distinguish inerts (blue) and combustibles, consisting of pulp (red) and plastics (green).

enrichment ratio was calculated to assess the separation efficiency of combustibles and inerts. For this purpose, the purity of a product fraction (inert content in inert product or combustibles content in combustibles product) was divided by the content of the respective material class in the input fraction. If the enrichment ratio was 1 or higher, the material class was enriched via sorting.

Spectra used for the constituent separation of PP, PE and PVC from PS, PET and residuals based on recipe 2 are given in Figure 3. In total, 7 spectra of PP (blue), PE (red) and PVC (green) were applied. Wavelength ranges included in the classification cover 1120-1242 nm, 1339-1414 nm and 1636-1671 nm (areas marked red in Figure 3). To prevent erroneous classification of PVC as PP or PE, soot-blackened plastic was always classified as PVC, based on the low intensity of radiation reflected by such particles. Neither spectra of PS nor of PET or residuals were stored in recipe 2.

For assessing the correct classification rates based on recipe 2, both mass-based (wt%) and particle-based (p%) approaches were examined. For the mass-based approach, the weight of each material class in the eject and reject fraction entered the calculation of yield by material. For the particle-based approach, the numbers of particles in the eject and reject fractions were counted and entered the calculation of yields by material type in a product. Multiple experiments were performed to reduce the effects of outliers due to atypical positioning and mechanical errors from atypical motions of objects during the sorting process. For each trial, all particles were analysed and sorted anew.

3. RESULTS AND DISCUSSION

The results of trials separating inerts and combustibles are given in Table 1. Data on the composition of input and output fractions is given separately for trials based on water contents of 0 wt% and of 15 wt%. Comparing the composition of all input fractions demonstrates that the inert content rises with decreasing particle size range for both water content levels. Especially material in the frac-

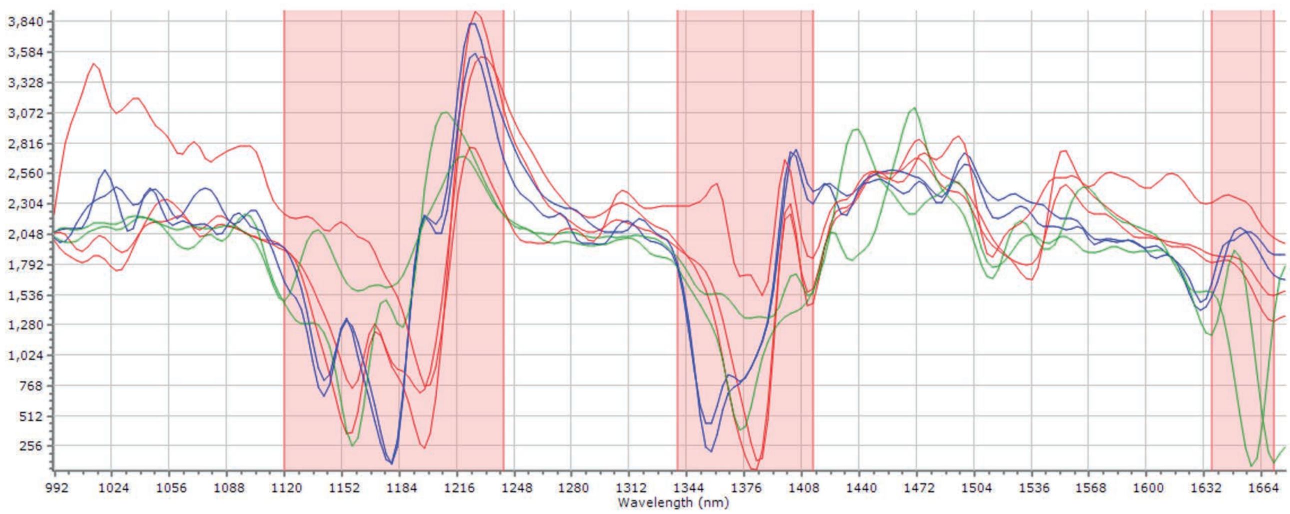

FIGURE 3: Spectral data stored in recipe 2 for distinguishing PP (blue), PE (red) and PVC (green) from PS, PET and residuals.

TABLE 1: Results of sorting trials with Recipe 1.

Water Content [wt%]	0			15		
Particle size range [mm]	90-30	30-10	10-4.5	90-30	30-10	10-4.5
Input						
Inert content [wt%]	88.2	97.1	98.1	85.6	93.7	98.8
Combustibles content [wt%]	11.8	2.9	1.9	14.4	6.3	1.2
Reject - Inert Fractions						
Purity - Inert [wt%]	99.3	99.7	98.8	99.6	97.7	99.4
Yield of inerts [wt%]	99.7	97.8	99.0	99.8	95.1	99.5
Enrichment ratio	1.126	1.027	1.007	1.164	1.043	1.006
Eject - Combustibles Fractions						
Purity - Combustibles [wt%]	62.0	26.3	36.3	68.9	31.3	49.5
Yield of combustibles [wt%]	94.7	72.2	99.0	97.4	66.8	53.6
Enrichment ratio	5.254	9.069	19.105	4.785	4.968	41.250

tion of 90-30 mm contains significantly more combustibles than material in smaller particle size ranges.

The relative content of combustibles in the input is higher for the water content of 15 wt% in particle size ranges of 90-30 mm and of 30-10 mm than in samples containing a water content of 0 wt%. This observation can be explained by higher water absorption in combustibles compared to inerts and increased content of adhesive fines due to the wet surface of combustibles.

All trials achieved high purities for inert fractions (97.7-99.7 wt%) while the purity of combustible fractions was comparatively low (26.3-68.9 wt%). Yet enrichment ratios are low for inert (1,006-1,164) and high for combustibles (4.785-41.250). These results can be attributed to the low content of combustibles in all input fractions, enabling high enrichment ratios, while high inert contents in the input fractions limit enrichment ratios. Rather low purities of combustibles in the respective product fractions can be explained by inert losses wrongly classified and ejected as combustibles. Due to the high amounts of inerts in input fractions, even minor loss of inerts can strongly impact the combustible fractions. The purity of generated combustible fractions could be further improved by applying multiple sorting stages, enriching the combustible fraction even more; the economic feasibility of this approach, however, is in doubt.

The enrichment ratios for inert fractions decrease with particle size while the enrichment ratios of combustibles increase. This is mainly associated with the decreasing relative amount of combustibles in the input for small particle size ranges. For instance, a water content of 0 wt% produced neither a lesser yield of inerts nor of combustibles for the particle size range of 10-4.5 mm, compared to the sorting results of the 90-30 mm fraction. Still, the enrichment ratio of inerts drops with decreasing particle size while the enrichment ratio of combustibles rises.

Although a water content of 15 wt% produces a proportionally lesser yield of combustibles with decreasing particle size, the combustibles enrichment ratio for the particle size range of 10-4.5 mm is highest at 41.250, which is more than twice the respective value for a water content of 0 wt%. This is mostly due to the fact that the lower combustibles content in the input and the 0.5 wt% higher yield of inerts for a water content of 15 wt% have more impact on the sorting efficiency than the yield of combustibles would.

Misclassification, mostly of combustibles but to some extent also of inert, can mainly be referred to faulty classification of pulp material. These particles were characterised by high amounts of adhesive fines on the surface. Such adhesives can impair the spectra, e. g. of paper, resulting in a mixed spectrum of inert and paper. This effect can also be observed in Figure 2 as spectra of pulp-based particles look similar to inert spectra due to their contamination with adhesive fines. The adverse effect of adhesives on SBS is best demonstrated for a water content of 15 wt%, due to increased amounts of adhesives. The yield of combustibles drops with particle size. Especially for the fraction of 10-4.5 mm, major quantities of adhesive fines could be observed on particle surfaces (compare Figure 4).

The sorting results of all trials involving Halbenrain material are given in Table 2. To evaluate the classification and sorting efficiency independently from input composition, only the yield of PP, PE and PVC products is given for each fraction. Results show the average yield of each material class over five runs.

Results of sorting trials show that the particle- and mass-related yields correlate significantly. Differences can be attributed to differing particle masses.

While generating the PP product, neither PVC, PS nor PET were wrongly classified and sorted. Less than 0.8 p% (0.6 wt%) of PE and residuals were falsely ejected.

When sorting out the PE product, the primary misclassification observed was that of PP as PE (3.2 p%/2.1 wt%) while the ejection of PVC, PS and residuals stayed always below a level of 0.8 p% (0.3 wt%), partly due to gliding particles (PS). No misclassification of PET was observed.

While producing the PVC product, an increased misclassification of PP (4.4 p%/2.7 wt%), PE (11.0 p%/8.8 wt%) and residuals (5.1 p%/3.5 wt%) was observed which can be attributed to the classification of soot-blackened particles as PVC. No PET was misclassified and discharged as PVC.

An overall distinction between plastic types and separation of such material fractions from LFM could be per-

FIGURE 4: Reject (left) and eject (right) from SBS trial, water content 15 wt%, fraction 10-4.5 mm.

formed using SBS technology. The relatively high accuracy rates achieved, despite misclassification due to soot-blackened particles, can be explained by the preceding comprehensive biological and mechanical processing as well as by the coarse particle grain size of the examined material, resulting in low amounts of water and adhesives on particle surfaces and enabling mostly correct classification and sorting results.

However, further multiple sorting stages and treatments (cleaning, drying, etc.) will be necessary to meet the requirements of recycling and not only those of WtE.

4. CONCLUSIONS

NIR-based SBS trials using pre-treated landfill material show promising results for the application of this technology in ELFM. The separation of inert and combustibles and the distinction between specific types of waste plastic was successfully demonstrated. Sorting efficiency is affected by the level of pre-processing, the water content and the relative amount of adhesive fines, the material composition and the range of particle sizes of the input material at the SBS stage.

While a decent identification of plastic types (except for soot-blackened plastics) using NIR spectroscopy is possible, detecting pulp-based particles and distinguishing them from inerts was sometimes impaired for particle size ranges <30 mm due to adhesive fines, particularly when water was present.

Whether any long-term stability of a sufficiently effective SBS process can be achieved under plausible processing conditions has to be tested at large-scale. Problems, e. g. due to dust formation or various degradation states of plastics, may decrease the efficiency rates attained so far. In such cases it might be necessary to adapt the algorithm for material classification.

TABLE 2: Yield of PP, PE and PVC products from Halbenrain LFM air-classified heavy fraction – average of 5 runs.

	PP product	PE product	PVC product
PP yield	94.1 p%	3.2 p%	4.4 p%
	94.8 wt%	2.1 wt%	2.7 wt%
PE yield	0.8 p%	92.9 p%	11.0 p%
	0.6 wt%	91.8 wt%	8.8 wt%
PVC yield	0.0 p%	0.2 p%	95.2 p%
	0.0 wt%	0.1 wt%	99.7 wt%
PS yield	0.0 p%	0.8 p%	0.8 p%
	0.0 wt%	0.3 wt%	0.7 wt%
PET yield	0.0 p%	0.0 p%	0.0 p%
	0.0 wt%	0.0 wt%	0.0 wt%
Residuals yield	0.4 p%	0.4 p%	5.1 p%
	0.3 wt%	0.2 wt%	3.5 wt%

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RELATING MAGNETIC PROPERTIES OF MUNICIPAL SOLID WASTE CONSTITUENTS TO IRON CONTENT – IMPLICATIONS FOR ENHANCED LANDFILL MINING

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ABSTRACT

Ferrous metals are a main recyclable waste fraction in Enhanced Landfill Mining (ELFM) projects. However, prior to mining, the metal content of municipal solid waste (MSW) landfills is unknown. We investigate if the metal content of MSW landfills can be estimated by inverse modeling of geophysical measurements as the magnetic properties of the subsurface are particularly sensitive to ferromagnetic metal enrichments. We conducted magnetic total-field measurements on a MSW landfill in Austria and estimated the bulk magnetic susceptibility (MS) of the subsurface by inverse modelling. For validation of the subsurface MS values, 32 drill-core samples from multiple locations and depths within the landfill were obtained and manually sorted into 12 waste fractions including ferrous metals (2.3 ± 1.4 wt.%, 1σ). To investigate if bulk MS could be accurately predicted from inverse modeling when the exact composition of the waste is known, the MS of iron and other expected waste fractions were investigated in laboratory analysis using reference samples from waste treatment plants and another ELFM project. Laboratory analyses partly yielded significantly larger MS values for waste materials than those given for virgin materials in literature. The bulk MS for each sample from the ELFM project was computed using a weighted mean with respect to the waste composition derived from manual sorting. The bulk MS derived from inverse modelling of the field data (0.06 to 0.11 SI) exceeded the bulk MS derived from the material composition of waste samples and the MS values of reference samples (0.01 to 0.05 SI).

1. INTRODUCTION

Landfill mining (LFM) (Krook, Svensson, & Eklund, 2012) and more recently Enhanced Landfill mining (ELFM) (Jones, et al., 2013), the recovery of resources from landfills, have gained increasing attention in the last few decades. The main focus of mining municipal solid waste (MSW) landfills is calorific fractions (20-30 wt.% dry matter) for energy recovery and metals (1-5 wt-% dry matter) for direct material recovery (Krook, Svensson, & Eklund, 2012). Whereas, decomposed organic and weathered mineral materials, often summarized under the term fine fractions (50-60 wt.-%) (Hernández Parrodi, Höllen, & Pomberger, 2018), and inert material (10 wt.-%) are mostly not recyclable (Wolfsberger, et al., 2015). MSW metal content can vary significantly within, and for different, landfills (Krook, Svensson, & Eklund, 2012), as it depends mainly on the fresh waste composition (Burnley, 2007) and mechanical-biological pre-treatment (Leikam & Stegmann, 1999). From an economic point of view, a relatively high proportion of metals in a landfill is the decisive parameter for a positive assessment of an ELFM project (Winterstetter, Laner, Rechberger, & Fellner, 2015). Therefore, it is relevant to investigate the metal content of a landfill prior to deciding the potential for ELFM.

Often, invasive methods are applied to estimate the material composition of landfills, this mainly includes drilling, and trenching, followed by manual sorting of the sampled material (Mor, Ravindra, de Visscher, Dahiya, & Chandra, 2006). However, invasive techniques are expensive and time-consuming. Due to these disadvantages, often only small volumes of a landfill are explored (McCann, 1994). This is problematic due to the strong heterogeneity of MSW, which can cause the results to be unreliable and/or poorly representative.

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Using non-invasive geophysical methods, larger sections of landfilled waste can be investigated in terms of their in-situ physical properties, and they can be scanned quickly, and hence, at relatively low expense. Such methods have been applied for decades in exploration of geogenic deposits like ores or petroleum (Dobrin & Savit, 1960), and have also proven to be useful for the investigation of landfills. However, most geophysical studies of landfills focus on environmental problems arising from the landfilled waste, and material composition is often not explored (Orlando & Marchesi, 2001) (Hermozilha, Grangeia, & Senos Matias, 2010) (Porsani, et al., 2004). However, recently attempts towards landfill characterisation in the context of ELFM projects have included the determination of leachate saturation levels and thickness of the waste by vertical electric sounding, electrical resistivity tomography, induced polarization and seismic diffraction techniques (Cardarelli & di Filippo, 2004). Additionally, since the development of the concept of ELFM, a few studies have focused on geophysical exploration through electromagnetic induction in view of supporting the decision of, and relevance of, an ELFM scenario (e.g., (Bobe, Van De Vijver, & Van Meirvenne, 2018); (Van De Vijver & Van Meirvenne, 2016)).

Magnetic methods are mainly applied to identify individual objects (e.g. drums) in landfills (Prezzi, Orgeira, Ostera, & Vasquez, 2005). On the contrary, in ELFM the average material composition of an entire landfill, or at least a compartment, is crucial (Krook, Svensson, & Eklund, 2012), not individual objects. For example, in one simpler non-MSW ELFM case study, landfilled foundry sands were distinguished from iron-rich materials in an industrial landfill by the absence of magnetic anomalies (Zanetti & Godio, 2006). In general, iron scrap in MSW landfills does not form whole compartments or layers, but occurs as individual objects of various sizes (µm - m) which are mixed with other MSW constituents like plastics, wood, inert materials and decomposed organics. For this reason, for ELFM of MSW landfills, the approach to identify individual objects is not valuable, but rather average MS values of whole landfills or larger parts of a landfill are required. In the recent ELFM study of Yannah, Martens, van Camp, & Walraevens (2019), metal enrichments were differentiated qualitatively to plastic enrichments. However, in general, quantitative interpretation of geophysical measurements in terms of waste composition remains challenging.

Iron is the most common metal in landfills (García López, et al., in press), and the magnetic susceptibility (MS), which is the degree of magnetization of a material in response to an applied magnetic field, of iron is large compared to other materials (Schenck, 1996). Thus, geophysical prospecting with magnetic methods is a logical method to apply in ELFM evaluation. Magnetic surveying allows one to derive subsurface MS values, and as the MS of the bulk of landfill materials is negligible compared to the contribution from iron, it might be possible to estimate iron content from magnetic data.

In magnetic exploration, there are two different approaches.

In gradiometer magnetic measurements, the strength of the Earth´s magnetic field with respect to an accurate location and time is measured. By comparing the geomagnetic reference field with temporal changes of the field during the survey time, a local magnetic anomaly is derived. Such anomalies can be linked to magnetisable objects in the subsurface. The magnetic susceptibility of a material gives a measure of its degree of magnetizability. In general, the interpretation of magnetic anomalies is not straightforward due to various complications arising from the strongly variable nature of waste materials and dynamic landfill conditions. Relating to the former, objects with two types of magnetization may occur: (i) induced magnetization due to the present Earth´s magnetic field, and (ii) remanent magnetization due to an earlier exposure to a magnetic field, which might have had a different orientation than the currently prevailing Earth magnetic field. The interaction of these magnetization effects, for the entire collection of individual magnetic objects present in landfilled waste, impedes unique interpretation of magnetic data in terms of waste properties. For MSW landfills, it is assumed that the remanent magnetization is even harder to capture than for rocks, as landfills usually consist of many small objects whose remanent magnetization is randomly distributed and therefore, levels off at a larger scale, not locally, such that magnetic data interpretation of landfills is more complicated than interpretation of geological magnetic anomalies.

To derive surface MS, active induction measurements are commonly applied. In these measurements an artificial alternating electromagnetic field is applied to the surface. The response of the illuminated volume to the alternating field is then recorded and can be linked to its magnetic susceptibility.

MS values of inorganic materials like copper (Cu, -9.63 x 10^6), iron (Fe, 2.00 x 10^5 SI), water (H_2O, -9.05 x 10^{-6} SI) and air (3.60 x 10^{-7} SI) were summarized by Schenck (1996). With respect to the prediction of the content of iron and other ferrous metals, it has to be stressed that the MS values for different iron alloys can vary around eight orders of magnitude. Data for organic waste constituents like polyethylene (PE, 4.34 x 10^{-5} SI), polyethylene terephthalate (Selwood, Pardo, & Pace, 1950) and for wood (-3.88 x 10^{-7} SI) (Phaovibul, Loboda-Cackovic, Hosemann, & Balta-Calleja, 1973) are limited and for some organic polymers like cotton or polypropylene (PP) no data was found. All relevant and available literature values are summarized in Table 1. It can be seen that due to its ferromagnetism, iron content can be expected to dominate magnetic anomaly data. Thus, we expect to find a relationship between the absolute values of magnetic anomalies and the landfill iron content.

For ELFM projects the values for virgin substances are of restricted applicability as material groups which are derived from mechanical processing, still contain fine-grained contaminations of other materials and are degraded in landfills. For an economic assessment of ELFM, only the share of metals that is obtained by mechanical processing or manual sorting can be compared to magnetic data and is therefore relevant to consider. Degraded and contaminated waste fractions, which are obtained after mechanical processing of fresh or landfilled MSW, have undergone

TABLE 1: Literature values for magnetic susceptibility of pure materials (1) = (Schenck, 1996), 2 = (Phaovibul, Loboda-Cackovic, Hosemann, & Balta-Calleja, 1973), 3 = (Rakos, Murin, Kafka, Varga, & Olcak, 1984), 4 = (Selwood, Pardo, & Pace, 1950)

Material	Formula/Abbreviation	Magnetic Susceptibility (SI)
Pure iron (1)	Fe	$2.00 \cdot 10^{5}$
Magnetic stainless steel, martensitic (1)	(Fe,Cr)	$4.00 \cdot 10^{2} – 1.10 \cdot 10^{3}$
Magnetite (1)	Fe_3O_4	$7.00 \cdot 10^{1}$
Stainless steel, austenitic (1)	(Fe,Cr)	$3.52 \cdot 10^{-3} – 6.70 \cdot 10^{-3}$
Goethite (1)	FeOOH	$1.46 \cdot 10^{-3}$
Polythylene (2)	PE	$4.34 \cdot 10^{-5}$
Air (1)	78% N_2 + 21% O_2	$3.60 \cdot 10^{-7}$
Quartz (1)	SiO_2	$-1.63 \cdot 10^{-5}$
Water (1)	H_2O	$-9.05 \cdot 10^{-6}$
Copper (1)	Cu	$-9.63 \cdot 10^{-6}$
Polyethylene terephthalate (4)	PET	$-6.74 \cdot 10^{-7}$
Cellulose (3)		$-3.37 \cdot 10^{-7}$
Fir wood (3)		$-3.88 \cdot 10^{-7}$

significant material changes, and might therefore, have a different magnetic susceptibility compared to that of virgin materials from industry. Therefore, a calibration of the magnetic model of a landfill (McCann, 1994) can only be validated when MS values of respective waste fractions are known. As such, the aim of magnetic exploration is to relate its results to the percentage of the ferrous metals fraction obtained by manual or mechanical sorting. Furthermore, for ELFM applications it has to be considered that ferrous metals may also occur as contaminations in other waste fractions.

In summary, two key issues hinder a quantitative iron content estimation in landfills using magnetic exploration methods: the change of the material during degradation in landfills and waste processing, and the large variation in intrinsic MS values of ferrous metals. These points motivate the two research questions addressed in this work: (i) to which degree do the MS values of virgin materials differ from those of waste fractions containing these materials, but which have been altered and have been contaminated by other materials, and (ii) can laboratory measurements of the MS of (mixtures of) individual waste fractions be related to total field magnetic measurements on site, and can the combination of laboratory and field measurements predict the iron content of a landfill?

To the best of our knowledge, quantitative studies for geophysical iron content exploration have not been presented yet. In order to predict the share of iron in landfills, the observed deviation of the total magnetic field intensity and the thereof derived bulk subsurface MS must be associated to the MS of different waste fractions.

To investigate our research questions, we approach the iron content estimation problem in three steps. First to answer research question (i) we conduct MS analyses on reference samples (i.e. waste fractions, and if not available, samples of materials which also occur in landfilled waste) and compare them to literature values of corresponding relevant materials. Second, we calculate the MS of waste mixtures based on compositions derived from manual sorting of samples obtained in a drilling campaign from an Austrian MSW landfill. The bulk MS calculation is based on MS values derived from the reference samples and the ratio of materials present. Third, we use the anomalous magnetic data from a survey at the respective Austrian MSW landfill to estimate the bulk MS of the waste along a profile by inverse modelling. The calculated mean MS for the drill-cored material was compared to the inverse-modelled MS to answer research question (ii).

2. MATERIALS AND METHODS

2.1 Magnetic Lab Tests

In total, magnetic properties of twelve reference samples representing materials relevant in ELFM projects were investigated (Table 2). Six waste fractions were obtained from a landfill mining project (Muras, Küppers, Höllen, & Rothschedl, 2018), five of them produced by manual sorting (polypropylene (PP), polyethylene (PE), paper & paperboard, textiles and wood), and one from mechanical biological treatment (MBT), i.e. the light fraction of a windsifter. Shredded bottle lids (PE + PP + polyethyleneterephthalate, PET) and black plastics were obtained from a plastic recycling plant, two further samples, i.e. copper scrap and granulated pig iron (GPI, d = 5 mm), were obtained from industrial partners, and building sand and potting soil were commercial products. In summary, 10 of 12 samples are waste samples characterized by contaminations of other materials, e.g. the iron fraction.

Sample bodies of all investigated reference samples were produced in non-compacted form in two open vessels of different size, a cube of 8 cm³ volume, and a cylinder (h = 9 cm, 10 cm, V = 707 cm³). Overall, four different magnetic sensors were used to check data reliability. These were: the Exploranium KT-9 (sensitivity: 10^{-5} SI units, measuring time 0.5 s, frequency 10 kHz); the Bartington MS2 with MS2EI sensor (2 kHz); the Agico MFK1-FA Kappa bridge; and the KLF-3 Minikappa. The Explonarium and Bartington sensors were used for the larger test specimens and the Agico and the KLF-3 sensor for the smaller test specimens. Meas-

TABLE 2: Reference samples for lab tests for the determination of magnetic properties.

No.	Material	Sample Codes	Origin
1	Polypropylene (PP)	PP-1 to PP-6	Halbenrain Landfill
2	Light fraction windsifter	LG-1 to LG-6	Halbenrain Landfill
3	Polyethylene (PE)	PE-1 to PE-6	Halbenrain Landfill
4	Shredded bottle lids	PET-1 to PET-6	Recycling Plant
5	Paper & Paperboard	K-1 to K-6	Halbenrain Landfill
6	Textiles	TX-1 to TX-6	Halbenrain Landfill
7	Wood	H-1 to H-6	Halbenrain Landfill
8	Black plastics (PE + PP)	KS-1 to KS-6	Recycling Plant
9	Iron	FE-1 to FE-6	Granulated pig iron (GPI), Steel Plant
10	Copper	CU-1 bis CU-6	Chair of Nonferrous Metallurgy, MUL
11	Quartz sand, 0.3-1mm	S-1 bis S-6	Quester BauProfi Quartz sand, lime-free, fire-dried
12	Potting soil	BE-1 to BE-6	Hornbach Universalblumenerde

urements with the Explonarium and Bartington sensor (12 measurements per material, per sensor and 24 measurements for light fraction (LG) samples), were conducted in direct contact with the material surface. The Bartington MS2 sensor was applied at different spots on the sample cube, whereas the Exploranium KT-9 sensor was applied at one single measurement spot due to its required larger sample size, which did not allow measuring different spots on one plane. Measurements with the Agico sensor were conducted in triplicate at two frequencies (976 Hz, 3904 Hz). For measurements of the GPI sample, the cubes were only partly filled to 15 ± 2% as MS values of completely filled cubes exceeded the calibration range.

The measured bulk MS values of partly filled cubes were converted to intrinsic MS values by considering the volume share of the investigated sample. To account for the MS of the sample cube, the average susceptibility of an empty sample cube was used as a blank and subtracted from the respective sample values.

The natural remanent magnetization (NRM; Mr) of the reference samples was investigated in free space and in the absence of any external magnetic field. The measurements were performed using two magnetometers, the Bartington Mag-01H Fluxgate for Cu and Fe, and the 2G Enterprises for all other samples. The samples were inserted in a cube with 8 cm edge length. For Cu and Fe, the NRM was determined in three spatial directions and the average values were used to calculate the resulting magnetization vectors. The induced magnetization (Mi) of Fe and Cu was calculated according to equation 1 from the magnetic susceptibility determined by the AGICO sensor, as this sensor has the largest sensitivity. For Cu and Fe, MS values for completely filled cubes were beyond the measuring range and therefore extrapolated from partly filled cubes.

$$Mi = 0.7958\ H \bullet \chi \qquad (1)$$

H = magnetic field at the site; x = magnetic susceptibility, the factor 0.7958 is due to the conversion in A/m.

The contribution of Mr and Mi is expressed by the Königsberg factor Q = Mr/Mi.

The viscous remanent magnetization (VRM) was determined after subjecting the samples to the Earth's natural magnetic field for 30 days and a subsequent measurement of the magnetization using the Bartington Mag-01H Fluxgate for Cu and Fe, and the 2G Enterprises for all other samples. The change in direction of the vector of the magnetization was used to calculate the VRM using Remasoft 3.0 software.

2.2 Magnetic Landfill Exploration

An Austrian MSW landfill, landfill site 1 in (Wolfsberger, et al., 2015), was selected for geophysical exploration. The survey locations were recorded with an accuracy of <1 m using a Trimble Total-station TK GPS. The survey area is outlined in Figure 1. The magnetic total-field measurements were conducted following a grid setup with a spacing of 2 m in the east-west direction and 1 m in the north-south direction, resulting in 4696 measurement points. Two GEM 19-OH proton-precession magnetometer sensors were placed at 1 and 2 m heights, the total magnetic field intensity and the gradient of the two sensors were recorded. In order to obtain the local magnetic anomalies, the measured total intensities were reduced by the mean International Geomagnetic Reference Field (IGRF) and filtered to eliminate long-wavelength features. The diurnal variation of the magnetic field was recorded using a second magnetometer (GEM 19-T) at a base station north and beside the landfill. Surface MS was mapped in a 2 by 2 m grid (2343 measuring spots) using an Explonarium KT-9 susceptibility sensor.

2.3 Landfill sampling campaign

A drilling campaign was conducted at the same Austrian MSW landfill and consisted of six drill-sites (Figure 1). For each borehole, several samples of 240 L were taken at 2 m depth intervals (Table 3). Waste samples from the drill-cores were then screened at 40 mm. The resulting sieve overflow (>40 mm) and underflow (<40 mm) were manually sorted into twelve material groups as defined in the Austrian Waste Management Plan (Austrian Federal Ministry of Agriculture, 2017), namely: ferrous metals, non-ferrous metals, plastics, paper/paperboard, inert, glass, com-

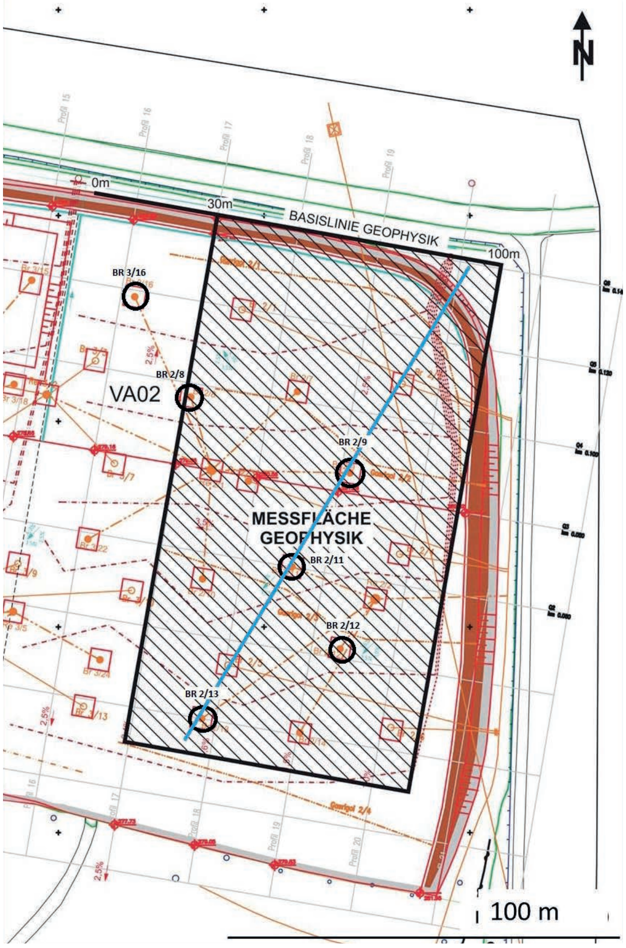


FIGURE 1: Measuring area (black) for magnetic exploration with indication of the profile for inverse modelling (blue), black circles indicate drilling locations.

pounds, problematic substances, wood, textiles, others and sorting residue. Samples from this campaign had been discarded by the time magnetic lab tests were conducted (Chapter 2.1), as such, the waste fractions obtained here by manual sorting are not the same as the samples used for lab tests in chapter 2.1. Finally, the resulting samples were weighed to derive the weight percentage of each fraction.

2.4 Magnetic Modelling

Bulk MS values for the investigated landfill material which comprises of individual constituents with distinct intrinsic MS values, were approximated in two ways, (i) from the waste composition, and (ii) deduced from magnetic anomalies.

Firstly, the MS values of individual waste reference samples that were obtained with the MFK1-FA sensor were multiplied by the percentage of individual waste fractions according to manual sorting of landfilled waste. Thus, the MS value obtained should be representative of the true bulk MS value for the excavated material and be directly comparable with the MS values derived from magnetic anomalies. Some approximations were used, in particular, the MS value for Cu scrap was used to represent the entire non-ferrous metal fraction, the PE reference sample was used for the plastics fraction, a value for quartz sand was used for the glass and the inert fractions, and for composite materials a value of 3.54 x 10^{-5} SI was used. This last value was derived from the composition of 75% paper, 20% PE and 5% aluminium (Al). The MS value for Al was taken from (Nave, 2019) and the MS value for potting soil was used for the sorting residue. For the water content, a MS value of 8.72 x 10^{-6} SI was used. Problematic materials (e.g. batteries, syringes etc.) and other materials were not considered.

TABLE 3: Samples from drilling used for manual sorting.

Borehole	Depth [m]	Number of samples
BR 3/16	7.70	3
BR 2/8	15.20	7
BR 2/11	17.20	8
BR 2/12	12.40	5
BR 2/13	10.40	4
BR 2/9	11.30	5
Sum		**32**

Secondly, magnetic anomalies observed in the field survey were used to model the bulk MS values for distinct prismatic bodies along a profile (Figure 1) using POTENT software from Geophysical Software Solutions Pty. Limited (Australia). Estimating bulk MS of distinct volumes simplifies the problem that within landfills, not all objects present can be resolved by inverse modelling. This is due to the abundance of different objects within the landfill. Reduced magnetic anomaly values were inverted without contributions from remanent magnetization and a geological background susceptibility of 5 x 10^{-4} SI. The Earth's induced magnetic field was set to that of the IGRF field at the time of the data acquisition, namely 48591 nT with a declination of 4° and an inclination of 65°.

The two approaches, prediction from waste composition, and inversion from magnetic anomalies, yielded different MS values. The bulk MS of the waste mixture is dominated by the MS of iron, as the MS of other materials is negligable (Table 1). Furthermore, the MS of iron is not constant. Thus, we investigated what intrinsic MS value for iron would yield a bulk MS value that corresponds to the MS modelled from measured magnetic anomalies of the MSW.

3. RESULTS AND DISCUSSION

3.1 Magnetic Lab Tests

The MS values of waste reference samples are summarized in Figure 2 for all sensors. In Table 4, data from the MFK1-FA sensor is listed separately. Sample statistics suggest that the data are not normally distributed, thus, median and quartile values are given.

The MS laboratory results are considered reliable as measurements of the MS2 sensor were generally in agreement with those of the MFK1-FA sensor. However, measurements with the MS2 sensor yielded several outliers. Values obtained by the MFK1-FA sensor were used for the prediction of the magnetic anomalies in chapter 3.2.

It was expected that positive outliers for one KS, one

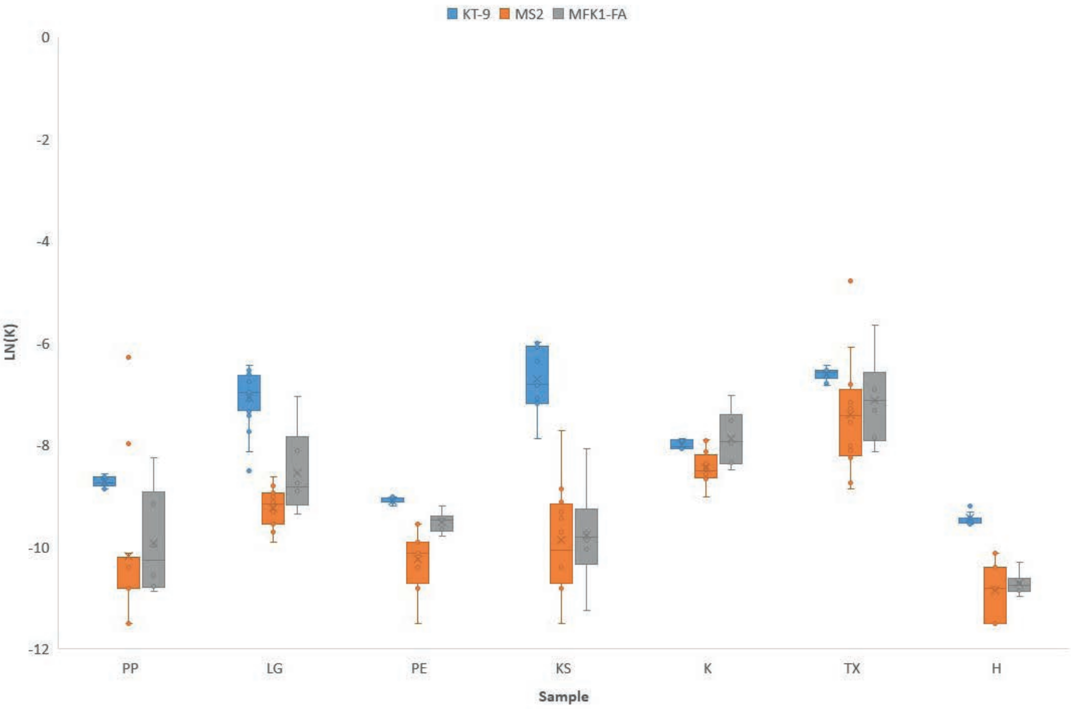


FIGURE 2: Magnetic susceptibility of reference samples.

TX and one PP sample were due to contamination by magnetic particles. This was confirmed by their removal using a hand magnet and subsequent re-measurement. For PET, Cu and Fe, the MS values were below and above the technical measuring range of the Exploranium KT-9 and Bartington MS2 sensors, respectively. The magnetic susceptibility in all quartz sand measurements, and 10 of the 12 potting soil measurements, were below -2.10×10^5 SI and within the measurement uncertainty of the Bartington MS2 sensor. These data are therefore not shown. The measured MS values of Fe and Cu were above the measuring range of the KLF-3 Minikappa and Agico MFK1-FA sensors.

Magnetic susceptibility is a frequency dependent quantity. Frequency dependence of magnetic susceptibility is defined as the percentage deviation between the susceptibility at 3904 and 976 Hz. This dependence was in the range of 4% for all measurements, which is far below the variation within each material group.

The NRM of individual materials shows large variations within each material group. For most materials the median values are between 0.02 and 0.42 SI. We found deviations from this range only for shredded bottle lids (NRM = $1.56 \cdot 10^{-4}$ SI) and quartz sand (NRM = $4.05 \cdot 10^{-4}$ SI) (Figure 3, Figure 5). Cu shows values in the same order of magnitude as the remaining non-metallic materials. Fe shows values located at the higher end of the spread, but not significantly higher than values derived for paper and textiles, which were expected to be significantly lower. No clear correlation between NRM and magnetic susceptibility was observed. This is due to the random orientation of grains

TABLE 4: Reference samples for lab tests for the determination of magnetic properties.

Sample	Q1	Median	Q3
K	$2.28 \cdot 10^{-4}$	$3.56 \cdot 10^{-4}$	$6.10 \cdot 10^{-4}$
LG	$1.03 \cdot 10^{-4}$	$1.46 \cdot 10^{-4}$	$3.89 \cdot 10^{-4}$
PE	$6.17 \cdot 10^{-5}$	$7.56 \cdot 10^{-5}$	$8.23 \cdot 10^{-5}$
PP	$2.03 \cdot 10^{-5}$	$3.48 \cdot 10^{-5}$	$1.32 \cdot 10^{-4}$
S	$1.38 \cdot 10^{-6}$	$2.21 \cdot 10^{-6}$	$3.30 \cdot 10^{-6}$
TX	$3.62 \cdot 10^{-4}$	$8.04 \cdot 10^{-4}$	$1.40 \cdot 10^{-3}$
BE	$2.93 \cdot 10^{-5}$	$3.54 \cdot 10^{-5}$	$4.31 \cdot 10^{-5}$
H	$1.87 \cdot 10^{-5}$	$2.13 \cdot 10^{-5}$	$2.46 \cdot 10^{-5}$
KS	$3.20 \cdot 10^{-5}$	$5.52 \cdot 10^{-5}$	$9.46 \cdot 10^{-5}$
PET	$-1.22 \cdot 10^{-7}$	$-1.17 \cdot 10^{-6}$	$-1.07 \cdot 10^{-6}$
FE (calculated from partly filled cubes)	$0.90 \cdot 10^{-1}$	$9.31 \cdot 10^{-1}$	$9.61 \cdot 10^{-1}$
CU (calculated from partly filled cubes)	$2.62 \cdot 10^{-4}$	$3.31 \cdot 10^{-4}$	$5.52 \cdot 10^{-4}$

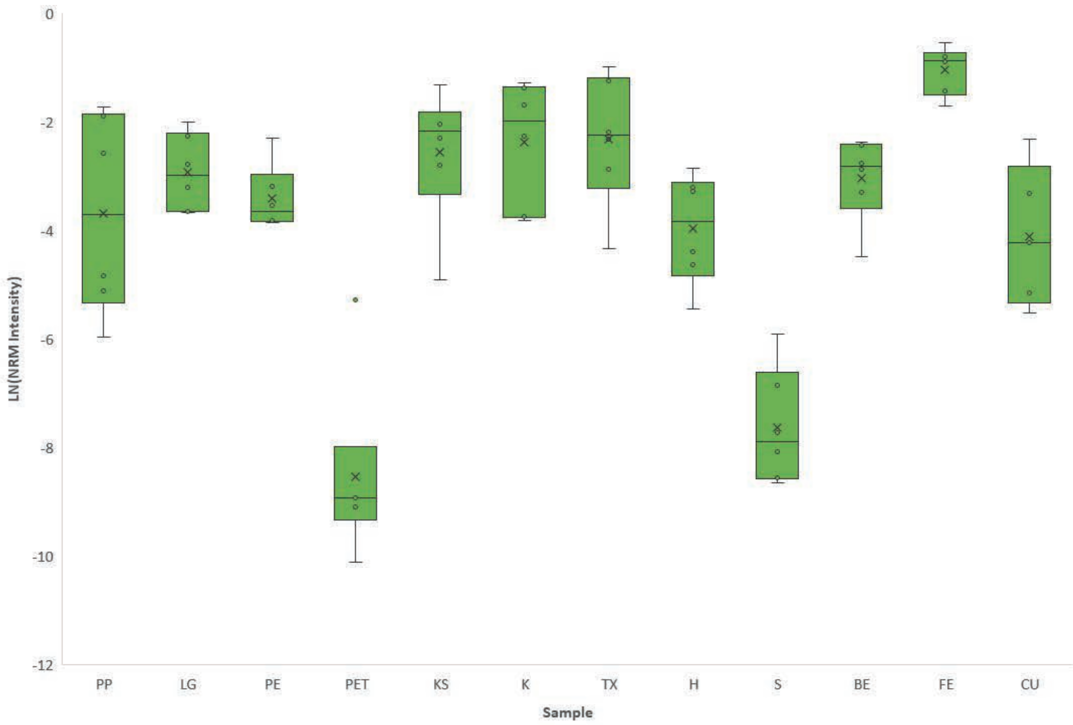


FIGURE 3: Natural remanent magnetization (in A/m) of reference samples.

within the cubes resulting in an overall compensation of remanence.

The VRM of individual reference samples is shown in Table 5. Paper shows a significant VRM, whereas textile samples are characterised by weak viscosity. In all other samples no VRM was observed. The VRM of the paper fraction, and subordinately the textile fraction, might be explained by metallic contaminations attached due to the recycling process.

In contrast to the assumption that remanent magnetization of different pieces levels out in a landfill, most waste samples show mainly remanent and only subordinately induced magnetization, e.g. for paper and paperboard the remanent magnetization is six times higher than the induced magnetization (Q = 6.26). The contribution of remanent magnetization to the total magnetization of non-metallic waste fractions can be explained by fine grained iron contaminations. As building sand was neither derived from ELFM, nor from a waste treatment plant, it is believed that no metal contamination is present in this fraction. However, it remains unclear why the building sand shows such strong remanent magnetization.

A comparison of the MS values of waste materials with those of fresh materials (Table 1) indicates that the MS of iron scrap ($9.31 \cdot 10^{-1}$) is five orders of magnitude lower than the MS for pure iron, two orders of magnitude lower than the MS for martensitic steel, but two orders of magnitude higher than the MS for austenitic steel (Schenck,

TABLE 5: Induced and remanent magnetization of reference samples.

Sample	Median Induced Magnetization [A/m]	Median Remanent Magnetization [A/m]	Median Q
K	1.42E-02	1.45E-01	6.26
LG	5.84E-03	5.16E-02	6.19
PE	3.04E-03	2.65E-02	10.75
PP	1.45E-03	4.23E-02	14.04
S	8.83E-05	4.05E-04	6.10
TX	3.27E-02	1.06E-01	3.18
BE	1.41E-03	6.03E-02	37.16
H	8.48E-04	2.50E-02	22.41
KS	2.20E-03	1.16E-01	36.49
PET	-4.66E-05	1.56E-04	-3.51
FE	3.71E+01	4.23E-01	0.01
CU	1.32E-02	2.55E-02	2.48

1996). Values for fresh and waste PE are in the same order of magnitude. The MS of the PET samples is one order of magnitude higher, and copper scrap and wood samples are two orders of magnitude higher than that of pure materials.

3.2 Magnetic Landfill Explorations

The magnetic anomaly map (Figure 4) shows several positive and negative anomalies in the order of 1000 nT. The anomalous areas might represent Fe enrichments. However, these could not be confirmed as no boreholes were drilled there.

The vertical gradient of the total intensity of the Earth´s magnetic field reveals strong positive anomalies in the east and strong negative anomalies in the north (Figure 5). These can be interpreted as polarised near-surface metal pieces. The positive and negative sign of the anomalies, respectively, mean a normal and reverse alignment of the remanent magnetic field inside the iron pieces. These results suggest there are near-surface iron pieces in borehole BR 2/8, which could not be verified by the drilling campaign as the uppermost sample was taken at a depth of 2 m. The high susceptibility of near surface iron pieces can be an additional explanation for the magnetic anomaly in the area of borehole BR 2/8.

Magnetic susceptibility data for the upper five centimeters (Explonarium sensor), indicates that the landfilled construction and demolition (C&D) waste contains partial enrichments of iron, whereas other areas at the surface are free of iron (Figure 6). This was confirmed by macroscopic observations, i.e. the presence of metal objects, at certain spots of the landfill surface. The corresponding small-scale magnetic anomalies especially occur in the area around bore hole BR 2/12 where also metal enrichments were found at the surface.

3.3 Waste Characterisation

Material composition of 32 samples taken at different locations and depths at the MSW landfill indicate an aver-

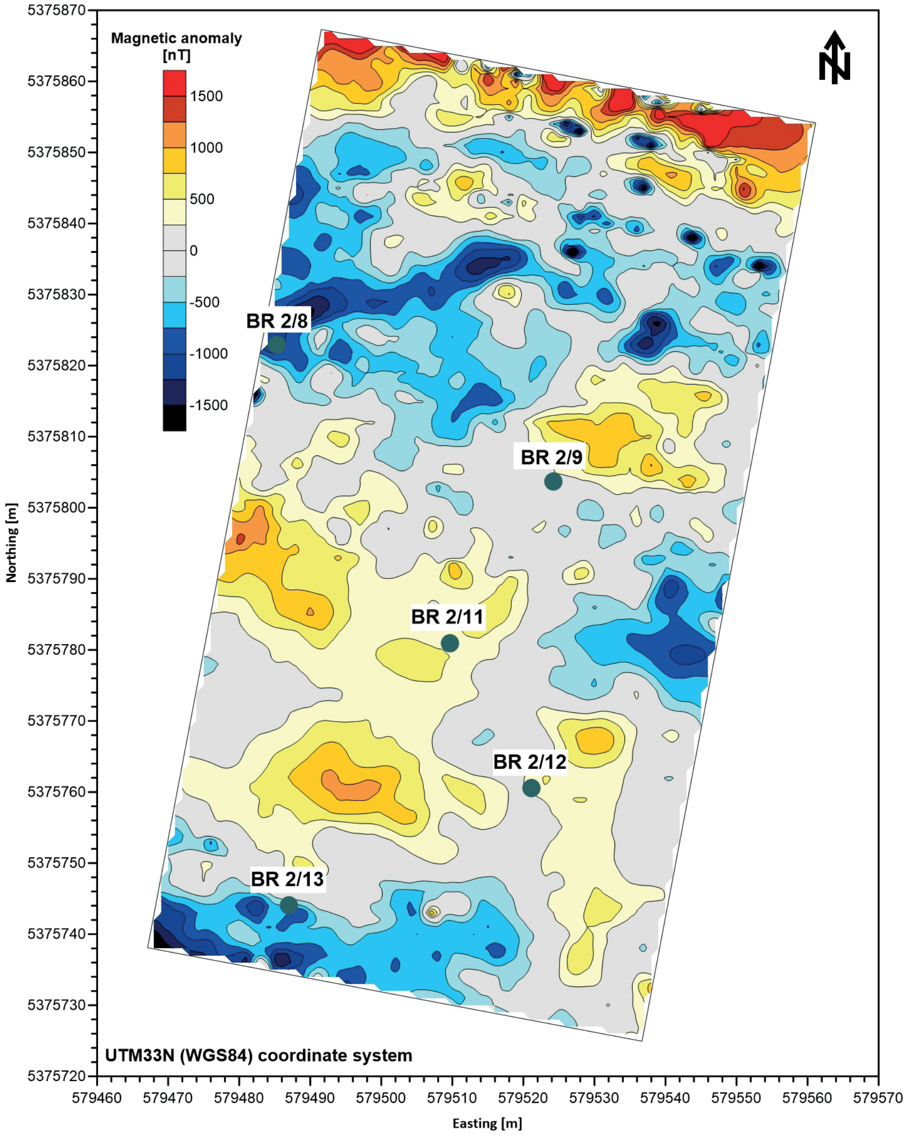


FIGURE 4: Magnetic anomalies at an Austrian landfill (lower sensor = 1 m above ground).

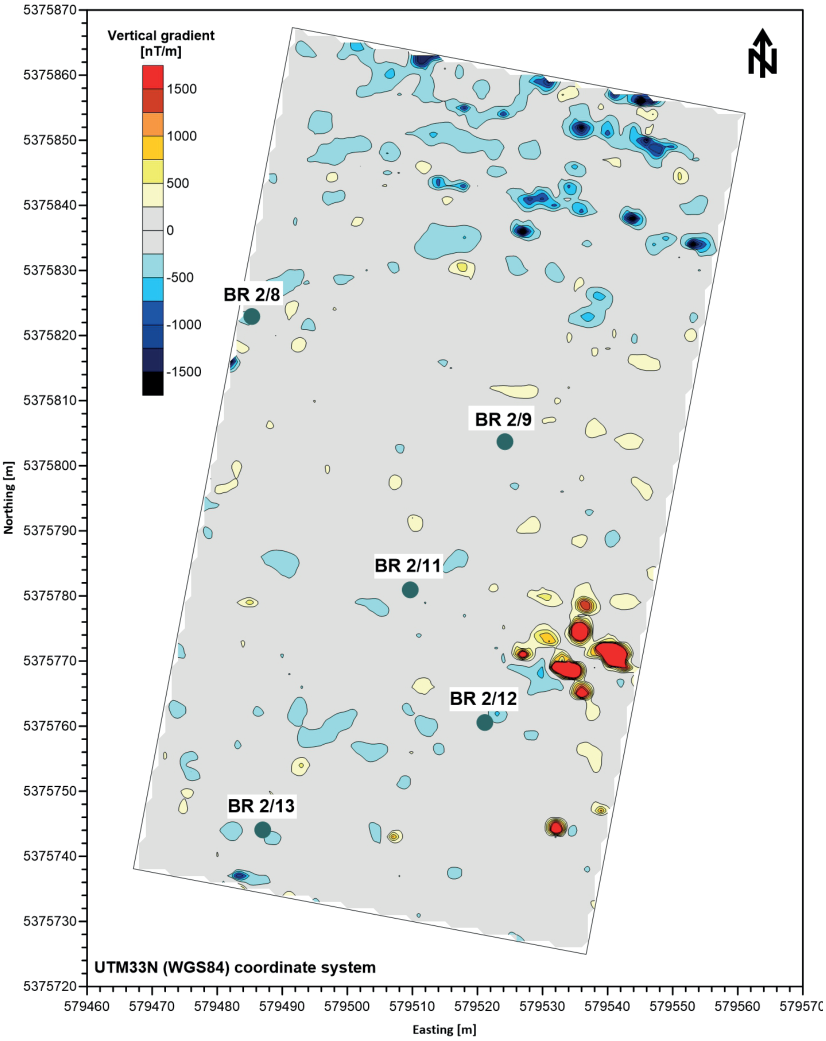


FIGURE 5: Vertical gradient of the total magnetic field at an Austrian landfill.

age composition (incl. water) of 42.2 ± 5.8 wt.% water, 26.2 ± 8.4 wt.% sorting residue, 10.7 ± 4.2 wt.% plastics, 5.8 ± 3.2 wt.% wood, 3.5 ± 2.6 wt.% textiles, 3.2 ± 1.2 wt.% inerts, 2.3 ± 1.4 wt.% iron, 2.2 ± 2.2 wt.% compounds, 2.0 ± 2.0 wt.% paper and paperboard, 0.6 ± 0.4 wt.% glass, 0.6 ± 0.4 wt.% nonferrous metals, 0.1 ± 0.1 wt.% problematic substances and 0.9 ± 0.7 wt.% others. No correlation between iron content and depth was found, although a metal enrichment in larger depths was expected due to worse waste separation in earlier times. Data for individual boreholes is shown in Figures 7a-c. According to the determined waste compositions, the bulk MS values for the individual waste samples are expected in the range of 0.01 to 0.05 SI.

3.4 Magnetic Modelling

Using MS values of the reference samples and the material composition of individual samples from the Austrian MSW landfill, a linear relationship between the iron content and the magnetic susceptibility value of samples was found. This relation predicts MS values for mixed MSW of 0.01 to 0.05 SI with iron contents between 1 and 5 wt.%. However, as the average MS value of iron might be higher for iron pieces in the landfill than that of the reference sample, the effect of different MS values for iron, i.e. 0.931 SI and 5 SI, has on the predicted iron content is demonstrated (Figure 8).

Subsequently, the measured total magnetic intensity along profile 1 was modelled using prismatic bodies with varying MS values (Figure 9). The model suggests that the MS of each body (0.06 to 0.11 SI) is significantly higher than the expected geologic background MS. Furthermore, areas of higher (≥ 0.10 SI) and lower susceptibility (≤ 0.08 SI) can be identified. These areas are assumed to correlate with higher and lower iron contents, respectively. The trend between the two drill cores along the modelled profile, i.e. BR 2/11 (modelled MS = 0.10 SI, average iron content 4.1 wt.% dry matter) and BR 2/9 (modelled MS = 0.075 SI, average iron content 3.9 wt.% dry matter), speculatively might support this assumption. In the southwest of the landfill we see an area with a MS close to zero, and just beside an area with increased MS. The different heights of the columns with respect to the landfill surface might contain information about the presence or absence of near-surface iron pieces.

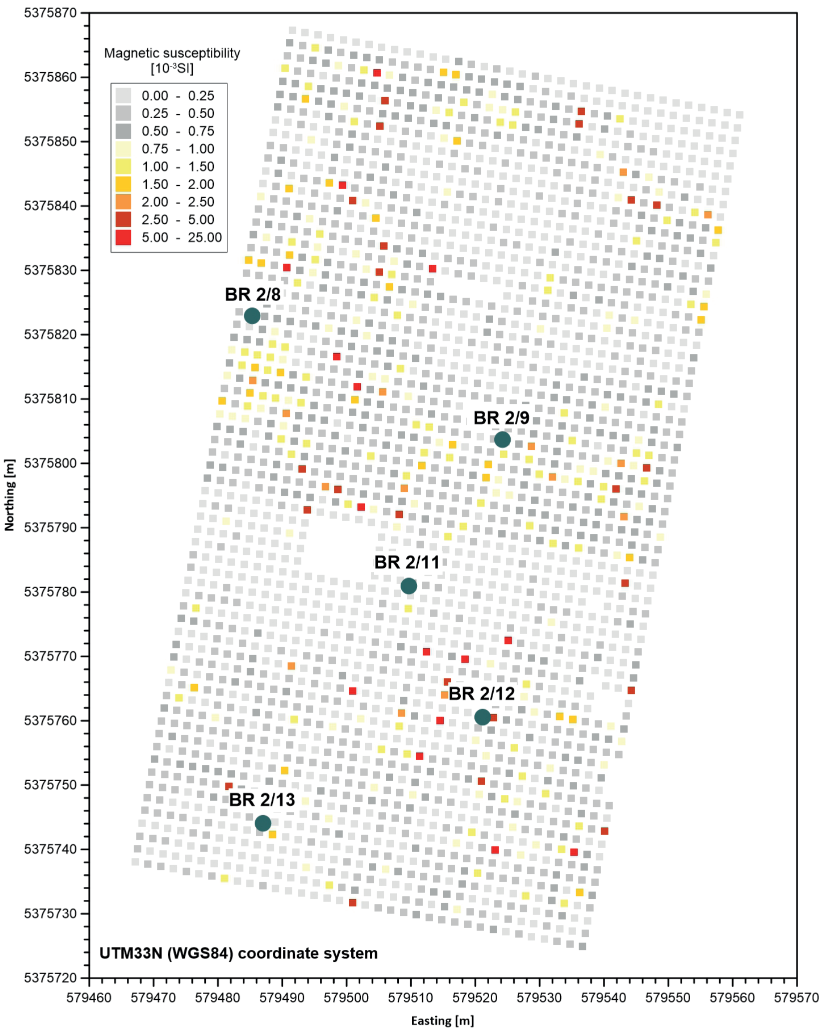


FIGURE 6: Magnetic surface susceptibility at an Austrian landfill.

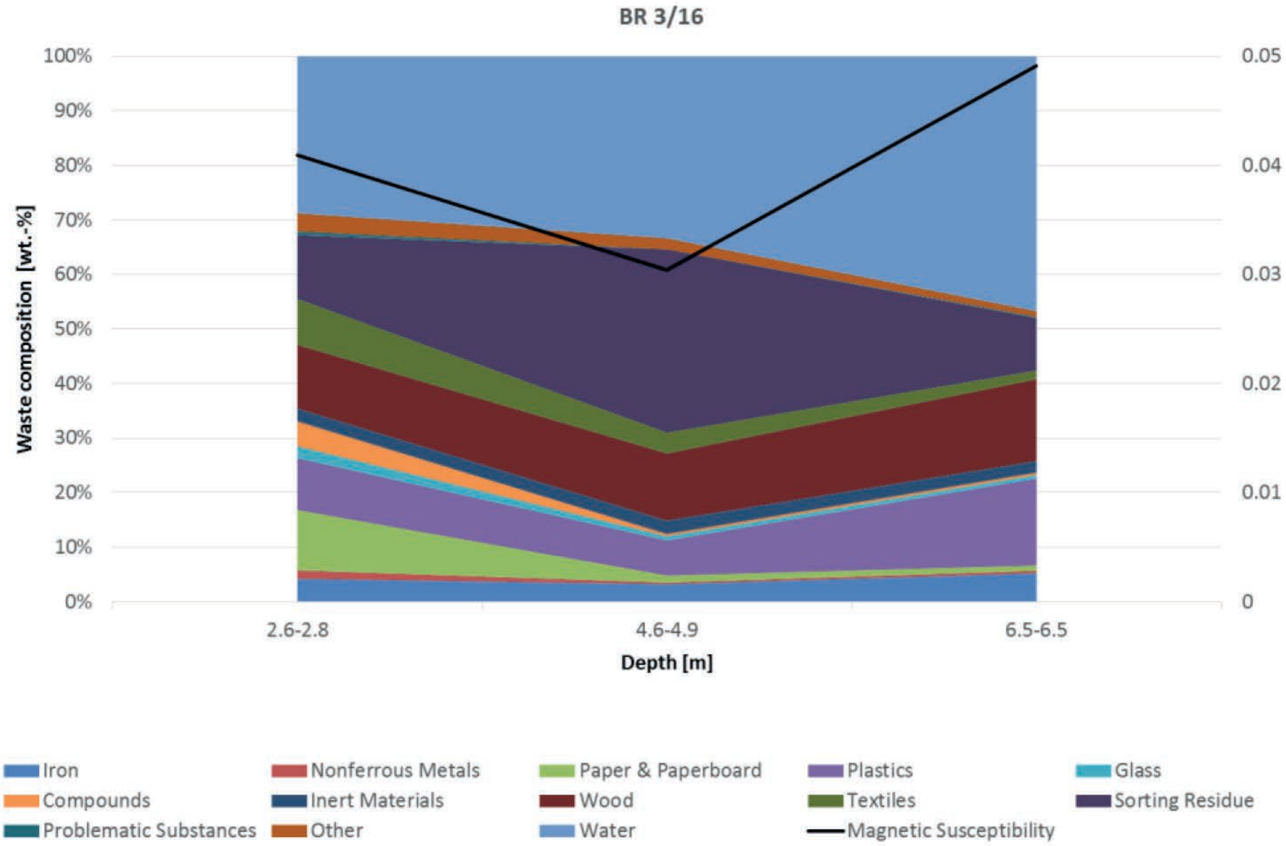
A comparison between the two approaches, i.e. prediction from waste composition, and inversion from magnetic anomalies, demonstrates that of MS estimation results reveals that MS values derived from magnetic surveying data (0.06-0.11 SI) are higher than those derived from laboratory measurements of reference samples (0.01-0.05 SI). This might be explained by differences in MS of individual iron alloys. Consequently, a multiplication of magnetic data and iron content was used to estimate the magnetic susceptibility of the landfilled iron scrap, and is in the range of 5 SI (Figure 10).

4. CONCLUSIONS

In this study we investigated two research questions, (i) for material recovery, to which degree do the MS values of defined materials differ from those corresponding to respective waste fractions, and (ii) can laboratory measurements of the MS of (mixtures of) individual waste fractions be related to the total field magnetic measurements on site, and can the combination of laboratory and field measurements predict the iron content of a landfill?.

In order to answer research question (i), we conducted MS analyses on reference samples, mainly produced by mechanical processing of MSW and compared them to literature values for virgin materials. To answer research question (ii), we measured the total magnetic intensity at an Austrian MSW landfill and inverted the data to obtain the MS of the buried waste. Afterwards, we took samples by drilling and manually sorted the samples. Then we calculated the MS of the obtained waste mixtures which would be expected from the MS values determined for the reference samples. The latter are representative of individual waste fractions. Finally, we compared the expected MS and the MS obtained from inversion of magnetic anomalous data.

Regarding research question (i), MS values of individual waste materials could be reproduced in repeated measurements, showing significant variations within each material fraction. As profound knowledge of the possible variation is crucial to the reliable interpretation of landfill magnetic anomalies, further research on magnetic waste properties is needed. The impact of metallic defilements, which remained even after removal by a magnet, was investigated by comparing the MS values of virgin materials and waste fractions. MS values for virgin and waste PE are in the same range, for almost all other fractions, the



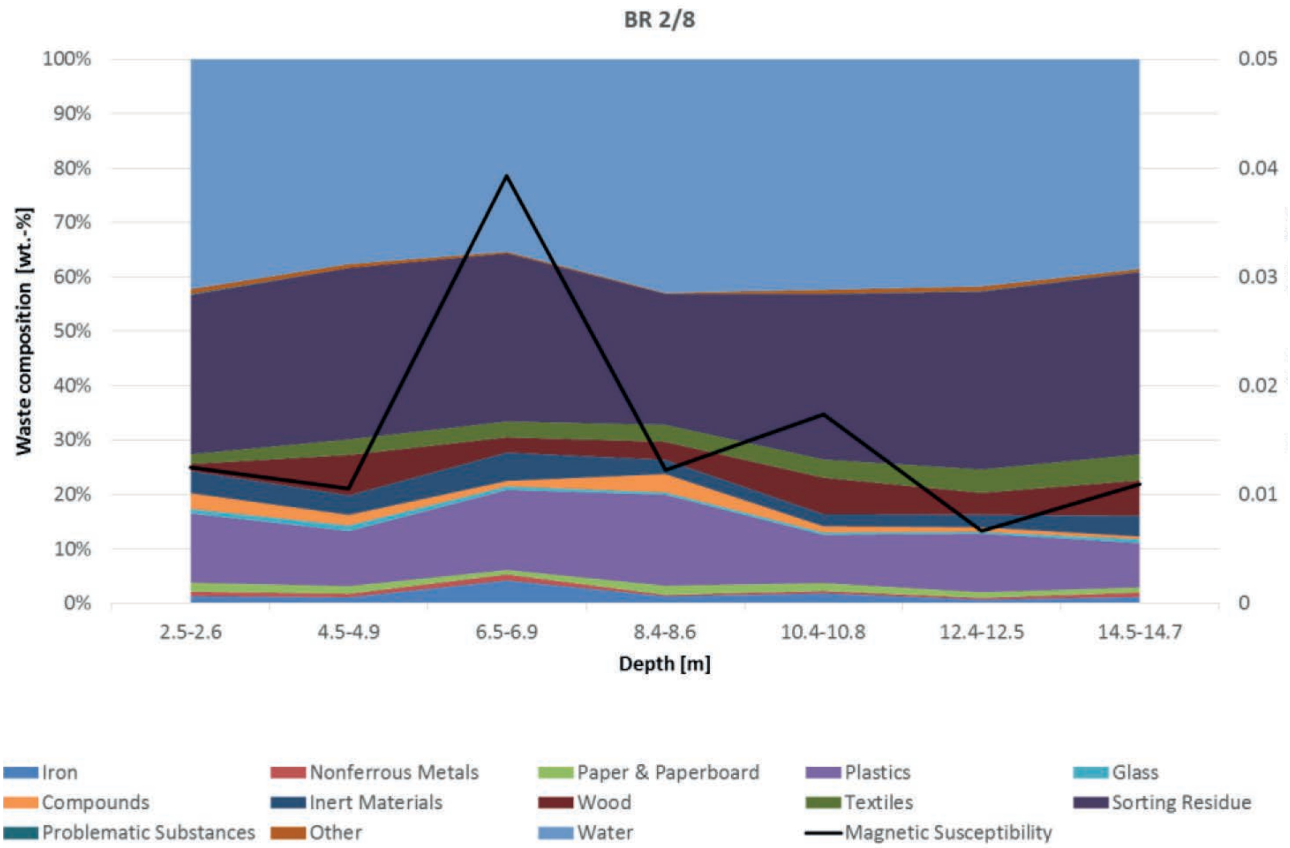


FIGURE 7a: Waste composition and predicted magnetic susceptibilities calculated from lab values for individual waste fractions at bore holes 3/16 and 2/8 at an Austrian landfill.

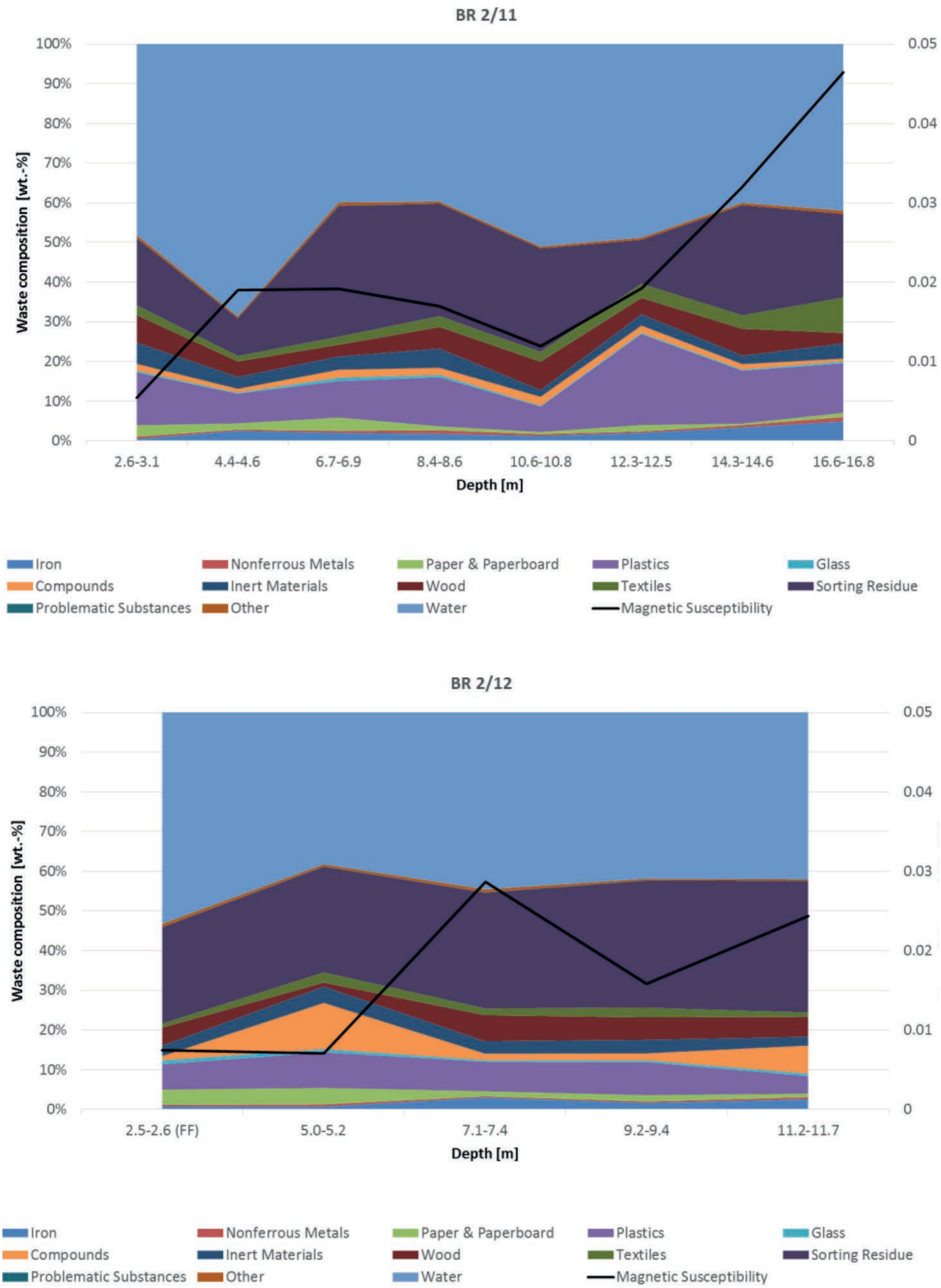
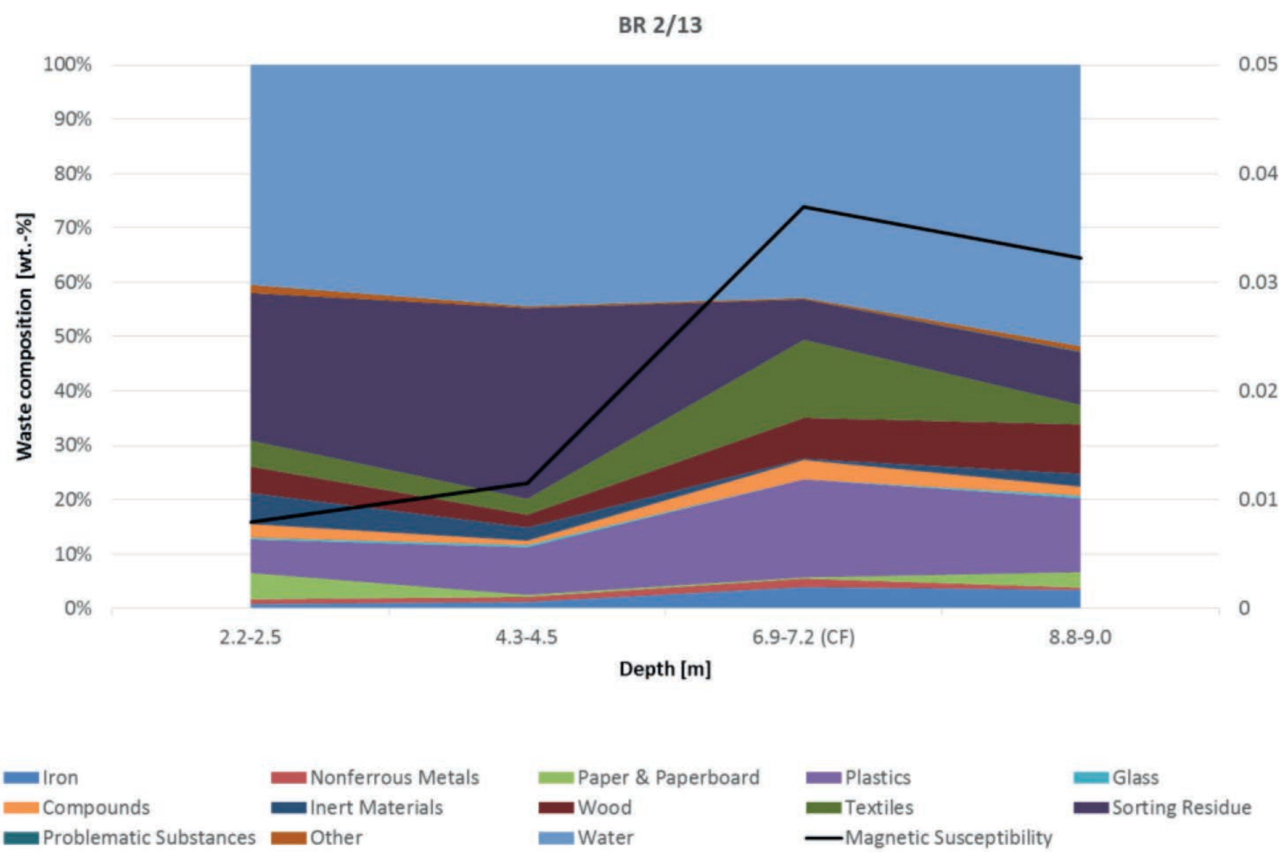


FIGURE 7b: Waste composition and predicted magnetic susceptibilities calculated from lab values for individual waste fractions at bore holes 2/11 and 2/12 at an Austrian landfill.



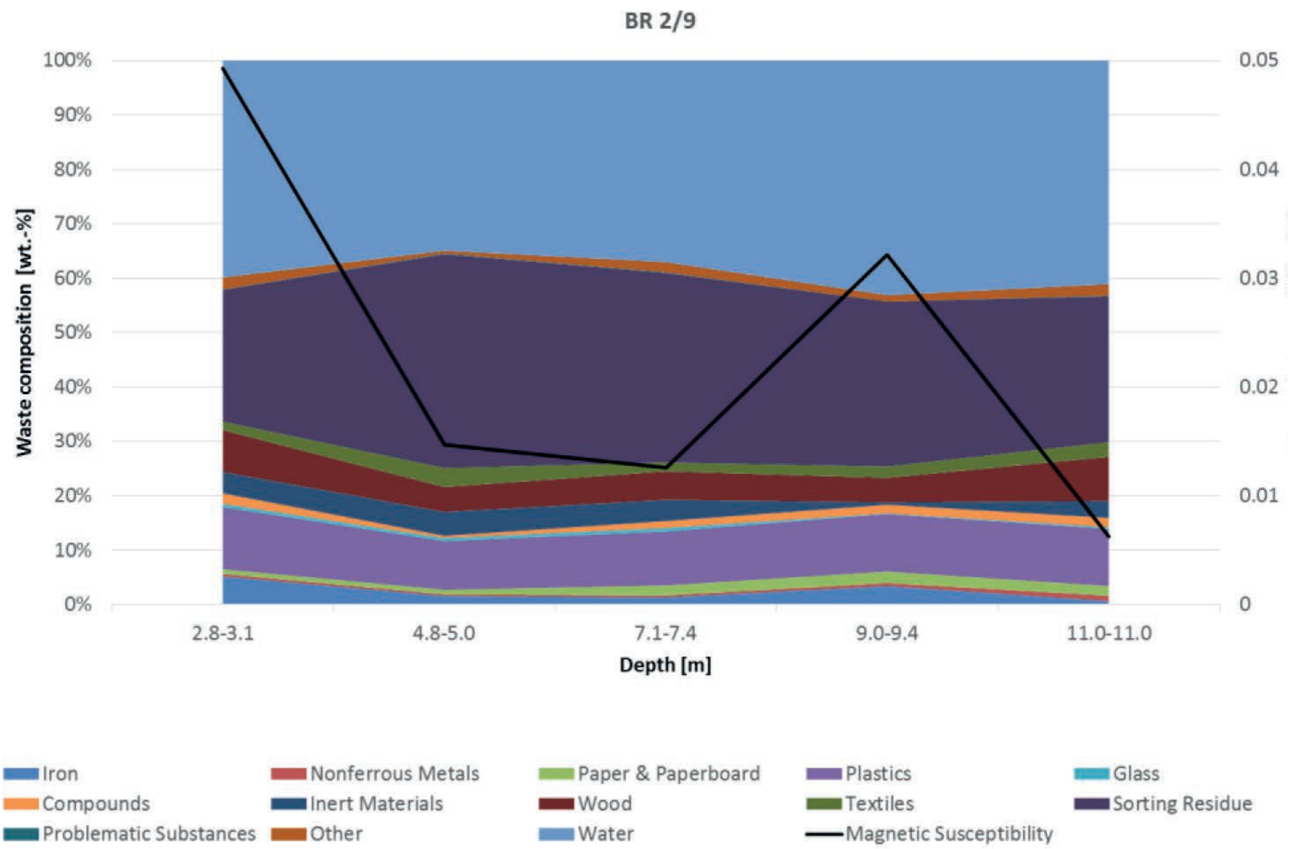


FIGURE 7c: Waste composition and predicted magnetic susceptibilities calculated from lab values for individual waste fractions at bore holes 2/13 and 2/9 at an Austrian landfill.

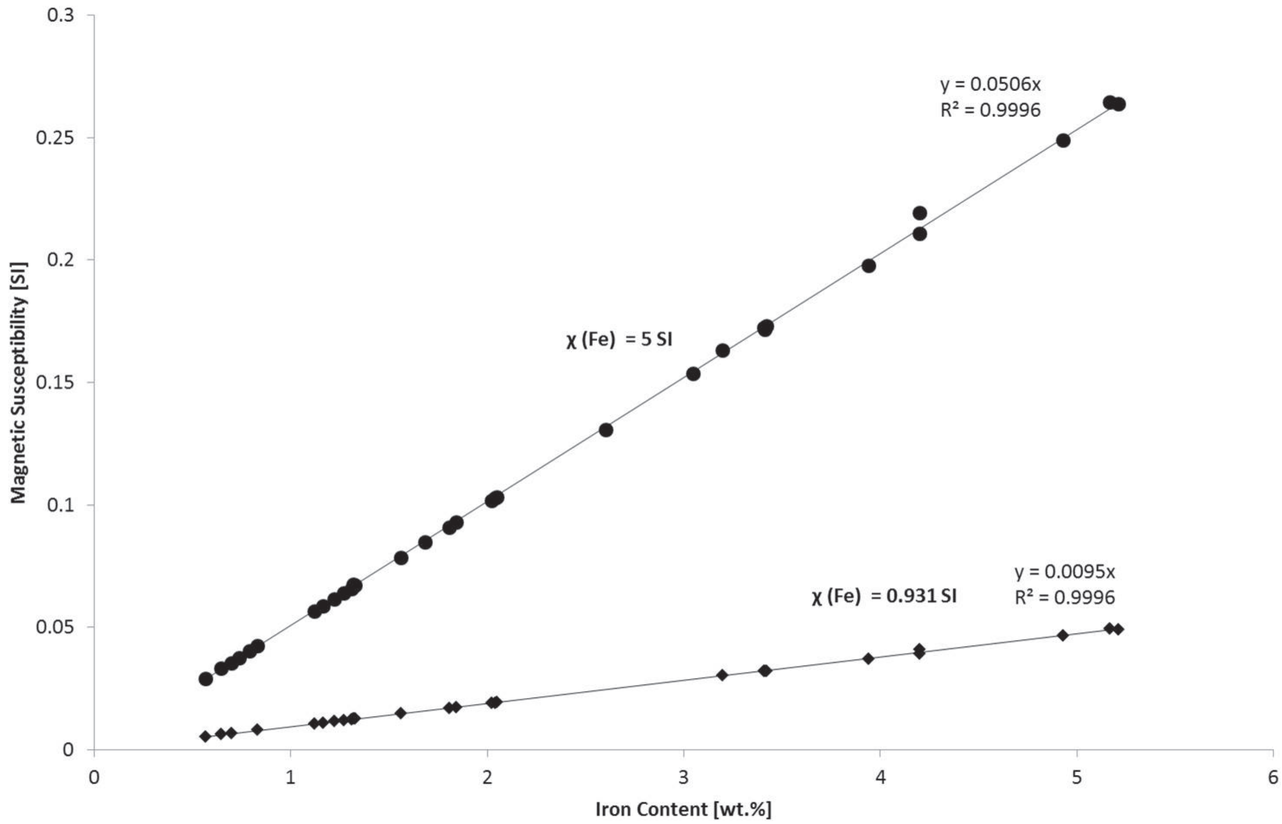


FIGURE 8: Predicted magnetic susceptibility of waste mixtures for x(Fe) = 0.931 SI (from lab measurements) and x(Fe) = 5 SI.

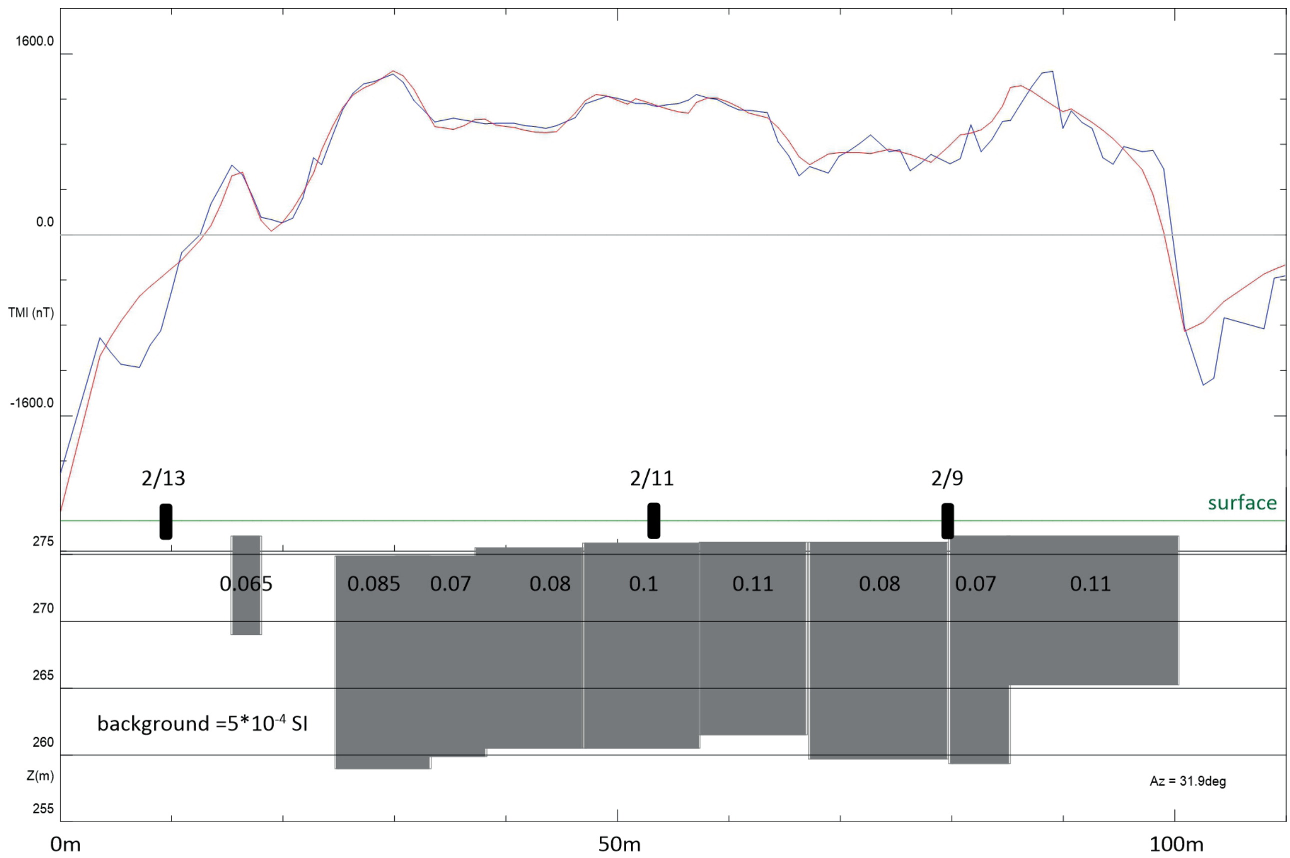


FIGURE 9: Magnetic susceptibility of the investigated MSW landfill, modelled from the total magnetic intensity.

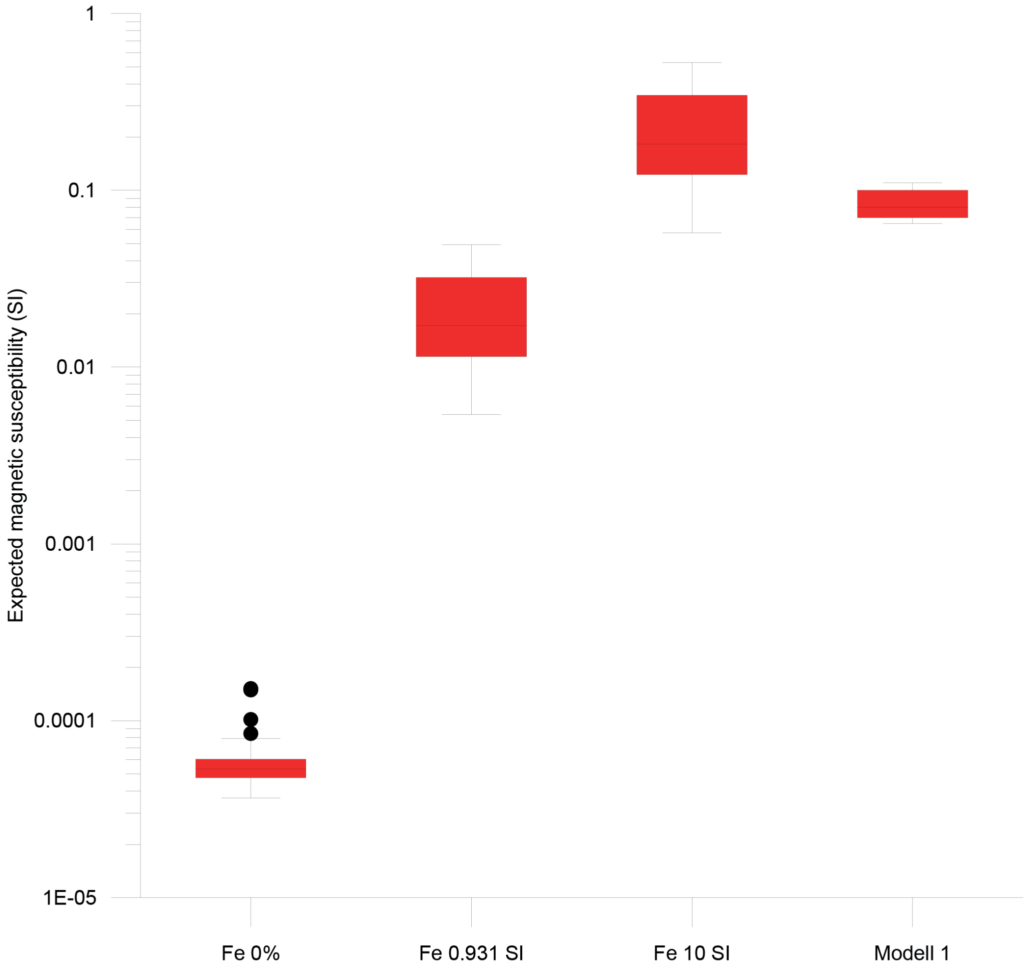


FIGURE 10: Expected magnetic susceptibility of a landfill without iron and with those amounts of iron obtained by manual sorting at the Austrian landfill for different magnetic susceptibilities of iron.

MS values for virgin materials and the respective waste fractions differed. For example, for PET MS values are one order, and for copper scrap and wood samples even two orders of magnitude above the MS values for virgin materials.

Regarding research question (ii) it was found that using MS values for the GPI and material composition of landfilled waste yields lower MS values than predicted by modelling magnetic survey data. In the MSW landfill case study, the comparison of the material composition of the samples from the two drill cores along the modelled profile with the bulk MS values from inverse modelling might suggest a rather speculative positive relation. BR 2/11 showed modelled MS of 0.10 SI and an average iron content of 4.1 wt.% dry matter, and BR 2/9 showed modelled MS of 0.075 SI, and an average iron content of 3.9 wt.% dry matter. However, this should be further supported by more extensive experiments.

In summary, it is not straightforward to establish a direct relationship between geophysical magnetic measurements and the iron content of landfilled waste. Modelling this relationship requires additional calibration data obtained from different types of geophysical measurements or prior knowledge on the waste composition.

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CASE STUDY ON ENHANCED LANDFILL MINING AT MONT-SAINT-GUIBERT LANDFILL IN BELGIUM: CHARACTERIZATION AND POTENTIAL OF FINE FRACTIONS

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ABSTRACT
Within the framework of the "EU Training Network for Resource Recovery through Enhanced Landfill Mining – NEW-MINE", around 374 Mg of waste were excavated from a landfill site in Mont-Saint-Guibert, Belgium. Parameters such as bulk density, water content, particle size distribution and material composition of the fine fractions (material <90 mm) were determined and analyzed. The present investigation has the main objective to document and disseminate the findings on the material characterization of the fine fractions obtained in this case study, since such information is of critical relevance for the design of an effective and efficient mechanical processing in (enhanced) landfill mining projects. Additionally, the potential of the fine fractions for material and energy recovery is discussed. The fine fractions in question were obtained through the implementation of a ballistic separation process with simultaneous screening directly after excavation, from which about 77 wt.% of the total amount of processed material in raw state corresponded to the fine fractions. These fractions presented an overall bulk density range of 720-1000 kg/m^3 in raw state and a total water content range of 25-30 wt.%. In dry state, the material showed a more uniform particle size distribution than in raw state, and results confirm that water content has a large impact on the particle size distribution of the fine fractions, as well as on the content of impurities in the material fractions "Combustibles", "Inert", "Total metals" and "Others" and on the presence of agglomerates. Results on the material composition in dry state reveal that amounts of 2.1-19.7 wt.% "Combustibles", 31.1-35.4 wt.% "Inert" and 0.6-1.8 wt.% "Total metals" could be recovered from the fine fractions 90-10 mm, while 37.8-55.6 wt.% "Fine fractions <10 mm" could be processed further in order to increase the recovery amounts of the previous material fractions and produce a substitute material for soil in construction applications.

1. INTRODUCTION

In general terms, (enhanced) landfill mining ((E)LFM) aims for the mitigation of pollution originating from landfill sites, reduction of aftercare and closure costs, land reclamation in urban areas, material recovery and, among many others, regaining landfill capacity (Hernández Parrodi et al., 2019; Hull et al., 2005; Jones et al., 2013; Jones et al., 2010; Jones and Tielemans, 2010; Krook et al., 2012). Particularly, the recovery of materials from the excavated material for recycling and production of alternative fuels has been included into the scope of many recent investigations in an attempt to raise the overall economic feasibility of (E)LFM projects (Hernández Parrodi et al., 2018a, 2018b). Nevertheless, mainly the coarse fractions (material with a particle size ≥10 mm to ≥60 mm, depending on the investigation) have been used for waste-to-material (WtM) and waste-to-energy (WtE) purposes in those investigations.

Concurrently, it has been identified in previous research that fine fractions (material with a particle size <60 mm to <10 mm, depending on the investigation), which represent about 40-80 wt.% of the total amount of excavated material, can be a relevant source for material and energy recovery (Hernández Parrodi et al., 2018a, 2018b). These fractions have been re-directed to the landfill, to a large extent, with poor or without any treatment at all (Bhatnagar et al., 2017; Münnich et al., 2013) and, therefore, the exploitation of their potential is of utmost relevance to increase the overall

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material and energy recovery in (E)LFM projects.

In order to assess the potential for material and energy recovery from the fine fractions of a particular landfill site, it is necessary to determine their material composition and main characteristics, such as bulk density, water content and particle size distribution (PSD). The present study has the main aim to document and disseminate the findings of the material characterization obtained through this investigation, as well as to discuss the potential of the examined fine fractions for WtM and WtE.

2. MATERIALS AND METHODS

2.1 Site description

The investigated landfill site in this case study is the "Centre d´enfouissement Technique de Mont-Saint-Guibert (CETeM)" located in the municipality of Mont-Saint-Guibert (MSG), Belgium (Figure 1a). Founded on a former sand quarry, which was exploited from 1937 to 1985, this landfill has served as one of the main disposal sites of municipal solid waste (MSW), non-hazardous industrial waste (IW) and construction & demolition waste (C&D) to the Belgian province of Walloon Brabant from 1958 (Bureau d´études greisch, 2002) to 2014. Nowadays, this site is going through its closure process, which is expected to conclude by the end of 2020.

MSG landfill has a total area of around 44.3 ha (D´Or, 2013), which is delimited in yellow in Figure 1b. The total area is divided into two main parts, namely the modern part and the old part of the site. The modern part has an area of about 26.5 ha (IGRETEC, 1994), which is delimited in blue in Figure 1b, while an estimated area of 14 ha corresponds to the old part, which is delimited in red in Figure 1b.

The present investigation was carried out at the old part of the landfill, which has an estimated depth between 30 m and 60 m and where at least 5.7 million m^3 of waste were disposed of between 1958 and 1985 (Gaël et al., 2017; IGRETEC, 1994). The old part of the landfill had a biogas collection system of vertical wells during the 1980s, which was removed after the stabilization of the waste, and has a functioning leachate collection system that consists of a lateral drain still in place.

Multi-sensor geophysical data from the subsurface of the old landfill´s body were gathered from three different areas during July-August 2017, from which one of them was selected for further geophysical exploration. Some decisive criteria for the selection of the exploration area were lateral and vertical extents, as well as the largest possible

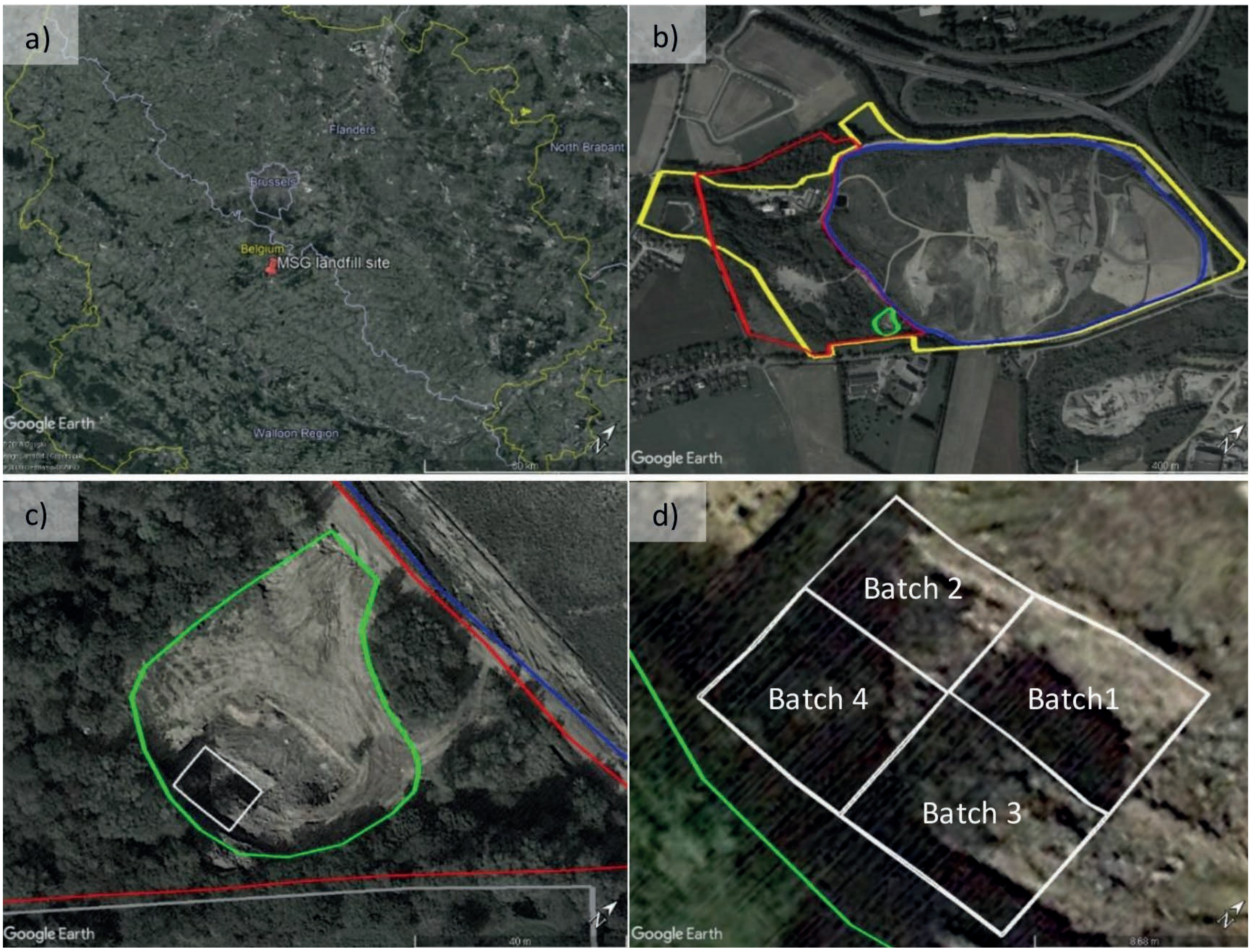

FIGURE 1: a) Location of landfill site in Belgium, b) MSG landfill site, c) exploration area and excavation zone and d) excavation zone with batches.

range of apparent surface electrical conductivity. The selected geophysical exploration area was around 2150 m^2, which is delimited in green in Figure 1b,c. Based on the results of the geophysical measurements, an excavation zone was selected within the selected exploration area due to its potential to validate measured geophysical properties of various unknown materials, since it was assumed a priori that a large variation range in electrical conductivity would coincide with a large variation in type of materials. The selected excavation zone had an area of about 130 m^2 and is delimited in white in Figure 1c. This zone is shown in greater detail in Figure 1d as well.

The concessioner for the operation of the landfill site is the company Renewi Belgium SA/NV, which has operated the site for over 25 years and provided most of the necessary equipment and manpower to realize this study.

2.2 Excavation works and material pre-processing

The material excavation and mechanical pre-processing were carried out during the last week of August and first week of September, 2017. The total excavated volume was divided into four sub-volumes, which are henceforth referred to as batches in this study (Figure 1d). The excavated volume was about 10 m long, 10 m wide and 4 m deep, while batches were around 5 m long and 5 m wide and had varying depths. The batches were classified in-situ according to their composition, since a clear waste stratification was identified visually within the excavated volume. As shown in Figure 2b, the first 2 m of the excavated volume consisted of a layer of mainly C&D, whereas below this layer the material corresponded to a layer of about 2 m of mostly MSW with daily cover layers of clay in between. Batch 1 and batch 2 were both excavated completely and, therefore, they were mainly composed of a mixture of MSW and C&D. For batch 3 and batch 4 the layers of C&D (upper layer) and MSW (bottom layer) were excavated individually in such a way that they predominantly consisted of C&D and MSW, respectively. Batches 1 to 4 were handled separately with the main purpose of studying their characteristics and mechanical processability as a mixture of C&D and MSW (batch 1 and batch 2) and as mostly C&D (batch 3) and MSW (batch 4), which allowed to obtain a more detailed insight into the performance of the mechanical processing with significant variations in the input material.

For the extraction of the waste material from the landfill, Hitachi 250LC and Liebherr 934 excavators with toothed digging type buckets of 2.5 m^3 were employed, while for the manipulation of the excavated material a Caterpillar 972K wheel loader equipped with a 3.5 m^3 bucket was utilized

FIGURE 2: a) Material excavation, b) excavation volume, c) ballistic separator and d) fine fractions.

(Figure 2a,c). During the excavation works the cover layer was removed first, which was mainly composed of clay and had an estimated average thickness of about 4 m (Figure 2b), in order to keep it separated from the landfilled waste. The cover layer material was not further processed. Subsequently, the excavated material was transported with Caterpillar 730C articulated trucks of 13.9 m³ (each) from the excavation zone to the mechanical pre-processing area (Figure 2a,c). The weight of the excavated material was determined with a weighing bridge of 50 Mg capacity and a resolution of 50 kg.

The fine fractions were obtained through the implementation of a two-step ballistic separation process (Figure 2d) with simultaneous screening (screening paddles). The ballistic separator (Stadler model STT 6000) sorted the input material into three-dimensional (3D), two-dimensional (2D) and under-screen fractions according to density, shape and particle size differences. In the ballistic separation process the screening paddles were firstly set to sieve the input material at 200 mm and subsequently at 90 mm. In this case study the fine fractions were defined as the excavated landfill material with a particle size <90 mm obtained from the second processing step of the ballistic separator. Figure 3 provides a schematic overview of the ballistic separation process, as well as of the outputs from which samples were taken. Additional details about the ballistic separator and the mechanical pre-processing and processing of the coarse fractions (≥90 mm) can be found in García López et al., 2019.

After being pre-processed with the ballistic separator, the fine fractions were loaded into containers of 25 m³ with a Manitou MRT 2150 telescopic handler equipped with a 2.5 m³ bucket to be stored until further processing.

2.3 Material samples and laboratory analysis

Due to the large amount of material excavated and pre-processed, representative single samples of the output fractions of the ballistic separator were taken for further analysis (Figure 4a). In this way, the quantity of material to be characterized and the amount of laboratory analysis can be reduced without compromising the reliability of the results. For this, the German guideline for procedures for physical, chemical and biological testing in connection with the recovery/disposal of waste (LAGA PN 98) was followed, which specifies the amount and size of the samples to be taken, according to the type, amount and particle size of the material to be sampled.

The sampling of the obtained fine fractions was done directly at the output chute of the ballistic separator. Two single samples of 10 l (each) were taken every 7.5 minutes in order to accumulate an amount of sixteen single samples per batch of processed material in 1 h. Eight composite samples (Figure 4c) of 20 l (each) were prepared on-site from every set of sixteen single samples, using the quartering method, as shown in Figure 4b.

The composite samples were used to determine the material composition and physical properties of the fine fractions at the raw materials laboratory and technical facility of the Department of Processing and Recycling (IAR) of the RWTH Aachen University. The material characterization was done in dry and raw states in order to allow direct comparison of the results. A circular vibratory box

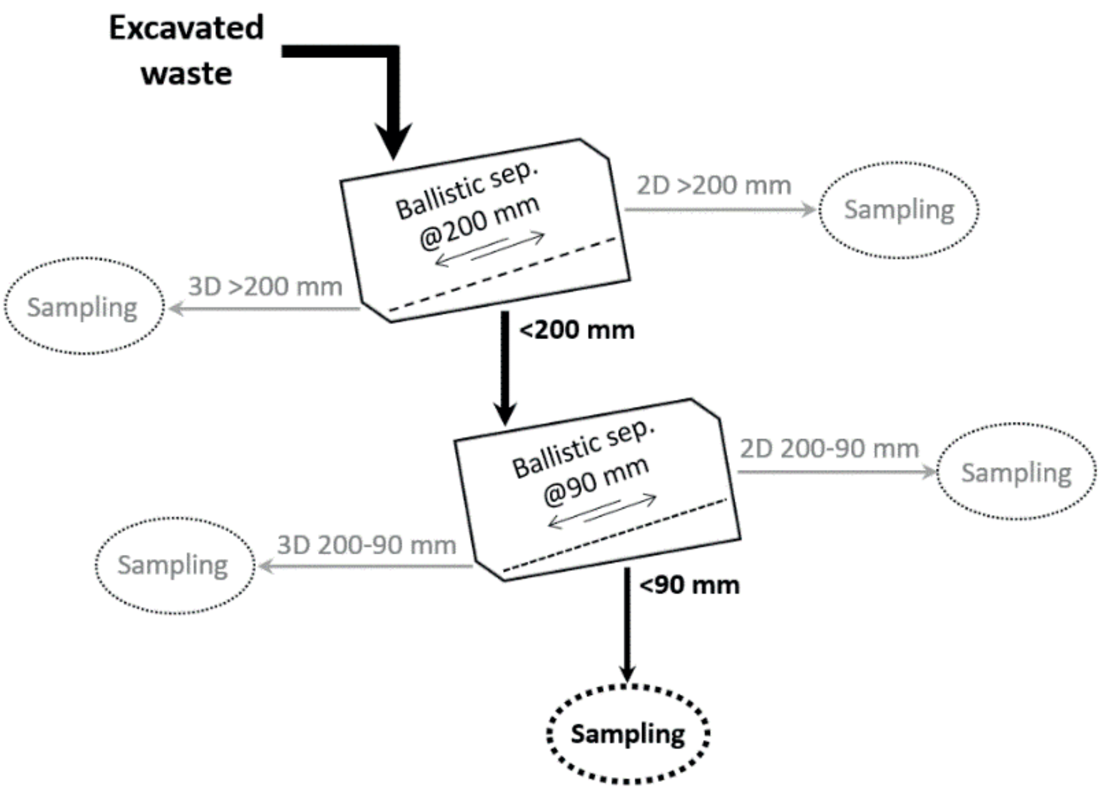

FIGURE 3: Pre-processing of excavated waste with ballistic separator.

FIGURE 4: a) Single samples, b) quartering method and c) composite samples.

sieve (Siebtechnik, 500 mm x 500 mm) with circular reticle sieves of 63 mm, 31.5 mm and 10 mm and a circular vibratory sieve tower (Siebtechnik, Ø 400 mm) with squared reticle sieves of 6.3 mm, 3.15 mm, 1.0 mm, 0.63 mm, 0.315 mm and 0.16 mm were used to determine the PSD, which was performed according to the norm DIN EN 15415-1:2011. Water and dry mass contents were determined according to the norm DIN EN 14346 (modified: drying at 75°C ± 5°C to prevent loss of volatile matter and melting of certain plastics). The drying of the samples was done in a Heraeus industrial drying oven. For the determination of the bulk density, the norm DIN SPEC CEN/TS 15401:2010 was followed. An industrial platform scale (Kern DS 150K1, resolution of 1.0 g) and a precision balance (Kern KB 2400-2N, resolution of 0.01 g) were used to determine the corresponding weights of all samples.

For the determination of the material composition, manual sorting was performed to particle size ranges ≥10 mm following the procedure described by the German guideline for uniform waste analysis in Saxony (Richtlinie zur einheitlichen Abfallanalytik in Sachsen). Grouped material fractions, such as "Combustibles" (i.e. "3D plastics", "2D plastics", "Textiles", "Wood", "Leather" and "Paper, Paperboard and Cardboard (PPC)"), "Inert" (i.e. "Bricks/Concrete/Stones", "Ceramics" and "Glass"), "Total metals" (i.e. "Fe metals" and "Non-Fe metals"), "Others" (i.e. bones, shells, sponges and unidentifiable materials), "Agglomerated fines <10 mm" (i.e. material <10 mm that stuck together due to the presence of moisture and formed material agglomerates) and "Fine fractions <10 mm" (i.e. soil, organic and weathered inert materials) were employed for the classification of the material composition.

3. RESULTS AND DISCUSSION

A total amount of about 374 Mg of waste was excavated and pre-processed at the MSG landfill, from which around 77 wt.% (raw state) corresponded to the fine fractions. The excavated pit (Figure 1d) had a total volume of about 425 m^3. As described in Section 2.2, the excavated volume was divided into four batches, which were about 140 m^3 (batch 1), 100 m^3 (batch 2), 120 m^3 (batch 3) and 65 m^3 (batch 4). Batch 1 and batch 2 consisted mainly of a mixture of MSW and C&D (Figure 5a,b), while batch 3 (Figure 5c) and batch 4 (Figure 5d) were mostly composed of C&D and MSW, respectively.

The total weights of the excavated batches in raw state were about 111 Mg for batch 1, 59 Mg for batch 2, 149 Mg for batch 3 and 55 Mg for batch 4, from which about 80 wt.%, 76 wt.%, 73 wt.% and 80 wt.% corresponded to

FIGURE 5: Fine fractions in raw state - a) batch 1, b) batch 2, c) batch 3 and d) batch 4.

the fine fractions, respectively. Table 1 summarizes the amounts and characteristics of the fine fractions obtained of each batch from the mechanical pre-processing.

The total mass and volume of the excavated material, as well as of the obtained fine fractions were determined by means of the articulated trucks and weighing bridge. Bulk density, water content, PSD and material composition were determined from the analysis of composite samples at IAR´s technical facility.

3.1 Bulk density and water content

Bulk density and water content are key parameters for the design of an effective and efficient mechanical processing of fine fractions, since the type of processing methods (i.e. processing approach) and size/number of processing units to be employed depend on them to a great extent. Hence, both bulk density and water content belong to the foremost parameters to be determined from fine fractions in (E)LFM. To this end, an amount of eight composite samples (n=8) was used to determine the bulk density in the raw state of each batch. The results were used to calculate different percentiles, as percentiles are a useful measure to identify the variation range in a set of values, while excluding a certain percentage of them that are less likely to occur according to pre-selected low and high limits. The 25th, 50th (median) and 75th percentiles were found to be the most useful for depicting the fluctuations of the bulk density. These results, together with the

TABLE 1: Amounts of excavated waste and fine fractions in raw state.

Batch	Material type	Amount of excavated waste [Mg]	Volume of excavated waste [m³]	Amount of fine fractions from excavated waste [wt.%]
1	MSW + C&D	111	140	80
2	MSW + C&D	59	100	76
3	Mostly C&D	149	120	73
4	Mostly MSW	52	65	80
	Total	374	425	77

maximum and minimum values, are shown as box-and-whisker plots in Figure 6a.

Results show that the bulk densities of batches 1 and 2 were quite similar, while those of batches 3 and 4 were very different from each other. Batch 3 presented a median bulk density around 1230 kg/m^3, which is comparable to that of C&D, with a very narrow variation between 25th and 75th percentiles accounting for the homogeneity of the material. Batch 4 had a median bulk density of about 630 kg/m^3, which corresponds to that of MSW, with a wider variation between 25th and 75th percentiles accounting for the heterogeneous nature of MSW. The determined bulk densities of batches 1 and 2 were located in between those of batches 3 and 4 with medians around 850 kg/m^3 (batch 1) and 810 kg/m^3 (batch 2) and a stronger overall variation between 25th and 75th percentiles, which logically correlates to a mixture of both C&D and MSW. In general, the bulk density of the fine fractions (all batches) varied between 720 kg/m^3 and 1000 kg/m^3 with a median of about 810 kg/m^3.

Regarding water content, four composite samples (n=4) were used to determine the moisture content of each batch, since the remaining amount of composite samples were used to determine the material composition in raw state. Analogously to bulk density, the 25th, 50th and 75th percentiles, and the maximum and minimum values of the water content were used to plot the corresponding box-and-whisker diagrams, which are displayed in Figure 6b.

Batches 1 and 2 showed very similar results with regard to water content, both with medians of about 27 wt.% and very slight variations between 25th and 75th percentiles. On the contrary, batches 3 and 4 presented water content medians of around 18 wt.% and 37 wt.%, respectively, with relatively larger variations between 25th and 75th percentiles. Altogether, the water content of the fine fractions had a median of about 27 wt.% and fluctuated between 25 wt.% and 30 wt.%. Thus, an overall bulk density range of 720-1000 kg/m^3 and a total water content range of 25-30 wt.% can be used to describe the fine fractions of MSG landfill.

Additionally, four (n=4) and eight (n=8) composite samples were used to determine the variation of bulk density with particle size in raw and dry states, respectively. This information is plotted in Figure 7, in which it can be observed that bulk density had slight variations between particle sizes of 1 mm and 6 mm, but strong ones between 6 mm and 90 mm in both states. The curves for raw state (shown in blue in Figure 7) begin at 1 mm because in this state particle sizes mostly <1 mm tend to adhere to each other and form agglomerates with diameters above 1 mm. From these results it can be said that, practically, bulk density decreased as particle size increased for both raw and dry states. Nonetheless, a decrease in bulk density was observed below 0.16 mm in dry state, probably because most of the remaining materials with high densities, such as ceramics, stones, glass and metals, were found above that particle size range.

The comparison of the medians of the raw data between raw and dry states, for several particle size ranges, reveals that bulk density decreased about 12% in the particle size range between 90-63 mm, 30% in 63-31.5 mm, 33% in 31.5-10 mm, 21% in 10-6.3 mm, 23% in 6.3-3.15 mm and 16% in 3.15-1 mm after drying with respect to raw state. The overall median of the reduction in bulk density of the fine fractions after drying was around 21%, which can be very useful information for the design of the mechanical processing.

Furthermore, four composite samples (n=4) from each of the batches 1, 3 and 4 after manual characterization were used to determine the water content per type of material, for three particle size ranges (90-63 mm, 63-31.5 mm

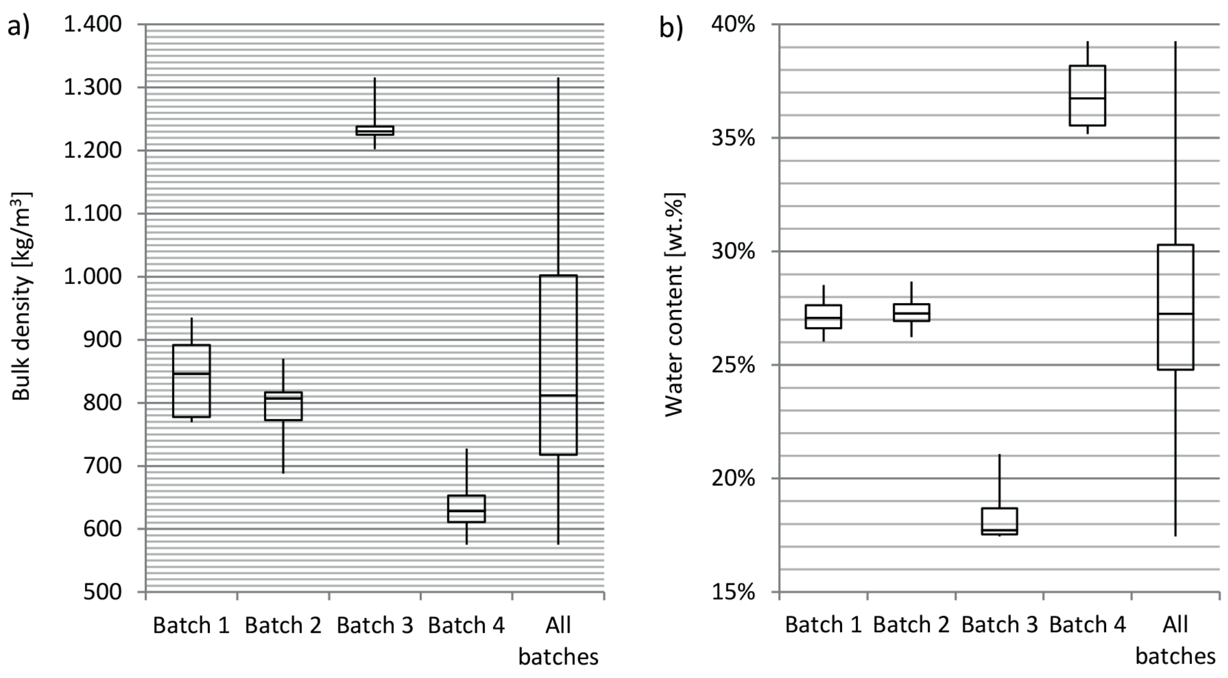

FIGURE 6: a) Bulk density in raw state and b) water content of the fine fractions.

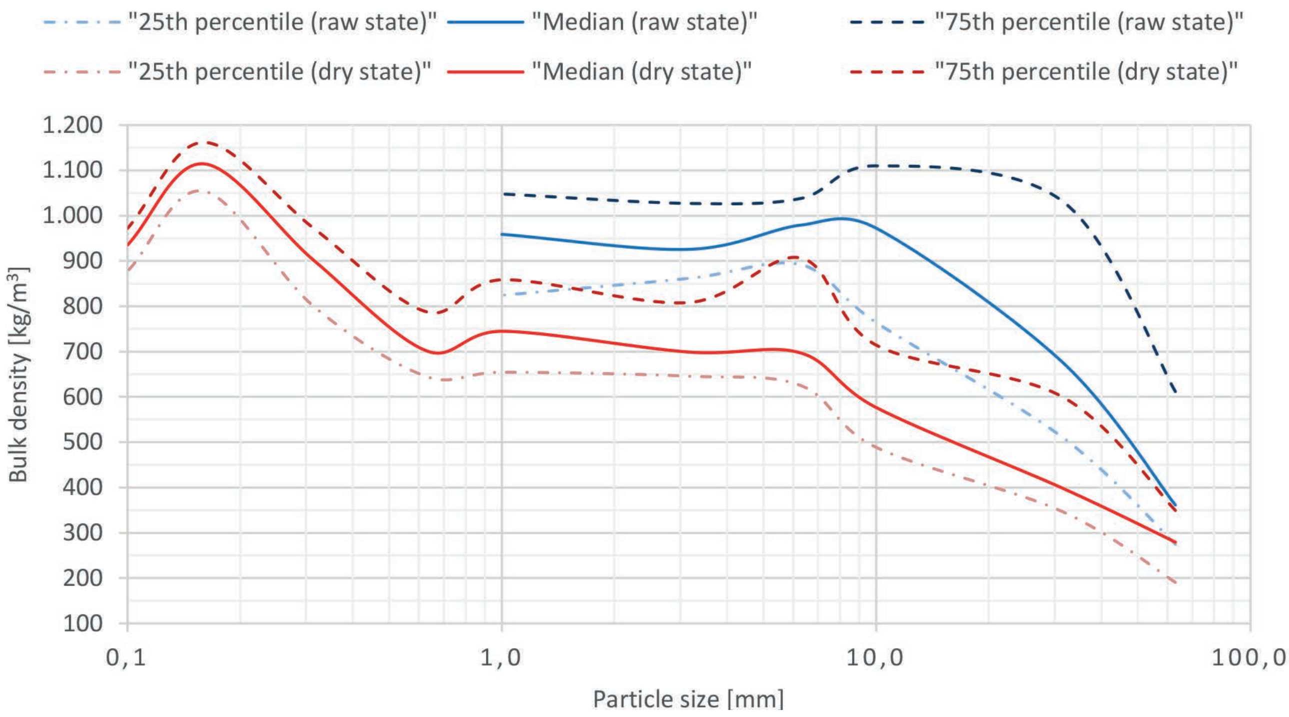

FIGURE 7: Variation of bulk density with particle size in the fine fractions in raw (blue) and dry (red) states.

and 31.5-10 mm), in order to study the water distribution among them. This information is shown in Table 2.

The materials presented in Table 2 can be classified into those with high, medium and low water contents. "PPC" and "Wood" showed the highest values, while "Bricks/Concrete/stones", "Ceramics", "Glass", "Fe metals" and "Non-Fe metals" presented the lowest values. Despite being among the materials with the lowest water content, "Ceramics", "Glass", "Fe metals" and "Non-Fe metals" show considerable amounts of water, which can be explained by the presence of impurities. In this case study impurities are regarded as fine particles (mainly <1 mm) of organic and inorganic matter that attach to the surface of coarser particles (also known as surface defilements). Impurities are

TABLE 2: Water content per material type of the fine fractions 90-10 mm.

Material type	Particle size range								
	90-63 mm			63-31.5 mm			31.5-10 mm		
	25th percentile	Median	75th percentile	25th percentile	Median	75th percentile	25th percentile	Median	75th percentile
3D plastics	16.2	19.9	25.2	20.3	22.5	27.3	18.3	20.5	24.2
2D plastics	18.4	26.4	34.4	25.8	31.4	33.2	24.4	27.8	32.7
Leather	N.A.	N.A.	N.A.	11.5	20.3	29.0	8.1	12.6	16.2
PPC	55.1	60.4	63.4	54.6	57.8	59.3	45.9	50.8	53.9
Textiles	26.9	32.9	34.6	22.3	33.7	39.1	22.6	26.3	30.6
Wood	57.8	60.1	60.7	53.1	56.1	59.7	43.7	47.2	56.0
Bricks/Concrete/ Stones	8.4	12.3	12.6	12.7	13.5	14.2	14.4	15.4	16.4
Ceramics	4.5	5.8	7.0	3.0	4.6	6.3	2.2	6.2	13.2
Glass	0.0	0.0	0.0	0.9	2.1	3.1	3.3	3.7	5.0
Fe metals	0.2	0.3	0.7	3.2	5.3	8.9	3.1	8.8	13.7
Non-Fe metals	0.5	1.0	1.4	1.5	5.8	8.6	2.5	10.0	15.5
Others	43.9	45.9	48.0	17.8	21.5	37.3	19.8	23.7	31.0
Agglomerated fines <10 mm	19.5	27.8	29.8	25.7	32.6	39.1	21.7	29.1	38.5

Notes: Figures are given in wt.%. Materials for which no sample was obtained are denoted as not available (N.A.).

associated with an increase of water content due to their capability to absorb and adsorb water. “3D plastics” and “2D plastics”, “Leather”, “Textiles”, as well as “Others” and “Agglomerated fines <10 mm”, belong to the materials with medium water contents. Similarly to “Ceramics”, “Glass”, “Fe metals” and “Non-Fe metals”, the moderate water contents of “3D plastics” and “2D plastics” are most likely to be due to the presence of impurities. This type of information results very useful during the design of the mechanical processing, as the effectivity of density separation methods can be greatly influenced by by the water content.

Moreover, no conclusive trend was identified regarding water content and particle size range, except for “Glass”, “Fe metals” and “No-Fe metals”, whose water contents increased as the particle size range decreased. The latter can be explained by the fact that the impurities vs. material type mass-ratio increases with the reduction in particle size of the material type.

After sieving in raw state, four composite samples (n=4) from each of the batches 2, 3 and 4 were employed for the determination of the water content per particle size range, which allowed to identify if some tendency of water being retained by a certain particle size range was to be found. This information was correlated with the mass distribution throughout the particle size ranges in question, so that its influence in the water distribution could be studied as well. This correlation is depicted in Figure 8.

Figure 8 shows that the water content was more or less evenly distributed throughout the fine fractions, as well as that most of the material and water were concentrated in the particle size range 31.5-10 mm. It is important to emphasize that the presence of water, up to a certain extent, promoted the formation of agglomerates and increased the amount of surface defilements and, therefore, a relevant amount of small particle sized material (mainly <1 mm) was retained in bigger particle size ranges when the fine fractions were in raw state (i.e. in the presence of moisture).

3.2 PSD analysis

The PSD analysis determines the amount of material in a certain particle size/particle size range with respect to the total amount of material. Among bulk density and water content, PSD is of critical importance for the design of an adequate processing approach of the fine fractions, as it gives information about the distribution of the material throughput in particle size classification steps of the mechanical processing, which are required for an effective and efficient mechanical processing. Hence, the PSD of the fine fractions was determined for both raw and dry states, in which four composite samples (n=4) from each batch were analyzed for each state. The obtained data from all batches was studied as a whole to produce and analyze the overall PSD curves of the fine fractions in both raw and dry states. This information is presented (Figure 9) using the 25th, 50th and 75th percentiles in order to show the median and variation range of the PSD, as well as to allow direct comparison between both states.

As displayed in Figure 9, the material in dry state presented a more uniform PSD than in raw state, since, as already mentioned, in moist conditions (raw state) fine particles, mostly <1 mm, tend to form agglomerates and adhere to coarser particles as surface defilements. Taking the curve for dry state as a reference for the real PSD of the fine fractions, it can be said that water content substantially influences the PSD of the material in a direct way and,

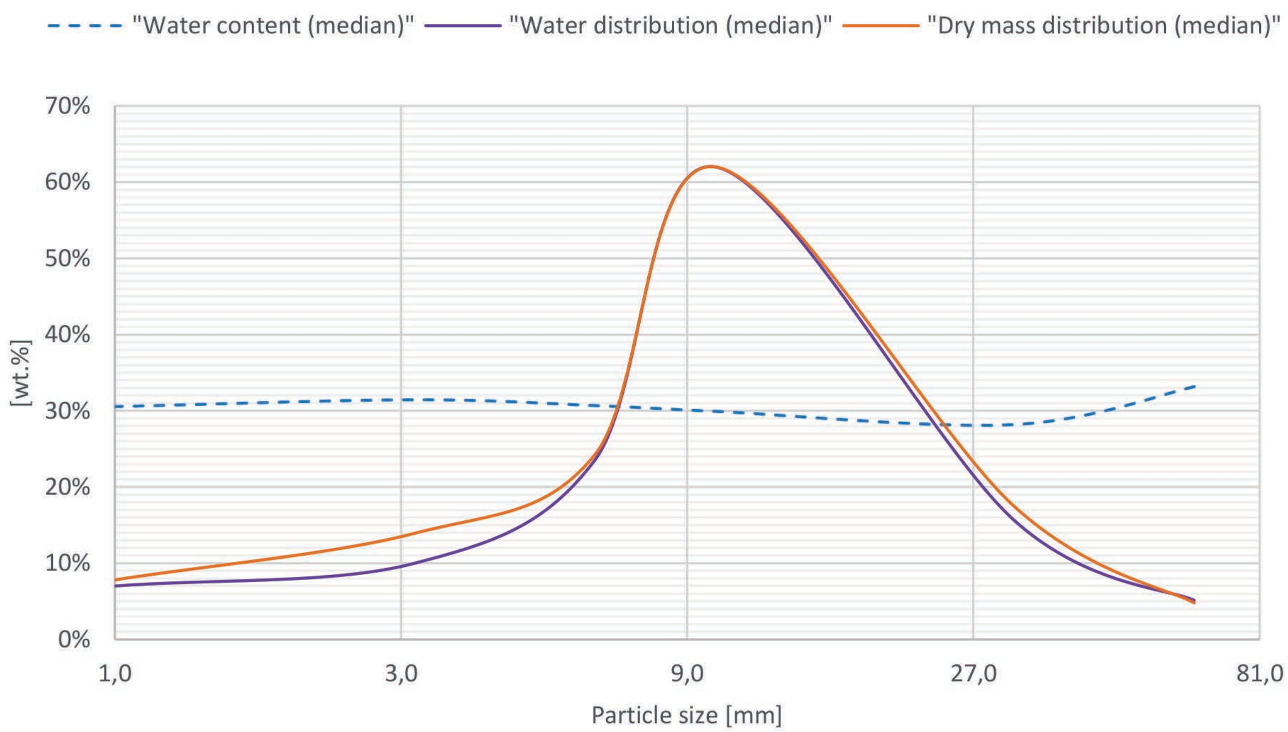

FIGURE 8: Water content, water and dry mass distribution in the fine fractions.

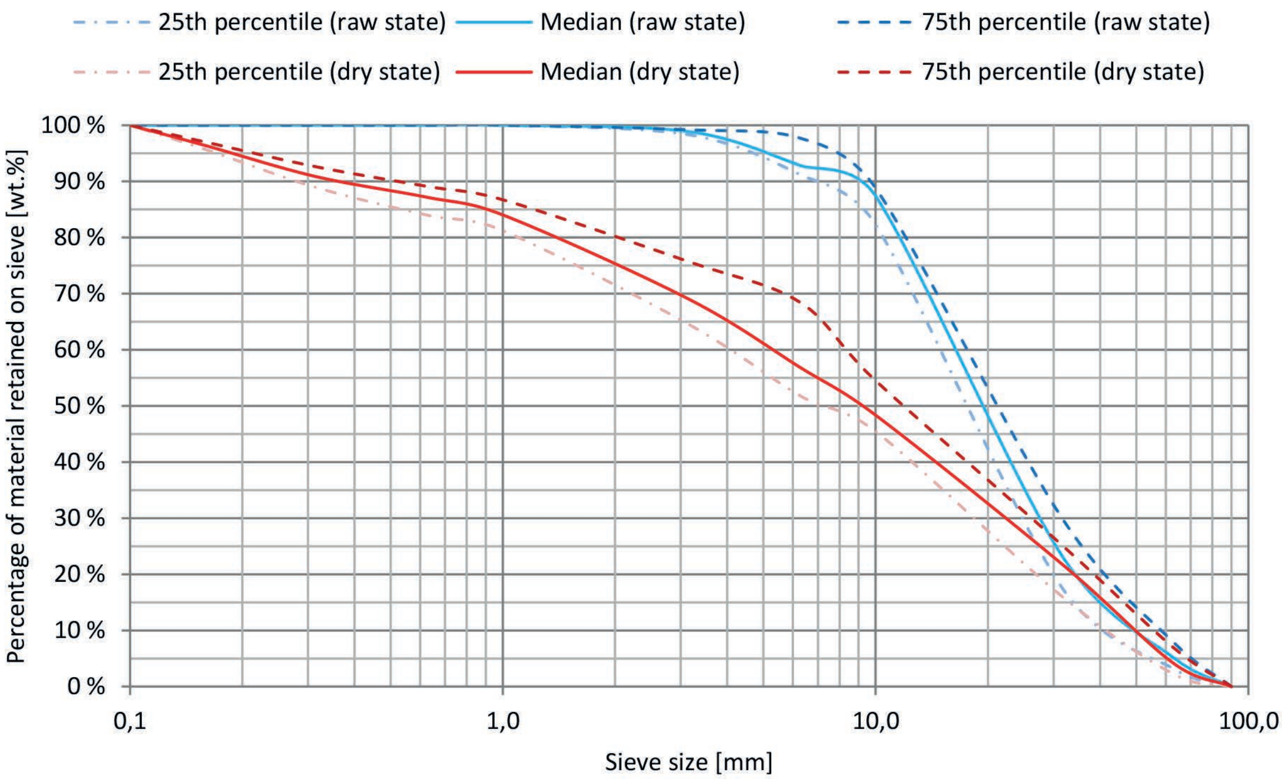

FIGURE 9: PSD of the fine fractions in raw (blue) and dry (red) states.

thus, the performance of sieving steps throughout the mechanical processing as well.

Moreover, about 80-90 wt.% of the fine fractions in raw state and around 45-55 wt.% in dry state were retained on the 10 mm sieve. On the contrary, fine fractions presented slighter differences between raw and dry states, regarding sieving performance, on the 31.5 mm sieve, in which 18-30 wt.% was retained in raw state and 16-25 wt.% in dry state. Therefore, it can be concluded that the PSD analysis of the fine fractions in raw and dry states can be useful to identify the required number and optimal cut-off diameter size of the sieving steps of the mechanical processing. Additionally, the PSD analysis can be used to identify the optimal moisture content to process the fine fractions in a dry mechanical processing in order to minimize dust generation and material loss, while maintaining a high sieving efficiency and without the need of complete drying. Besides, the PSD analysis can be employed to pinpoint the particle size from which the material might require a drying step or wet processing.

For example, in this case an initial sieving down to around 30 mm could be performed to the fine fractions directly after the ballistic separation process, without the necessity of any drying step. Nevertheless, the impact of processing the excavated material in raw state on the quality (amount of impurities and surface defilements) of the fractions to be subsequently recovered from both coarse and fine fractions needs to be taken into account for this as well. Subsequently, the moisture content of the material <30 mm could be either reduced or increased, according to the succeeding mechanical processing method (i.e. dry or wet), before applying an additional sieving at about 10 mm. The moisture adjustment would serve the purpose of increasing the sieving efficiency, reducing the amount of material agglomerates and surface defilements and, hence, improving the performance of the following mechanical processing.

3.3 Material composition

Besides bulk density, water content and PSD, the material composition of the fine fractions is of decisive relevance for the design of an appropriate mechanical processing approach, as well as for the selection of the WtM and WtE strategies to follow. The types of materials that could be recovered from the fine fractions are identified by the determination of the material composition and, in combination with PSD analysis, it provides information regarding the amount and location, in terms of particle size range, of the materials to be recovered, as well as information required for selecting an adequate processing method. Thus, the material compositions of batches 1, 3 and 4 were determined in both raw and dry conditions from a total of four composite samples (n=4) for each batch and state. Photographs of the material types that constitute the grouped material fractions (i.e. “Combustibles”, “Inert”, “Total metals”, “Others” and “Agglomerated fines <10 mm”) defined in Section 2.3 are displayed in Figure 10. Due to strong similarities between batches 1 and 2 regarding visual material composition, bulk density, water content and PSD, the composition of batch 1 has been assumed valid for batch 2 as well. The composite samples of batch 2 were used to obtain additional information, such as the variation of bulk density and water content according to particle size, which modified the initial conditions of the material and, thus, did not allow obtaining reliable results if manual sorting were to be performed a posteriori. The data obtained from the material characterization of batches 1, 3 and 4 was, hence, used to calculate the median material composition of the fine fractions as a whole.

In order to allow direct comparison between the results

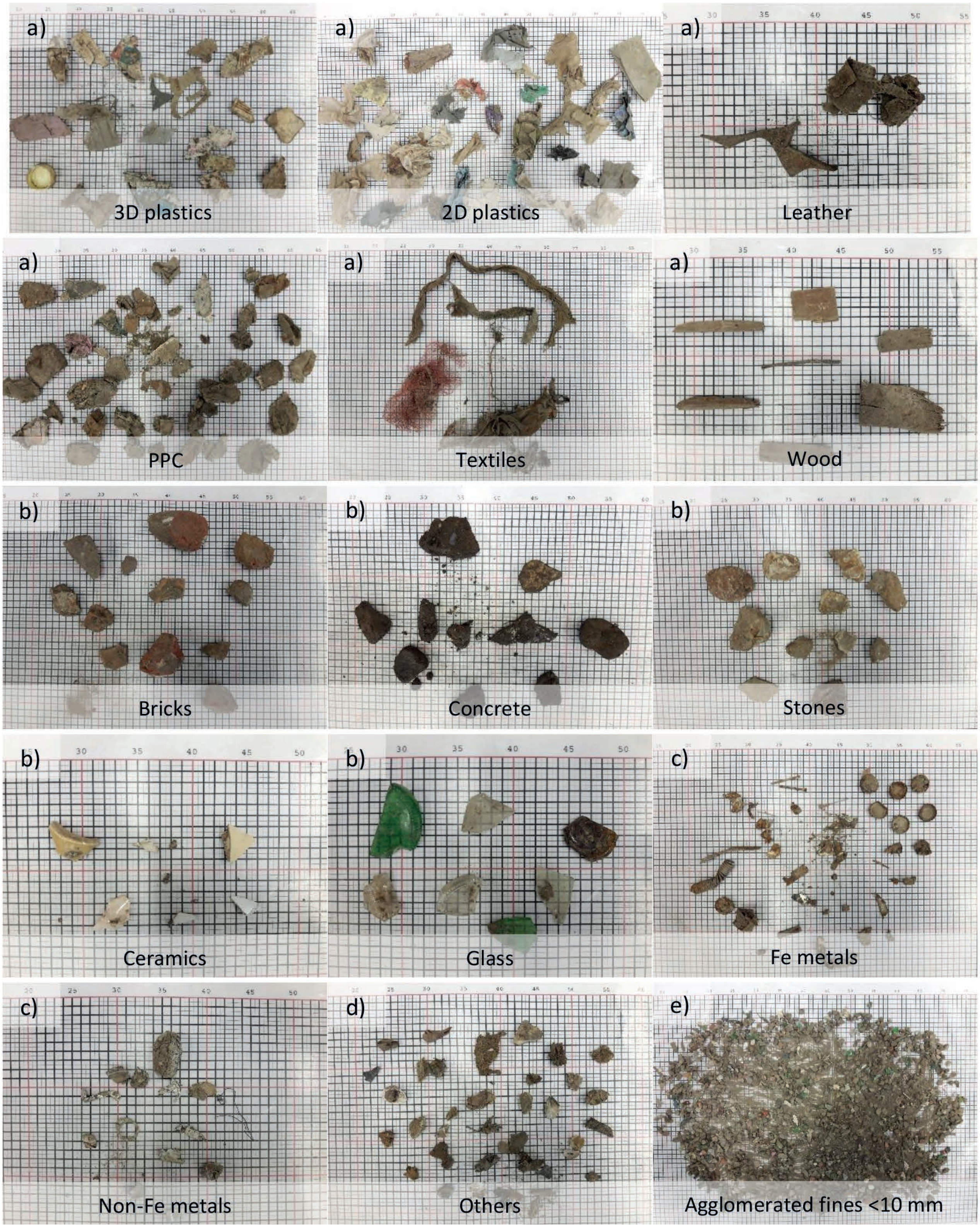

FIGURE 10: Photographs of the grouped material fractions: a) “Combustibles”, b) “Inert”, c) “Total metals”, d) “Others” and e) “Agglomerated fines <10 mm”.

of raw and dry states, the water content of each material type was determined and deducted from the results of the material composition in raw state (i.e. raw state in water-free conditions). In this way, the capacity of some materials to absorb and adsorb water does not play a role in the results of weight distribution, which could lead to relevant misinterpretations in some cases.

The results of the material composition, classified in five grouped material fractions (i.e. “Combustibles”, “Inert”, “Total metals”, “Others” and “Agglomerated fines <10 mm”) and according to the three particle size ranges (i.e. 90-63 mm, 63-31.5 mm and 31.5-10 mm), for raw, raw (water-free) and dry states are presented in Table 3. The material composition of the “Fine fractions <10 mm” was not determined in this study and, thus, this fraction was included in Table 3 as “Mixed materials”. It was visually detected that the fine fractions <3.15 mm in both states mostly corresponded to a relatively homogeneous soil-like material, which in turn was mainly composed of weathered inorganic and degraded organic matter. The presence of “Combustibles”, “Total metals” and “Others” could not be identified below 3.15 mm.

The overall material composition of the fine fractions showed amounts of “Agglomerated fines <10 mm” of about 1.1 wt.% and 1.1 wt.% in the particle size range 90-63 mm, 1.7 wt.% and 1.5 wt.% in 63-31.5 mm and 35.4 wt.% and 34.2 wt.% in 31.5-10 mm in raw and raw (water-free) states, respectively. The amounts of the same fraction in dry state were around 0.1 wt.% in the particle size range 90-63 mm, 0.2 wt.% in 63-31.5 mm and 3.5 wt.% in 31.5-10 mm, which means that the amount of agglomerated material is roughly ten times lower in dry state than in raw and raw (water-free) states. In turn, “Fine fractions <10 mm (Mixed materials)” presented an amount of about 12.3 wt.% and 11.8 wt.% in raw and raw (water-free) states, respectively, versus around 51.8 wt.% in dry state. This information confirms that a significant amount of “Fine fractions <10 mm” tends to form agglomerates and adhere to bigger particles in raw state (presence of moisture), which end up mixed with and adhered to coarser material fractions (i.e. “Combustibles”, “Inert”, “Total metals” and “Others”), potentially leading to a reduction in efficiency and performance along the mechanical processing. For instance, the variation of the amounts of “Total metals” between raw, raw (water-free) and dry states are likely to be mainly attributed to the presence of surface defilements, which still remained attached to that fraction after drying in most of the samples.

The presence of surface defilements was visually identified in the “Combustibles”, “Inert”, “Total metals” and “Others” fractions in raw, raw (water-free) and dry states. As it can be logically expected, the amount of surface defilements was considerably larger in raw and raw (water-free) states than in dry state. It is important to point out that the material composition to be taken as a reference in (E)LFM projects and investigations regarding material and energy recovery is the one determined in dry state, since in this state the material composition is least influenced by the presence of water, agglomerates and surface defilements

TABLE 3: Material composition of the fine fractions per particle size range in raw, raw (water-free) and dry states.

Particle size range / Grouped material fraction		Amount [wt.%]								
		Raw state			Raw state (water-free)			Dry state		
		25th percentile	Median	75th percentile	25th percentile	Median	75th percentile	25th percentile	Median	75th percentile
90-63 mm	Combustibles	0.6	1.1	3.6	0.5	0.9	3.2	0.3	1.0	3.0
	Inert	0.1	1.4	2.2	0.2	1.6	2.5	0.9	1.5	2.9
	Total metals	0.0	0.0	0.2	0.0	0.0	0.2	0.0	0.0	0.2
	Others	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Agglomerated fines <10 mm	0.4	1.1	1.9	0.4	1.1	1.7	0.0	0.1	0.1
63-31.5 mm	Combustibles	2.4	5.9	8.0	1.9	4.7	5.9	0.7	2.7	6.4
	Inert	5.6	7.6	8.0	7.2	8.5	9.9	8.7	9.7	13.2
	Total metals	0.0	0.4	0.7	0.0	0.4	0.9	0.2	0.7	0.9
	Others	0.1	0.3	0.8	0.1	0.4	0.8	0.0	0.2	1.6
	Agglomerated fines <10 mm	0.6	1.7	9.7	0.6	1.5	9.0	0.2	0.2	0.3
31.5-10 mm	Combustibles	2.9	3.7	12.4	2.5	3.2	9.4	1.3	3.7	9.6
	Inert	9.4	12.3	20.4	13.2	16.8	21.7	17.8	19.9	22.2
	Total metals	0.3	0.4	0.6	0.4	0.6	0.7	0.3	0.5	0.7
	Others	0.6	0.8	5.4	0.6	0.8	5.8	0.4	0.6	1.1
	Agglomerated fines <10 mm	25.5	35.4	42.0	23.9	34.2	41.0	2.9	3.5	3.8
Fine fractions <10 mm	Mixed materials	11.3	12.3	15.3	11.1	11.8	14.6	37.8	51.8	55.6

and, hence, reflects the material composition which is closest to reality. Therefore, discrepancies from the material composition in raw state with respect to the one in dry state can be explained by the presence of impurities and moisture. The latter being valid only if the quantities have not been calculated under water-free conditions. Nevertheless, more precise results of material composition might be achieved by implementing a washing step or wet sieving before the manual characterization, which will most likely require additional effort and time. The determination of the material composition in raw and raw (water-free) states is useful for the quantification and localization of material agglomerates and impurities, which are of critical relevance for the design of the mechanical processing.

In the case of the "Inert" fractions, amounts of about 1.4 wt.% and 1.6 wt.% in the particle size range 90-63 mm, 7.6 wt.% and 8.5 wt.% in 63-31.5 mm and 12.3 wt.% and 16.8 wt.% in 31.5-10 mm were determined in raw and raw (water-free) states, while amounts of 1.5 wt.%, 9.7 wt.% and 19.9 wt.%, respectively, were obtained in dry state. These figures show that most of the "Inert" fraction was found between 31.5 mm and 10 mm in all three states and that the amount of "Inert" increased with the decrease in particle size. Nonetheless, the influence of moisture (agglomerates and surface defilements) might lead to misinterpretations if not taken into account properly, considering that the real amount of "Inert" is higher than the one shown by the compositions in raw and raw (water-free) states.

The amounts of "Combustibles" and "Others" showed inconsistent fluctuations between the different states and particle size ranges, which might be explained due to the fact that both correspond to the most heterogeneous grouped fractions and are composed of various materials with different properties.

The previous outcomes show that "Total metals" were less affected, in terms of impurities, by the presence of moisture and, therefore, they could be recovered in raw state of the fine fractions, provided that the original water content allows an effective particle size classification (sieving) a priori. On the other hand, other fractions, such as "Combustibles" and "Inert" fractions, might need either partial/complete drying or washing step before material recovery can be implemented efficiently.

In general, these results show that most of the amount of the grouped fractions is located between 63 mm and 10 mm and, therefore, omitting the sieving step at 63 mm and sieving directly at around 30 mm would be advisable for full scale processing of the fine fractions. Particle size classification, as well as most mechanical processing steps in (E)LFM, performs best with narrow differences in particle sizes, so a direct sieving from 90 mm to 10 mm would most likely lead to sieve clogging and bad performance of the equipment along the subsequent mechanical processing.

3.4 Potential for material and energy recovery

The findings presented above document the presence of materials in the fine fractions that could have potential for WtM and WtE (i.e. "Combustibles", "Inert" and "Total metals"), as well as the most important characteristics of the fine fractions (i.e. bulk density, water content and PSD) that are to be taken into account for the design of an effective and efficient mechanical processing. Provided that such mechanical processing is implemented, the fine fractions from the excavated area at MSG landfill could yield medians of about 8.0 wt.% "Combustibles", 32.4 wt.% "Inert", 1.3 wt.% "Total metals", 0.9 wt.% "Others", 3.8 wt.% "Agglomerated fines <10 mm" and 51.8 wt.% "Fine fractions <10 mm" (Table 4). As the fraction "Others" was mostly composed of organic matter (e.g. bones, shells, sponges, among others), which could be valorized thermally together with the "Combustibles" fraction, the total amount of combustibles could be slightly increased.

If processed in dry or reduced moisture state (optimal water content) most of "Agglomerated fines <10 mm" would most likely end up in the "Fine fractions <10 mm", raising the amount of the latter as well. This fraction could be processed further in order to recover additional amounts of "Combustibles", "Inert" and "Total metals", since it was identified through PSD analysis and manual characterization that some of those material fractions were still present above 3.15 mm. These additional amounts are most likely to be low; nonetheless, this is a necessary step in order to reduce the amount of undesired materials if the production of a soil substitute material is envisaged. As proposed in Hernández Parrodi et al., 2018b, "Fine fractions <10 mm" could be processed further to produce a material that can be used as soil substitute in construction applications, whereas the "Inert" fraction could be used for the production of construction aggregates and "Total metals" could be sent to recycling, following the WtM pathway. In turn, "Combustibles" (together with "Others") could be suitable to produce refuse derived fuel (RDF) and, thus, incorporate to a WtE scheme. Nonetheless, the mechanical processing approach is to be designed in such a way that the applicable specifications for the usage of such materials in the intended purposes are met as well.

It is relevant to note that the previous amounts were taken directly from the results of the manual characterization and PSD analysis without considering efficiencies of mechanical processing and material losses. Therefore, these figures may vary considerably in full scale processing.

TABLE 4: Material composition of the fine fractions in dry state.

Particle size range / Grouped material fraction		Amount [wt.%]		
		25th percentile	Median	75th percentile
Fine fractions 90-10 mm	Combustibles	2.1	8.0	19.7
	Inert	31.1	32.4	35.4
	Total metals	0.6	1.3	1.8
	Others	0.6	0.9	3.4
	Agglomerated fines <10 mm	3.3	3.8	4.2
Fine fractions <10 mm	Mixed materials	37.8	51.8	55.6

4. CONCLUSIONS

The determination of the main characteristics of the fine fractions, such as bulk density, water content, PSD and material composition, is of utmost importance for the design of a successful mechanical treatment process, as well as for assessing the potential for material and energy recovery from fine fractions in (E)LFM projects. Bulk density, water content, PSD and material composition, as well as the correlations between them, are necessary information in order to be able to predict the behavior of fine fractions in a certain mechanical processing method. Furthermore, they turn out to be critical parameters to be taken into account for an appropriate selection of processing methods. Moreover, material composition serves as a basis to identify the strategies to follow regarding WtM and WtE in (E)LFM.

In this case study about 77 wt.% of the total landfill-mined material in raw state corresponded to the fine fractions (material <90 mm), which had an overall bulk density range of 720-1000 kg/m^3 and a total water content range of 25-30 wt.%. In general, bulk density appeared to increase as particle size decreased in both raw and dry states. Nevertheless, a decrease in bulk density was observed in the particle size range <0.16 mm in dry state. Furthermore, the overall bulk density was reduced about 21 wt.% after drying the fine fractions, which can be very useful information for the design of the mechanical processing.

The amount of moisture contained in the fine fractions substantially influences the presence of “Agglomerated fines <10 mm”, as well as the amount of impurities in the fractions “Combustibles”, “Inert”, “Total metals” and “Others”. “Total metals” seemed to be less influenced by water content and, therefore, their recovery could be done in raw state, given that the original water content allows an adequate particle size classification a priori. For the recovery of the “Combustibles”, “Inert” and “Others” fractions and further processing of the “Fine fractions <10 mm”, additional drying/moisture reduction or washing step/wet processing might be required in order to achieve adequate mechanical processing and obtain acceptable material qualities.

In addition to the determination of the relative mass distribution according to size, the PSD analysis in both raw and dry states can be used to identify the particle size from which the fine fractions might require a drying step or wet processing. Moreover, such analysis can also be utilized to determine the optimal water content in order to minimize dust generation and material loss during dry mechanical processing without the need of complete drying.

Results on the material composition in dry state reveal that amounts of 2.1-19.7 wt.% “Combustibles”, 31.1-35.4 wt.% “Inert” and 0.6-1.8 wt.% “Total metals” could be recovered from the fine fractions 90-10 mm, while 37.8-55.6 wt.% “Fine fractions <10 mm” could be processed further in order to increase the recovery amounts of the previous fractions and produce a substitute material for soil in construction applications. For this, applicable specifications for the usage of such materials in the foreseen purposes need to be taken into account.

It is highly important to highlight that the findings of this study are only valid for the investigated area at the landfill site and, therefore, additional studies covering the complete landfill area of the site are to be done in order to determine the overall material composition and characteristics of the MSG landfill, as well as to assess the global potential for WtM and WtE. Additionally, it is of critical relevance to point out that one of the greatest challenges faced by (E)LFM remains to be the processing of the fine fractions for the recovery of valorizable fractions in an economically feasible manner, since the implicated capital and operating expenditures continue to outweigh the revenues obtained by the valorization of those fractions. However, this situation is more related to policy and market aspects than to its technical feasibility.

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CASE STUDY ON ENHANCED LANDFILL MINING AT MONT-SAINT-GUIBERT LANDFILL IN BELGIUM: MECHANICAL PROCESSING OF FINE FRACTIONS FOR MATERIAL AND ENERGY RECOVERY

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ABSTRACT
(Enhanced) landfill mining ((E)LFM) projects have been mainly driven by land reclamation, environmental pollution mitigation and remediation of old landfills and dumpsites, among others. However, previous studies have also shown that these sites may be a relevant source of secondary raw materials. In this respect and within the framework of the "EU Training Network for Resource Recovery through Enhanced Landfill Mining – NEW-MINE", around 374 Mg of waste was excavated from a landfill site in the municipality of Mont-Saint-Guibert, Belgium, as part of a case study to evaluate the full implementation of ELFM. The excavated landfilled material was pre-processed with a ballistic separator onsite directly after excavation, with which the fine fractions (material <90 mm) were obtained. Subsequently, samples of the fine fractions were characterized in order to determine their main properties and material composition, which in turn were used to define the material and energy recovery strategies to be followed. According to these strategies a chain of mechanical processing steps was selected and tested in the processing of the fine fractions in the optimal water content (15 wt.% WC) and dry states. The mechanical processing consisted of particle size classification, ferrous and non-ferrous metals extraction, density separation and sensor-based sorting steps. For the recovery of materials (waste-to-material), fractions of a soil-like material (fine fractions <4.5 mm), inert, ferrous and non-ferrous metals were targeted. These fractions might be suitable for replacing soil in construction applications (e.g. embankments), substituting construction aggregates (e.g. construction gravel) and recycling, respectively. For the recovery of energy (waste-to-energy), a fraction composed of combustible materials was aimed for, which might be suitable for the production of an alternative fuel (e.g. refuse derived fuel). The mechanical processing in the dry state yielded total amounts of 41.9-43.9 wt.% DM fine fractions <4.5 mm, 35.9-39.0 wt.% DM inert materials, 7.4-10.0 wt.% DM combustible materials, 1.2-1.8 wt.% DM ferrous metals and 0.2-0.4 wt.% DM non-ferrous metals. These figures suggest that a significant share of the fine fractions could be recovered through the tested mechanical processing approach, which might contribute to the overall economic and environmental feasibility of the project in case of implementing full scale (E)LFM at the studied landfill site.

1. INTRODUCTION

Early research shows that (enhanced) landfill mining ((E)LFM) projects have been mainly driven by land reclamation, environmental pollution mitigation and remediation of old landfills and dumpsites, among others (Hernández Parrodi, Höllen, & Pomberger, 2018a). However, many LFM projects have faced strong difficulties or even failed to achieve immediate economic feasibility, which does not take into account long-term environmental passive costs, such as landfill aftercare and air, water and soil pollution remediation, among others. Furthermore, this has been frequently accompanied by social, political and legislatorial resistance, despite the pollution remediation and mitigation nature of LFM. Altogether, such circumstances have created skepticism towards the viability of the LFM concept and hampered its widespread practice. Therefore,

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LFM has evolved over the last decade into a sound concept known as ELFM, which seeks to meet the environmental, social and economic constraints in LFM; incorporating maximum material and energy recovery, while complying with the most stringent environmental and social criteria (Jones, Geysen, Rossy, & Bienge, 2010). Within the framework of ELFM, a research project from the European Union started in 2016 and has been studying the full implementation of this concept. This project is the "EU Training Network for Resource Recovery through Enhanced Landfill Mining – NEW-MINE", which studies, among others, additional sources of revenue, such as high value-added products (i.e. hydrogen, methane, synthetic polymers and glass ceramics), as well as the involved social, environmental and policy implications to promote not only economic but overall feasibility of ELFM and its successful implementation in current waste management systems (Hernández Parrodi et al., 2019b).

NEW-MINE is related to a handful of landfill sites in Europe, including the Mont-Saint-Guibert (MSG) landfill in Belgium. The latter was chosen to perform the assessment of a whole case study on ELFM. Landfill waste was excavated from the MSG landfill and pre-processed onsite directly after excavation. Two ballistic separation steps were employed as pre-processing, in which the excavated material was divided into different outputs. The relevant output stream for the present study are the fine fractions, which was the material with a particle size <90 mm obtained after the second step of ballistic separation. Representative single samples were taken at the underscreen outlet of the ballistic separator. Those single samples were used to prepare composite samples, which were employed to perform the material characterization of the fine fractions reported in Hernández Parrodi et al., 2019a. According to the results of the material characterization, the waste-to-material (WtM) and waste-to-energy (WtE) strategies for the fine fractions were defined and a specific mechanical processing approach was selected.

The main purpose of the selected mechanical processing was to separate the fine fractions into five different fractions: combustibles, inert, ferrous (Fe) metals, non-ferrous (non-Fe) metals and fine fractions <4.5 mm. Combustibles were intended to produce a fraction with high calorific value that could be used as refuse derived fuel (RDF). This type of material recovered from old landfill sites is usually very heterogenous and presents undesired characteristics for traditional recycling, such as significant water content, great amount of impurities and high state of degradation. Hence, thermo-chemical processes (i.e. incineration, gasification and pyrolysis) might result to be some of the few presently available and feasible alternatives to valorize landfill-mined materials with high calorific value, transforming them into a potential source of energy for WtE applications. Inert, Fe and Non-Fe metals, as well as the fine fractions <4.5 mm, targeted towards WtM. The recovery of Fe and non-Fe metals was envisaged for recycling, while the production of a substitute for construction aggregates was foreseen with the inert fraction. A substitute for soil in construction applications was targeted with the fine fractions <4.5 mm.

To this end, composite samples of the fine fractions from the MSG landfill were processed with a series of mechanical processing equipment. The equipment employed, as well as the results obtained are presented herein. The main objective of this study is to evaluate and discuss the performance of the selected mechanical processing approach for optimum material and energy recovery in (E)LFM.

2. MATERIALS AND METHODS

2.1 Site description, excavation works and material pre-processing

The landfill site "Centre d´enfouissement Technique de Mont-Saint-Guibert (CETeM)" is located in the municipality of MSG in Wallonia, Belgium, about 33 km south of Brussels capital city. This landfill was one of the main disposal sites of municipal solid waste (MSW), non-hazardous industrial waste (IW) and construction and demolition waste (C&D) in the province of Walloon Brabant from 1958 to 2014 (Bureau d´études greisch (beg), 2002). For this investigation about 425 m^3 (374 Mg) of landfill waste were excavated (~10 m long, ~10 m wide and ~4 m deep) and the excavated volume was divided into four sub-volumes, which are henceforth referred to as batches. These batches were visually classified according to their main composition, as a clear stratification of the material was identified during excavation. The C&D and MSW layers had a thickness of about 2 m each. Hence, the dimensions of each batch were ~5 m in length, ~5 m in width and either ~2 m or ~4 m in depth, depending on the targeted material to be excavated. Directly after excavation, each batch was pre-processed individually with a ballistic separator (Stadler model STT 6000) in two steps; first with screen paddles of 200 mm and subsequently with screen paddles of 90 mm in cascade arrangement. The fine fractions processed in this case study correspond to the underscreen fraction <90 mm obtained after the second ballistic separation step. Information about the coarse fractions (material ≥90 mm), as well as further details about the ballistic separation process have been reported by García López et al., 2019.

During the pre-processing of each batch representative single samples from the underscreen output fraction of the ballistic separator were taken, and composite samples were prepared from the single samples. This study focuses solely on the mechanical processing of the composite samples of batches 1 and 2, from which a total of 32 single samples (16 single samples for each batch) of 10 l were taken and 16 composite samples (8 composite samples for each batch) of 20 l were prepared using the quartering method, according to the German guideline for procedures for physical, chemical and biological testing in relation to the recovery/disposal of waste (LAGA PN 98).

Further information regarding the landfill site, excavation works, material pre-processing and sampling procedures are reported in Hernández Parrodi et al., 2019a. The results of all composite samples, material fractions and particle size ranges (i.e. material characterization, (optimal) water content and mechanical processing) presented herein are based on mass percentage (wt.%), for which an industrial platform scale (Kern DS 150K1, resolution of 1.0 g) and a precision balance (Kern KB 2400-2N, resolution of 0.01 g) were employed. Water and dry mass contents were determined according to the DIN EN 14346 (modified: drying at 75 °C ± 5 °C to avoid melting of certain plastics and material losses), while the drying processes were carried out in a Heraeus industrial drying oven.

2.2 Fine fractions

The material characterization of the fine fractions from the MSG landfill was performed by Hernández Parrodi et al., 2019a, who reported overall ranges for bulk density and water content of 720-1 000 kg/m^3 (median of 810 kg/m^3) and 25-30 wt.% (median of 27 wt.%), respectively. The data of the material composition of the fine fractions 90-10 mm in dry state from that study was reclassified in 3 particle size ranges (i.e. 90-31.5 mm, 31.5-10 mm and Fine fractions <10 mm) and 7 grouped material fractions (i.e. "Combustibles", "Inert", "Fe metals", "Non-Fe metals", "Others", "Agglomerated fines <10 mm" and "Mixed materials") in this study, in order to allow direct comparison with the results of the mechanical processing. The reclassification consisted in separating the grouped material fraction "Total metals" from the material characterization into the fractions "Fe metals" and "Non-Fe metals", as well as in joining the particle size ranges 90-63 mm and 63-31.5 mm to form a particle size range of 90-31.5 mm. This information is shown in Table 1.

Figures in Table 1 were calculated using the 25th, 50th (median) and 75th percentiles in order to depict the variation range of the amount of each grouped material fraction, which can be used as reference to evaluate the recovery of "Combustibles", "Inert", "Fe metals" and "Non-Fe metals"

TABLE 1: Material composition of the fine fractions in dry state (modified from Hernández Parrodi et al., 2019a).

Particle size range/Grouped material fraction		Amount [wt.%]		
		25th percentile	Median	75th percentile
90-31.5 mm	Combustibles	0.9	4.0	9.9
	Inert	9.7	13.3	15.1
	Fe metals	0.2	0.3	0.7
	Non-Fe metals	0.0	0.2	0.5
	Others	0.0	0.2	1.9
	Agglomerated fines <10 mm	0.2	0.3	0.4
31.5-10 mm	Combustibles	1.3	3.7	9.6
	Inert	17.8	19.9	22.2
	Fe metals	0.2	0.3	0.5
	Non-Fe metals	0.1	0.1	0.2
	Others	0.4	0.6	1.1
	Agglomerated fines <10 mm	2.9	3.5	3.8
Fine fractions <10 mm	Mixed materials	37.8	51.8	55.6

Notes: Total amounts do not account for 100 wt.% due to the utilization of quantiles.

from the fine fractions of the MSG landfill in the tested mechanical processing. These figures show that median amounts of about 4.0 wt.% "Combustibles", 13.3 wt.% "Inert", 0.3 wt.% "Fe metals" and 0.2 wt.% "Non-Fe metals" could be recovered from particle size range 90-31.5 mm, while around 3.7 wt.% "Combustibles", 19.9 wt.% "Inert", 0.3 wt.% "Fe metals" and 0.1 wt.% "Non-Fe metals" could be obtained from particle size range 31.5-10 mm. Additionally, about half of the total amount of the fine fractions (median of 51.8 wt.%) can be expected to be <10 mm.

2.3 Particle size distribution and water content

A study on the particle size distribution of the fine fractions from batches 1 and 2 with different water contents was carried out in order to identify the optimal water content. The optimal water content is that with which the fine fractions are still able to be handled efficiently in a dry mechanical process without the need of complete drying. Theoretically, less resources (i.e. energy and time) would be needed to reach adequate mechanical processing in this way, while material losses and dust generation would be decreased significantly. Eight composite samples of batch 1 (n=4) and batch 2 (n=4) were used to determine the range of the particle size distribution with water contents of 10 wt.% and 20 wt.%, respectively. Initially all composite samples were dried completely according to the DIN EN 14346 (as described in Section 2.1) and, subsequently, the corresponding water contents were set following the procedure described in Section 2.4.1. Afterwards, the composite samples were sieved in a circular vibratory box sieve (Siebtechnik, 500 mm x 500 mm) with circular reticle sieves of 63 mm, 31.5 mm and 10.0 mm, followed by a sieving with a circular vibratory sieve tower (Siebtechnik, Ø 400 mm) with squared reticle sieves of 6.3 mm, 3.15 mm, 1.0 mm, 0.63 mm, 0.315 mm and 0.16 mm, according to the DIN EN 15415-1:2011. Finally, the obtained curves were compared to the particle size distribution ranges from both batches in raw (n=8) and dry (n=8) states from the material characterization (Hernández Parrodi et al., 2019a) in order to identify the optimal water content.

2.4 Mechanical processing

A specific process chain of mechanical equipment was selected to process the fine fractions according to the results of the material characterization and the defined strategies for WtM and WtE reported in Hernández Parrodi et al., 2019a. This arrangement was selected with the main objective of optimizing the recovery of certain fractions from the fine fractions; such as a fraction with high calorific value that could be used as an alternative fuel (e.g. RDF/Solid Recovered Fuel (SRF)), an inert fraction that could be used as substitute for construction aggregates (e.g. construction sand/gravel) and a soil-like fraction (i.e. fine fractions <4.5 mm) that could be used as substitute for soil in construction applications (e.g. dykes/embankments), as well as fractions composed of Fe and non-Fe metals that could be recycled. At the same time, the selected arrangement aims at reducing the amount of the fine fractions to be re-landfilled or stored until more adequate technologies are developed for its valorization or further utilization. To this end, a dry mechanical processing approach was chosen, since wet processing methods tend to be more elaborate and complex, and have been associated with higher capital and operational costs in the past (Bunge, 2012). Additionally, wet methods involve sending a certain share of impurities and contaminants to an aqueous medium, which must be treated at some point as well. Furthermore, in the context of (E)LFM most of the outputs of a wet processing approach will need a significant reduction in moisture before being suitable for WtM and WtE schemes.

Figure 1 presents an overview of the mechanical processing approach implemented in this study. The whole mechanical processing was investigated using composite samples of batches 1 and 2 in the optimal water content state (n=8) and in the dry state (n=8), in which 4 composite samples of each batch were processed in the optimal water content state and 4 composite samples in the dry state. Each composite sample was processed individually in order to study the variation between samples as well. Figure 1 shows two main material flows (black arrows): one for the composite samples in the optimal water content state and one for those in the dry state.

In order to avoid further alteration of the composite samples (e.g. moisture gain/loss, material loss, weathering and fractionation), the calibration of all processing equipment, except for the sensor-based sorter, was performed by means of artificial samples, which were composed of similar type of materials in new state (i.e. hard and soft plastics, paper, wood, glass, stones, metals and soil) and prepared in such a way that the main characteristics of the material (i.e. composition, density, particle size range and water content) were simulated. The sensor-based sorting equipment was calibrated with real representative pieces of each material type selected by hand, since the actual spectra of each material were needed for the calibration of the equipment.

Most of the materials with high calorific value (e.g. plastics, textiles, leather, paper and wood) were expected to be found in the light fractions and, hence, a fraction called "Combustibles", which is marked in purple in Figure 1, was generated with those materials to produce an alternative fuel, which might be suitable for thermal valorization. To this end, thermogravimetry has proven to be a promising method to determine the composition of such materials and study their decomposition in thermo-chemical conversion processes, such as incineration, pyrolysis and gasification, and, thus, can be helpful for selecting the most appropriate option to be employed (Burlakovs et al., 2019). Inert materials (e.g. bricks, concrete, stones, glass and ceramics) were anticipated in the heavy fractions and, thus, a fraction denominated "Inert", which is marked in grey in Figure 1, was generated with these fractions in order to produce a substitute for construction aggregates. In addition, seashells, which are not scarce in Belgian MSW, were sorted out of the "Inert" fraction by means of sensor-based sorting and were incorporated in the "Combustibles" fraction, since they have proven to act as an effective antichlor agent (Tameda et al., 2018) due to their high calcium carbonate content. Fe and non-Fe metals, which are

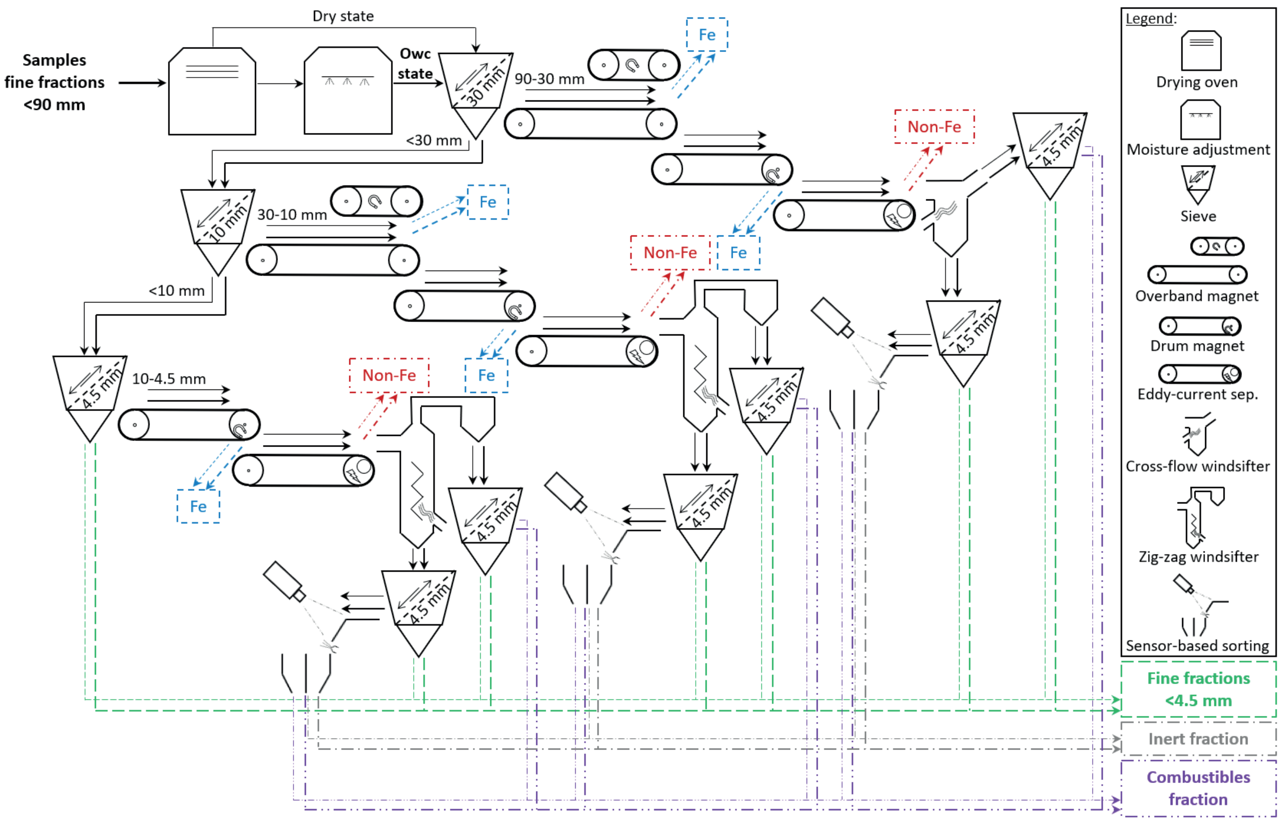

FIGURE 1: Mechanical processing flow chart of the fine fractions in the optimal water content (owc) and dry states.

respectively marked in blue and red in Figure 1, were also extracted from the fine fractions, since they can represent a substantial share of the revenues from LFM (Van Vossen & Prent, 2011; Winterstetter, Laner, Rechberger, & Fellner, 2015). The results on the quality assessment of the recovered non-Fe metals for recycling purposes are reported in Lucas et al., 2019. However, the recovery of ferrous metals from landfilled material is nowadays regarded as technically possible, since it has been successfully performed in previous LFM investigations (Van Vossen & Prent, 2011; Wagner & Raymond, 2015). Hence, the quality of the extracted ferrous metals was not investigated in the present case study.

Additionally, a fraction named "Fine fractions <4.5 mm", which is marked in green in Figure 1, was created with the recovered surface defilements and agglomerates liberated by the coarser particle size ranges (i.e. 90-30 mm, 30-10 mm and 10-4.5 mm) along the mechanical processing, as well as with the underscreen fraction from the sieving step at 4.5 mm. The properties of the "Fine fractions <4.5 mm" will be studied in order to determine if a substitute for soil in construction applications could be produced with the whole or a certain amount of this fraction.

The mechanical processing until sieving steps at 4.5 mm was carried out at the technical facilities of the Department of Processing and Recycling (IAR) of the RWTH Aachen University, whereas the sensor-based sorting steps were performed at the technical laboratory of the Chair of Waste Processing Technology and Waste Management (AVAW) of the Montanuniversität Leoben. For explanatory purposes, the mechanical processing was organized in five stages: i) material conditioning, ii) particle size classification, iii) extraction of Fe and non-Fe metals, iv) separation of light and heavy fractions and v) quality improvement of light and heavy fractions. These stages are described in the following sections of this chapter.

2.4.1 Material conditioning

There are several industrial options for drying or reducing the water content of landfill-mined material, such as aeration pile, biodrying and drum furnace, among others. Each of those options has its own advantages, limitations and cost implications towards (E)LFM, which can be very relevant and, thus, must be carefully assessed beforehand. However, the drying process is not the main focus of the present study and, therefore, it is not discussed further on.

In order to set the target water contents (i.e. optimal water content state and dry state) for the 2 scenarios of the mechanical processing, the composite samples of both batches were completely dried in the industrial drying oven according to the DIN EN 14346 (as described in Section 2.1). Subsequently, water was added to half of the composite samples (n=4) of each batch until the optimal water content was reached. For this, tap water was gradually and uniformly sprinkled in layers of about 3 cm with a manual pressurized water sprayer (GLORIA prima 3 l – 3 bar) in a 90 l container. The material was thoroughly mixed and was left to rest for 24 h, in such a way that the water addition was evenly distributed throughout the whole sample. The remaining half of the composite samples (n=4) of each batch was kept in dry state.

After setting the target water contents, the mechanical processing was conducted separately for the two groups of samples: i) composite samples (n=8) in the optimal water content state and ii) composite samples (n=8) in the dry state. Both groups of samples included composite samples of each batch (n=4) and, from this point on, all composite samples were processed identically.

2.4.2 Particle size classification

Directly after the adjustment of water content the composite samples were classified into the following 4 particle size ranges:

- 90-30 mm
- 30-10 mm
- 10-4.5 mm
- <4.5 mm

The previous particle size ranges were selected according to the results of the material characterization reported in Hernández Parrodi et al., 2019a, targeting a minimum amount of sieving steps and a maximum amount of recoverable material per particle size range. This particle size classification was done using two different types of sieves for waste materials.

The first sieving step was performed with a circular motion vibrating sieve for waste materials (iFE waste screen for waste treatment and recycling) with 30 mm squared sieve panels (Figure 2a). This type of sieve was used due to its vibrating circular motion operating principle, robustness and cascade arrangement of the screening panels, which make it adequate for sieving heterogeneous humid waste mixtures with minimum clogging. This equipment had a total sieving length of 2.0 m and width of 0.8 m and was operated with a fixed inclination of 15°. As it is shown in Figure 2a, the sieve was feed by means of a 5.0 m long conveyor belt with a slope of 42°. The median throughput was ca. 13 kg (1 composite sample) per run, for which around 1 minute processing time was needed in the optimal water content state. In the dry state ca. 11 kg (median) was processed using the same duration.

The second and third sieving steps were done with a flip-flow type of sieve (Hein Lehmann LIWELL® screening machine) with 10 mm and 4.5 mm squared screen mats, respectively (Figure 2b). A flip-flow sieve was selected as it can cope with materials difficult to sieve due to their small grain size, moist and/or sticky nature, which is the case for the fine fractions from (E)LFM. This equipment uses flexible screen mats to apply a trampoline-like movement that prevents the sieve from clogging, while breaking apart material agglomerates and sieving the input material in a uniform and continuous manner. The sieve had total functional length and width of about 3.0 m and 0.5 m, respectively, and was operated with an inclination of 25°. In the second sieving step ca. 10 kg (median) of material was sieved at 10 mm for 1 minute per composite sample in the optimal water content state, whereas ca. 8 kg (median) was sieved in the dry state. In the third sieving step

FIGURE 2: a) Circular motion vibrating and b) flip-flow sieves.

ca. 7 kg (median) of material was sieved at 4.5 mm for 1 minute in the optimal water content state and ca. 6 kg (median) in the dry state.

2.4.3 Extraction of Fe and non-Fe metals

Fe and non-Fe metals were removed by magnetic and eddy-current separators. Fe metals were extracted using an overband magnet (Figure 3a) followed by a drum magnet (Figure 3b), in a cascade arrangement, in particle size ranges 90-30 mm and 30-10 mm, while they were removed employing only the drum magnet in the particle size range 10-4.5 mm. The overband magnet was used to extract the largest particles of the Fe metals fraction with high quality (low content of impurities), as the magnet was located above the throughput flow and pulled Fe metals out of the stream. In turn, the drum magnet was located below the throughput flow and pulled Fe metals out of the stream downwards. Thus, the drum magnet was used to remove the remainder of Fe metals, which ranged from Fe metal pieces attached to other materials to the smallest Fe particles (incl. iron filings and iron oxides) present in the throughput, and which normally have a poor quality. Additionally, the drum magnet was employed to protect subsequent processing equipment, i.e. eddy-current separator, since the presence of Fe metals can lead to overheating and malfunction of such equipment.

The employed overband magnet was a permanent Steinert suspension magnet (750 mm long and 100 mm wide) placed transversely to the throughput flow. A spacing of 180 mm was used for the particle size range 90-30 mm with respect to the conveyor belt transporting the material, while one of 60 mm was utilized for the particle size range 30-10 mm in both the optimal water content and dry states. The overband magnet was operated with a constant speed of about 1.5 m/s and the conveyor belt was set to a constant speed of around 1.0 m/s in both states. As for the drum magnet, a permanent Steinert drum magnetic separator (300 mm diameter and 500 mm long) was utilized, which was operated in both states at 35 rpm and fed by a vibratory conveyor at an approximate rate of 0.5 kg/minute.

Non-Fe metals were extracted by means of a permanent Steinert eccentric eddy-current separator (500 mm diameter and 800 mm long) operated at 3 000 rpm with an eccentricity of 30° for the particle size range 90-30 mm,

FIGURE 3: a) Overband magnetic, b) drum magnetic and c) Eddy-current separators.

36° for 30-10 mm and 42° for 10-4.5 mm in both states (Figure 3c). This machine was equipped with a vibratory conveyor and a conveyor belt, with which the material was driven over the eddy-current magnetic wheel at a speed of about 1.5 m/s for all samples.

2.4.4 Separation of light and heavy fractions

Windsifting was used to split light from heavy materials. This density separation method uses material properties, such as density and shape to separate the throughput by means of a stream of air, which carries light materials to a different recipient.

A cross-flow windsifter (cross-flow air classifier, self-made by the IAR) was employed to process the particle size range 90-30 mm (Figure 4a). This equipment blows a bottom up stream of air across the throughput flow in a transversal way and light materials are transported by the air stream along a pipe to a container, while heavy materials fall down at the air stream contact area and are collected in a separate container. This equipment was utilized since it is relatively robust and can handle particle sizes up to around 200 mm. During the operation of the cross-flow windsifter, an airflow volume flow of about 7 000 m^3/h was employed and the input material was delivered at a rate of around 1 kg/minute in both states. The fixed angle with which the air stream was injected was 45° with respect to the horizontal plane.

For the processing of the particle size ranges 30-10 mm and 10-4.5 mm a zig-zag windsifter (Graf zig-zag air classifier, custom made for the IAR) was used (Figure 4b), as it can separate small grain-sized materials with high precision. In this equipment the input material is delivered by an airtight vibratory conveyor into a horizontal zig-zag shaped channel, where an air stream is blown from bottom to top. The zig-zag shaped channel creates a combination of cross- and counter-flow air streams along multiple steps that transport light particles into an aerocyclone and subsequently to a separate recipient. The heavy fraction slides down over the zig-zag shaped channel and is collected in a container. The zig-zag windsifter was operated with an airflow speed range of 7.5-8.5 m/s for the particle size 30-10 mm, whereas 6.5-7.5 m/s was used for 10-4.5 mm in both states. The zig-zag shaped channel was around 1.2 m long and the input material was fed at a rate of about 0.5 kg/minute for the particle size range 30-10 mm and of 0.3 kg/minute for 10-4.5 mm in both states.

FIGURE 4: a) Cross-flow and b) zig-zag windsifters.

2.4.5 Enrichment and quality improvement of light and heavy fractions

Surface defilements, agglomerates and fine particles from material weathering are loosened and released along mechanical processing. These fine materials can be removed in order to improve the quality of the output fractions. Moreover, certain combustible materials with high densities might still be found in the heavy fractions after density separation. Such materials can be removed from the heavy fraction by means of sensor-based sorting, so that the quality of the heavy fractions is improved and the amount of materials with high calorific value can be valorized together with the "Combustibles" fraction.

Light and heavy fractions from the cross-flow and zig-zag windsifters were sieved further in order to remove released fine particles along the whole mechanical processing. This sieving was performed with a circular vibratory sieve tower (Siebtechnik, Ø 400 mm) with a squared reticle sieve of 4.5 mm during 1 minute in both states, since the amount of light and heavy fractions obtained from the density separation steps did not allow the employment of a larger scale equipment. Additionally, this last sieving step served the purpose of preconditioning the heavy fraction for the sensor-based sorting step, in which the presence of dust and fine particles interferes with the correct recognition and classification of the input material. The reduction of impurities in the light fraction might lead to reduce the ash content and, thus, to raise the calorific value, as well as to decrease the amount of certain contaminants, such as heavy metals and organic pollutants. Moreover, the underscreen fraction below 4.5 mm from the light and heavy fractions could be jointly valorized or processed further with the fraction "Fine fractions <4.5 mm" in this manner.

Near infrared (NIR) was employed by the sensor-based sorter to measure the wavelength with which a certain material reflects infrared radiation. Such measurements are then used to compute the spectrum variation for each material, which is either left in the material stream or sorted out, according to the desired set up of the equipment. The sorting is done by means of a pulse of pressurized air released through a nozzle, which shoots the particles to be sorted out, sending them to a separate container. To this end, a pilot scale sensor-based sorter manufactured by binder + co with a hyperspectral imaging (HIS) chute was utilized (Figure 5), which was equipped with a vibratory conveyor.

FIGURE 5: Sensor-based sorting equipment.

In this last processing step of the heavy fractions from the windsifting steps, the material was fed to the sensor recognition area at a rate of about 1 kg/minute, 0.5 kg/minute and 0.3 kg/minute for the particle size ranges 90-30 mm, 30-10 mm and 10-4.5 mm, respectively, in both states. The pressurized air was set to different pressures as well, which were 3 bar for 90-30 mm, 2 bar for 30-10 mm and 1.5 bar for 10-4.5 mm in both states. Further details about this processing step are reported in Küppers, Hernández Parrodi, García López, Pomberger, & Vollprecht, 2019.

3. RESULTS AND DISCUSSION

3.1 Optimal water content

Particle size distribution curves were calculated for the composite samples of batch 1 and batch 2 with median water contents of 0 wt.% (dry state), 10 wt.%, 20 wt.% and 27 wt.% (raw state), and analogously as for the material composition (Section 2.2), quantiles (25^{th}, 50^{th} and 75^{th} percentiles) were used to determine the variation range in each water content. The particle size distribution curves (solid lines) for each water content, as well as their variation ranges (dash and dash-dot lines), are plotted in Figure 6.

The particle size distribution curves in Figure 6 show a slight alteration of the particle size distribution with water contents of 10 wt.% and 20 wt.% above 3 mm. This suggests that the structure of the material might most likely have experienced fragmentation, material losses and a sort of cleaning effect due to complete drying, remixing, re-moisturization and a second particle size classification (i.e. sieving), since the same composite samples used to determine the particle size distribution in raw state of both batches were used for the adjusted water contents. This was the case because there were no additional virgin samples from the fine fractions available for this purpose. Nevertheless, taking into account that relevant amounts of grouped material fractions other than “Agglomerated fines <10 mm” (i.e. “Combustibles”, “Inert”, “Fe metals”, “Non-Fe metals” and “Others”) were not identified below a particle size of 3.15 mm during the material characterization of the fine fractions (Hernández Parrodi et al., 2019a), the sorting of the fine fractions into “Combustibles + Others”, “Inert”, “Fe metals” and “Non-Fe metals” would only make sense above 3 mm. Thus, given that the curve for 10 wt.% water content could be expected to be very close to the one in dry state, it was concluded that a reduction of the original water content (median of 27 wt.%) to around 15 wt.% would suffice to allow an adequate mechanical processing of the fine fractions above 3 mm. The latter, assuming that the additional amount of surface defilements in comparison to dry state, which was visually determined as not quantitatively relevant, would not interfere significantly with the efficiency and effectiveness of the sensor-based sorting steps of the mechanical processing, nor to meet the quality standards of the targeted outputs (i.e. RDF, substitutes for construction aggregates and soil in construction applications, and ferrous and non-ferrous metals). Nonetheless, a further reduction of the water content might be required for adequate particle size classification below 3 mm; especially below 0.6 mm, where the needed reduction appears to be below 10 wt.%.

Therefore, in order to assess the potential for material and energy recovery from the fine fractions of the MSG landfill through the selected dry mechanical processing approach, 2 scenarios with different water contents were studied and compared. These scenarios correspond to the above determined optimal water content state of 15 wt.% and the dry state.

3.2 General mass balance

In order to obtain a full overview of the mass distribution in the tested mechanical processing of the fine fractions, the outputs of the whole mechanical processing were classified into 6 categories, namely “Fine fractions <4.5 mm”, “Inert”, “Combustibles”, “Fe metals”, “Non-Fe metals” and, depending on the state, either “Material & water losses” for the optimal water content state or “Material losses” for the dry state. The output “Fine fractions <4.5 mm” in this case study corresponds to the material that was generated by a sieving step at 4.5 mm along the mechanical proces-

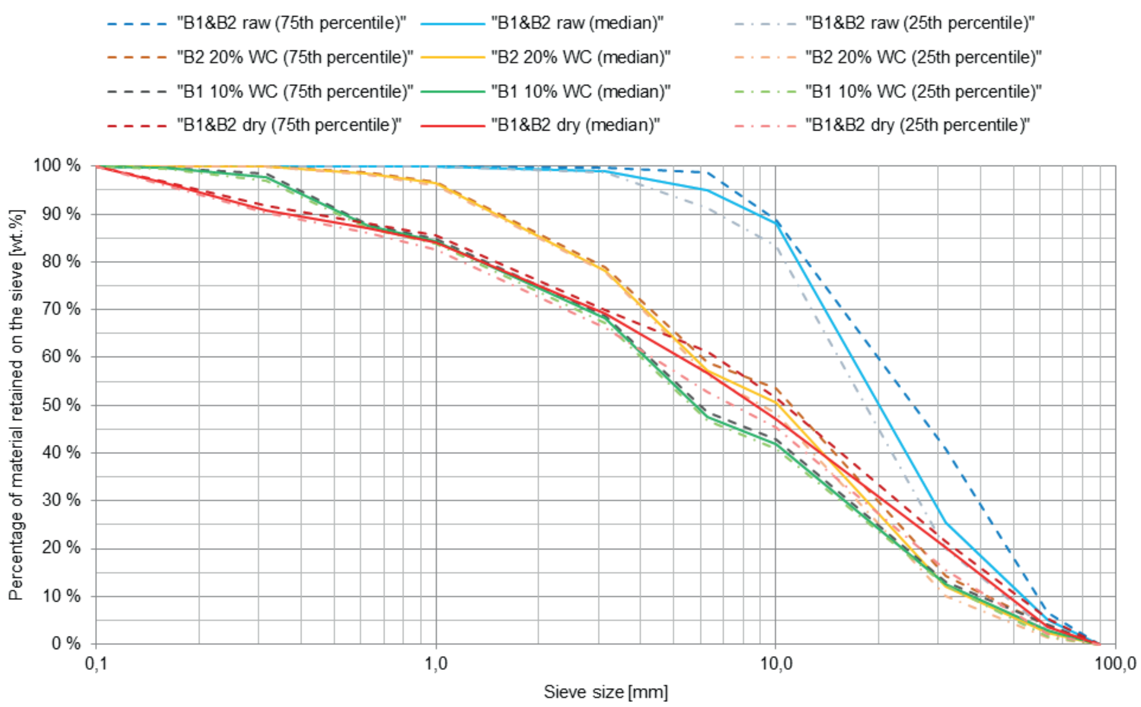

FIGURE 6: Particle size distribution of the fine fractions with different water contents.

sing, and it is referred to as "Soil-like material" concerning its apparent material composition. It is relevant to clarify that, as stated in Hernández Parrodi, Höllen, & Pomberger, 2018b, the term "Soil-like material" does not intend to rigorously classify this material as soil, but instead employs it for reasons of appearance, as well as because it is a commonly used term in the field. The amount of each category for each composite sample was determined for each state and the median was calculated (n=8). This information is displayed for each category and state in form of Sankey diagrams in Figure 7.

As shown in Figure 7, both the optimal water content state and the dry state presented the same tendency in terms of the amounts obtained from each output of the mechanical processing, in which most of the fine fractions corresponded to the "Fine fractions <4.5 mm" output, with amounts of 42.9 wt.% and 42.7 wt.%, respectively. That output was followed by "Inert", with amounts of 35.5 wt.% in the optimal water content state and 37.2 wt.% in the dry state. "Combustibles" output followed "Inert" with the respective amounts of 12.5 wt.% and 9.0 wt.% in the optimal water content and dry states. Subsequently, "Material & water losses" in the optimal water content state were slightly lower than "Material losses" in the dry state, with amounts of 7.6 wt.% and 7.9 wt.%, respectively. This may be explained by the fact that the presence of water increased the weight of certain materials to some extent and promoted the formation of surface defilements and agglomerates of fine particle sized material (<1 mm), which in turn decreased the loss of light and small particle sized materials (e.g. plastic foils and dust). However, it should be said that the material in the dry state might also have been influenced to a certain extent by the presence of water due to absorption/adsorption of humidity from the environment, which was not monitored throughout the whole mechanical processing nor taken into account in the mass balance of this state. Hence, the mass increase due to the influence of humidity from the environment might have compensated for a certain amount of material losses in the dry state, as well as decreased losses in the form of dust. Therefore, material losses in a strictly dry state might be higher than those reported in the present study. As for the amounts of "Fe metals" and "Non-Fe metals" outputs, a lower amount of "Fe metals" (0.9 wt.%) was obtained in the optimal water content state with respect to the dry state (1.4 wt.%), whilst the amount of "Non-Fe metals" obtained in the optimal water content state (0.4 wt.%) was slightly larger than in the dry state (0.3 wt.%). Discrepancies regarding the amounts between outputs "Fine fractions <4.5 mm", "Inert", "Combustibles", "Fe metals" and "Non-Fe metals" in the optimal water content and dry states are addressed in the following section.

3.3 Mass balance of grouped material fractions per particle size range

Regarding the different materials recovered from the fine fractions and, analogously to the results of the material characterization presented in Section 3.1, a mass balance of the obtained materials according to particle size range was performed using quantiles. The resulting information was organized according to the following grouped material fractions: "Combustibles", "Inert", "Fe metals", "Non-Fe metals" and "Fine fractions <4.5 mm (Soil-like material)", which are in accordance with the categories used to classify the outputs of the mechanical processing in the previous section (Section 3.2) and the particle size ranges generated along the mechanical processing (i.e. 90-30 mm, 30-10 mm and 10-4.5 mm). These grouped ma-

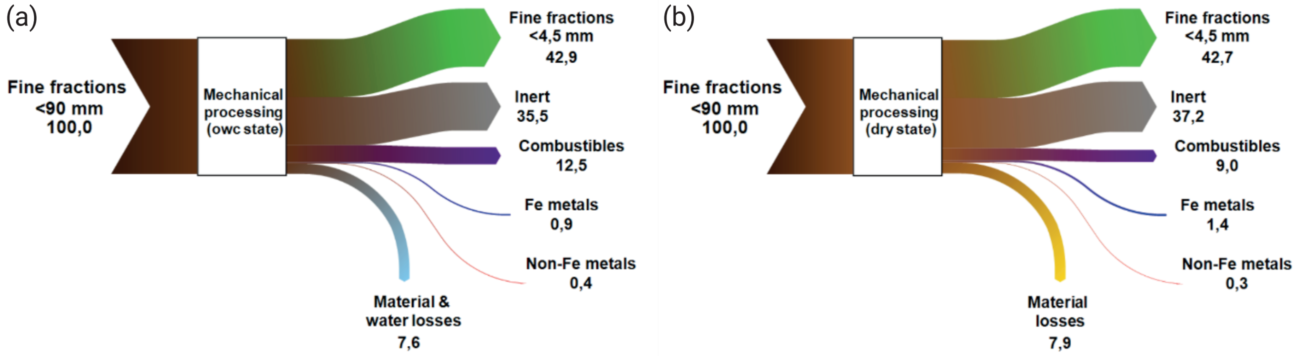

FIGURE 7: General mass balance of the mechanical processing in the a) optimal water content (owc) and b) dry states [figures in wt.%].

terial fractions are also in agreement with those of the material characterization presented in Table 1, except for the fraction "Fine fractions <4.5 mm (Soil-like material)", which has a particle size <4.5 mm and, hence, corresponds partially to fraction "Fine fractions <10 mm (Mixed materials)" and fraction "Others" of the material characterization, from which most part ended up in the fraction "Combustibles" of the mechanical processing due to its characteristics. Unlike material characterization, which was performed by hand down to a particle size of 10 mm, the mechanical processing was implemented down to 4.5 mm, since small amounts of recoverable materials were still visually identified below 10 mm and above 3.15 mm in the material characterization (Hernández Parrodi et al., 2019a). Thus, most of the fraction "Fine fractions <10 mm (Mixed materials)" is composed of the "Fine fractions <4.5 mm (Soil-like material)" fraction, whereas the remainder is expected to be distributed among the rest of the grouped material fractions (i.e. "Combustibles", "Inert", "Fe metals" and "Non-Fe metals") in the particle size range 10-4.5 mm of the mechanical processing.

The recovered amounts of each grouped material fraction according to particle size range for each state (i.e. the optimal water content and dry states) of the mechanical processing are summarized in Table 2, in which, in contrast to the rest of the figures in this article, two decimal figures were employed in order to depict the low amounts of non-Fe metals recovered from the particle size range 10-4.5 mm.

As is the case for the general mass balance discussed in Section 3.2, figures in Table 2 show a clear common trend with respect to the amount of each grouped material

TABLE 2: Amounts of grouped material fractions per particle size range from mechanical processing.

Particle size range / Grouped material fraction		Amount [wt.%]					
		15 wt.% water content			Dry state		
		25th percentile	Median	75th percentile	25th percentile	Median	75th percentile
90-30 mm	Combustibles	3.46	4.64	5.33	3.37	4.13	4.66
	Inert	13.57	15.26	16.87	15.06	15.47	16.10
	Fe metals	0.20	0.32	0.44	0.46	0.59	1.03
	Non-Fe metals	0.15	0.21	0.27	0.12	0.24	0.34
	Soil-like material	0.88	1.15	1.26	0.91	1.08	1.14
30-10 mm	Combustibles	4.57	4.96	6.72	2.34	2.67	3.16
	Inert	11.93	15.60	17.17	12.51	12.97	14.23
	Fe metals	0.26	0.46	0.55	0.27	0.38	0.55
	Non-Fe metals	0.11	0.15	0.15	0.05	0.07	0.08
	Soil-like material	0.79	0.81	0.87	1.00	1.28	1.37
10-4.5 mm	Combustibles	2.4	2.51	2.71	1.82	1.95	2.27
	Inert	5.84	6.31	6.83	6.06	8.06	9.90
	Fe metals	0.15	0.17	0.17	0.24	0.27	0.29
	Non-Fe metals	0.02	0.02	0.03	0.02	0.03	0.03
	Soil-like material	1.26	1.30	1.46	2.41	2.71	3.24
Fine fractions <4.5 mm	Soil-like material	36.88	39.54	40.27	36.90	37.50	39.12

Notes: Total amounts do not account for 100 wt.% due to losses of material and water (if the case) along mechanical processing and the utilization of quantiles.

fraction in particle size ranges 90-30 mm (in both states), 30-10 mm (in both states) and 10-4.5 mm (only in the optimal water content state), in which most of the material corresponded to the “Inert” fraction, followed by fractions “Combustibles”, “Soil-like material”, “Fe metals” and “Non-ferrous metals”. This information shows that the trend presented by the fine fractions at a general level (particle size range 90-4.5 mm) was also valid at a more specific level (particle size ranges 90-30 mm, 30-10 mm and 10-4.5 mm). Nevertheless, this was the case only to a certain extent, since the same tendency was not identified in the particle size range 10-4.5 mm in the dry state, in which most of the material was allocated to the fraction “Inert”, but, in contrast, the latter was followed by fraction “Soil-like material” instead of by fraction “Combustibles”. This may be the case because, in general, fractions presented a lower amount of surface defilements in the dry state, which in the case of “Combustibles” in a particle size range of 10-4.5 mm might represent a significant loss in terms of mass. However, fractions “Fe metals” and “Non-Fe metals” in the particle size range 10-4.5 mm maintained the same trend as particle size ranges 90-30 mm and 30-10 mm.

In the optimal water content state, the particle size ranges 90-30 mm and 30-10 mm presented similar total amounts of material, with 21.6 wt.% and 22.0 wt.%, respectively; whereas the particle size range 10-4.5 mm accounted for 10.3 wt.%. In the dry state, most of the material was present in the particle size range 90-30 mm with an amount of 21.5 wt.%, followed by the particle size range 30-10 mm with 17.4 wt.% and by 10-4.5 mm with 13.0 wt.%. These figures show that the total amount of material in the dry state tended to decrease according to particle size in the particle size ranges between 90 mm and 4.5 mm. Additionally, the presence of water affected the amount of the particle size range 30-10 mm the most, which altered such trend in the optimal water content. Nonetheless, the presence of water also affected the amount of material in the particle size ranges 90-30 mm and 10-4.5 mm, although to a lesser extent; the 90-30 mm range was the least affected.

As for the total amounts of the grouped material fractions according to particle size range, results show that most of fractions “Combustibles”, “Inert”, “Fe metals” and “Non-Fe metals” was extracted from particle size ranges 90-30 mm and 30-10 mm in both states. Furthermore, the particle size range 30-10 mm was mostly affected by the presence of water, as the difference between the amount obtained in the dry state and the one obtained in the optimal water content state was the greatest in that particle size range. Notwithstanding, the particle size range 10-4.5 mm could be a relevant source of “Inert” fraction and could also be used to obtain an additional amount of “Combustibles”. The amounts of “Soil-like material” increased as the particle size decreased from 90 mm to 4.5 mm in the dry state, whilst in the optimal water content state most of it was obtained in the particle size range 10-4.5 mm, followed by particle size ranges 90-30 mm and 30-10 mm. However, the fraction “Soil-like material” presented fair variations, in general, between the optimal water content and dry states, from which the highest corresponded to the particle size range 10-4.5 mm, followed by 30-10 mm. The grouped material fraction that showed a greater variation due to the presence of water was “Non-Fe metals” in the particle size range 30-10 mm, which showed a significant decrease in terms of amount in the dry state. In turn, fractions “Soil-like material” and “Ferrous metals” presented a significant increase in the dry state with respect to the optimal water content state. These variations are also likely due to the greater amount of surface defilements and agglomerates in the optimal water content state, which can affect the efficiency of separation processes and affect the mass of certain materials.

The previous information shows that the presence of water affected material types and particle sizes in similar and different ways at the same time, since it can increase the mass of a certain material by absorption/adsorption and/or the presence of surface defilements. This may alter the characteristics of that material, which might play a crucial role in a certain processing step (e.g. density separation and sensor-based sorting). Simultaneously, the presence of water can promote the formation of agglomerates that affect the particle size distribution of the fine fractions, which might also play an important role in mechanical processing steps (e.g. sieving and metals separation). In addition, dust generation and material losses were also affected by the presence of water, which presented lower amounts in the optimal water content state.

Comparing the amounts obtained from each grouped material fraction of the mechanical processing in particle sizes 90-30 mm and 30-10 mm in the dry state with those of the material characterization in particle sizes 90-31.5 mm and 31.5-10 mm in Table 1 shows that there were slight deviations among the amounts of both. This might mainly be attributed to the fact that the amounts of the material characterization were the result of the manual sorting of all four batches excavated at the MSG landfill, while those of the mechanical processing were the result of processing batch 1 and batch 2. However, such deviations are minor and, thus, it can be said that the amounts of each grouped material fraction obtained in the mechanical processing are in agreement with the expected quantities.

In order to summarize and evaluate the total obtained amount of each grouped material fraction from the fine fractions by means of the tested mechanical processing in the optimal water content and dry states, the amounts from particle size ranges 90-30 mm, 30-10 mm and 10-4.5 mm were accumulated in a single particle size range (i.e. 90-4.5 mm), while the amounts of “Soil-like material” from those particle size ranges were congregated in the grouped material fraction “Fine fractions <4.5 mm (Soil-like material)”. This information is displayed in Table 3.

In the optimal water content state, total amounts of 40.0-43.6 wt.% “Soil-like material”, 34.1-39.1 wt.% “Inert”, 11.8-12.9 wt.% “Combustibles”, 0.6-1.2 wt.% “Fe metals” and 0.2-0.5 wt.% “Non-Fe metals” were obtained. In turn, total amounts of 41.9-43.9 wt.% “Soil-like material”, 35.9-39.0 wt.% “Inert”, 7.4-10.0 wt.% “Combustibles”, 1.2-1.8 wt.% “Fe metals” and 0.2-0.4 wt.% “Non-Fe metals” were obtained in the dry state.

Generally, it can be concluded that the higher recovered amounts of “Combustibles” and “Non-Fe metals” in the

TABLE 3: Total amounts of grouped material fractions from mechanical processing.

Particle size range / Grouped material fraction		Amount [wt.%]					
		15 wt.% water content			Dry state		
		25th percentile	Median	75th percentile	25th percentile	Median	75th percentile
90-4.5 mm	Combustibles	11.8	12.5	12.9	7.4	9.0	10.0
	Inert	34.1	35.5	39.1	35.9	37.2	39.0
	Fe metals	0.6	0.9	1.2	1.2	1.4	1.8
	Non-Fe metals	0.2	0.4	0.5	0.2	0.3	0.4
Fine fractions <4.5 mm	Soil-like material	40.0	42.9	43.6	41.9	42.7	43.9

Notes: Total amounts do not account for 100 wt.% due to losses of material and water (if the case) along mechanical processing and the utilization of quantiles.

optimal water content state with respect to the dry state can be attributed to the absorption/adsorption of water by some of the materials present in those fractions, such as textiles and leather in the "Combustibles" fraction and "Fine fractions <4.5 mm (Soil-like material)" in the form of surface defilements and impurities in both fractions, rather than to a better performance of the mechanical processing in the optimal water content state. Controversially, almost equal amounts of "Fine fractions <4.5 mm (Soil-like material)" were obtained in both states, while a lower amount of that fraction would have been expected in the optimal water content. This was likely the case because most of the water was absorbed/adsorbed by the "Fine fractions <4.5 mm (Soil-like material)" fraction, which compensated for the amount of the latter lost to the "Combustibles", "Inert", "Fe metals" and "Non-Fe metals" fractions in the optimal water content state. In addition, a larger amount of "Inert" was obtained in the dry state than in the optimal water content state, which may also be explained by the influence of water in all fractions in the optimal water content state. Moreover, results suggest that the recovery of "Fe metals" can be increased by processing the material in the dry state, while the quality of the recovered "Non-Fe metals" can be improved in the same manner.

Furthermore, it can be said that the results from the mechanical processing in both states are in agreement with the total amounts of the material characterization of the fine fractions obtained by Hernández Parrodi et al., 2019a, in dry state: in that study amounts in the ranges of 37.8-55.6 wt.% "Fine fractions <10 mm (Mixed materials)", 31.1-35.4 wt.% "Inert", 2.1-19.7 wt.% "Combustibles", 3.3-4.2 wt.% "Agglomerated fines <10 mm", 0.6-3.4 wt.% "Others" and 0.6-1.8 wt.% "Total metals" were reported. One should take into account the following considerations: i) the fine fractions were segregated to a greater extent in the mechanical processing than in the material characterization (i.e. 4.5 mm vs. 10 mm, respectively), ii) most of the fraction "Others" of the material characterization is expected to be distributed among the fractions "Combustibles" and "Inert" of the mechanical processing, iii) most of the fraction "Agglomerated fines <10 mm" of the material characterization is expected to be in the fraction "Inert" of the mechanical processing, iv) material losses were greater in the mechanical processing than in the material characterization (i.e. ca. 8 wt.% vs. <2 wt.%, respectively), v) amounts of surface defilements and agglomerates were most likely affected by the fact that the same composite samples were used firstly for the material characterization and secondly for the mechanical processing, and vi) the results of the material characterization take into account the 4 excavated batches at the MSG landfill, while the material processing was performed with 2 batches. Moreover, it is relevant to highlight that an additional total amount of over 10 wt.%, distributed among grouped material fractions "Combustibles", "Inert", "Fe metals" and "Non-Fe metals", could be obtained in both states by processing the fine fractions down to a particle size of 4.5 mm.

The results of the mechanical processing in the optimal water content state (water content of ca. 15 wt.%) were not compared to those of the material characterization in raw state (water content of ca. 27 wt.%) due to significant differences in water content. Furthermore, the optimal water content state is considered as an alternative to process the fine fractions in dry state with lower energy demand, material loss and dust emissions. Therefore, the mechanical processing tested in this study can be regarded as a successful approach to separate the fine fractions into sub-fractions in an effective and efficient manner, which facilitate WtM and WtE schemes. Nevertheless, this is to be verified by means of laboratory analysis as a next step.

3.4 Physical appearance of output fractions

In order to document and discuss the physical appearance of all grouped material fractions obtained from the tested mechanical processing, photographs of each output fraction in both the optimal water content and dry states were taken on a grid of 1 cm per 1 cm. Figure 8 displays pictures of the grouped material fractions "Combustibles", "Inert", "Fe metals" and "Non-Fe metals" obtained in the optimal water content state, while images of the same fractions obtained in the dry state are shown in Figure 9.

The comparison of the images in Figure 8 to those in Figure 9 shows that the recovered grouped material fractions "Combustibles", "Inert", "Fe metals" and "Non-Fe metals" presented a greater amount of surface defilements and agglomerates (i.e. impurities), both mainly composed of fraction "Fine fractions <4.5 mm (Soil-like material)", in the optimal water content state than in the dry state. This could be remediated by the implementation of one or se-

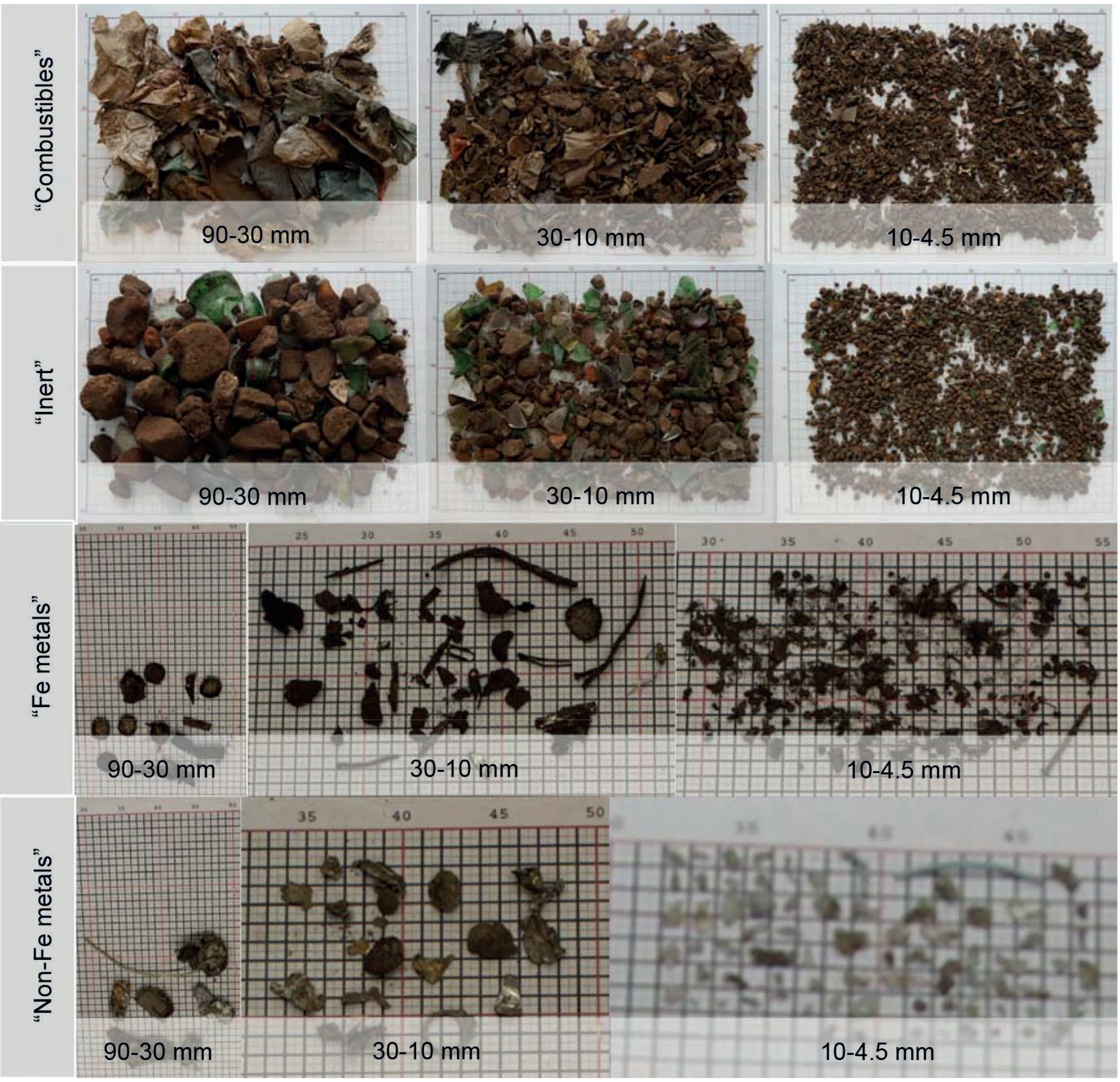

FIGURE 8: Grouped material fractions recovered in the optimal water content state.

veral washing steps, which could significantly reduce the amount of impurities (Tameda et al., 2018) in those fractions. However, the effectiveness of the mechanical processing in terms of its capability to separate the material throughput into the different grouped material fractions did not seem to be greatly affected, since there was no significant discrepancy of materials present in an incorrect grouped material fraction between both the optimal water content state and the dry state.

Regarding the physical appearance of fraction "Fine fraction <4.5 mm (Soil-like material)" in the optimal water content and dry states, Figure 10 shows that its visual characteristics did not differ significantly between both states. Nonetheless, a greater amount of agglomerated material <1 mm could be expected in the optimal water content state and, therefore, a reduction of the water content would be necessary for an adequate further dry mechanical processing of this fraction.

It is important to reiterate that laboratory analysis of the fractions "Combustibles", "Inert" and "Fine fractions <4.5 mm (Soil-like material)" are to follow the present study, in order to determine quantitatively if the applicable specifications for the foreseen purposes have been met by either one or both states, and therefore it cannot be yet assured, that the obtained outputs can be subject to valorization schemes of WtM and WtE.

4. CONCLUSIONS

In this study total amounts of 40.0-43.6 wt.% "Fine fractions <4.5 mm (Soil-like material)", 34.1-39.1 wt.% "Inert", 11.8-12.9 wt.% "Combustibles", 0.6-1.2 wt.% "Fe metals"

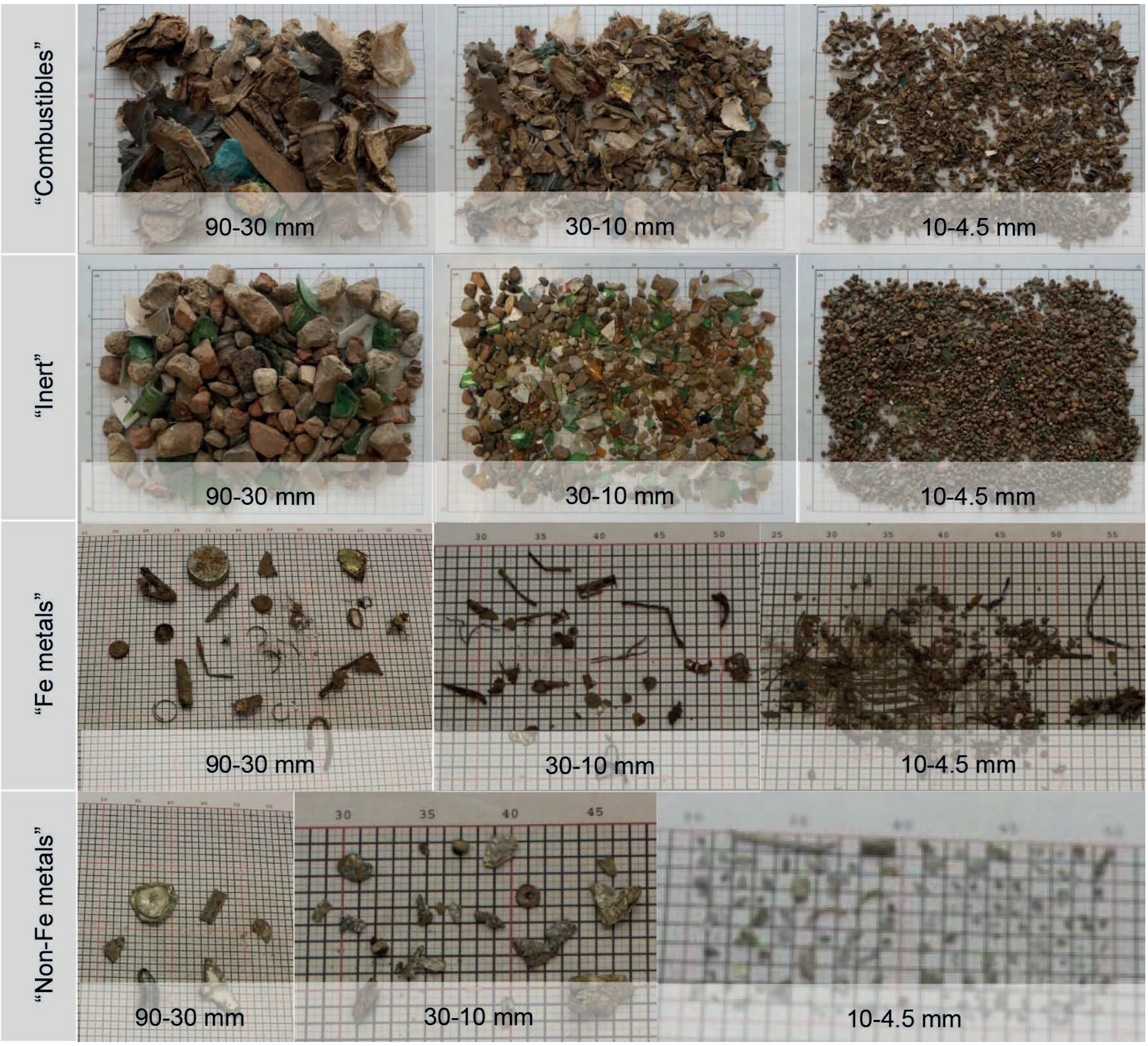

FIGURE 9: Grouped material fractions recovered in the dry state.

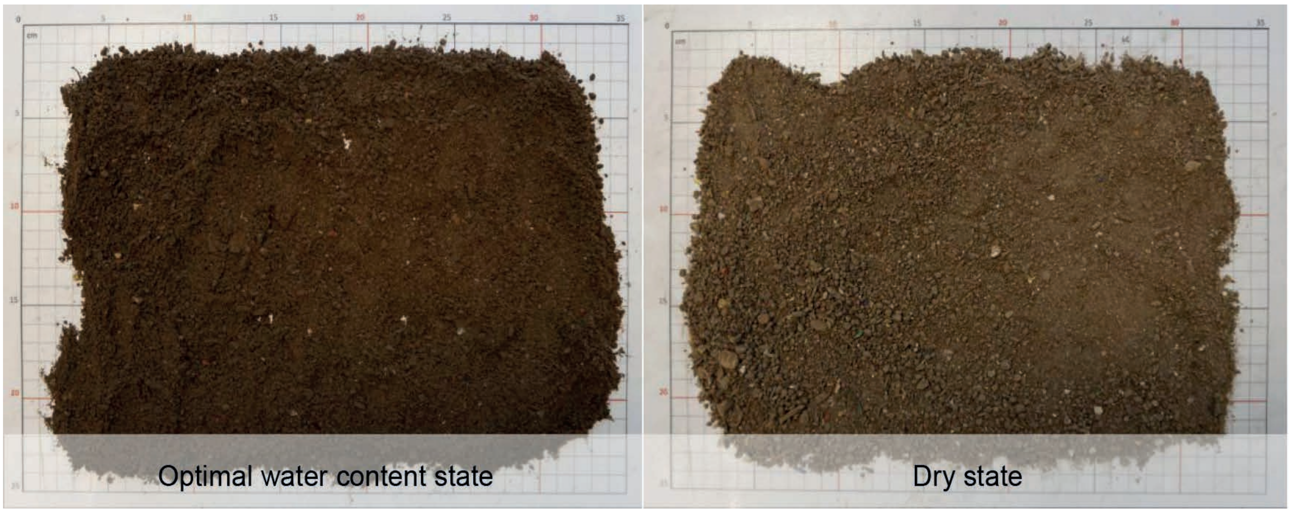

FIGURE 10: "Soil-like material" fraction in the optimal water content and dry states.

and 0.2-0.5 wt.% "Non-Fe metals" were obtained in the optimal water content state, while amounts of 41.9-43.9 wt.% "Fine fractions <4.5 mm (Soil-like material)", 35.9-39.0 wt.% "Inert", 7.4-10.0 wt.% "Combustibles", 1.2-1.8 wt.% "Fe metals" and 0.2-0.4 wt.% "Non-Fe metals" were generated in the dry state. These figures agree with the amounts determined in the material characterization of the fine fractions. Hence, it can be stated that the tested mechanical processing succeeded in sorting the fine fractions into the targeted grouped material fractions in an effective and efficient manner. Additionally, results suggest that a significant total amount of the fine fractions could be recovered through the implemented mechanical processing approach, which might contribute to the overall economic and environmental feasibility of the project in case of implementing full scale (E)LFM at the MSG landfill.

In general, the grouped material fractions recovered in the optimal water content state presented a higher amount of surface defilements and agglomerates (i.e. impurities) than in the dry state, from which fractions "Fe metals" and "Combustibles" seemed to be the most affected. Particle size range 30-10 mm appeared to be the most affected by the presence of water, while particle size range 90-30 mm was least affected. Dust generation and material losses were also influenced by the presence of water, which presented a slightly lower amount in the optimal water content state than in the dry state. Particle size ranges 90-30 mm and 30-10 mm yielded most of the recovered material, and particle size range 10-4.5 mm could be a relevant source of "Inert" fraction, as well as provide an additional amount of "Combustibles". A total amount of over 10 wt.%, distributed among all grouped material fractions of particle size range 10-4.5 mm, was additionally obtained by processing the fine fractions until a particle size of 4.5 mm.

It can be concluded that the real amounts of each grouped material fraction to be recovered from the fine fractions correspond to those obtained in the dry state, as well as the real material distribution according to particle size range. Discrepancies between the amounts obtained in the optimal water content and dry states can be mainly attributed to absorption/adsorption of water by the different types of materials present in each grouped material fraction. Additionally, these discrepancies can be due to the presence of surface defilements and agglomerates among the different grouped material fractions and particle size ranges, which in turn can affect the properties of certain materials (e.g. shape, mass and density), as well as the performance of sorting processes (e.g. particle size and density classification, magnetic and eddy-current separation and sensor-based sorting). The presence of impurities can also decrease the quality of certain materials, since they can be associated with the presence of heavy metals and organic pollutants. This could undermine the potential for the valorization of such materials in WtM and WtE schemes. However, discrepancies between both the optimal water content and dry states in this study were found to be negligible with respect to the success of the mechanical processing to separate the fine fractions into the different grouped material fractions. Nevertheless, laboratory analyses are yet to be performed in order to evaluate the effects, in terms of quality, of the greater amount of impurities present in the fractions obtained in the optimal water content state than those in the dry state. Moreover, laboratory analysis of outputs "Combustibles", "Inert" and "Fine fractions <4.5 mm (Soil-like material)" will determine if these fractions can be used for the intended purposes (i.e. alternative fuel, substitute for construction aggregates and substitute for soil in construction applications, respectively) or if further treatment might be necessary.

It is important to note that the mechanical processing approach tested in this study was carried out by means of small- and pilot-scale equipment, and results may differ substantially in large-scale machinery. Furthermore, the results of this study are case specific and much attention must be paid to several factors when transposing this information for the purposes of future investigations and full-scale applications. Moreover, it is worth stressing that the current market value of secondary raw materials, such as substitutes for construction aggregates and soil in construction applications, can be very low or even have negative values, as is the case with RDF in some countries. Additionally, the extent of the mechanical processing of the fine fractions is directly proportional to its cost and highly concatenated with the quality of its outputs. Besides, usually most of the landfill-mined material corresponds to fine fractions and, hence, they can hardly be left out of the scope of (E)LFM projects. Therefore, the profitability of (E)LFM is directly linked to a successful recovery of materials and energy from the fine fractions. Hence, the mechanical processing of the fine fractions is to be designed in such an optimal way that the applicable quality standards of the desired outputs can be met, and capital and operational expenditures do not hinder the viability of the whole project.

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QUALITY ASSESSMENT OF NONFERROUS METALS RECOVERED BY MEANS OF LANDFILL MINING: A CASE STUDY IN BELGIUM

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ABSTRACT

Nonferrous metals (NFMs) provide a major contribution to potential revenues generated by the implementation of landfill mining (LFM). However, metals present in landfills undergo stronger degradation than during regular use, likely resulting in a lower quality compared to conventional scrap. Nowadays, information relating to the most common metals found in LFM projects is readily available, although no consistent quality data can be attained. In general, excavated landfill material is processed mechanically through a series of different steps, including screening and metal separation by magnetic and eddy current separators. This study focuses on the characterisation of NFMs recovered from a specific landfill site in Belgium, with the aim of assessing the quality of each NFM for marketing purposes. The study also addresses the issue of metal concentration and defilements detected, with a preliminary evaluation indicating a total of 5 kg of NFMs per ton of excavated material processed at the Mont-Saint-Guibert landfill. In addition, the application of thermal treatment enabled the observation that, on average, only 70 wt% of the nonferrous fraction is metallic. The majority of surface defilements (30 wt%) are represented by a combination of organic and inorganic impurities that are strongly bound to NFMs. Consequently, the different scraps extracted and the eventual destination of each were technically assessed using two separate approaches. The first approach facilitated the potential recovery of seven types of scraps from NFMs, including two different qualities of Al scrap, two of Cu, one of Pb, one of Zn, and one of stainless steel. In line with the second, and perhaps more realistic approach, NFMs may be directly marketable from the landfill as mixed nonferrous scrap.

1. INTRODUCTION

NFMs such as Cu, Al, Zn, Pb, Cr, Ni, Ag and Au, are distinguished from ferrous metals (FMs) based on their low or zero magnetization when in the proximity of a magnetic field. NFMs are perhaps the most valuable secondary raw material found in several types of waste, including electronic waste, construction and demolition (C&D) waste, industrial waste, and municipal solid waste (MSW). Many NFMs are considered strategic metals in Europe (Roadmap to a Resource Efficient Europe) and used in countless applications. Accordingly, the European Commission has prioritised sustainable access to critical raw materials through the recycling and reuse of waste.

In the field of NFM sorting, eddy current separators (ECSs), dense media separators (Barker, 2014) and hand sorting (Capuzzi and Timelli, 2018) are commonly employed. Furthermore, sorting technologies have been developed to automate and optimise the sorting processes: X-ray fluorescence (XRF), colour sorting and X-ray tomography (XRT) can be used for different qualities of Cu, Al, Pb, Zn and stainless steel scrap (Dürkoop et al., 2016; Schlesinger, 2013; Schlesinger et al., 2011).

The valorisation of scraps from non-conventional sources such as, for instance, MSW, C&D, or even landfilled waste (LFW) indicates the need for a detailed understanding of both the concentration and quality of the metals and the number of steps or techniques applied to separate the latter into different metal categories and grades. For example, Soo et al. (2019) investigated the influence of different sources of Al-scraps from an aluminium recycling facility in Belgium demonstrating how the quality obtained was linked to particle size and metal source.

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Several authors have highlighted the accumulation over time in a large number of landfills of a vast quantity of materials, which might be suitable for use as potential secondary resources (Kapur and Graedel, 2006; Lifset et al., 2002; Muller et al., 2006; Quaghebeur et al., 2013). For example, according to Krook et al. (2012), the amount of Cu landfilled worldwide is comparable to the existing stock in use within the technosphere. Likewise, Cohen-Rosenthal (2004) reported an amount of Al and steel stored in a site investigated corresponding to approx. 1,000 tons and 12,000 tons, respectively, per excavated hectare.

According to Winterstetter et al. (2015), and Van Vossen and Prent (2011), in addition to the most common economic drivers such as reclaimed land or avoidance of repeated landfilling costs, NFMs contribute extensively to the revenues of LFM. However, metals remain buried for several decades, potentially subjected to corrosion and pollution before being excavated from a landfill. There is a clear lack of information in the literature relating to metal concentration, quality and marketability of these scraps. Table 1 illustrates the number of metals found in several LFM projects, not all of which indicated the proportion of FMs and NFMs; indeed, only the Austrian LAMIS project indicated the actual concentration of the ferrous fraction obtained by mechanical processing after a pyrometallurgical trial using an induction furnace (Wolfsberger et al., 2015).

Therefore, the main research questions addressed by this study are:

- What is the actual concentration of NFMs, and what is the extent of surface defilements ?
- Can the recovered NFMs be commercialised and, if so, under what standard?

2. MATERIALS AND METHODS

2.1 Site description

NFMs analysed in this study were excavated from a landfill site located in the municipality of Mont-Saint-Guibert (MSG) in the province of Walloon Brabant, Belgium (Figure 1). This site covers an area of approx. 44 ha, which has been in operation since 1937 as a sand quarry and was transformed in 1958 into a disposal site for MSW, C&D waste and non-hazardous industrial waste (ISSeP, 2011). The excavation took place in the oldest part (red delimited area in Figure 1a) which covers a surface of 14 ha, storing circa 5.7 million m^3 of waste (Hernández Parrodi et al., 2019; IGRETEC, 1994).

For the purpose of this study, a small zone of circa 130 m^2 from the old part of the landfill (white delimited area in Figure 1b) was selected for excavation based on the results of the geophysical exploration (García López et al., 2018; Hernández Parrodi et al., 2019a).

2.2 Excavation works and material pre-processing

The selected area was excavated to a depth of approx. 5 m (excluding 4 m of cover layer) and roughly 370 ton of LFW extracted (Figure 2a). The excavated volume (425 m^3) was divided into four sub-volumes (batches 1-4) of 140 m^3, 100 m^3, 120 m^3 and 65 m^3, respectively. The batches were classified in situ according to type of waste. Batches 1 and 2 were mainly composed of MSW and C&D, while batch 3 was largely comprised of C&D and batch 4 MSW. These batches had previously been processed using a ballistic separator (Figure 2b) in two steps, producing three different outputs: 3D, 2D and under-screen fractions. In the first step, the ballistic separator used a screen of 200 mm, while

TABLE 1: Concentration of metals in previous LFM investigations (Hernández Parrodi et al., 2018).

Bibliography	Van Vossen and Prent, 2011 (various countries)	Kaartinen et al., 2013 (Kuopio, Finland)	Quaghebeur et al., 2013 (REMO, Belgium)	Wolfsberger et al., 2015 (Lower Austria, Austria)	Jani et al., 2016 (Högbytorp, Sweden)	Bhatnagar et al., 2017 (Kudjape, Estonia)	Garcia Lopez et al. 2018 (MSG, Belgium)
Type of waste disposed	Various	MSW	MSW	MSW	MSW+C&D	MSW	MSW+ C&D
Total metals	2.0%	3.0-4.0%	2.8 ± 1.0%	2.1-4.7%	1.0%	3.1%	2.9%

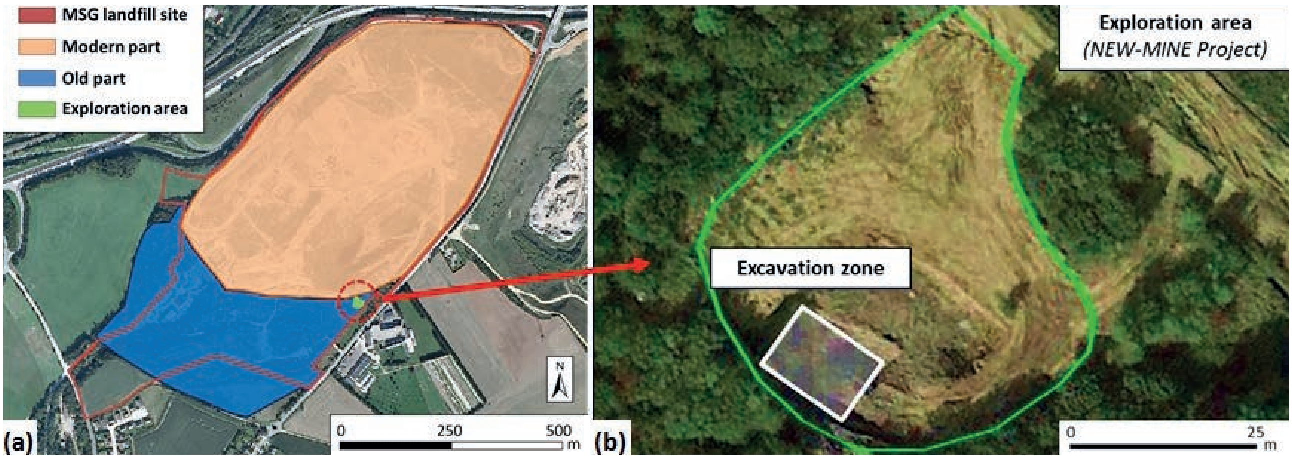

FIGURE 1: MSG landfill (a) and excavation zone (b) (Hernández Parrodi et al., 2019a).

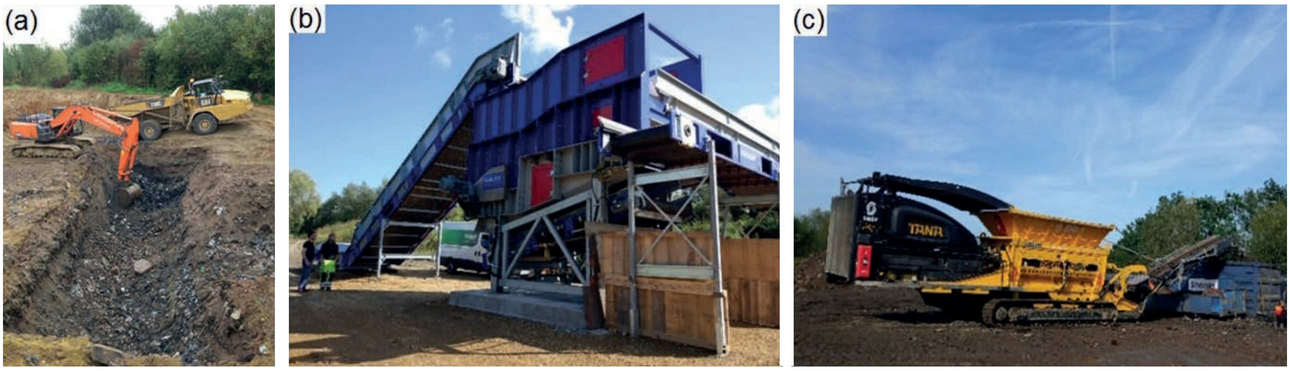

FIGURE 2: (a) Excavator and dumpster; (b) ballistic separator; (c) mobile shredder.

the second step was performed with a screen of 90 mm.

The under-screen fraction below 90 mm was defined as the fine fraction, whereas materials with a particle size ≥90 mm (3D ≥200 mm, 2D ≥200 mm, 3D 200-90 mm and 2D 200-90) corresponded to the coarse fraction. The 2D fraction from the first step of the ballistic separation (≥200 mm) was processed in situ using a mobile shredder equipped with a built-in over-belt magnetic separator (Figure 2c) to recover FMs. The rest of the material was sampled and subsequently processed. Further information about this landfill site and the characteristics of the excavated material can be found in García López et al., 2019, and Hernández Parrodi et al., 2019a.

2.3 Material processing

During the excavation, samples were obtained as prescribed by the German Directives LAGA-PN78 and LAGA-PN98. With the exception of the 3D fraction ≥200 mm, manually sorted in situ, the remaining fractions were first dried at 75°C (based on DIN CEN/TS 15414-1 to prevent loss of volatile matter and degradation of certain plastics) and then processed at the Department of Processing and Recycling (IAR) of RWTH Aachen University. Table 2 summarises the sampling of different fractions. A detailed description of the methodology and material composition can be found in García López et al. 2019 and Hernandez Parrodi et al., 2019b.

Different methods of metal extraction were chosen according to particle size. NFMs in the coarse fractions (200-90 mm and ≥200 mm) were retrieved manually following the recovery of FM using different types of magnetic separators. On the other hand, the fine fraction (<90 mm) was subdivided into three particle size ranges, i.e. 90-30 mm, 30-10 mm and 10-4.5 mm to enhance the recovery of FMs and NFMs using magnetic separators (over band and drum magnetic) and ECSs, respectively. The fraction <4.5 mm was not processed further for recovery of FMs and NFMs. Further details regarding the mechanical processing of fine fractions are reported in Hernández Parrodi et al. 2019b.

TABLE 2: List of processed samples.

Fraction	Nº of samples	Mass processed [kg]	Batches analysed
3D≥ 200 mm	Processed in-situ		
2D≥200 mm*	30	413	1, 2, 3 & 4
2D 200-90 mm	21	474	1, 2, 3
3D 200-90 mm	23	203	1, 2, 3
<90 mm	16	200	1 & 2

** Note: fraction shredded down to 275 mm*

NFMs from both coarse and fine fractions were separated further by manual sorting and divided into different categories: non-magnetic Fe scrap, Cu scrap, Al scrap, heavy scrap and unknown metals.

Following this preliminary separation, a portable XRF analyser (Thermo Fisher NITON XL3t 600) and a digital balance were used to analyse the chemical composition and weigh each metallic particle respectively in order to perform a quantitative analysis.

On average, almost 240 different metallic pieces were analysed, of which 74 wt% originated from the fine fractions. Based on their chemical composition and morphology, metal particles were organised into the categories listed in Table 3.

Figure 3 summarises the characterisation process carried out in this study from the excavation and mechanical processing outputs to the assessment of metallic scraps.

2.4 Assessment of metal quality

To assess defilements bound to NFMs and actual metal concentration, three different approaches were tested. The first method was based on ultrasonic cleaning as most of the impurities remained attached to the metals after cleaning for 20 minutes. The second approach was the same method used in the Austrian LAMIS Project (Wolfsberger et al., 2015) based on the smelting of scraps in a raw state. Smelting provides a detailed insight into the metal grade and alloy content, although providing only a rough estimation of defilements, as was the case with Al scrap recovered from MSG (Lucas et al., 2019). For example, Al has a high affinity for oxygen, and during smelting and casting lost around 50% of its mass as a result of oxidation (Samuel, 2003).

Carbon-rich defilements may act as reductants of metal oxides or favour the formation of carbides during smelting of different NFMs, which in many cases is critical and undesired. As an example, metal oxides such as Ti (i.e. from

TABLE 3: Classification of metal categories from NFM samples of MSG landfill.

Category name	Details
Al-foils	Foils of aluminium usually used for food preparation
Al-packaging	Used beverage cans (UBC), Tetrapack© and other aluminium packaging such as aerosol cans, aluminium wrapping, etc.
Al-alloy	Remaining aluminium particles found in waste, not included in Al-foils and Al-packaging
Brass	Metallic particles mainly composed of Cu and Zn
Cu-wires	Electric wires
Pb-alloy	Metallic particles in which Pb was the main element
Stainless steel	Nonmagnetic Fe-scrap containing elements such as Cr and Ni
Zn-alloy	Metallic particles in which Zn was the main element
Other metals	Rest of the metals found with a low frequency such as bronzes, silver alloys, iron scraps (mainly Fe-Sn food cans), nickel alloys, etc.

the coating of Al cans), Fe, Si, Zn, found in a vast number of natural minerals and soils can be easily reduced by Al during smelting and end up in the metallic phase as alloying (Schlesinger, 2013; Schmitz et al., 2006)

Finally, the best results were obtained using thermal treatment based on the de-coating process used by the Al industry in the treatment of used beverage cans (UBC) prior to smelting (Schmitz, 2006; XIAO et al., 2005). Al is potentially the NFM most heavily affected by thermal treatment due to its high affinity for oxygen. Indeed, literature reports

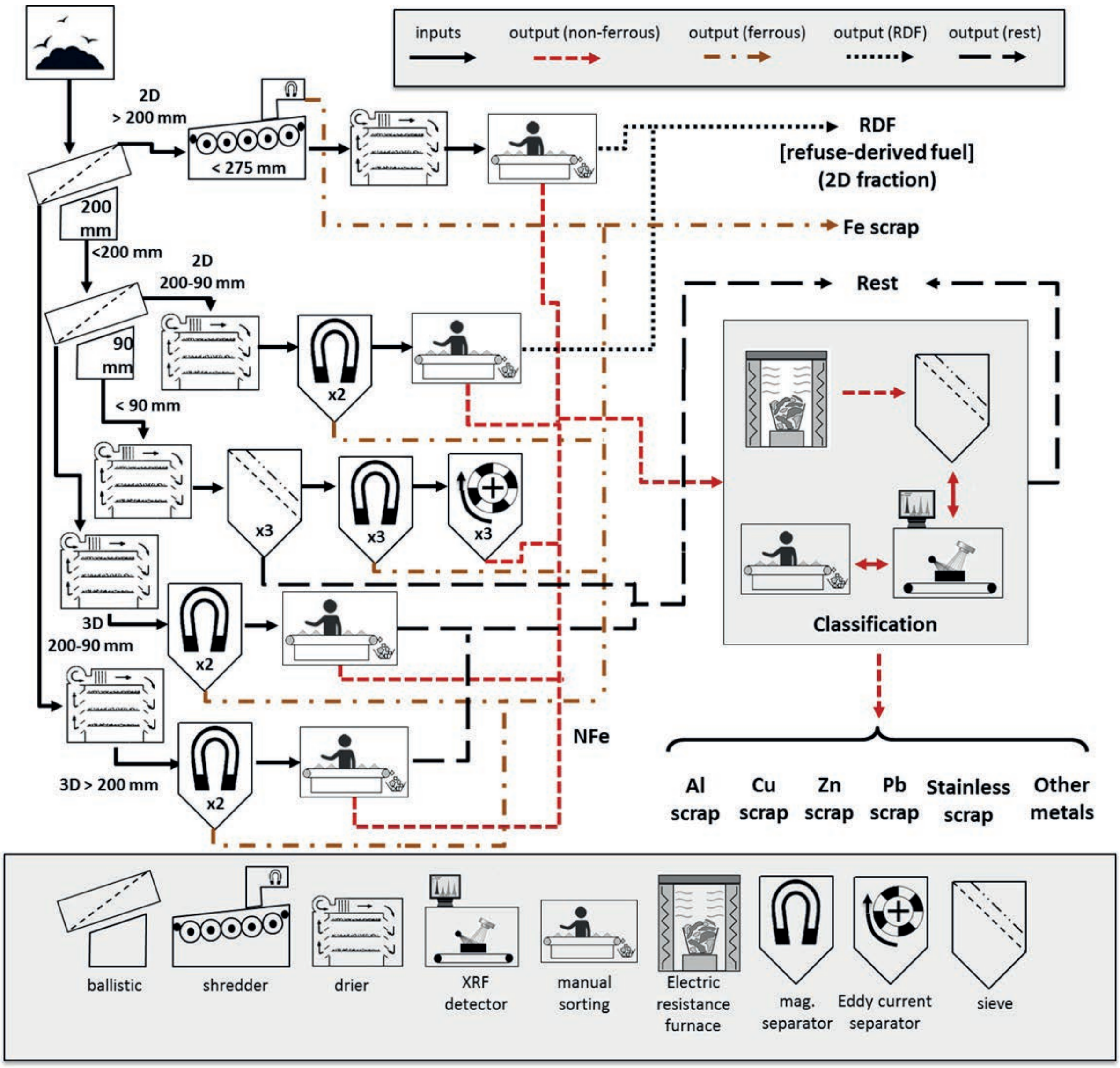

FIGURE 3: Scheme of NFM processing of excavated LFW.

recommend temperatures ranging between 400-450°C to minimise metal losses during this step (Schmitz et al., 2006). The treatment temperatures applied to NFMs were invariably below boiling point, with Al, Fe, Cu and other less common scraps, for instance, Ni or Ag, remaining unmolten during treatment. In addition to Al, the most critical scraps were those compounded mainly by Pb and Zn, which are liquid at 450°C. These materials, smelted separately, showed no critical degradation during treatment. With regard to other possible volatile compounds such as water or salt, scraps had previously been dried during the material processing, and as the melting point for the majority of salts is above 750°C, these compounds were not expected to volatilise during thermal treatment.

For the reasons stated above, the scraps listed in Table 3 were treated at 400/450°C for 30 min in an air atmosphere using an electric resistance furnace. A thermocouple type K placed inside the crucible guaranteed temperature control.

Scraps incinerated inside 0.4 litre-clay crucibles were weighed before and after thermal treatment using a high precision balance. Weight loss registered between the input and output of each incinerated scrap category was interpreted as the organic content. Following thermal treatment, outputs were sieved at 1 mm, washed and finally dried at 100°C for 24 hours. Large non-metallic particles such as rock, ceramics or glass were removed by hand. The weight difference between incineration output and cleaned metals was taken as inorganic content.

With regard to the marketability of metal scraps, it should be highlighted how industry standards are used as references in scrap trading. Several of these standards and their denomination codes are summarised in the Scrap Specifications Circular which is updated every year. However, no specific denomination has been coined for unconventional sources of mixed nonferrous scraps such as those originating from MSW or LFW. Hence, prices are subject to agreement between buyers and sellers, and essential aspects such as the variety of metals contained in the scraps, and the concentration and nature of defilement should be given due consideration, particularly as the number of separation steps and final destination of these metals are heavily dependent on the latter. For example, Soo et al. (2019) studied the influence of different Al fractions recovered from a Belgian recycling facility. In this study, fractions <12 mm and ≥40 mm exhibited a large number of undesired alloys (Fe, Cu, Zn, Si, among others) after smelting; moreover, a particular fraction mixed with Fe scrap was separated manually, increasing not only operative costs but also producing low Al grades with a marginal profit.

To date, the market is devoid of reference standards for scraps recovered from landfills. Using information collected from incineration outputs, seven potential marketable scraps that adhered closely to the standards applied in the scrap market were defined and are listed in Table 4.

Further to dividing NFMs into a series of different scrap categories, metal concentration should also be taken into account in order to estimate the potential value. In this study, the price of each scrap category ($P_{scrap}^{landfill}$) extracted from landfill was defined as the product between the price of the closed standard scrap ($P_{scrap}^{standard}$) listed in Table 5, divided by its concentration of metals ($M^{standard}$ pre-established in the Scrap Specifications Circular), and multiplied by the metal concentration of each pre-defined category ($M^{landfill}$). This methodology is summarised in equation 1.

$$P_{scrap}^{landfill} = P_{scrap}^{standard} \times M^{standard} \; / M^{landfill} \qquad (1)$$

3. RESULTS

3.1 Mechanical processing and manual sorting

Manual and mechanical sorting of ballistic separation output fractions revealed that metals represented 2.9 wt% of input material, with NFMs constituting only 16.5%.

Notably, the distribution of metals across particle size was largely similar to the distribution registered for input material (see Table 6), i.e. no particular abundance of metals across any of the particle size fractions.

Figures 4a and 4b illustrate the results obtained for NFMs classified according to the category of metals detected most frequently from fine and coarse fractions, respectively.

The category "Others" includes minor and less common scraps such as bronze, steel-tin cans, Ni scraps and the considerably less common Ag scraps (silverware). All these scraps together represented less than 1.9 wt% of total NFM fraction.

TABLE 4: Categories of potentially marketable NFM scraps.

Category name	Details
Al-scrap	
Al-scrap I	High grade: fraction 10-90 mm and which is mainly composed of UBC and to a lesser extent Al-foils.
Al-scrap II	Low grade: rest of Al scraps.
Cu-scrap	
Cu-scrap I	High grade: the category Cu-wires from the fraction ≥90 mm
Cu-scrap II	Low grade: rest of NFMs, except Zn, Pb and stainless-steel scraps
Pb-scrap	Mixed Pb scrap
Zn-scrap	Mixed Zn scrap
Stainless-steel scrap	Mixed stainless steel scrap

TABLE 5: Prices of nonferrous scrap in Jun 2019.

ISRI code	Category	Price (euro/ton)
SCORE	Zn mixed scrap	600
SABOT	18/8 stainless steel solids	720
RACKS/RADIO	Pb soft scrap	1,100
NA	Pb scrap	1,000
RAINS	Pb scrap from auto batteries	820
BERRY	Dry bright wire	4,000
CANDY	Copper Wire and Tubing Scrap	3,450
DRUID	Insulated Cu wire (85% recovered scrap)	1,900
BIRCH	Copper wire N°2 with a metal purity > 96%	3,250
ZEBRA	Mixed brass	2,400
NA	Heavy brass	2,400
ZORBA	Zorba90 (90% of NFMs)	500
DROID	Insulated copper wire N°2 with a metal purity > 96% (45% recovered scrap)	1,100
TAKE, TALC, TALCRED	UBC	1,400
TAINT/TABOR	Aluminium foil	200

Source: www.recycling.com; www.scrapmonster.com

Cu and Al scraps represented approx. 80% of NFMs recovered from this landfill. Al scraps, mainly present as foils and packaging, were the most common metals found in the fine fraction (44.5 wt%) followed by Cu scraps (Brass and Cu-wires) (33.4 wt%). In the coarse fraction, Cu scraps were the most widely detected NFMs with 50.2 wt%.

Pb and Zn scraps were mainly detected in fine fractions; Pb scraps originate from a series of sources, including automotive battery parts, old plumbing, and roof covers, among other unknown sources, whilst Zn is mainly used as anticorrosive layers in Fe scraps and as an alloy in Cu. However, Zn scraps within NFMs were also found frequently as mechanical components or in alkaline batteries.

Stainless steel scraps were recovered largely from the coarse fraction by hand-sorting, whilst ECSs retrieved only small amounts in the fine fraction. The fine fraction may however contain more stainless steel than the quantities retrieved. The lift-off for stainless steel in ECSs is the lowest of all NFMs (76 times lower than Al), with small pieces tending to remain in the waste when impurities have adhered to the scraps (AbdAlla et al., 2019; Kristian Kahle, Ramboll et al., 2015; Spencer and Schlömann, 1975).

3.2 Thermal processing

Figures 5 and 6 summarise the results obtained from the thermal processing and chemical analysis of each metal sample.

TABLE 6: Mass distribution of LFW processed with ballistic separators.

Screening results [wt%]	Input material	Ferrous (2.39 wt% input)	Non-ferrous (0.47 wt% input)
≥200 mm	6 %	8 %	8 %
200-90 mm	16 %	24 %	18 %
< 90 mm	78 %	68 %	74 %

In general, the fine fractions contained more impurities than coarse fractions (Figure 7). Large metallic particles, largely from the category of C&D waste, did not contain more than 15 wt% of defilements (see all categories in Figure. 5b, with the exception of Al-foils and Al-pack). The only exception was Cu-wires, for which polymeric insulating cover accounted for almost 50 wt%.

With the exception of categories Al-foils, Al-pack and other metals which contained degraded biological matter, the remaining burnable defilements were mostly polymers and carton used in association with various metals.

The chemical compositions presented in Figures 6a and 6b reveal how the two main fractions analysed (< 90mm and ≥90 mm) displayed similar ratios of defilements (circa 30 wt%). In contrast, the proportion of NFMs differed, with Al representing the most widely present traditional metal in the fine fraction, followed by e.g. Cu and Zn, both of which commonly found in MSW (Morf et al., 2013).

Contrary to expectations, the typical chemical composition of scraps from the coarse fraction (originating largely from C&D waste) was dominated by Cu (34.1 wt%), followed by Al (15.6 wt%), rather than vice versa. The important chemical elements Fe, Ni and Cr were also detected due to the presence of stainless steel.

3.3 Assessing the marketability of NFMs

Assessment of the marketability of scraps based on the results obtained by mechanical separation may lead to an inaccurate conclusion as to the potential profitability of these recovered metals. One clear example is represented by Cu. In the coarse fraction, the most frequently detected scrap was Cu-wires (39.2 wt%) (see Cu-wires in Figure 3b). However, when applied to this category, thermal treatment revealed a metal concentration of only 47.8 wt% (see Cu-wires in Figure 5b), thus indicating a lower concentration of actual Cu in the coarse fraction.

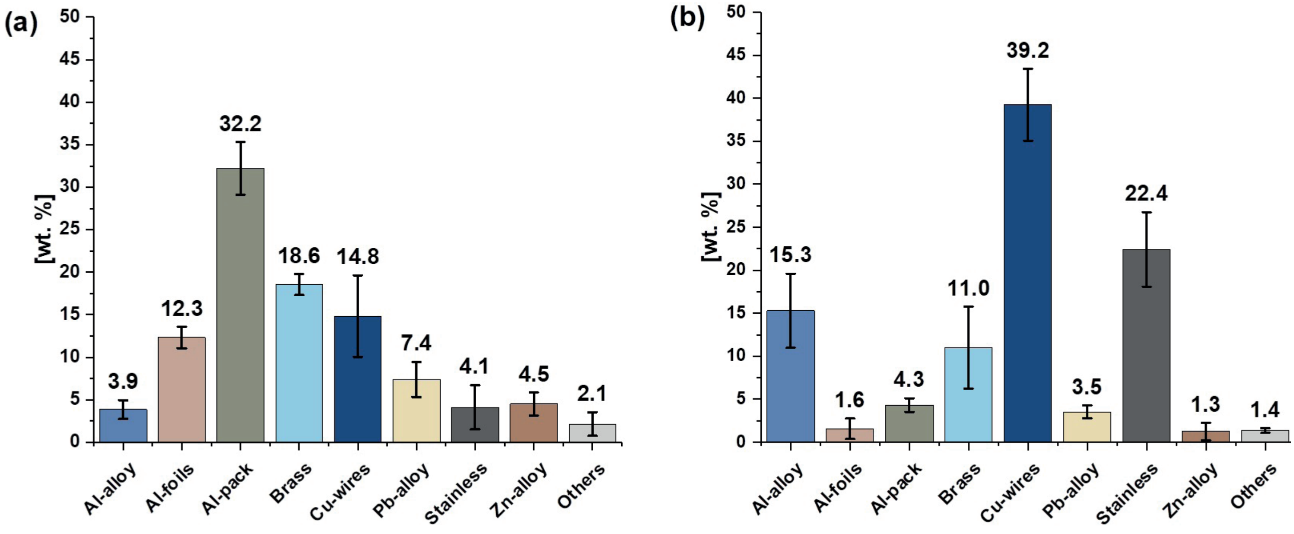

FIGURE 4: Categories of NFMs after mechanical separation [wt%]: (a) fine fractions (<90 mm), (b) coarse fractions (≥90 mm).

3.3.1 Metal concentration and surface defilements

To focus on the first question asked at the start of this paper, relating to metal concentration and surface defilements in NFMs, the results obtained by thermal treatment indicated the presence of approx. 27.3 wt% defilements (Figure 6c), of which 20.8 wt% organic (burnable) and 6.5 wt% inorganic. Both fine and coarse fractions displayed similar trends of defilements (see Figures 6a and 6b). Nevertheless, defilements in fine sub-fractions (4.5-10 mm .10-30 mm and 30-90 mm) showed significant discrepancies, in particular the fraction 4.5-10 mm with impurities amounting to approx. 40-50 wt% (Figure 7).

The origin of these impurities however varied. Defilements in the fine fraction were related to landfilled MSW in which polymers present in packaging and decomposed organic matter in contact with soil material had become bound to the metals over the years.

On the other hand, in the coarse fraction, organics were represented mainly by polymers which had formed composites with metals such as Cu-wires, with a polymeric fraction of roughly 41 wt%. Residual soil materials agglomerated on the metal surface represented the main source of inorganic defilements. Glass and ceramics used in electrical applications, such as switches, fusible plugs, and light bulb holders, among other materials, were detected bound to several metals.

The concentration of metals ranged from 56.5 wt% in the fraction 4.5-10 mm to 74.1 in the fractions ≥90 mm (Figure 7). The fraction 30-90 mm yielded the highest number of metals, driven by Al packaging such as UBC, Al foils and different types of Cu alloys.

3.3.2 Marketable scraps

To answer the last question raised, relating to the commercialisation of the scraps identified, two approaches were used in order to assist decision-makers during the

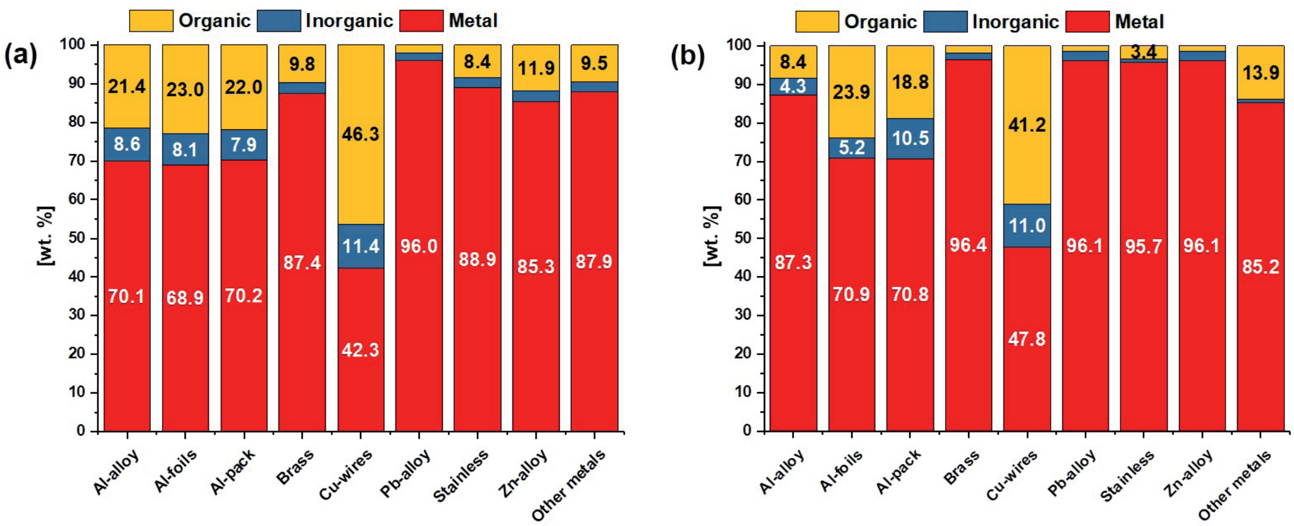

FIGURE 5: Concentration of metal, organic and inorganic matter present in each NFM [wt%]: (a) fine fractions (<90 mm), (b) coarse fractions (≥90 mm).

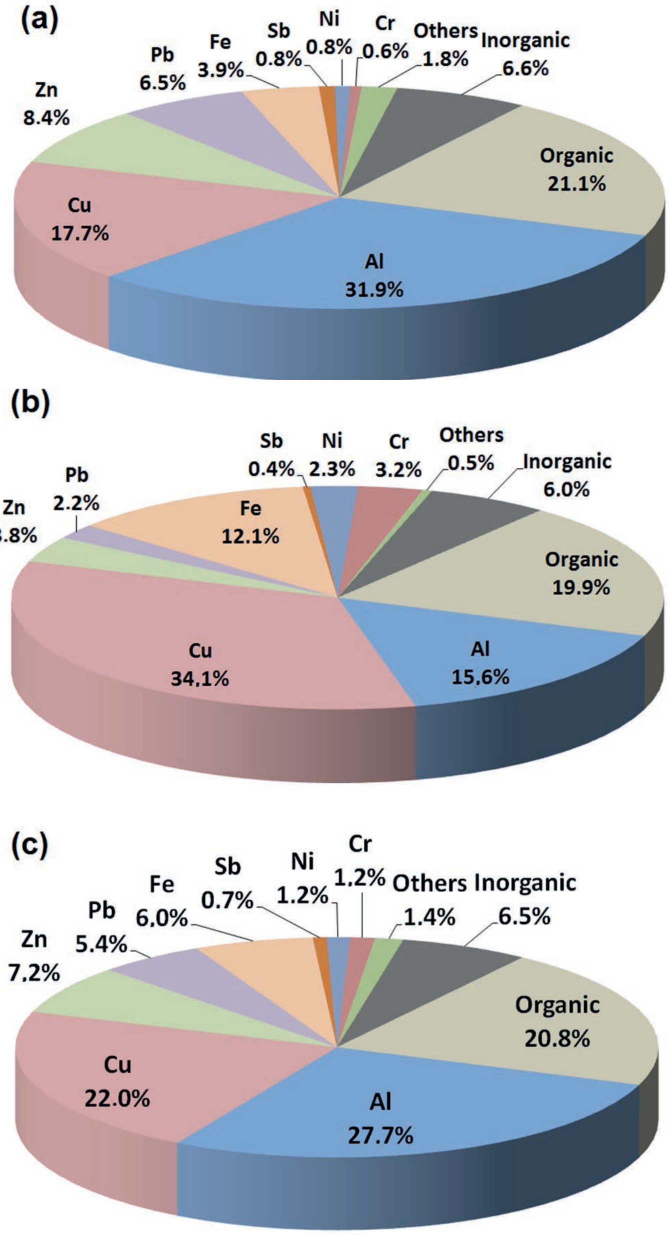

FIGURE 6: Chemical composition of NFMs: (a) fine fractions (<90 mm), (b) coarse fractions (≥90 mm), and (c) global composition.

assessment of upcoming LFM projects.

During the excavation and treatment of LFW in situ, magnetic separators and ECSs may be installed for use in metal separation. On applying these technologies, the extracted NFMs would then be suitable for marketing as a mixed nonferrous scrap. In the presence of Al as the majority metal contained in NFMs, as was the case in the present study, NFMs are traded under the standard identified as Zorba (Scrap Specifications Circular, 2017). This scrap category also requires the addition of two numbers affirming metal concentration.

The price of Zorba90 (90 refers to 90% metal concentration), typically detected in C&D and MSW, is in the range of 500 euros per ton (Table 5). Within the framework of the analysis carried out here, NFMs from MSG was defined as Zorba70, where 70 is an approximate representation of the metal concentration found in this study. Under these terms, NFMs from MSG are routinely marketed for no more than 400 euros per ton.

Whether it is the company carrying out the LFM project or a specialised recycling company to deal with the NFMs is of little concern, the different fractions can still be separated and valorised and the sum remunerated may at times even increase two or three-fold. Under this optimistic approach, Figure 8a, 8b and 9 summarise the results of the seven marketable scraps in terms of proportion detected, contribution of each fraction (<90 mm and ≥90 mm) and metal concentration, respectively.

Al scrap

As stated previously, Al is the most commonly detected NFM, with Al pieces frequently containing Mg, Mn, Si, Fe,

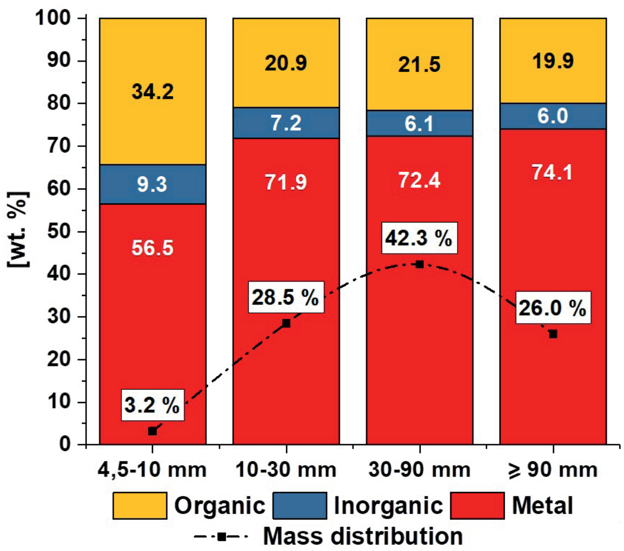

FIGURE 7: Concentration of metal and defilements in NFMs according to particle size [wt%].

and Cu as alloying. The type and amount of alloying in the metal composition is related to the origin of the scraps. Al scraps in the fraction <90 mm originate largely from packaging and foils, in which the metals have a negligible alloying content. On the other hand, the coarse fraction (≥90 mm) contains particles of Al with a wide range of alloying and metallic pieces such as mechanically assembled screws. The characterisation assay performed revealed that the coarse fraction also contained a small proportion of Al packaging and foils. It is important to highlight that Al foils in the coarse fraction were found bound to other materials which had initially been absent. Al packaging detected in the coarse fractions was seven to eight-fold lower than in the fines (see the category Al-pack in Figures 4a and 4b).

As shown in Figure 7, the fraction 4.5-10 mm contained the most significant number of impurities; accordingly, Al was retrieved together with other metals such as Zn, Pb or Cu. The fractions 10-30 mm and 30-90 were made up largely of Al packaging and foils. During thermal treatment, defilements were removed almost entirely; these two fractions therefore are expected to produce a relatively high-grade aluminium during refining.

Although no studies have been conducted to date to investigate the obtaining of Al from LFW, experience gained with MSW indicates the possibility of using dense media separators and XRT sorting technology to separate Al from other NFMs (Capuzzi and Timelli, 2018; Lucas et al., 2019; Schmitz et al., 2006).

Irrespective of whether or not the <10 mm and >90/100 mm fractions are separated by screening, a scrap similar to UBC may be produced from the total fraction of Al scrap (see Figure 8b). UBC scrap is traded as TAKE, TALC or TALCRED (Scrap Specifications Circular, 2017); in Europe, the going rate on the scrap market is up to 1,400 euros per ton (Table 5). These potentially high-grade Al scraps from LFW, known as Al-scrap I, represent 26.4 wt% of NFMs (Figure 7a) and, as shown in Figure 8b, are made up almost totally of Al from the fine fraction (<90 mm). In line with the results presented in Figure 9, 70.7 wt% of this scrap can be valorised, with the potential price of Al-scrap I with defilements being in the range of 990 euros per ton.

On the scrap market, the price for Al foils may reach up to 200 euros per ton (see TAINT/TABOR standard in Table 5), particularly as these foils are highly sensitive to oxidation during refining. According to Soo et al. (2019), the price of Al with high Fe content is 1,000 euros per ton. The category Al-scrap II contains Al from the fraction below 10 mm (mainly Al foils mixed with other metals) and Al from the coarse fraction (Al alloys and Al with mechanically linked metals such as screws and nuts). Consequently, on considering the concentration of impurities (28 wt%), the expected prices would drop to less than 500 euros per ton.

Al scraps are treated exclusively in the secondary production circuit (Bever, 1976) and, due to the presence of undesirable alloying elements in the scrap, are used in combination with primary Al to produce alloys for specific applications, for example, parts and engines for the car industry (Paraskevas et al., 2015).

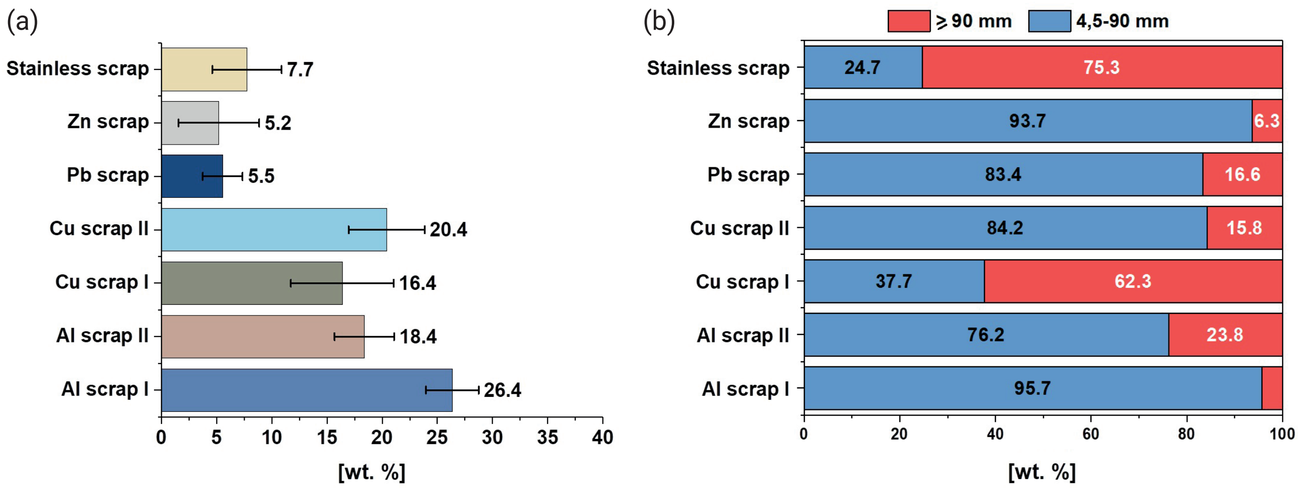

FIGURE 8: Type of marketable NFM scraps: (a) proportions (b) contribution of each fraction.

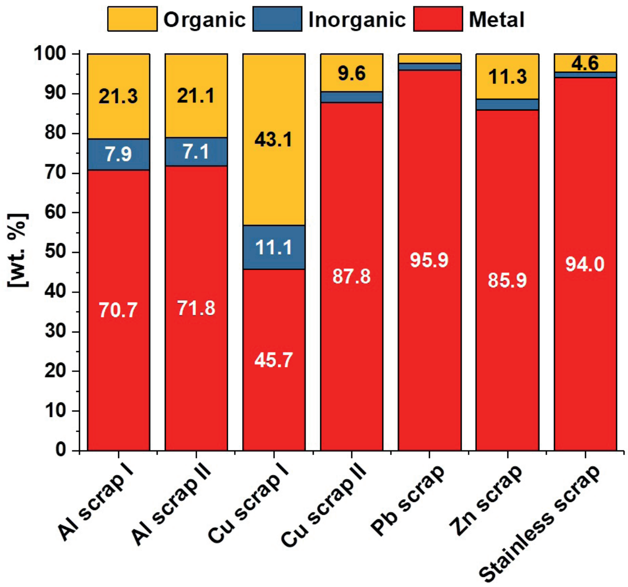

FIGURE 9: Concentration of metal and defilements in marketable NFMs [wt%].

Cu scrap

Cu scraps are the second most common type of NFM detected in the investigated fractions. The amount of Cu found was higher than expected due to the presence of C&D and industrial waste. Two different Cu scraps complying to a large extent with market standards were identified; Cu-scrap I obtained entirely from the category Cu-wires, and Cu-scrap II containing metals from the categories Brass and Other metals. Sorting technologies or dense-media separators may be used to separate wires from other Cu scraps. Almost 55 wt% of Cu wires are non-metallic (Figure 8), therefore featuring a low relative density compared to other heavy NFMs (lead, zinc, stainless steel and brass).

The purity of Cu in wires usually exceeds 96% and can be sold on the scrap market in the Droid category (Insulated Cu wire scrap N°2). The standard applied for Droid normally stipulates a metal concentration of 45%, similar to the trends observed in this study (Figure 9); rates currently offered per ton are in the range of 1,100 euros (see Table 5). Providing a chopping process (Schlesinger et al., 2011) is carried out, the metal obtained may reach prices of 3,250 euros per ton under the Birch standard.

Cu-scrap II contains 63 wt% of Cu, in addition to a series of other alloying materials and impurities such as Zn, Pb, Ni, Sn, Fe, Ag and Zn, with its alloys also being included in this category. This topic will be discussed further below in "Zn and Pb scrap". Accordingly, Cu-scrap II, consisting of a mixture of heavy NFMs, reaches requirements for the standard Zebra or Heavy brass (Table 5). This category features only 12. wt% defilements (Figure 9) and is marketed at 2,400 euros per ton.

Indeed, Cu-scrap I is valorised by undergoing a chopping process in either the primary or secondary Cu production circuit during the first or second refining stage to produce "Anode-copper". Potential applications for Cu-scrap II include use in a matte smelter or converter furnaces, which use scraps having a Cu concentration of less than 80% (Habashi, 1998; Schlesinger et al., 2011).

Pb and Zn scrap

Pb-scrap represents approx. 5.5 wt% of all NFMs identified (Figure 8a) and, as shown in Figure 8b, is found mainly in the fraction <90 mm. Analysed Pb pieces featured the presence of less than 5% defilements (Figure 9), indicating the suitability of these scraps for direct marketing as "mixed Pb scrap" or "Pb scrap" (Table 5). The current rate for Pb scrap, separated from NFMs by means of sorting technologies, is up to 1,000 euros per ton.

The scraps undergo treatment in the secondary Pb industry initially via pyrometallurgical treatment and subsequently electro-refining in the same way as Cu (Habashi, 1998).

Zn-scrap is tradable as Score (Scrap Specifications Circular, 2017) with a marked price of 600 euros per ton. However, only Zn scraps from the coarse fraction can potentially be separated using sorting technologies in view of the relatively high purity. The recovery of Zn from fine fractions is complex, being frequently bound to other metals and polymers. Cu-scrap II contains Zn as the main alloy, and during the refining of Cu, Zn oxide is recovered from the off-gas of converter furnaces. The latter might therefore also be included as Cu-scrap II.

Old stainless steel scrap

Approximately 75% of stainless steel scraps originating largely from C&D or industrial waste were recovered from the coarse fraction. In terms of chemical composition, more than 85 wt% of the stainless steel recovered was AISI 304 or 316 (austenitic stainless steel). The steel industry is extremely severe with regard to the nature of pre-existing alloying elements in these scraps, indicating the need to apply sorting technologies to separate the scraps into different stainless steel categories. On separation from other NFMs, austenitic scrap is traded under the ISRI code Sabot (Scrap Specifications Circular, 2017), with a market price of 720 euros per ton (Table 5).

Table 7 summarises calculation of the feasible price ranges for all NFMs, including defilements.

4. DISCUSSION

Prior to the advent of thermal treatment, separation (i.e. cleaning) of organic matter and soil from the surface of the metal was an arduous task. The majority of the metals had been buried, pressed and compacted under the weight of overlying waste for a period of 40 to 60 years. During this time, defilements had frequently become strongly bound to the metals. Accordingly, when assessing the quality of metal retrieved from landfills, studies conducted to investigate the effect generated during anaerobic and humic phases (Belevi and Baccini, 1989; Bozkurt, 1998; Bozkurt et al., 1999; Martensson et al., 1999) should be given due consideration. During the anaerobic phase, metals are affected by corrosion due to the presence of organic acids. The anaerobic phase is followed by slow mineralisation of organic matter, which might explain the difficulties encountered when cleaning the metals.

TABLE 7: Calculated potential prices of NFMs recovered from MSG.

Category	Proportion [wt%]	Potential Price [euro/ton]	ISRI code	Details
Realistic approach				
NFM	100	400	Zorba70	Mixed nonferrous scrap with a metal concentration of 70%
Total	400 euros per ton			
Optimistic approach				
Al-scrap				
Al-scrap I	26.4	900	TAKE, TALC or TALCRED	UBC with a small amount of Al-foils
Al-scrap II	18.4	500	NA	Mixture of Al-foils priced at 200 euros per ton and Al polluted with Fe scrap with a price of 1000 euros per ton (Soo et al. (2019)
Cu-scrap				
Cu-scrap I	16.4	1,100	Droid	Insulated Cu wire scrap N°2 (metal purity >96 %)
Cu-scrap II	20.4	2,400	Zebra	Mixed heavy nonferrous metals
Pb-scrap	5.5	1,000	NA	mixed lead scrap does not have an ISRI code, but is commercialised as is.
Zn-scrap	5.2	600	Score	Zn scrap from the coarse fractions
Stainless steel scrap	7.7	720	Sabot	Austenitic stainless steel scraps
Total	1,141.24 euros per ton			

In spite of a large number of surface defilements attached to these scraps, the majority of NFMs show no severe signs of deterioration compared with ferrous scraps, not included in this study. Nevertheless, some Cu alloys and all steel-tin cans analysed showed clear signs of deterioration.

The use of mobile technology is mandatory in the context of LFM projects in order to process LFW in situ. For this purpose, magnetic separators and ECSs are used in combination with other equipment such as ballistic separators and shredders to extract metals and other materials.

The use of more sophisticated in-situ technologies to separate scraps into different types should be evaluated in terms of cost and potential benefits to be obtained. Realistically however, in an LFM project, "mixed nonferrous scrap" alone is retrieved and traded direct from the landfill.

An intermediate approach aimed at separating NFMs into a light fraction (composed mostly of aluminium) and a heavy fraction employing dense media separators in situ may also be conceivable. Should this be the case, the prices paid will be intermediate, ranging from 500 to 700 euros per ton.

5. CONCLUSIONS

Preliminary results obtained by means of mechanical and manual processing revealed a concentration of NFMs corresponding to 4.8 kg per ton of LFW. Defilements however amounted to 27.3 wt%, thus, the actual amount of recoverable NFMs wass closer to 3.5 kg per ton.

Application of a thermal process following the guidelines issued by the Al industry for de-coating treatments proved useful in eliminating and separating the majority of defilements from NFMs in a raw state. It should however be highlighted that prior to incineration metal scraps should be separated according to metal categories, e.g. Al, Cu, Pb, Zn and stainless steel.

The steel and Al industries are extremely severe in relation to the pollutants and alloying content of scraps. Accordingly, these two metals will need to be subjected to particular care in separation and assessment; the use of sorting technologies such as those based on XRF and XRT sensors is recommended.

The Cu industry is marginally flexible with regard to the type of defilements and metals mixed with Cu scraps in view of the possibility of their re-use in a wide range of processes in both primary and secondary circuits. Primary production circuits apply stricter limitations for pollutants and alloying content, dependent on whether these scraps enter into the first or second refining steps; however, less severe constraints are applied if the scraps are used in matte smelters.

The current rates paid in the commercialisation of NFMs may vary from 400 euros for a mixed-nonferrous scrap to more than 1,100 euros per ton for NFMs that have been separated and divided by categories and grades. To conclude therefore, appropriate technical and economic assessment should be undertaken with the aim of determining the most suitable strategy in order to maximise profitability of the recovered scraps.

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INCREASING THE DIMENSIONAL STABILITY OF CaO-FeO_X-AL_2O_3-SIO_2 ALKALI-ACTIVATED MATERIALS: ON THE SWELLING POTENTIAL OF CALCIUM OXIDE-RICH ADMIXTURES

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ABSTRACT

Advanced thermochemical conversion processes are emerging technologies for materials' recovery and energetic conversion of wastes. During these processes, a (semi-)vitreous material is also produced, and as these technologies get closer to maturity and full-scale implementation, significant volumes of these secondary outputs are expected to be generated. The production of building materials through the alkali activation of such residues is often identified as a possible large-scale valorization route, but the high susceptibility of alkaliactivated materials (AAM) to shrinkage limits their attractiveness to the construction sector. Aiming to mitigate such a phenomenon, an experimental study was conducted investigating the effect of calcium oxide-rich admixtures on the dimensional stability of CaO-FeO_x-Al_2O_3-SiO_2 AAMs. This work describes the impacts of such admixtures on autogenous and drying shrinkage, porosity, microstructure, and mineralogy on AAMs. Drying shrinkage was identified as the governing mechanism affecting AAM volumetric stability, whereas autogenous shrinkage was less significant. The reference pastes presented the highest drying shrinkage, while increasing the dosage of shrinkage reducing agent (SRA) was found to reduce drying shrinkage up to 63%. The reduction of drying shrinkage was proportional to SRA content; however, elevated dosages of such admixture were found to be detrimental for AAM microstructure. On the other hand, small dosages of calcium oxide-rich admixtures did not induce significant changes in the samples' mineralogical evolution but promoted the formation of denser and less fractured microstructures. The results presented here show that calcium oxide-rich admixtures can be used to increase AAM's volumetric stability and an optimal dosage is prescribed.

1. INTRODUCTION

Alkali-activated materials (AAMs) are emerging as potential alternatives to cementitious materials, due to a less energy-intensive production process with lower environmental burdens associated. Alkali activation can be described as the reaction of a solid aluminosilicate material with an alkaline medium - which is usually a concentrated aqueous solution of alkali hydroxide, silicates, carbonates or sulfates - to produce a hardened binder (Provis, 2014). Unlike cement manufacturing, alkali-activation does not involve high-temperature processes, being AAMs manufactured at room or slightly elevated temperatures (<100°C). The use of industrial by-products and residues in AAMs design aims to further reduce their environmental footprint and production cost making them very interesting in the context of circular economy and increasing their competitiveness relative to ordinary Portland cement (OPC)-based systems. The research in this field has been mostly dedicated to metallurgical slags (Onisei et al., 2015; Komnitsas et al., 2019) but a wide group of yet unexplored waste streams can be used as AAMs precursors.

As gasification technologies of refuse-derived fuel (RDF) obtained via ELFM will reach maturity and are implemented worldwide, significant volumes of CaO-FeO_x-Al_2O_3-SiO_2-rich vitreous residues are expected to be generated in those thermal conversion processes. Currently, the use of such residues is limited to low-value applica-

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tions (e.g. road paving) but alkali-activation technology can represent a promising large-scale and fruitful valorization route.

Apart from allowing to reintegrate significant volumes of residues into the materials cycle, AAMs can also present several technical advantageous features, such as superior compressive strength, fire resistance and chemical attack resistance (Cartwright et al., 2014). However, the acceptance and large-scale implementation of AAMs will greatly depend on their long-term performance in which volumetric stability is a key factor. In fact, high susceptibility to shrinkage of AAMs is frequently reported (Cartwright et al., 2014; Lee et al., 2014), which presents a major compromising factor to the widespread adoption of the alkali activation technology. During the alkali hydrolysis that initially disrupts the precursor's glassy structure, a significant amount of water is consumed. As dissolution continues and the concertation of dissolved species in the medium increases, oligomers start to be formed, which later further crosslinks to create polymers. The recombination and reorganization of these polymers result in a significant volume contraction and release the water consumed during dissolution (Duxson et al., 2007). This water remains in the pore structure of the binder without being chemically bonded (Pacheco-Torgal et al., 2012). The existence of significant amounts of free-water in IPs systems later results in severe shrinkage, leading to the appearance of multiple micro to macro-cracks in the binder matrix.

In isothermal conditions, different shrinkage mechanism contributes to the IPs volumetric instability. Autogenous shrinkage offsets once the material initiate to consolidate and is driven by self-desiccation (Neto et al., 2008). Self-desiccation is generated in the pores as the pore solution continues to react with the pore surface to form additional binding phases. The saturation of the pores reduces and capillary pore tensions are generated (Jensen et al., 2001). Additional shrinkage results from evaporation processes when the materials are exposed to an open environment (drying shrinkage).

Methods to mitigate shrinkage have been extensively developed for OPC-based systems, however much fewer successful investigations have been reported in field alkali-activated systems, particularly when (FeO_x)-CaO-rich slags are used as precursors. In the cement industry, the swelling potential of oxide-based admixtures has been successfully used for decades to increase volumetric stability and counteract shrinkage effects (Ono et al., 1971; Chanvillard et al., 2007; Corinaldesi et al. 2015; Yang et al. 2019). Hydroxide formation from quicklime hydration has been broadly reported (Chen et al., 2012) but cement and AAMs' chemistry are entirely distinct, and the use of calcium compounds to compensate for shrinkage may affect the polymerization process and products formed (Guo et al., 2015). Van Deventer et al. (2007) investigated the effect of Ca^{2+} during the alkali activation of fly-ash and suggested that such cationic species rapidly dissolve and precipitate, providing a large number of extra potential nucleation sites. Kellermeier et al. (2010) investigated the progress of calcium carbonates in inorganic silica-rich environments and proposed that growing amorphous calcium carbonate particles can promote the spontaneous polymerization of silica.

These accelerating effects will compete with the removal of the OH^- ions from the solution, which has the net effect of lowering its pH. A decrease in solution pH will in turn affect further dissolution and precipitation processes, thus reducing supersaturation, which is the primary driving force for polymerization (Van Deventer et al., 2007). Additionally, the stability of calcium-containing precipitates also depends upon the medium composition and pH (Van Deventer et al., 2007), which render it difficult to define a general kinetic reaction model and to predict the effects of calcium-rich admixtures in specific mix designs. Nonetheless, Salman et al. (2015) observed very early exothermic peaks during the alkali activation of Ca-rich slags, from which the authors concluded that they must be related to the enhancement of reaction kinetics due to the hydrolysis of Si-O-Ca and Ca-O-Ca linkages.

However, the scarce literature on the effects of Ca^{2+} during alkali activation is focused on systems with low-iron content (Yuan et al., 2014; Velandia et al., 2016), whereas appreciable levels of iron will increase the complexity of the mechanisms involved. Daux et al. (1997) studied the dissolution of basaltic glasses and observed that Fe species are dissolved and precipitated faster than Si and Al. Van Deventer et al. (2007) reported that reactive Fe species will most likely behave similarly to Ca^{2+}, precipitating as hydroxide or oxy-hydroxide and decreasing the pH of the solution. Gomes et al. (2010) used ^{57}Fe Mössbauer spectroscopy to determine the fate of iron during alkali activation and found that a small portion of Al^{3+} could be replaced by Fe^{3+} in octahedral sites. Nonetheless, Simon et al. (2018) demonstrated that AAMs produced from Fe-rich slags are structurally different from "Fe-enriched" aluminosilicate geopolymers, in more than just Fe-Al substitution in tetrahedrally-coordinated sites. In recent work, Peys et al. (2019) used in situ X-ray total scattering and subsequent pair distribution function (PDF) analysis to describe the formation mechanism of Fe-rich alkali-activated materials. They found that the atomic rearrangements undergone by Fe-silicate species are heavily dependent on the alkali cation provided by the activating solution. In the case of sodium, the atomic correlations of the parent precursors are maintained in the intermediate products, while potassium solutions induce an increase in the coordination number of Fe species. Moreover, it was reported that Fe^{2+} and Fe^{3+} oxidation states coexist in the binder phase under different forms. The Fe^{2+} state was observed to be present as trioctahedral layers, while the participation of Fe^{3+} in the polymerized silicate network seemed most likely. In addition, PDF analysis of matured specimens has shown significant modifications, suggesting the long-term oxidization of the Fe^{2+} species present in the trioctahedral layers.

From the above, it seems evident that some questions remain unanswered to enable a full understanding of the structural role of iron during alkali activation and, to a larger extent, to describe the effects of calcium "enrichment" in Fe-rich systems.

Hence, in the present study, CaO-FeO_x-Al_2O_3-SiO_2 alkali-activated materials were produced and the impact of calcium-rich admixtures on its autogenous and dry shrinkage, mineralogy, microstructure, and porosity was assessed. Combined with previous analyses on setting time and mechanical features (Ascensão et al., 2019b), a holistic overview was possible and an optimal admixture dosage is prescribed.

2. EXPERIMENTAL

2.1 Methods

X-ray fluorescence (Bruker AXS S8 Tiger spectrometer) was used to determine the bulk chemical composition of the materials used in this work; its surface area was determined via nitrogen adsorption/desorption methods, according to ISO 9277:2010.

All pastes and monolithic samples analyzed during this work were produced following the procedure described by Ascensão et al. (2017), in which two extra minutes were added to mix the shrinkage control agent. The samples were produced by pouring the pastes into metallic molds and curing them for 24 h in controlled conditions (20±0.5°C and 95±1% relative humidity). Afterward, the samples were demolded and kept at room conditions (20°C and 65 % relative humidity).

In order to investigate the dynamics of autogenous shrinkage, a laser measurement system was used (as shown in Figure 1). The set-up consisted of two laser units horizontally aligned and directed to lightweight reflectors placed on top of the pastes. A polypropylene foil was used to isolate the pastes from direct contact with the metallic mold and to minimize water evaporation. The distance between the lasers and the reflectors was recorded continuously for ten days and the sum of the relative displacements was converted into linear deformation.

To evaluate the impact of Ca-rich compounds on drying shrinkage, two types of samples were produced. For preliminary measurements, six samples (2x2x16 cm^3) were produced for each formulation, having a metallic spindle (type II) attached in each extremity. On selected formulations, two samples (4x4x16 cm^3) were produced with a metallic spindle (type I), according to EN 12617-4:2002. The length variation of samples was monitored using a dial gauge with a sensitivity of 0.001 mm for up to 56 days after casting. Weight variation (with a precision of 0.01 g) was recorded during the same period.

The mineralogical composition of hardened samples was assessed by X-ray diffraction (XRD), conducted on a conventional Bragg-Brentano Bruker D8 Advance diffractometer, equipped with Lynxeye detector (Cu Kα radiation λ=1.54059 Å, divergence slit 0.5°, Soller slit set 2.5°+2.5°, 5-70°2θ, step/size 0.02° and t/step 0.04s.), and phase identification using EVA software. All samples were collected from mechanical testing, then ground, sieved and low vacuum dried (40°C) for 3 h prior to testing. The mineralogical evolution of the samples was monitored on the 1st, 3rd, 7th and 28th days. Scanning electron microscopy (SEM - EVO® MA 15) equipped with energy dispersive X-ray spectrometry (EDS, AZtecEnergy) was used to evaluate the differences in morphology and microstructure of the produced samples after 28 days of curing. All backscattered electron images (BSE) were acquired using a 20 kV acceleration voltage and a work distance of 10.0 mm.

The pore size distribution of the samples was investigated by mercury intrusion porosimetry (MIP). Fractured samples from mechanical tests at 28 days of curing were collected and dried in a vacuum chamber for 5 hours (45°C) prior to testing. A mercury surface tension of 0.48 N/m and a contact angle of 141.0° were set for the MIP measurements.

2.2 Materials

A synthetic CaO-FeO_x-Al_2O_3-SiO_2 rich vitreous material (referred to as plasmastone; hereafter PS) was used as the main precursor. Its detailed production process is described in Machiels et al. (2017). PS was dried, homogenized and milled, and it is composed of SiO_2 35.1, CaO 22.9, FeO_x 22.8, Al_2O_3 16.1, MgO 1.4, K_2O 0.6, TiO_2 0.6, Na_2O 0.3, Mn_2O_3 0.1, SO_3 0.1 and 1.9 wt% loss on ignition (LOI). Its BET surface area after milling was determined to be 1120 m^2/kg and its particle size distribution is given in previous studies (Ascensão et al., 2019a).

Densified silica fume (hereafter, SF) was purchased from ELKEM® (Microsilica Grade 940) and used as an admixture to provide a secondary source of SiO_2. The specific surface area (BET; 22210 m^2/kg) and composition (XRF)

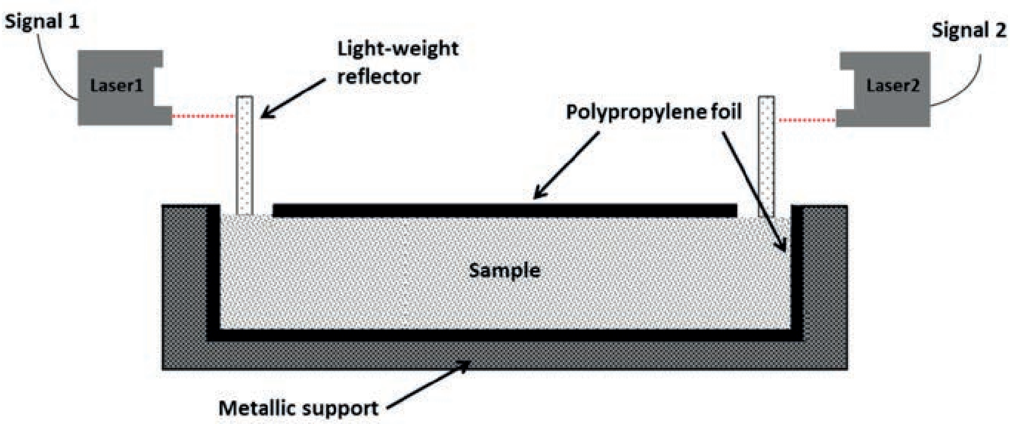

FIGURE 1: Schematic representation of autogenous shrinkage measurement apparatus.

of the SF were determined by laboratory experimentation. The SiO_2 content of the SF was 95.0 wt%, and minor elements present, such as CaO, Fe_2O_3 and Al_2O_3, were considered negligible (<0.5 wt%). A commercially available calcium-oxide rich material (Expandex C-NEW, Sika, Italy) was used as a shrinkage reducing agent (SRA). The SRA particle size was found to range between 0.1–67 µm, with d10, d50 and, d90 of 0.6, 4.3 and 27 µm, respectively. SRA was determined to be mainly composed of CaO 73.0, SiO_2 3.6 and MgO 1.5 and minor contents of Fe_2O_3, Al_2O_3, K_2O, TiO_2, Na_2O, Mn_2O_3, SO_3, SrO, P_2O_5 (all less than 1.0 wt%), with 21.0 wt% loss on ignition (LOI). X-ray diffraction patterns showed that PS and SF patterns exhibited a pronounced hump between 20-40° and 15-30° 2θ respectively, confirming their predominantly amorphous nature (Figure 2a). In the SF's XRD pattern only one crystalline peak was detected and identified as moissanite (SiC; PDF 02-1464). As can be seen in Figure 2b, the main minerals detected on the SRA pattern were calcite ($CaCO_3$; PDF 00-005-0586), lime (CaO; PDF 00-037-1497) and portlandite ($Ca(OH)_2$; PDF 00-004-0733).

Potassium hydroxide (14 M) and potassium silicate solutions (23.8 wt% SiO_2, 9.5 wt% K_2O and 66.7 wt% H_2O) were prepared by dissolving potassium hydroxide beads (reagent grade, 85%, Carlo Erba, Italy) and anhydrous potassium silicate (-48 mesh; 2.5 SiO_2:K_2O wt%, Alfa Aesar, Germany) in demineralized water. The solutions were prepared in advance to allow them to cool down prior to the preparation of the pastes. The detailed description of the prepared pastes can be seen in Table 1. It should be noted that the analyzed pastes differ only by the amount of SRA, whereas the other mix components were kept constant.

3. RESULTS AND DISCUSSION

3.1 AAM shrinkage characteristics

3.1.1 Autogenous shrinkage

As briefly stated, alkali-activated materials are known to be several times more prone to shrinkage than OPC-based systems (Thomas et al., 2017). The swelling potential of calcium-rich admixtures can be used to compensate for this, thus enhancing AAM's long-term performance. In open conditions, autogenous and drying shrinkage occurs simultaneously, though the mechanisms involved are rather different. Autogenous shrinkage can be defined as a physico-chemical phenomenon that results from chemo-mechanical and hygro-mechanical interactions (Mounanga et al., 2011). The former is due to the difference between the absolute density of the reaction products and the starting materials (also known as chemical shrinkage), while the latter results from tensile stresses generated during the emptying of the pores as hydration reactions progress (also known as self-desiccation). In AAM synthesis, a high solid content is often used to achieve good mechanical performances and enhanced durability. Imposing such synthesis conditions, however, leads to the refinement of the pore structures, which inevitably enlarges capillary stress in partially filled pores, and so contributes to high autogenous shrinkage (Lee et al., 2014). Moreover, the high viscosity of silicate solutions, like the ones used in this work, considerably increases the viscosity of the activating solution, contributing to a further rise in surface tension in partially filled pores (Sakulich et al., 2013).

Figure 3 shows length variation due to autogenous shrinkage of the analyzed pastes as a function of SRA

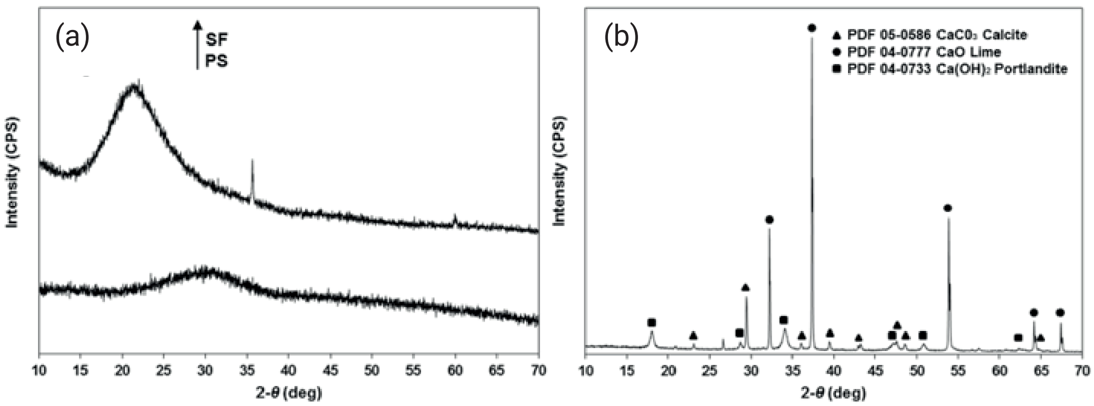

FIGURE 2: X-ray diffraction patterns of a) AAM raw materials, plasmastone and silica fume and b) shrinkage reducing agent.

TABLE 1: Classification of the landfilled material by categories.

Code	Mixture portion (wt%)					solid/liquid wt ratio	Admixtures (wt solid %)
	PS	SF	k-silicate (aq.)	KOH (aq.)	H_2O		SRA
RPa	72.1	4.1	6.2	5.3	12.3	3.2	0.0
SRA1							1.0
SRA2							2.0
SRA3							3.0

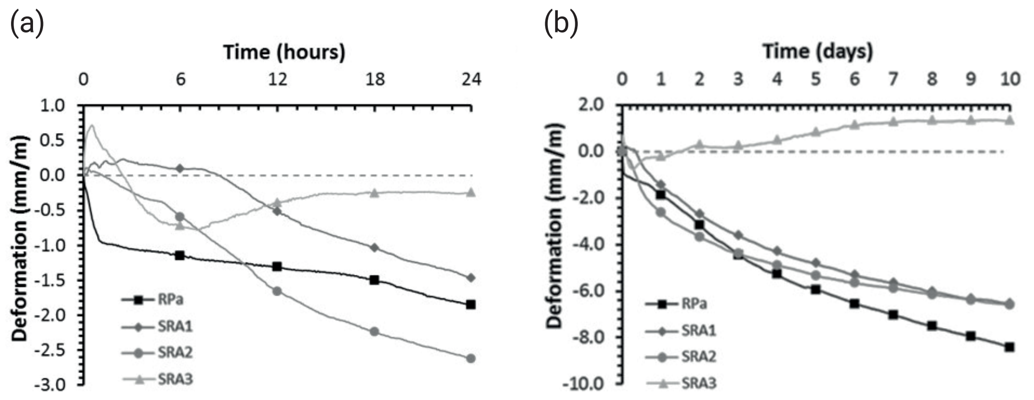

FIGURE 3: Autogenous shrinkage as a function of shrinkage reducing agent content: a) initial 24 hours and b) evolution until the tenth day of curing.

content and time. It can be seen that the reference paste presented high linear deformation during the testing period. In the initial stage of the reaction, considered to be the timeframe from mixing until the initial setting time (≈42 minutes) (Ascensão et al., 2019b), capillary stresses can be negligible, and autogenous shrinkage is mainly attributed to chemical contraction. Thus, it is interesting to notice that shortly after that period, the autogenous shrinkage progression rate drastically reduces in the reference paste, which may indicate the point at which chemical shrinkage ceases and self-desiccation begins to dominate.

The addition of calcium-rich admixtures was expected to urge the formation of calcium hydrates species and/or increase the polymeric gel formation (Van Deventer et al., 2007; Guo et al., 2015) or promote the spontaneous Si-polymerization (Kellermeier et al., 2010), which could potentially affect both chemical and self-dissection processes. Figure 3b shows that the addition of calcium oxide reduced autogenous shrinkage during the test period. In fact, samples with 1.0 and 2.0 wt% SRA underwent approximately 20.0% less length variation than the reference paste, while with higher SRA dosages a slight expansion occurred.

As calcium oxide was introduced, its finer fraction immediately hydrated and induced an initial expansion, as can be seen in Figure 3a. Depending on the SRA content, such expansion is able to partially mitigate or overcome early age autogenous shrinkage. The reasons for the distinct behavior of SRA1 and SRA2 at very early ages are not clear. However, the similar results of SRA1 and SRA2 samples after 10 days seems to suggest the existence of a critical SRA dosage. As the reaction progresses, self-desiccation becomes the dominant shrinkage process and the addition of 1.0 and 2.0 wt% SRA could only minimize its effects.

When 3.0 wt% SRA was used, the initial chemical expansion was followed by a period in which self-desiccation imposed a considerable volumetric contraction. The continuous SRA hydration was later able to mitigate such a contraction, with only a slight expansion visible after 10 days.

These results indicate that calcium oxide-rich admixtures can, in fact, be used to control AAM's autogenous shrinkage, but more detailed analysis should be conducted to assess the possible influence of factors such as i) CaO homogeneous dispersion, b) early age relaxation effects and c) expansion restriction due to a rigid set-up.

3.1.2 Drying shrinkage

Drying shrinkage occurs due to the loss of internal water to the external environment through evaporation processes and has been identified as a relevant shrinkage mechanism in alkali-activated materials (Cartwright et al., 2014, Lee et al., 2014). It should be mentioned that the results discussed in this section comprise the contributions of all forms of shrinkage and should be understood as the total shrinkage.

Figure 4a and b show the linear deformation and specific mass variation of 2x2x16 cm^3 samples up to 56 days. In all samples drying shrinkage was a continuous process, with the highest value recorded in the reference paste after 56 days of curing (21.4 mm/m). As calcium oxide content rose, drying shrinkage was progressively reduced to a minimum of 7.9 mm/m when 3.0 wt% SRA was used (-63%). In all samples, between 76% and 85% of the total drying shrinkage was observed within the initial seven days of curing and a high degree of dimensional stability was reached after 28 days. Mass loss varied from 3% to 4% and occurred mainly in the initial seven days of curing. Total shrinkage has shown a direct correlation with specific mass variation. The reference samples presented the highest shrinkage and mass loss while increasing the SRA dosage progressively reduced the magnitude of those values.

In alkali-activated systems, calcium compounds can possibly contribute to the formation of calcium hydrate species and/or increase the formation of polymeric Si-rich materials. If the former occurs, expanded reaction products are formed and the free water available in the polymeric structures reduces. If the latter, calcium compounds contribute to an increase in the formation of polymeric gel and more stable structures are obtained, thus enhancing their volumetric stability. The microstructural changes imposed by SRA addition along with its effect on pore size distribution will be discussed in the following sections.

Given that surface area may considerably interfere with evaporation processes and consequently affect drying shrinkage, probes of selected samples with different superficial area per unit of volume ratios (h) were produced and

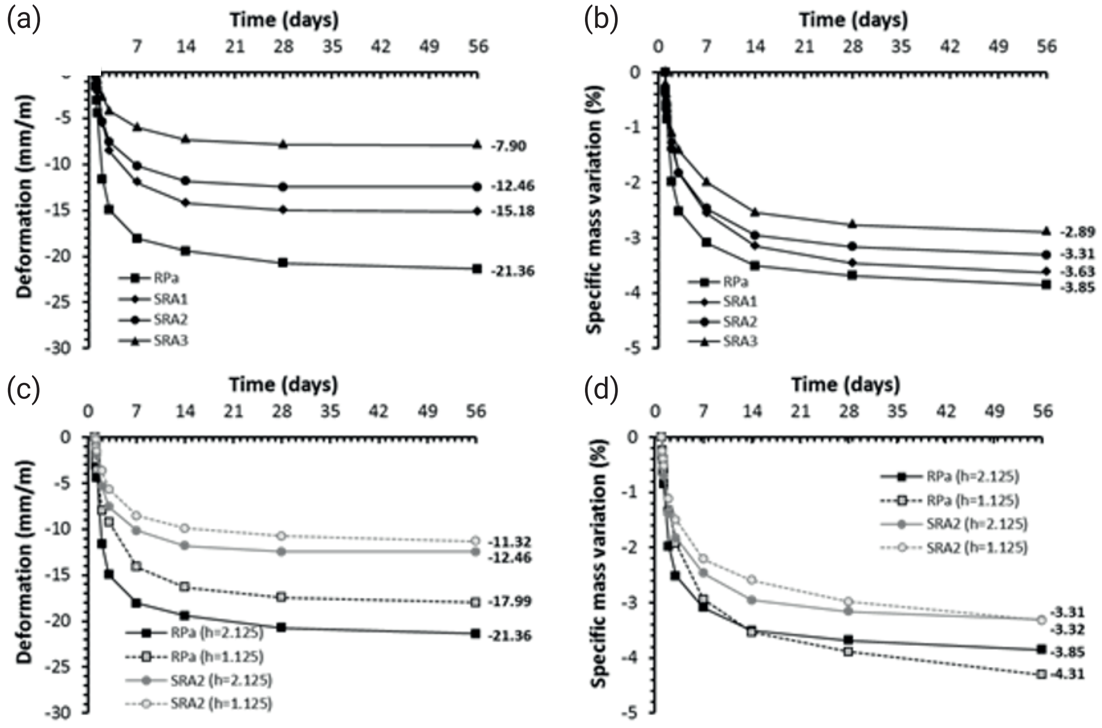

FIGURE 4: Linear deformation (a,c) and specific mass variation (b,d) as function of time, SRA content and surface-to-volume ratio (h).

monitored over 56 days. The results show that by reducing h, linear deformation decreases, but the magnitude of such reduction mainly depends on SRA presence (see Figure 4c and d). In the reference samples, final linear deformation was decreased by 16% while its specific mass variation increased by 12%. On SRA2 samples, reducing h resulted in comparable specific mass variations after 56 days, but linear deformation diminished by 9%. In both cases, a more progressive mass loss was promoted, which contributes to a decrease in capillary stress at early ages and an increase in the polymeric structures' volumetric stability.

Therefore, independently of the geometry of the samples produced, commercially available CaO-rich admixtures were found to be an effective SRA that can be used to control AAMs shrinkage. Combined with the previous analysis on setting time and mechanical features (Ascensão et al., 2019b), an optimal admixture dosage of 2.0 wt% is recommended. At this dosage, autogenous and linear drying shrinkage were reduced by 20% and 42%, respectively, while a reasonable fluidity and setting time was maintained to guarantee samples were cast properly. Curing at slightly elevated temperatures could increase the volumetric stability of AAMs (Mastali et al., 2018) and if combined whit CaO-rich admixtures led to minimal shrinkage values. However, it is also known that slightly elevated temperature modifies the reaction kinetics and products formed which would have intermingled the effects of these two shrinkage mitigation strategies. Moreover, from the environmental and economic point of view, curing AAMs at room temperature is always preferable. In the context of industrial production, avoid energy-intensive processes that require the implementation of dedicated infrastructures represents significant savings that can dictate the sustainability and economic viability of alkali-activated products. In addition, curing regimes that require slightly elevated temperatures limit the fields of application of the developed shrinkage mitigation strategies. While such curing conditions can be replicated in some applications (e.g. pre-cast) in other common construction practices such may be challenging (e.g. ready-mix concrete).

3.2 Reaction products

3.2.1 XRD analysis

Figure 5a and Figure 5b shows a comparison between precursors and AAM patterns, in which it is evident that the produced binders retain a mainly amorphous structure. No significant shift of the broad hump center towards higher 2θ values was observed. A sharp peak was visible in all patterns, revealing calcite (CaCO3; PDF 00-005-0586) as the prevalent crystalline phase formed. At later ages, and with increasing SRA content, vaterite ($CaCO_3$; PDF 01-074-1867), a metastable phase of calcium carbonate, was identified (Figure 5b-d) but the pronounced broad hump and the low magnitude of vaterite's secondary peaks make it hard to postulate its presence. The almost exclusive formation of calcite may indicate silica concentrations near saturation levels and low Ca^{2+} to CO_3^{2-} activity ratios as the work of Lakshtanov et al. (2009) shows that polymeric silica serves as nucleation sites for calcite and inhibits vaterite formation. Moreover, the formation of calcium carbonate species in AAMs has been associated with the high pH of the pore solutions that facilitates the reaction of calcium species with atmospheric CO_2 (Salman et al., 2105). Along with the presence of calcium species and highly alkaline pore solutions, AAM microstructure also has a determinant effect on the formation of calcium carbonates as it con-

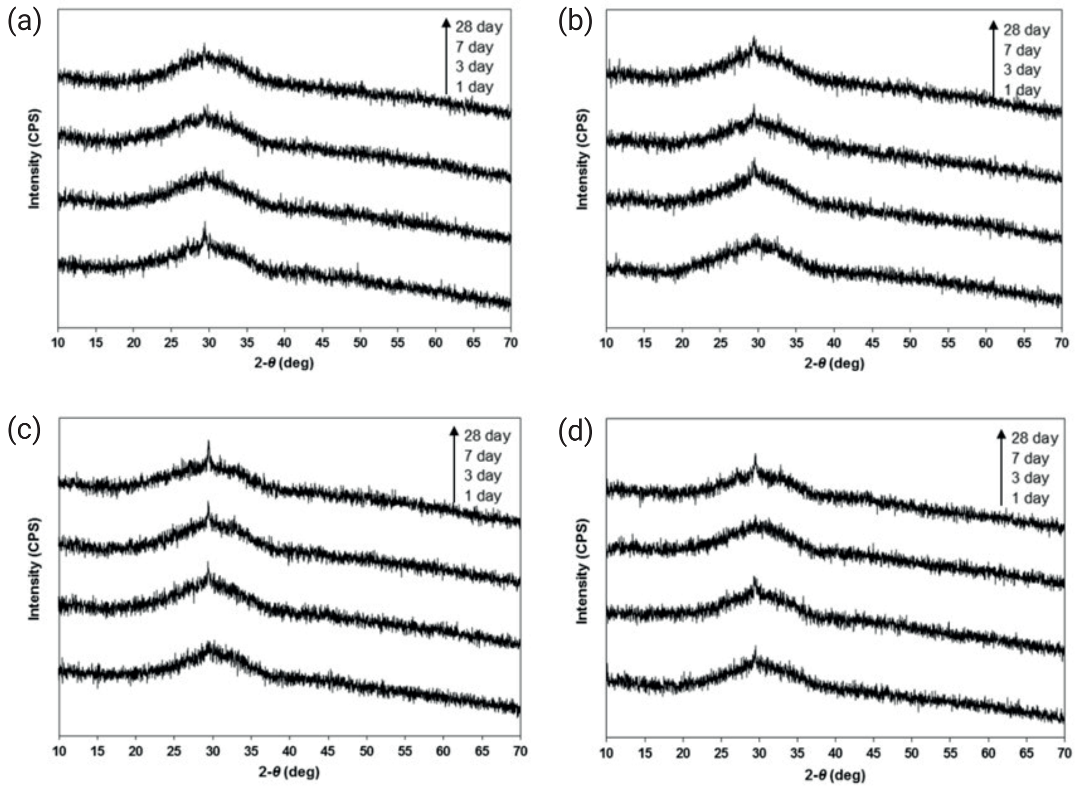

FIGURE 5: XRD patterns of AAM samples and their temporal evolution as a function of shrinkage reducing agent content: a) reference paste; b) 1.0 wt% SRA; c) 2.0 wt% SRA and d) 3.0 wt% SRA.

trols the access of atmospheric CO_2 to the pore solution and inner reaction products.

Thus, although the addition of SRA increases the availability of calcium species, an increase in calcium carbonates was not observed, as XRD patterns show (Figure 5). In fact, the increasing availability of calcium compounds leads to the formation of denser structures when SRA content does not exceed 2.0 wt% (detailed in the "Microstructure" and "Porosity" sections), which restricts atmospheric CO_2 diffusion and so limits the advance of carbonation. However, this does not apply to the SRA3 samples, where an increase in calcium oxide availability is followed by a slight increase of porosity (Figure 6a), even though carbonation levels remain roughly the same. One explanation may lay in the fact that SRA3 samples present a considerably different pore size distribution (Figure 6b) comprising considerably larger pores that diminish pore-specific surface area and contact area with atmospheric CO_2.

Nonetheless, the formation of calcite, which was verified in all samples, can contribute to densification by filling the porous structures and thus enhancing samples' volumetric stability and mechanical properties as previously reported (Ascensão et al., 2019b).

3.3 Microstructure

Figure 7 shows representative backscattered electron imaging micrographs of samples produced with distinct amounts of SRA. A binder phase was formed in all samples, but the existence of unreacted particles confirms that complete dissolution was not achieved (Figure 7a). The high solid-to-liquid ratio used (3.20) may have limited precursors' dissolution, but all samples show a homogeneous matrix in which undissolved particles acted as small-sized aggregates (Figure 7a-f). These results are in agreement with previous findings (Machiels et al., 2014), which reported samples with similar S/L ratios as having a degree of precursor's dissolution of 76 wt%. Although the quantification of unreacted particles has not been performed, a slight decrease in the number of unreacted particles seems to be promoted as SRA content rises.

Depending on solutions' saturation level with respect to amorphous silica, calcium carbonates, can urge the polymerization of silica in their vicinity (Kellermeier et al., 2010), thus accelerating AAMs' polymerization. On the other hand, the formation of Ca-precipitates decreases the activating solution pH due to the removal of OH^- ions, limiting the precursors' dissolution and reduces the medium's supersaturation level. As the authors previously reported (Ascensão et al., 2019b), increasing SRA content led to shorter setting times, which, combined with the reduction of unreacted particles, suggests that the former is favored when small amounts of calcium oxide are added to an already Ca-rich system.

EDS analysis was performed on selected unreacted particles revealing a homogeneous composition constituted by the chemical elements of the main solid precursor, PS (not shown here for the sake of brevity). Some metallic artifacts were detected (as individuated in Figure 7b), which were mainly composed of FeO_x>90.0 wt%.

The binder phase of the reference sample has shown

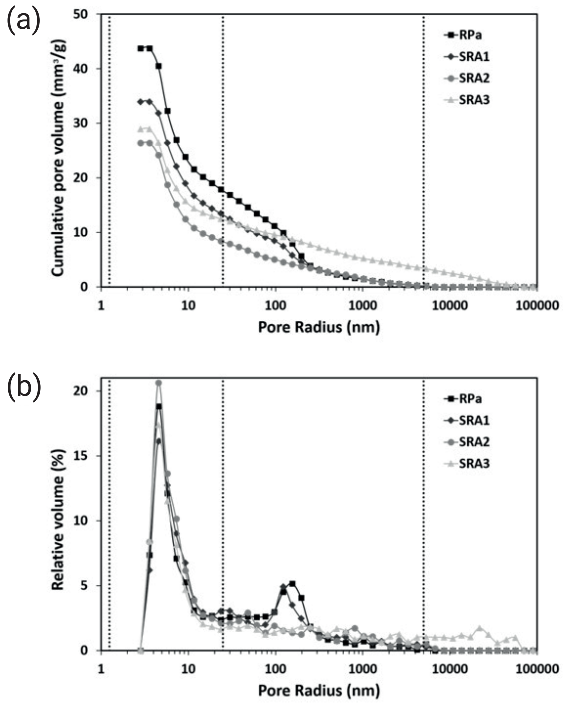

FIGURE 6: Cumulative pore volume (a) and relative pore size distribution (b) of samples after 28 days of curing.

to be composed of approximately SiO_2: CaO: Fe_2O_3: Al_2O_3 : K_2O = 2:1:1:0.5:0.2, with the exception of some particular circular areas where an increased content of Si elements was observed (e.g., circular areas identified in Figure 7b).

The higher Si content in these areas may suggest that silica fume particle dissolution occurs at these locations and, due to mobility restrictions imposed by the high viscosity and short open time of the pastes, local heterogeneities were formed in the gel phase (Figure 7b-e). These small circular areas were detected in all samples, however, with a growing Ca content as SRA rose. Furthermore, SRA-containing samples exhibited areas where a Ca-rich gel phase predominated. Figure 7f shows one of those areas in which the binder phase is mainly constituted by a Ca-rich gel (approx. SiO_2: CaO: Fe_2O_3: Al_2O_3: K_2O = 1:2:0.1:0.2:0.03). In all samples, cracks pierced the binder phase as the undissolved particles limited their propagation. Crack formation and development was particularly severe in Ca-rich binder phases, of which Figure 7f is a representative example. Yet, Ca-rich areas represent a small portion of the binder phase; the cracks formed on those areas are being compensated by the formation of denser and less fractured structures on a macroscopic level. As can be seen in Figure 7b-e, as SRA content increases up to 2.0 wt%, fewer and finer cracks were formed, while with higher SRA dosages more cracks were induced. These results are corroborated by MIP data (detailed in the "Porosity" section), which revealed higher and broader porosity in SRA3 samples. Further, some spherical pores were observed, especially in the reference and SRA1 sample, which can be attributed to air trapped during mixing. Those pores nearly vanished in SRA2 and SRA3 samples, further contributing to the production of dense and mechanically strong polymeric structures as shown previously (Ascensão et al., 2019b).

3.4 Porosity

Pores could be grouped into four main categories according to size: i) micropores, <1.25 nm; ii) mesopores ranging from 1.25-25 nm; iii) macropores ranging from 25-5000 nm; and iv) entrained and entrapped air voids and pre-existing microcracks >50000 nm (Collins et al., 2000). Micropores are inherent to reaction products whereas capillary pores (comprising both meso- and macropores) can be seen as the residual unfilled spaces between them. Drying shrinkage will greatly depend upon capillary pore size distribution, as it determines the extension and stresses generated by water loss during curing.

Figure 6 presents the cumulative and relative pore size distribution of AAM samples as a function of SRA content. Micropores were outside of the range of measurement and therefore their relative volume is considered negligible. In all samples, the majority of the pores have a pore radius <25 nm and a higher volume of mesopores is concentrated around 5 nm. The amount of mesopores (vol%) increased as calcium oxide dosage rose, from 59 vol% in RPa sample to 68.0 vol% in sample SRA2, and decreased to 57 vol% in sample SRA3. SRA3 samples' hastened viscosity and setting may have compromised their proper confinement and particle packing, increasing the number of voids and cracks in these samples. In fact, Figure 6b shows that SRA3 samples were the only ones containing a proportion of pores with dimensions higher than 10000 nm. These results are in good agreement with the increased amount of visible cracks present in these samples relative to SRA2 (Figure 7b-e) and with the reduction of strength development previously reported (Ascensão et al., 2019b).

4. CONCLUSIONS

The use of a calcium oxide-rich admixture to increase the dimensional stability of CaO-FeO_x-Al_2O_3-SiO_2-rich alkali-activated materials and its effect on porosity, autogenous and drying shrinkage, mineralogy, and microstructure were studied in this research. The main findings are summarised as follows:

- Drying shrinkage was identified as the governing mechanism affecting AAM volumetric stability, whereas autogenous shrinkage was less significant.
- Calcium oxide-rich admixtures can be effectively used to increase AAMs dimensional stability.
- Shrinkage reduction was proportional to SRA content, but elevated dosages of the latter have a detrimental effect on AAMs microstructure.
- The SRA addition did not induce significant mineralogical changes.

These results demonstrated that commercially available CaO-rich admixtures can be effectively used to control autogenous and drying shrinkage of AAMs. Considering the impacts of CaO-rich admixtures on the remaining AAM properties, a dosage of 2.0 wt% is suggested by the authors.

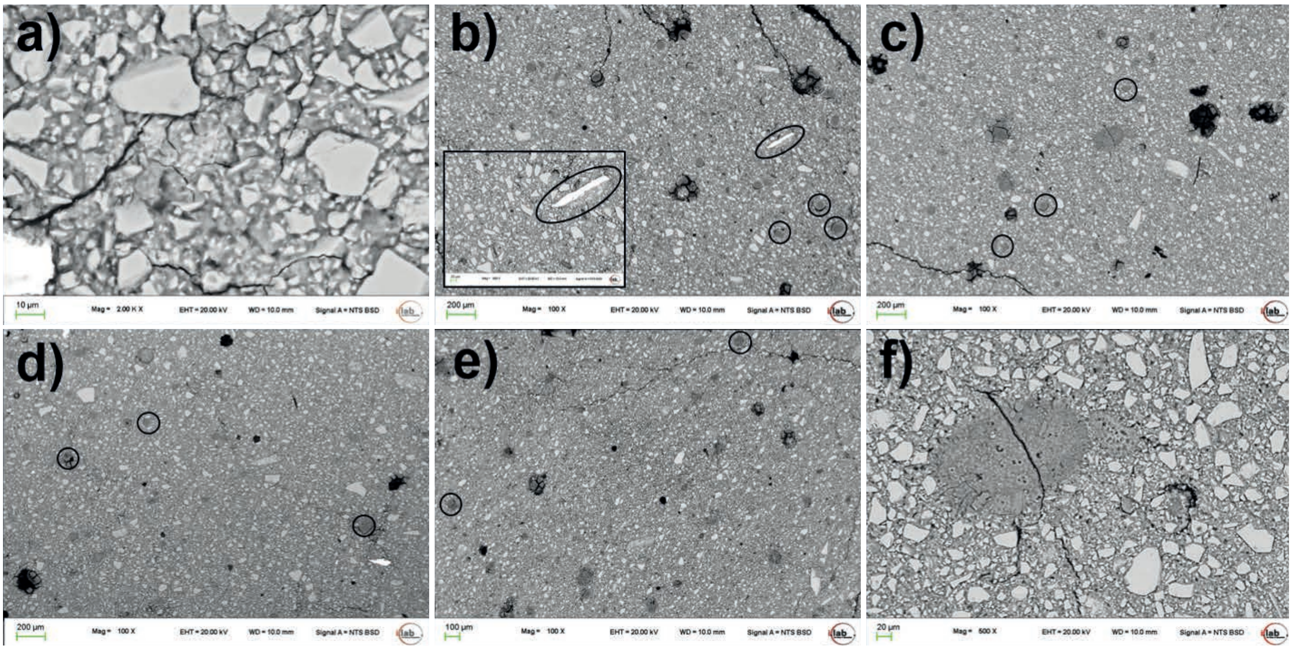

FIGURE 7: Backscattered electron imaging micrographs of samples' microstructure and close-ups of selected areas after curing for 28 days: a) reference binder phase and unreacted particles; b) reference paste microstructure; c) 1.0 wt% SRA; d) 2.0 wt% SRA; e) 3.0 wt% SRA and f) Ca-rich binder phase. The insert circles in 7 b-e) show Si-rich binder phases.

Apart from the significant shrinkage reductions obtained, the reduced cost and simple addition method make the use of CaO-rich RSA a promising shrinkage mitigation strategy to increase the performance and competitiveness AAMs relative to benchmark materials.

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STRONG POROUS GLASS-CERAMICS FROM ALKALI ACTIVATION AND SINTER-CRYSTALLIZATION OF VITRIFIED MSWI BOTTOM ASH

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ABSTRACT

Vitrification of municipal solid waste incineration (MSWI) bottom ash is an effective method to produce a chemically stable glass, with metal recovery. In order to justify the high costs of this process, the vitrified residue can then be upcycled into potential marketable products. In this study, vitrified bottom ash was successfully converted into strong and chemically stable porous glass-ceramics by the combination of alkali activation and sintering. After the activation of the glass in a NaOH solution of low molarity, foams were easily produced by intensive mechanical stirring, with the aid of a surfactant, and stabilized by gelation. The obtained open-celled material was further consolidated by a sintering treatment, at 800-900 °C. The addition of recycled soda-lime glass allowed activation at low molarity and sintering at lower temperature, but it reduced the mechanical properties and the stabilization of heavy metals. On the other hand, the increase in molarity of the alkaline solution increased the porosity and also the strength of foams from vitrified bottom ash.

1. INTRODUCTION

The quantity of municipal solid waste (MSW) produced has never been higher. It has been estimated that around 1.3 billion tonnes of solid waste were produced in the world in 2012, which may dramatically increase to 2.2 billion tonnes by 2025 (Hoornweg & Bhada-Tata, 2012). In the EU 28, almost 30% of MSW produced is still being landfilled, with significantly different rates among the European countries (Cucchiella, D'Adamo, & Gastaldi, 2017). An alternative to landfilling of MSW is represented by incineration with energy recovery, in which up to 90 vol% of waste can be reduced (Tillman, Vick, & Rossi, 1989). Besides the exhaust gas (which is used to generate energy), municipal incinerators produce two types of residues, fly ash and bottom ash. Bottom ash represents more than 98% of the incineration outputs (Joseph, Snellings, Van den Heede, Matthys, & De Belie, 2018) and it is currently treated mechanically using screeners, crushers, magnets, eddy current separators, sorting technologies and washers to extract the metallic fraction and clean as maximum as possible the mineral fraction. In the EU, the rest of bottom ash is mostly landfilled, but in some other instances it can be used as aggregate for road paving or construction (Sabbas et al., 2003). In addition, previous studies reported that the treated bottom ash can also be valorised into new products such as tiles, bricks and alkali activated materials (R. V. Silva, de Brito, Lynn, & Dhir, 2017). However, as this residue can still contain hazardous metals, chlorides, sulphates and other pollutants, it is crucial to perform an environmental impact assessment of the developed material before its commercialization (R. V. Silva, de Brito, Lynn, & Dhir, 2019). In fact, environmental issues that can be caused by bottom ash lies as one of the main reasons why this ash is still being mainly landfilled (He, Pu, Shao, & Zhang, 2017). Another option of managing the bottom ash is through the vitrification of the residue, which generates a chemically stable and homogeneous glass (Bassani et al., 2009). However, as vitrification is an high demanding energy process, it is only economically viable if the glass can then be upcycled into high added value products, such as glass-ceramics (Colombo, Brusatin, Bernardo, & Scarinci, 2003).

Upcycling of vitrified residues into marketable products has been extensively referred to in the literature. Examples include tiles, aggregates for reinforcement of concrete and glass foams for thermal and acoustic insulation (Rincón, Marangoni, Cetin, & Bernardo, 2016). The latter can offer

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a series of interesting properties for building construction such as low density, high compressive strength, flame resistance and nontoxicity (Rincón et al., 2016; Scarinci, Brusatin, & Bernardo, 2006). Unlike the extensively used polymeric foams, glass foams consist in a much safer option for building insulation in case of fire. However, these foams are still quite expensive to be produced due to specific process and additives used (Monich, Romero, Höllen, & Bernardo, 2018).

An alternative technique, aimed at decreasing the high cost involved in the production of glass foams and based on alkali activation, has been recently developed (Rincón, Giacomello, Pasetto, & Bernardo, 2017). According to this approach, that could be defined of 'inorganic gel casting', a suspension of fine glass powders, in an alkaline solution of low molarity, undergoes progressive hardening due to the formation of surface gels (Elsayed et al., 2017; Garcia-Lodeiro, Aparicio-Rebollo, Fernández-Jimenez, & Palomo, 2016; Monich et al., 2018; Rincón et al., 2017), in turn due to the partial dissolution of the same glass. Before complete setting, a surfactant is added to the suspension, later subjected to intensive mechanical stirring. The trapping of air bubbles, favoured by the surfactant, determines a substantial foaming. When stirring is ceased, the ongoing gelation prevents the collapse of the foamed 'green' structure. The material is then extracted from the container, dried and sintered by viscous flow (Rincón et al., 2017). This technique has already been successfully applied in the production of highly porous and strong waste derived materials (Monich et al., 2018; Rincón, Desideri, & Bernardo, 2018; Rincón et al., 2017; Rincon Romero, Salvo, & Bernardo, 2018) as well as bioactive glass-ceramics (Elsayed et al., 2017).

In this study, the process of alkali activation, gelation and sintering was extended to produce porous glass-ceramics made with vitrified bottom ash (VBA). This glass residue was obtained from the smelting of bottom ash by using an electric arc furnace, followed by quenching. Electric arc furnace is a simple built technology with low thermal losses and high output, frequently employed in the vitrification of residues (Colombo et al., 2003). Four different types of porous glass-ceramics were developed in order to minimize costs of production, especially related to the alkaline solution and heating treatment, without compromising the mechanical properties and chemical stability of the foams. Recycled soda-lime glass (SLG) was used to aid the gelation and sintering process of two groups of materials (Monich et al., 2018).

The overall approach does not strictly match with the concept of 'landfill mining' (Monich et al., 2018), according to which landfill remediation is performed by excavation, removal of directly recyclable components (e.g. metallic objects, plastics), pyrolysis (with transformation of organic compounds into combustible gas) and conditioning of the inorganic residue. It should be noted, however, that the conversion of vitrified bottom ash into glass-ceramics may represent a model for the last operation of landfill mining. Once the inorganic residue is melted, and valuable metals are separated, a full 'circularity' (i.e. "enhanced landfill mining") will be achieved only in the hypothesis of reuse also of the non-metallic fraction.

2. MATERIALS AND METHODS

MSWI bottom ash was gently provided by the company AVR (Netherlands) and dried at 200°C for 24h. Thereafter, the dried bottom ash (up to 3 kg by trial) was added to a graphite crucible and smelted in a lab-scale electric arc furnace operating in DC at around 1500°C for 60 minutes. A graphite electrode of 50 mm was used on the top (Figure 1). After the smelting, the melt residue was quenched in water, dried and crushed (by means of a jaw crusher) in order to separate a metal fraction (up to 14 wt%). The non-metallic fraction was dry ball milled, until the particle size was below 75 µm.

The chemical composition of the obtained vitrified bottom ash (Table 1) was assessed by a PANanalytical WDXRF spectrometer it is quite similar to the one of "slag sitals" (specially concerning amounts of SiO_2, CaO, Na_2O, MgO, Fe_2O_3 and Cr_2O_3). "Slag sitals" consisted on strong and chemically stable slag derived glass-ceramics developed in the late USSR (Höland & Beall, 2012). Table 1 also shows the chemical composition of soda-lime glass which was employed in the development of two groups of porous glass-ceramics. This glass (medium particle size equal to 30 µm) was gently provided by SASIL SpA (Brusnengo, Biella, Italy) after colour selection and removal of metallic and polymeric residues from the glass cullet. This fraction is usually not recycled due to ceramic contaminations (Rincón et al., 2017).

Thermal analysis (DSC/TGA, 3+ STARe System, Mettler Toledo, Columbus, OH, USA), with heating rate of 10°C/min, was done on fine powders (particle size < 75 µm) as well on coarse powder (particle size ~ 1 mm) of vitrified bottom ash in order to determine the physico-chemical transformations occurring during heating.

Porous glass-ceramics were produced by firstly mixing at 400 rpm fine powders of the waste glasses to an alkaline solution of NaOH for 3h. The overall solid loading content used was of 70 wt%. After the partial dissolution of the fine powders, 4 wt% of surfactant (Triton X-100, (polyoxyethylene octyl phenyl ether – C14H22O(C2H4O)n, n = 9–10, Sig-

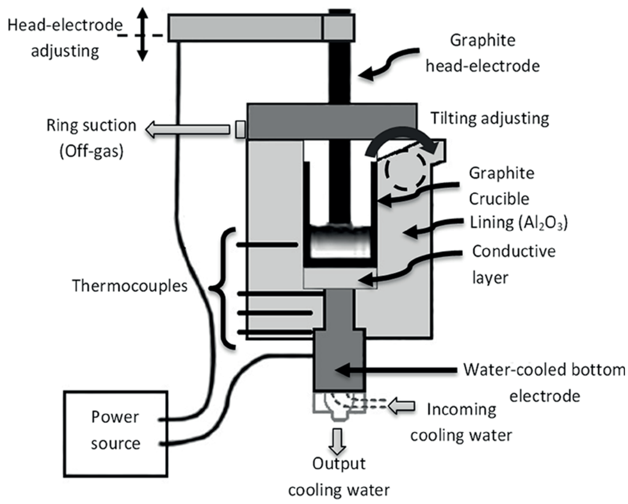

FIGURE 1: Sketch of the lab-scale submerged arc furnace (SAF).

TABLE 1: Chemical composition of the glasses employed in this study (wt%).

	Vitrified bottom ash	Soda-lime glass
SiO_2	50.32	71.9
CaO	20.90	7.5
Al_2O_3	19.03	1.2
Na_2O	4.59	14.3
MgO	2.65	4
TiO_2	0.85	0.1
K_2O	0.75	0.4
CuO	0.21	
BaO	0.18	
Fe_2O_3	0.14	0.3
MnO	0.13	
SrO	0.11	
ZrO_2	0.06	
Cr_2O_3	0.05	
P_2O_5	0.02	
Cl	0.02	

ma-Aldrich, Gillingham, UK) was added to the suspension, which was then submitted to an intensive mechanical stirring at 2000 rpm. The foamed suspension was subsequently dried at 40°C for 48h, demoulded and fired at 800°C or 900°C, with heating rate of 10°C/min and a holding time of 1h. Table 2 presents the conditions applied to produce the four different groups of samples.

The mineralogical analysis of crushed fired foams was performed by X-ray diffraction (XRD) (Bruker D8 Advance, Karlsruhe, Germany), using CuKα radiation, 0.15418 nm, 40 kV–40 mA, 2θ = 15-60°, step size 0.05°, 2 s counting time. High resolution X-ray diffraction analysis was done on fine powders of vitrified bottom ash and on not fired crushed foams. In this case, a position sensitive detector was used, with step size of 0.02° and counting time of 2 s. This generated a distinctive high signal-to-noise ratio, which allowed to identify the crystalline reaction products of alkali activation. The Match!® program package (Crystal Impact GbR, Bonn, Germany), supported by data from Powder Diffraction File (PDF)-2 database (International Centre for Diffraction Data, Newtown Square, PA, USA) was used for phase identification.

Fourier-transform infrared spectra were collected with Jasco 4200 FTIR spectrometer (Jasco, Japan) equipped with an attenuated total reflection (ATR) attachment (ZnSe crystal) on the powdered samples of vitrified bottom ash and on the samples from group B before and after sintering. For each measurement 32 scans were coded at a resolution of 4 cm^{-1}, in the range of 700 cm^{-1} to 4000 cm^{-1}.

TABLE 2: Approaches applied in the production of samples.

Group of samples	A	B	C	D
Molarity of the alkaline solution	1 M	1.5 M	1 M	1 M
Composition	100% VBA	100% VBA	90% VBA/	
10% SLG	90% VBA/			
10% SLG				
Firing temperature	900°C	900°C	900°C	800°C

The fired foams were cut into cubes (side of approximate 10 mm) and used for further characterizations. The bulk density of the fired foams was calculated by the ration of the mass (measured with a digital balance) to the volume (measured by using a caliper) of the samples. A gas pycnometer (Micromeritics AccuPyc 1330, Norcross, GA) was employed to measure the apparent and true densities of the foams and of the finely crushed samples, respectively.

The compressive strength of 10 porous glass-ceramics of each group was determined by using an Instron 1121 UTM (Instron Danvers, MA). The mechanical test was done at room temperature with a cross-head speed of 1 mm/min.

The morphological and microstructural characterizations of the fired foams was assessed by means of an optical stereomicroscopy (AxioCam ERc 5 s Microscope Camera, Carl Zeiss Microscopy, Thornwood, New York, USA).

The chemical stability of vitrified bottom ash and of each group of fired foams was evaluated by means of leaching test, based on norm EN 12457-4 (“Norm EN 12457-4,” 2002). The materials were firstly crushed and sieved below 4 mm. Thereafter, the sieved fragments were added to a plastic flask with pure distilled water (liquid/solid ratio of 10), which was submitted to mixing for 24h at room temperature. The suspension was then filtered and centrifuged, obtaining the eluate. Inductively coupled plasma mass spectrometry (ICP-OES, Spectro Genesis, Germany) was used to measure the heavy metals of the eluate. The leachate values allowed for waste acceptable at landfills for inert waste and non-hazardous waste (Directive 2003/33/EC, 2003) was used as a reference.

3. RESULTS AND DISCUSSION

The crystallization temperature (Tc) of fine powder of vitrified bottom ash lies around 925°C, according to the differential scanning calorimetry (DSC) curve of the fine powder of vitrified bottom ash (Figure 2a). This temperature was used as a reference for firing experiments, which were performed at 900°C. Foams made with addition of soda-lime glass were also fired at 800°C, in an attempt to decrease energy consumption during sintering. It was not possible to detect the crystallization temperature of the coarse powder, which indicates that this glass is sensitive to surface crystallization (E. Bernardo, 2008). The particle size of the glass did not influence the thermogravimetry analysis (TGA, Figure 2b): the TGA curves show a decrease in mass of less than 0.5% at higher temperatures, for fine vitrified bottom ash.

The FTIR spectra provided information on the hardening mechanism (Figure 3). Even considering the strongest activation (Figure 3a, B green), the formation of C-S-H compounds, at the basis of the obtainment of glass foams from ‘inorganic gel casting’ (Rincón et al., 2017), is hardly visible: peaks at 3458 cm^{-1} and at 1680 cm^{-1}, attributed to O-H stretching and O-H bending, remained very slight. The main peak at 1450 cm^{-1}, visible in all groups of green foams, cor-

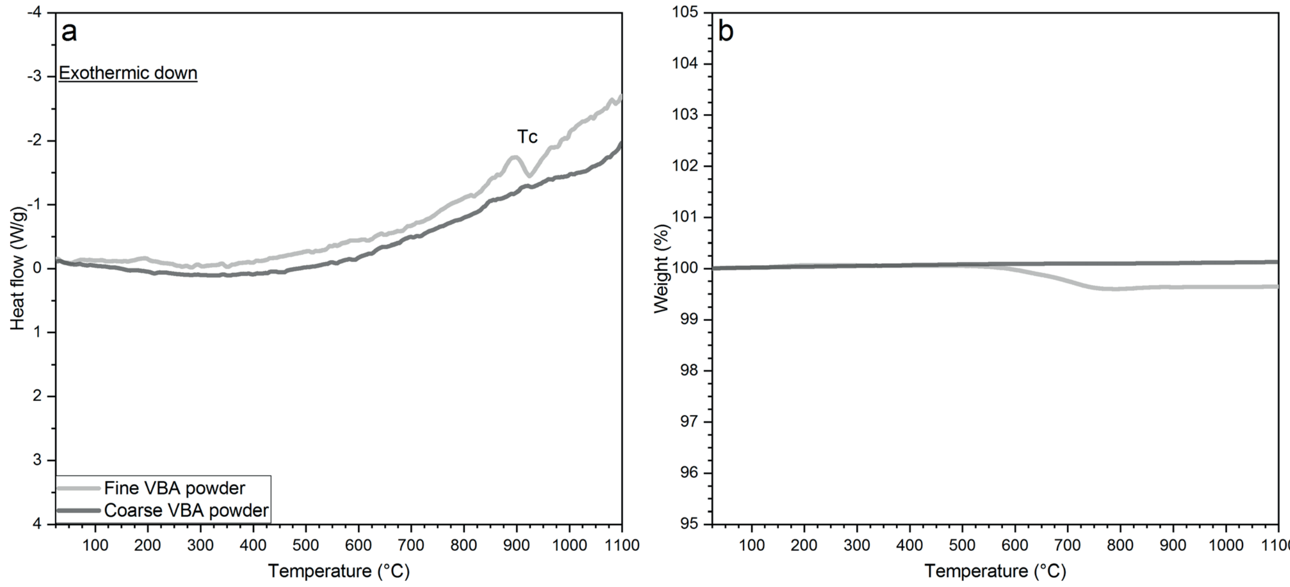

FIGURE 2: DSC (a) and TGA (b) curves of fine and coarse powder of vitrified bottom ash.

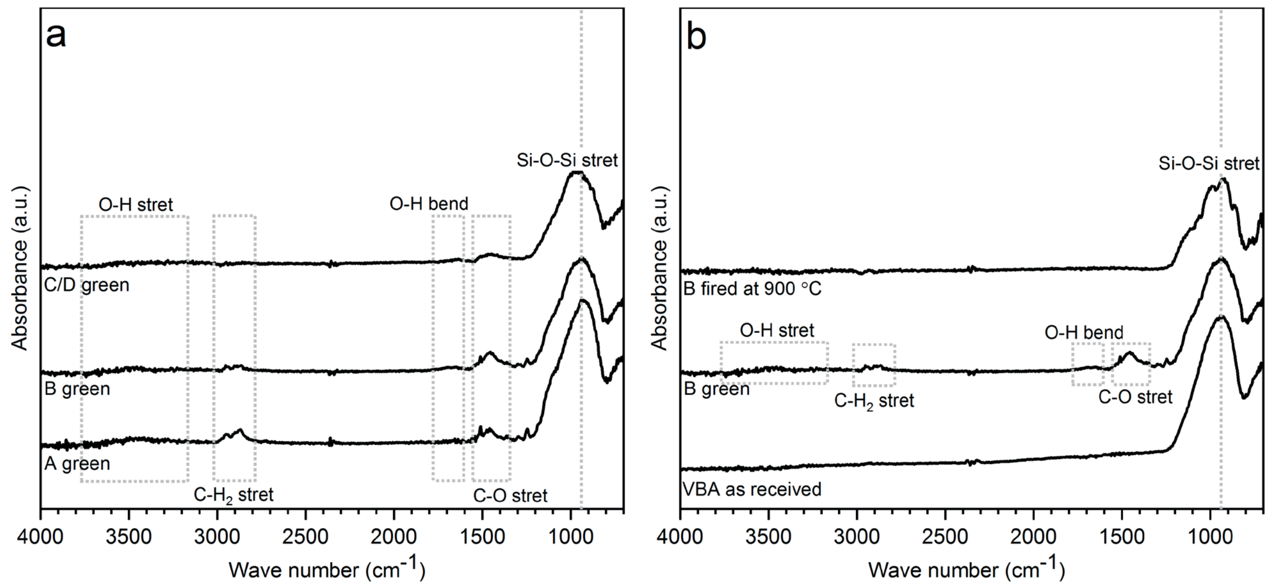

FIGURE 3: FTIR spectra of: a) foams from groups A, B and C/D before firing; b) vitrified bottom ash and foams from group B before and after firing.

responds to the stretching vibration of C-O (Rincon Romero et al., 2018). This finding confirms recently reported experiences (Rincon Romero et al., 2018), according to which the hardening of the activated vitrified bottom ash is mainly due to carbonation. Furthermore, the peak at 2900 cm^{-1} is associated to C-H$_2$ stretching due to the addition of the surfactant (Monich et al., 2018). Regarding the spectrum of vitrified bottom ash (Figure 3b), the band between 800 cm^{-1} and 1260 cm^{-1} corresponds to the asymmetric Si-O-Si stretching vibration (Paola Pisciella & Pelino, 2005). This bands becomes slightly narrower after alkali activation and it is separated in more peaks after the firing treatment, probably due to crystallization (Rincon Romero et al., 2018).

High resolution X-ray diffraction analysis (Figure 4) allowed to identify which carbonated and hydrated phases were formed, according to alkali activation. Trona ($Na_3H(CO_3)_2 \cdot 2(H_2O)$, PDF#00-029-1447) was detected as the only newly formed phases, in green foams (i.e. after foaming and drying) from group A, in agreement with previous findings in alkali activated vitrified bottom ash (Rincon Romero et al., 2018), made with a stronger activating solution (2.5 M NaOH).

Unlike in previous experiences, in order to favour the handling of foams upon demoulding (green 'A' foams were particularly weak), pre-foaming and curing step (aimed at enhancing the dissolution) were not applied. Instead, a slight increase in molarity of activating solution was considered. Passing from 1 M (Figure 4, group A) to 1.5 M (Figure 4, group B) favoured the formation of more phases, contributing to the hardening. The X-ray signals are consistent with meionite ($(Ca_3.4Na_{0.64})(Al_{5.43}Si_{6.59})O_{24}(CO_3)_{0.88}O_{0.12}$, PDF#75-1222), sodium carbonate (Na_2CO_3, PDF#86-0315), tilleyite ($(Ca_5Si_2O_7(CO_3)2$, PDF#73-2117) and sodium aluminium silicate carbonate ($Na_8Al_6Si_6O_{24}CO_3$, PDF#00-024-1045). The stronger activation evidently determined some dissolution of the glass (in turn favouring the incorporation of Ca^{2+}, Al^{3+} and Si^{4+} in carbonates), but it did not lead to any practical formation of non-carbonate phases.

Significant changes occurred in foams made with addition of 10 wt% soda-lime glass (Figure 4, C/D groups). This addition had been conceived to yield stronger foams, in the green state, by keeping a low molarity of activating solution (1M NaOH). The low molarity did not cause the formation of C-S-H compounds (previously observed with waste glass/soda lime mixtures (Monich et al., 2018), but turned the newly formed phases from being sodium based to being calcium based. Calcium carbonate ($CaCO_3$, PDF#86-2339) was clearly detected. The remaining peak is consistent with the presence of traces of the alumino-silicate zeolite gismondine ($CaAl_2Si_2O_8·4H_2O$, PDF#20-0452). Gismondine is interesting, being found in geopolymers from granulated blast furnace slag (Zhang, Zhao, Li, & Xu, 2008), i.e. in products from very strong alkali activation.

The different formulations had some impacts after firing. The XRD patterns (Figure 5) of foams fired at 900°C showed signals consistent with those of labradorite ($Ca_{0.64}Na_{0.35}(Al_{1.63}Si_{2.37}O_8)$, PDF#83-1371) and gehlenite ($Ca_2(Al(AlSi)O_7)$, PDF#74-1607). Labradorite and gehlenite have already been previously detected in glass-ceramics made with plasma vitrified MSWI fly ashes (Bernardo et al., 2011). We cannot exclude the presence also of an Al-rich pyroxene (augite, $CaMg_{0.7}Al_{0.6}Si_{1.7}O_6$, PDF#78-1392). Pyroxene solid solutions are quite typical in waste-derived glass-ceramics (Park, Moon, & Heo, 2003), as well as plagioclase and melilite solid solutions (comprising labradorite and gehlenite, respectively).

Figure 5 also indicates that the increase of molarity of the alkaline solution and the introduction of soda-lime glass had a 'symmetrical' effect on the crystallization: compared to 'A' foams, both foams from stronger activation ('B' type) and from glass addition ('C' type) exhibited more marked peaks. However, it may be seen that in the first case all peaks became more intense; in the second, on the contrary, gehlenite had a more significant increase.

The enhanced crystallization is reasonably due to the increase of overall alkali content in both groups B and C, which may have lowered the apparent activation energy for crystal growth, as already observed for alkali rich glasses (Watanabe, Hashimoto, Hayashi, & Nagata, 2008)). The enhancement of crystallization was found at 900°C; firing below the crystallization temperature of vitrified bottom ash (at 800°C, group D) led simply to fully amorphous foams.

The glass-ceramic foams presented porosity higher than 58 vol%, mainly open, as shown by Table 3. Table 3 also indicates that the increase of molarity from 1 M to 1.5 M enabled the increase of almost 10% in porosity. The reason probably lies on the fact that the more "gelified" suspension (group B, as shown in Figure 4) could prevent more efficiently the collapse of the bubbles entrapped after foaming. Due to their high porosity, the foams from group B could be potentially applied as thermal or acoustic insulators in buildings. Regarding the mechanical properties,

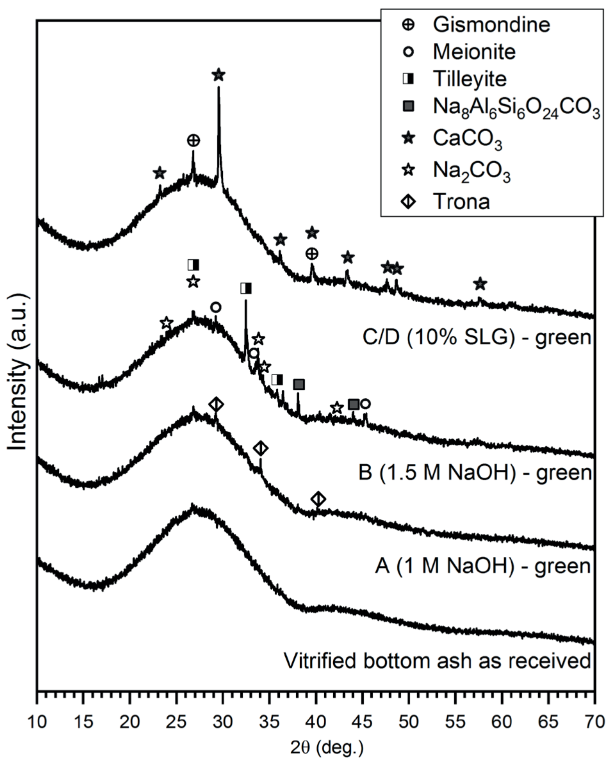

FIGURE 4: High resolution XRD patterns of vitrified bottom ash and not fired foams.

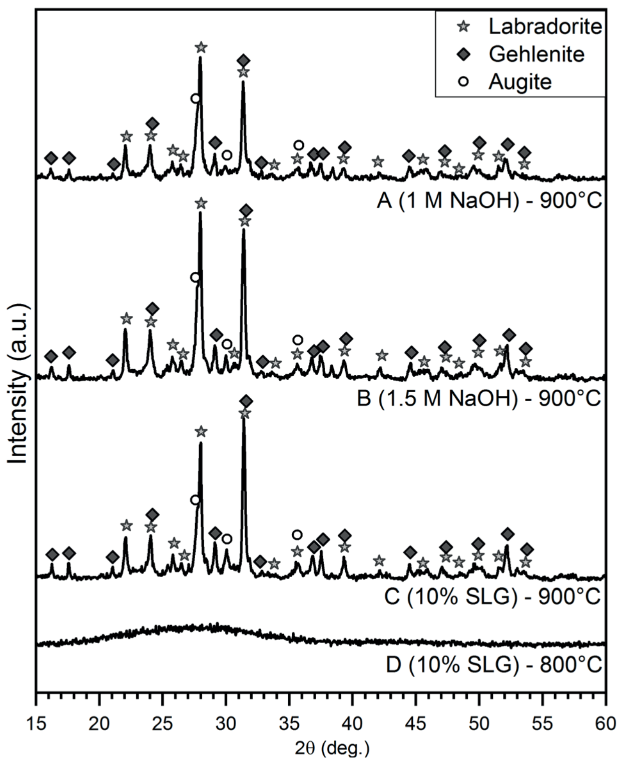

FIGURE 5: XRD patterns of the fired vitrified bottom ash derived foams.

the compressive strength reached a maximum of 8.1 MPa for foams made with a stronger alkaline solution (group B). This value is well above the typical crushing strength for commercial glass foams, which lies between 0.4 and 6 MPa (Scarinci, Brusatin, & Bernardo, 2006); in addition, the strength-to-density ratio compares well with that of commercial foams (e.g. alumina foams with the similar density hardly exceed 8 MPa) (CES EduPack, 2018).

Foams made with soda-lime glass (groups C and D), on the other hand, did not present an increase of porosity. In addition, the increase in firing temperature from 800°C to 900°C, enabling crystallization, determined 38% increase in the strength of the foams.

The micrographs of the porous materials are shown in Figure 6. As already indicated by Table 3, the foams present high porosity, mainly open. The increase in the molarity of the alkaline solution from 1 M (group A) to 1.5 M (group B) decreased substantially the pore size. The decrease in pore size could be one of the reasons lying behind the increase in compressive strength in foams from groups B, despite presenting a higher porosity. As already observed in another study, foams with smaller macro-pore size presented higher compressive strength than foams with a larger macro-pore size up to a certain level of porosity (Liu, 1997).

Regarding the foams made with addition of soda-lime glass (groups C and D), it may be observed the influence of the firing temperature on the pore size distribution: foams fired at 800°C (group D) present a larger pore size than foams fired at 900°C (group C). As the foams were still amorphous at 800°C (Figure 5), the softened glass may have contributed to reshape the pores during firing, before crystallization. The precipitation of crystals increased then the viscosity of the softened glass, which prevented a further reshaping of pores at higher temperatures (Rincón, Giacomello, Pasetto, & Bernardo, 2017).

As vitrified bottom ash is originated from waste, it is essential to perform leaching test on the samples, in order to verify if the leaching of heavy metals is within the regulation. Table 4 shows that the vitrification of MSWI bottom ash effectively yielded a safe material with very low leaching of heavy metals. Concerning the glass-ceramic foams made with only vitrified bottom ash (groups A and B), the leaching of heavy metals was below the limit values for inert and non-hazardous waste. On the other hand, due to a high leaching of Sb, foams made with soda-lime glass (groups C and D) could only be accepted as non-hazardous waste.

The increase in the alkalinity of the residual glass, with

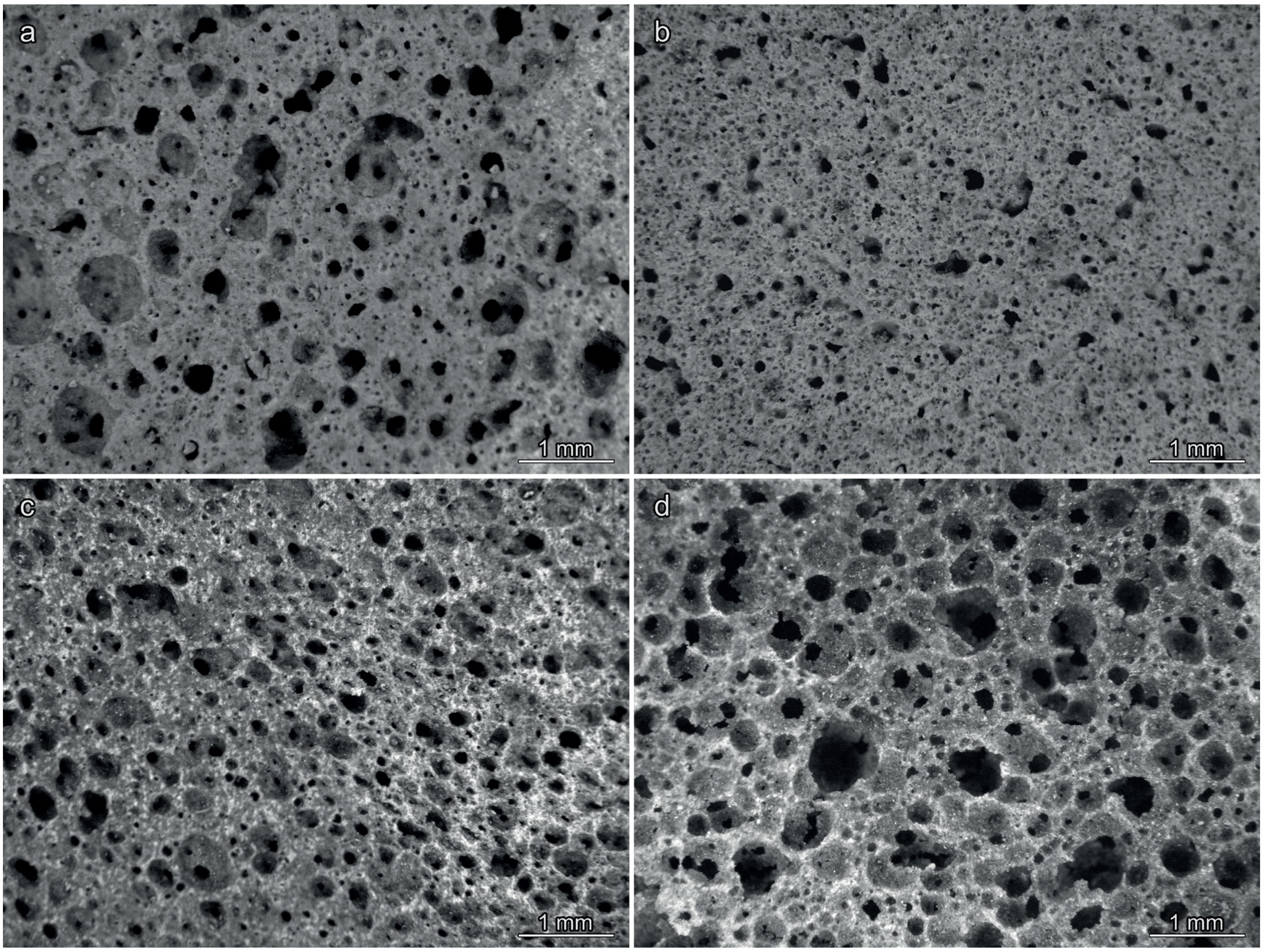

FIGURE 6: Micrographs of the four types of porous glass-ceramics developed: a) group A, 1 M NaOH; b) group B, 1.5 M NaOH; c) group C, 10% SGL fired at 900°C; d) group D, 10% SGL fired at 800°C.

TABLE 3: Porosity and mechanical properties of the four groups of porous materials produced.

Group of samples	A	B	C	D
Activation	1 M NaOH	1.5 M NaOH	1 M NaOH	1 M NaOH
Soda-lime glass (%)	-	-	10%	10%
Sintering Temperature (°C)	900	900	900	800
Density determinations				
ρ_{geom} (g/cm³)	1.04 ± 0.08	0.85 ± 0.04	1.03 ± 0.03	1.07 ± 0.09
$\rho_{apparent}$ (g/cm³)	2.28 ± 0.01	2.59 ± 0.03	2.45 ± 0.04	2.54 ± 0.04
ρ_{true} (g/cm³)	2.52 ± 0.00	2.62 ± 0.00	2.53 ± 0.01	2.57 ± 0.00
ρ_{rel}	0.412	0.326	0.407	0.414
Porosity distribution				
Total porosity, P (%)	58.8	67.4	59.3	58.6
Open porosity, OP (%)	54.4	67.0	58	58.1
Closed porosity, CP (%)	4.4	0.4	1.3	0.5
Strength determinations				
σ_{comp} (MPa)	7.0 ± 2.7	8.1 ± 1.1	7.3 ± 1.8	5.3 ± 2.0

TABLE 4: Results of the leaching test of vitrified bottom ash and of the four groups of foams (mg/kg) [*: above limit].

	Limits (Directive 2003/33/EC, 2003)		VBA	A	B	C	D
	Inert waste	Non-hazardous waste					
As	0.5	2	0.0076	<0.0049	<0.0049	<0.0049	<0.0049
Ba	20	100	0.0054	0.0354	0.0696	0.0041	0.0061
Cd	0.04	1	<0.0002	0.0004	<0.0002	<0.0002	<0.0002
Cr total	0.5	10	0.0066	0.0072	0.0021	0.0020	0.0146
Cu	2	50	0.0219	0.0128	0.0024	0.0003	0.002
Hg	0.01	0.2	<0.0004	<0.0004	<0.0004	0.0023	0.0015
Mo	0.5	10	0.0184	<0.0033	<0.0033	<0.0033	<0.0033
Ni	0.4	10	0.0017	0.0042	<0.0014	<0.0014	<0.0014
Pb	0.5	10	0.0068	0.0111	0.0072	<0.0047	<0.0047
Sb	0.06	0.7	0.0339	0.0151	0.0320	0.3518*	0.2316*
Se	0.1	0.5	<0.0122	0.0163	0.0221	<0.0122	<0.0122
Zn	4	50	<0.0203	<0.0203	<0.0203	<0.0203	<0.0203
		Final pH	**7.8**	**8.0**	**8.0**	**7.6**	**7.5**

the addition of soda-lime glass, may have favoured its dissolution (Monich et al., 2018; P. Pisciella, Crisucci, Karamanov, & Pelino, 2001). It must be observed, however, that the leaching tests were applied on glass foam fragments, i.e. on samples with huge specific surface. The conditions of chemical attack, as well the reference limits (intended for materials to be disposed in landfills), were probably excessive.

4. CONCLUSIONS

We may conclude that:

- The technique based on alkali activation, gelation, foaming and sintering could be applied to produce porous and strong glass-ceramics made with vitrified bottom ash, with limited costs (considering the limited alkalinity of activating solutions and low firing temperatures);
- The valorisation of vitrified bottom ash into porous and strong glass-ceramics by an economic process has the potential to produce potential marketable products. This could help to decrease the high costs of vitrification;
- The hardening of the suspension originates mainly from the formation of carbonates;
- The increase in molarity from 1 to 1.5 M produced stronger foams with higher porosity and smaller pore size;
- The introduction of soda-lime glass allowed the achievement of comparable compressive strength (at 900°C), with a reduced molarity of activating solution; however, this was accompanied by some degradation of the stabilization of pollutants.

'Declarations of interest: none'.

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DEVELOPING STAKEHOLDER ARCHETYPES FOR ENHANCED LANDFILL MINING

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ABSTRACT
Understanding the perspectives of different stakeholders on emerging technological concepts is an important step towards their implementation. Enhanced Landfill Mining (ELFM) is one of these emerging concepts. It aims at valorizing past waste streams to higher added values in a sustainable manner. Yet, assessment of ELFM mainly focusses on environmental and private economic issues, and societal impacts are rarely analyzed. This study uses semi-structured interviews to build understanding for different ELFM practitioners and researchers and develops five stakeholder archetypes for ELFM implementation: the Engaged Citizen, the Entrepreneur, the Technology Enthusiast, the Visionary and the Skeptic. The archetypes outline major differences in approaching ELFM implementation. The stakeholder perceptions are put into context with existing literature, and implications for ELFM implementation and future research are discussed. Results show that differences in regulatory changes and technology choices are affected by different stakeholder perspectives and more research is needed to balance inner- and inter-dimensional conflicts of ELFM's sustainability. The developed archetypes can especially be helpful when evaluating social impacts, whose perception often depends on opinion and is difficult to quantify.

1. INTRODUCTION

Growing pressure on environmental change has dominated the recent public discussion on climate-related issues. Yet, regulatory measures to reduce CO_2 emissions, for example, are not always perceived as fair and effective by all members of society. This can be seen in the recent 'Gilet Jaunes' movement in France, for which positive environmental change is perceived as conflicting with social needs (Amjahid and Raether, 2018). Nonetheless, the effective management of natural resources (NRM) plays an important role in avoiding future climate impacts. Making it compatible with social and economic needs is therefore essential for its implementation. NRM affects societal, environmental and economic change, connecting all dimensions of sustainability. The importance of NRM is reflected in the Paris Agreement, where signatories are obliged to build up the resilience of socio-economic and environmental systems through NRM (UN, 2016). To tackle this challenge, it is not only important to advance towards a renewable energy system and rethink major production processes. It also calls for new technologies and material sources, to integrate secondary raw materials into a circular economy. To do so, the European Union has developed an Action Plan for Circular Economy, covering elements of production, consumption, and waste management (EC, 2015). As implemented in the EU Landfill Directive, the Action Plan also calls for a waste hierarchy and focusses on the prevention and recycling of waste, integrating current streams into resource management (EC, 2015, 1999). However, past waste streams are mostly being ignored and have traditionally been landfilled (Krook et al., 2012), and with them valuable materials and resources.

Growing market and environmental pressures have led to the development of a rather new concept: Enhanced Landfill Mining (ELFM). ELFM aims to add value to past urban waste streams as materials (Waste-to-Material, WtM) and energy (Waste-to-Energy, WtE) using innovative technology in an integrated, environmentally and societally sound way (Jones et al., 2013). The concept originated from remediation projects and has since shifted to the creation

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of higher added values (e.g. hydrogen) (Krook et al., 2012, Jones et al., 2013). Potentially, ELFM could lead to the mitigation of primary resource production and therefore generate positive environmental effects (Danthurebandara et al., 2015a; Jain et al., 2014). Its economic performance, on the other hand, is still unclear. High investment and processing costs hinder ELFM implementation, despite environmental gains (Hermann et al., 2016a; Kieckhäfer et al., 2017). Societal factors are rarely considered in ELFM research or generalized to an extent where impacts an effects become entangled, often through monetization, making it difficult to draw conclusions (Damigos et al., 2015; Van Passel et al., 2013).

2. AIM AND SCOPE

A limited current knowledge base (Krook et al., 2018), and a lack of industrial experience emphasizes the need for research in the field. Krook, Svensson, and Eklund (2012) conclude further investigations on stakeholder perceptions including societal actors are "essential for understanding the capacity of technology and conditions for realization" of ELFM. While current studies usually focus on environmental risks or economic assessments, it remains unclear, how different stakeholders approach ELFM and how different perceptions affect ELFM implementation. The aim of the paper is to make differences and similarities of various stakeholders' perspectives on ELFM explicit. The findings are structured through the development of stakeholder archetypes. These archetypes provide a basis for ELFM research to integrate societal factors and enhance assessment methods as well as the scientific discussion on ELFM by integrating different opinions rather than a seeming objectivity. This is especially important considering the quantification of societal impacts, as results can be biased to some extent not only by the chosen method but also by the perspective taken by the assessor. The archetypes further provide information to different stakeholders involved in ELFM and can help nurture each other's understanding to avoid societal conflicts along the road of implementation. The developed archetypes can be used as educational material to explain and better understand inner- and interdimensional conflicts of sustainability when implementing new technology concepts. They are an easy-to-use tool to show differences in knowledge distributions amongst stakeholders and can provide valuable insights for policymakers. They give industry actors the opportunity to develop a better understanding of partners and market conditions and can help to avoid fears and worries in the general population.

The scope of this study is limited to ELFM implementation and conclusions to other industry sectors are not drawn. It focusses on a Belgium case at the Remo landfill in Flanders. The Remo case provides a well researched scientific basis to reasonably interpret results and a high degree of stakeholder involvement (Bosmans et al., 2013; Danthurebandara et al., 2013; Quaghebeur et al., 2010). It comprises an area of about 160 hectares dedicated to landfilling and stores about 16.5 million tons of municipal solid (MSW) and industrial (IW) waste. Leachate protection, as well as a gas collection system, are installed at the facilities. Within the "Closing the Circle" (CtC) project, initiated by the operators, ELFM operations are planned in the near future (Geysen, 2017; Group Machiels, 2018; Quaghebeur et al., 2013). Additionally, neighboring community members have formed a citizen initiative called DeLocals. Their goal is to understand the ELFM operations at Remo and distribute information about developments, accomplishments as well as problems to relevant stakeholders (Ballard et al., 2018). The study considers perspectives on landfilling in general, ELMF implementation, the different dimensions of sustainability in ELFM as well as regulatory issues. Misconceptions and knowledge gaps within stakeholder groups are discussed as well as implications for ELFM implementation. The study uses interviews for assessment and considers a brought range of stakeholders, including institutional, industrial, scientific and communal actors.

3. MATERIALS & METHOD

To develop the different stakeholder archetypes, semi-structured interviews were conducted. To do so, an interview guide was developed based on the relevant literature. The stakeholder selection process was based on an extended quadruple-helix (QH) framework (Arnkil et al., 2010; Kolehmainen et al., 2016). The analysis was based on the general inductive approach by Thomas (2006). Interviews were taken in person or on the phone with a total of 13 interviewees. The interviews were analyzed using a general inductive approach (Thomas, 2006) and from the findings, the stakeholder archetypes were developed. It is important to stress the qualitative nature of the study. Through in-depth interviews we intent to derive what motivates and drives different stakeholders with regards to their stakeholder classes of a specific ELFM case, i.e. the Remo landfill. This approach helps to avoid hypothetical biases as well as an over-representation of one stakeholder group. However, a restricted pool of potential interviewees due to the case specification, and additional temporal constraints limit the number of interviews and therefore the statistical representativeness of the study. Nonetheless, as the Remo case is of scientific interest, we believe this research adds a missing part and provides a basis for the future investigation of societal impacts.

3.1 The Interview Guide

From an initial review of the relevant literature, five major themes of scientific interest were identified. Assessed parameters and derived research needs were carefully analyzed. The themes included (i) "perspectives on landfills and their management" (e.g. Krook, Svensson, and Eklund 2012), (ii) "economic drivers and barriers for ELFM" (e.g. Danthurebandara et al. 2015; Frändegård, Krook, and Svensson 2015), (iii) "environmental benefits and risks of ELFM" (e.g. Gusca, Fainzilbergs, and Muizniece 2015; Laner et al. 2016), (iv) "societal challenges for ELFM implementation" (e.g. Van Passel et al. 2013; Lederer, Laner, and Fellner 2014), and (v) "the role and responsibilities of institutions and other stakeholders involved in ELFM activities" (e.g. Krook, Svensson, and Eklund 2012; Johansson 2016). While the first theme (i) was chosen to identify the general approach of participants to landfills and ELFM, the second, third and fourth themes (ii-iv) aim at analyzing the perceived sustainability of ELFM. The last theme (v) was chosen to investigate how different stakeholders are involved in ELFM projects and where they are able to influence processes along realization. The interview guide can be found in the Annex to this study.

3.2 The Extended Quadruple-helix Framework

The Quadruple-Helix (QH) framework is often used in the context of new technological development and distinguishes between various actors at different points of innovation processes. It aims to capture multiple and reciprocal relations between involved stakeholders (Arnkil et al., 2010; Kolehmainen et al., 2016). It consists of four major strands: The institutional, the scientific, the societal and the industrial strand. The industrial strand was further subclassified along the value-creation-chain of ELFM. This resulted in subclasses: operators, technology providers, and buyers. Additionally, attributes were added to the QH framework to further differentiate certain properties of the interviewees. The attributes included (i) level of operation, i.e. local, regional, federal or supranational, (ii) level of case-involvement and (iii) level of impact on overall ELFM implementation, both differentiated between high, moderate, and low, first evaluated by the researchers and consequently adapted through new findings from the interviews, as well as the stakeholder's (iv) organizational type, distinguishing governmental (gov.), non-governmental (n-gov.) or private (p) organizations. A schematic representation of the extended QH framework can be seen in Figure 1.

3.3 Analysis

The general inductive approach was used to derive concepts, models, and structures from the raw interview data. The unit of analysis used was "concepts"(Corbin and Strauss, 1990). In this study, a concept could comprise only one word or several sentences. The analysis was done in three main steps. First, the raw textual data was condensed into a brief summary format. In the second step, the summary findings were used to establish clear links and relations between various actors and concepts expressed during the interviews. Consequentially, a theoretical framework about the underlying structure of the research findings was developed, i.e. the stakeholder archetypes (Thomas, 2006). To put the general inductive approach into practice, concepts were coded according to the categories of the interview guide, providing a priori-coding and using QSR International's NVivo 11 software. Similar statements were joined into one coding category and related interviewees connected to the statements to derive clear differences between actors and concepts. Overlapping coding was allowed, opening up the possibility of one concept being assigned to several coding categories, hinting to links between them. Consequentially, concepts, interlinked through stakeholder class and/or content, were grouped and structured in a sensible manner by tabulating them and develop the stakeholder archetypes.

3.4 Stakeholder Selection

The study includes three actors from institutions and one scientific actor from a university. To include the societal community of the QH framework, three interviews were held with neighbors from the surrounding communities of the Remo landfill. The extended industrial strand included two managers from the operating company, two technology providers, and one technology incubator. The incubator was chosen to represent buyers of ELFM products. Since operations at the Remo site have not started yet, finding real buyers was not possible. Focusing also on waste management and operating in a similar region, the technology incubator was chosen as a proxy-representative for this stakeholder class. An overview of all participants can be found in Table 1.

4. RESULTS

The results are structured in two basic parts. First, the descriptive summary of the interviews is presented. Its purpose is to provide a more detailed overview of the different stakeholder perspectives and transparency to make results reproducible. The second part presents the developed stakeholder archetypes.

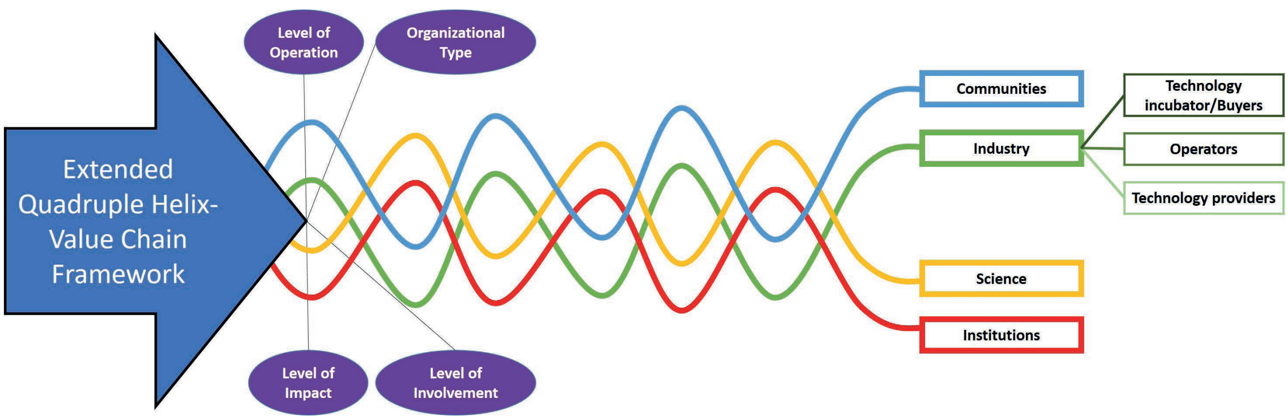

FIGURE 1: The Figure shows the extended quadruple-helix framework including sub-classes and attributes.

TABLE 1: Table 1 shows the interviewees sorted by stakeholder class.

QH/value chain-Class	Stakeholder	Attributes	No.
Community members			**3**
	De Locals	Non-governmental Local Medium/high influence Low/medium impact	3
Institutional actors			**4**
	Government	Governmental Local High influence Low impact	1
	Waste Agency	Governmental Regional High influence Medium impact	2
	European Commission	Governmental Supranational Medium influence High impact	1
Scientific actors			**1**
	Researcher	Non-governmental Supranational Low/medium influence Medium/high impact	1
Industrial actors			**5**
	Operators	Private Supranational High influence Medium/high impact	2
	Technology providers	Private Supranational Medium influence Medium impact	2
	Technology incubator	Private Regional Low influence Medium impact	1
Total			**13**

4.1 Descriptive summary

The descriptive summary entails the main concepts addressed by the stakeholders during the interviews. As ELFM is a relatively young field of research and lacks the assessment of societal factors, this descriptive part should help the reader understand and comprehend the findings, and could provide a basis for future research. The first subsection, Approach to landfills, describes how stakeholders perceive the functions and safety of the Remo landfill, as well as advantages and disadvantages. The second subsection, Concept, and attitude towards ELFM, describes how stakeholders approach ELFM in general and what differences they perceive in the concept. The next three subsections, economic drivers and barriers, environmental risks and benefits and societal challenges for ELFM, treat perceptions about the sustainability of ELFM. The last subsection, Key Actors of ELFM, describes who the different stakeholders perceive as playing the most influential role in ELFM implementation.

4.1.1 Approach to Landfills

All stakeholders perceive a temporary storage function of landfills. Yet, landfilling is considered the least favorable waste treatment option, but deemed necessary throughout most interviewees. The operating company emphasizes the offered service of waste disposal, whereas the scientific side also mentioned landfills as a source of pollution and, like the technology incubator, a land occupant. Institutional participants make an explicit distinction between "dumpsites" that pre-date the European Landfill Directive from 1999 and "landfills" that comply with it.

Technology providers perceived advantages of landfilling waste over incineration. They explained the storage function of landfills with a lack of technologies to handle certain waste streams in the past and made it clear that significant amounts of waste would still be landfilled in the future, passing the problems on to the next generation

All participants accentuated that a properly operated landfill under current legislation could be considered safe, but older landfills are often perceived as less safe. Nevertheless, distinctions in perceptions lay in the details: The landfill operators made a distinction of "very old landfills" justified by changes in landfilled waste streams: "…when you go back in the past there are not that many risky waste streams…" This is coherent with the experiences of institutional participants stating most landfills are in better conditions than estimated, and the expected toxic "time-bombs seem not to be a reality after all". Additionally, the institutional side stressed that changing circumstances, due to changes in climatic conditions, can affect the safety of a landfill, for example through higher flood risks. Flooding a landfill could potentially endanger groundwater reservoirs and the stability of landfills through soil movements. The technology incubator criticized illegal waste dumping as a major risk and perceived a lack of control mechanisms for waste disposal.

Perceived advantages of landfills are the potential for resource recovery and the removal of waste from the local communities. The institutional participants also stated an advantage in being able to control the process of waste disposal.

Perceived disadvantages, on the other hand, were approached differently. While all stakeholders mentioned a suboptimal use of land and environmental risks, operators also mentioned the installation of additional security measures against wildlife as well as risks coming directly from ongoing operations. Local communities further perceived risk for human health coming from toxic materials (e.g. mercury or asbestos), whereas the researcher stated a disadvantage of industrial landfills often containing toxic materials, in combination with a lack of control mechanisms.

Concerning the Remo site, all stakeholders had positive associations, although opposing groups to the project from local communities and politics were also mentioned. The operators were described as "thinking in a modern way" or "courageous". Problems from the past are perceived as mostly resolved and communication between stakeholders has improved. The most critique about the site came from operators themselves, where the need for optimization of processes and technology was expressed. Negative associations from local community members and the institutional side were mainly towards landfills in general, coming from experiences pre-dating the EU Landfill Directive. Table 2 gives an overview of the results from this section.

4.1.2 Concept and Attitude Towards ELFM

The concept of ELFM and distinctions to traditional LFM were perceived differently between stakeholders. Yet, all stakeholders stated a mostly positive attitude towards ELFM.

For operators, ELFM should be carried out as a private business activity. The main distinction of traditional LFM was presented by involving stakeholders. Local community members and institutional participants put a focus on material recovery using high-level recycling and sorting technologies, whereas the institutional side even expanded the concept of ELFM to Enhanced Landfill Management and Mining (ELFM^2), including managing an interim-use phase of landfills until mining activities would start. The local government has developed a code of conduct to communicate safety issues with the operators and police forces and is driven towards ELFM mainly for environmental reasons. Technology providers, in contrast to the operators, perceive ELFM as an environmental clean-up activity using advanced technology, where thermal treatment of waste could be an end-of-pipe solution, minimizing disposal costs for ELFM. The technology incubator focussed on maximizing the added value of materials, making reuse and recycling strategies a primary objective. Institutions and operators are convinced most landfills will be mined in the future, while it cannot be considered an option categorically. The scientific participant emphasized the importance of ELFM having almost no discharge flow and described it as an (economically) "risky recycling activity".

Operators are actively engaging in ELFM for profit-orientated reasons with environmental "spillovers", given a "clear, positive, net balance". Other stakeholders are motivated to engage in ELFM for environmental reasons. Nevertheless, ELFM should be able to stand economically independent from an institutional and industrial point of view. Table 3 gives an overview of the results from this section.

4.1.3 Perceived Sustainability

The perceived sustainability of ELFM is derived from the themes (ii), (iii) and (iv). Throughout the interviews, participants were asked to describe economic drivers and barriers, environmental risks and benefits and societal challenges. While economic drivers and barriers were

TABLE 2: Table 2 summarizes the Approach to Landfills.

QH/value chain-Class	Stakeholder	Approach to landfills	
		Specific Beliefs	**Common Beliefs**
Community members	De Locals	• The risk for human health • Problems with odor	• Landfills function as a temporary storage • Mostly positive associations with Remo site • Modern landfills are considered safe • Advantages: Potential for material recovery and waste removal • Disadvantages: Suboptimal use of land, environmental risks
Institutional Actors	Waste Agency	• Fewer risks than expected • Flood risks	
	European Union	• Distinction between "dumpsites" and landfills	
	Local Government	• Focus on permitting activities • Problems with operators in the past, followed by positive change	
Scientific Actors	Researcher	• Source of pollution	
Business Actors	Operators	• Waste disposal service • Process optimization needed	
	Technology Provider	• Long term risks are uncertain • Landfilling is preferred over incineration	
	Technology incubator	• Landfills as land occupant • Illegal waste dumping	

TABLE 3: Table 3 summarizes the Concept and Attitude Towards ELFM.

QH/value chain-Class	Stakeholder	Concept and attitude towards ELFM	
		Specific Beliefs	Common Beliefs
Community members	De Locals	• Focus on material recovery and advanced recycling technology • Approach from an environmental perspective	• Positive attitude towards ELFM • Not every landfill is suitable for ELFM
Institutional Actors	Waste Agency	• ELFM[2] and interim-use phase • Approach from an environmental perspective	
	European Union	• Increased resource independence • Economically independent	
	Local Government	• Close communication with operators • Environmental motivation	
Scientific Actors	Researcher	• Focus on low discharge flow and env. benefits • Risky recycling activity	
Business Actors	Operators	• The primary objective is recuperation of land, energy, and materials • Stakeholder involvement is essential for ELFM	
	Technology Providers	• Environmental clean-up activity using advanced technology • Thermal treatment is needed to avoid new disposal costs (end-of-pipe solution)	
	Technology incubator	• Focus on maximizing valorization of materials • Land recuperation as a secondary objective	

perceived similarly amongst stakeholders, the economic dimension has a different significance for different stakeholders within the sustainability framework. While most participants emphasized environmental aspects, institutions and operators focused on economic factors with environmental and social "spillovers". Environmental benefits are generally perceived through the reduction of risks through waste removal and the mitigation of primary resource production, whereas risks were described as being similar to those coming from operating landfills. The biggest societal challenge was considered the involvement of all stakeholders.

Operators and institutions both mentioned land-recuperation as the clear primary economic driver of ELFM. The industrial participants also stressed the driving force of "doing activities" in the form of large-scale pilot projects. One participant stated that "when we start mining the Remo site, from this one activity, many spin-offs will develop". They further emphasized the economic advantages of technological development in cost reductions. In agreement with the scientific participant and the technology incubator, operators are favoring the idea of combining public and private money for investment support. This could take the form of private-public partnerships, subsidies or public insurances. While institutional actors were not as fond of this idea, they perceived a driver in cost reductions for long-term monitoring through ELFM and an interim use-phase. Local communities identified the generation of employment, especially of low-skilled labor, as well as energy generation and material recovery as main drivers for ELFM. External factors, like market prices for primary and secondary raw material, could be driving ELFM projects, if rising but also hinder development if decreasing. Similarly, operators stated that technological development, generally perceived as a driver, could also be a barrier to investment if new technologies emerge before the planned return on investment. Institutions and operators described finding investors in general as one of the most difficult challenges for ELFM. This is explained partly by a lack of awareness in the relevant sectors and partly by (un)known risks in the development of market prices, new technologies and public acceptance: "You get investment support a bit here, a bit there. So, you have to puzzle all these small supports for your big investment, and this is, of course, time-consuming." Operators emphasized that high monitoring and sampling activities would drive up costs and could hinder implementation. Site-specific factors, like the location of the landfill and waste composition, could also be a relevant barrier or driver, depending on the context. Table 4 gives a more detailed overview of the economic perceptions about ELFM.

While most environmental benefits are perceived through the mitigation of risks, technology providers further mentioned that technological development could lead to improvements in future landfilling and recycling operations, and thereby have indirect environmental benefits. The main risks perceived were odor, noise, and risks for human health coming from dust or groundwater contamination. Formerly uncontrolled dumped waste could pose risks to ELFM operations when discovered and toxic materials could be brought back into the material cycles. Institutional and local community members also expressed their concerns about auto-combustion of gases initiated by the change of anaerobic to aerobic conditions in landfills. Additionally, the scientist believes bad execution could lead to bigger environmental problems than before: "These are huge risks, also on the environmental level the risk of creating a bigger environmental problem than before is still there." Operators also mentioned that the energy consumption of ELFM activities today is mainly fossil fuel based. Table 5 shows the main environmental risks and benefits according to the different stakeholder groups.

Stakeholder involvement, perceived as the biggest societal challenge, could affect ELFM implementation in va-

TABLE 4: Table 4 summarizes the Economic Drivers and Barriers.

QH/value chain-Class	Stakeholder	Economic Drivers and Barriers	
		Specific Beliefs	**Common Beliefs**
Community members	De Locals	• Material recovery and job generation • Long-term project costs	• Changes in market prices for primary and secondary raw materials affect the economic feasibility of ELFM • Location of the landfill and waste composition can be a driver or a barrier
Institutional Actors	Waste Agency	• Avoidance of long-term monitoring costs • Interim use of landfill can reduce costs • Lack of knowledge with investors is a barrier	
	European Union	• ELFM should be driven by private businesses	
	Local Government	• Industrial symbiosis is needed	
Scientific Actors	Researcher	• Emphasis on environmental aspects • Public financial support is important	
Business Actors	Operators	• Business activity with environmental benefits • Technological uncertainty can hinder investments • Lack of public financial support for pilot projects	
	Technology Provider	• Profitability of ELFM is in question • Sorting technology is not efficient enough • Financial uncertainty poses a long-term risk	
	Technology incubator	• Hydrogen production could be an essential driver • The flexibility of outputs ca drive ELFM • Material recovery is a long-term driver	

TABLE 5: Table 5 Summarizes Environmental Benefits and Risks.

QH/value chain-Class	Stakeholder	Environmental Benefits and Risks	
		Specific Beliefs	**Common Beliefs**
Community members	De Locals	• Risks of toxic materials being reintroduced into the material cycle Risks for natural habitat on top of old landfills	• Reduction of risks through waste removal and avoidance of primary resource consumption • Mitigation of groundwater pollution and soil • Risks of ELFM are similar to current/traditional landfilling operations • Risks for odor, noise and human health
Institutional Actors	Waste Agency	• The risk for auto combustion of gases	
	European Union	• Environmental benefits on global level Recuperation of construction materials is important env. factor	
	Local Government		
Scientific Actors	Researcher	• High operational risks	
Business Actors	Operators	• Risks for air and groundwater pollution	
	Technology Provider	• Uncertainty about long-term environmental impacts ELFM in combination with CCS can improve environmental performance	
	Technology incubator	• Waste composition is a risk Uncontrolled dumping poses risks	

rious ways. Operators fear public opposition by non-involvement, but also consider a need for more awareness of ELFM, in general, to make financing and permitting processes easier. All stakeholders identified a lack of public acceptance as a project's biggest societal barrier at this time: "That's the barrier number one." Operators, institutions and local community members explained this partly by knowledge and awareness gaps between the different parties involved, adding to concerns about the environmental risks. According to an institutional participant knowledge distribution should also include public authorities, stating, "[The] most important thing, from my point of view, is the transitioning of the mindsets, that's a policy aspect." Local community members also urged for the inclusion of politicians in this process and criticized the conflict of interest between short-term politics and long-term development. The participant from the local government, on the other hand, mentioned the organization of town hall meetings, being not very well visited, and explained that positive change by the operators is often not recognized within the community, while small mistakes are overemphasized. This view is congruent with the beliefs of local community members and operators, who see a barrier in small groups being able to hinder a project through legal procedures, overpowering a "silent" but supportive majority. A situation where "a small group talks for a large community that doesn't talk." In Table 6, an overview of the perceived societal challenges can be found.

4.1.4 Key Actors of ELFM

All stakeholders, but the operators themselves, who perceived investors as highly important, named the ope-

TABLE 6: Table 6 summarizes the Societal Challenges.

QH/value chain-Class	Stakeholder	Societal Challenges	
		Specific Beliefs	Common Beliefs
Community members	De Locals	• Fear of environmental impacts • Supporters of ELFM do not participate as actively as opponents	• Public involvement is perceived as the biggest challenge • Stakeholder involvement perceived as an advantage for ELFM implementation • Recuperation of land for recreational purposes can help to get acceptance for ELFM implementation • General legal framework can hinder ELFM implementation
Institutional Actors	Waste Agency	• Integration of political actors is necessary • Conflicts of interests between short-term (political) projects and long-term development	
	European Union	• Reuse and recycling is preferred over primary resource use	
	Local Government	• Complaints are often subjective • Positive change is rarely recognized • Natural habitat and safety concerns within citizens	
Scientific Actors	Researcher		
Business Actors	Operators	• Fear of public opposition • Regulations for non-ELFM production • Need for more awareness about ELFM in general public and investors	
	Technology Providers	• Societal and environmental pressures differ in location	
	Technology incubator	• Regulatory instruments are needed for ELFM implementation	

rating company as the most important key actor involved. Regulatory bodies should play a crucial role according to all stakeholders. The institutional and scientific side also stressed the importance of involving local communities. However, Institutions perceive the general public as even more important than local residents. Scientific bodies are mostly perceived as platforms for knowledge transfer between the involved parties but would play a secondary role in the realization of ELFM projects. Technology providers emphasized their own role by stressing needs for optimizing sorting technologies.

All stakeholders perceived the role of institutions as an overall positive. Most participants named the Flemish waste agency one of the key actors involved and were overall satisfied with their role. The subsidiarity principle of the EU was positively acknowledged by institutional participants, who also perceived their regional role as a platform for experimentation and trials. It was criticized by communal, scientific and institutional participants that advice from regulatory bodies is often not followed on a political level. Similarly, technology providers and the technology incubator, institutional participants and operators would appreciate regulations that “help and stimulate landfill mining activities” and make them easier to monitor, but could not identify any current regulations “hampering” ELFM implementation.

4.2 Stakeholder Archetypes

To structure the diverse and complex perspectives, stakeholder archetypes were developed. Each stakeholder type ought to be understood as a prototype for a distinct approach to ELFM implementation to facilitate the understanding of different stakeholders, and tailor research and industrial activities to stakeholder needs. In total five different stakeholder archetypes have emerged from the analysis: The Engaged Citizen, the Entrepreneur, the Technology Enthusiast, the Visionary and the Skeptic. If certain concepts in one coding category were interlinked with a dominant stakeholder class, they were grouped to represent a district archetype. Some archetypes share common beliefs, as overlapping coding was applied, but a new type was developed when a distinct property or belief differed substantially from other combinations or concepts were contradicting each other.

4.2.1 The Engaged Citizen

The Engaged Citizen approaches ELFM from an environmental perspective. Her or his main concerns are the safety and well-being of their community. The avoidance of odor, noise, and traffic, as well as the mitigation of environmental risks affecting human health, are a main priority. To achieve influence on a project, Engaged Citizens actively participate in the implementation process and seek to gain and distribute information. While their influence on a specific ELFM project can be quite high, their overall impact on ELFM implementation as an industrial sector is rather low. Engaged Citizens organize in a non-governmental form but have access to various resources due to the diversity of their group. A rather risk-averse attitude in combination with a curiosity for technology and innovation drive them. Because of their environmental approach to ELFM, financing models are considered less important. Problems often occur in communication with other stakeholders and are related to knowledge gaps about technologies, regulations and project details. Yet, through engagement, the Engaged Citizen gains information and establish a moderate knowledge base.

4.2.2 The Entrepreneur

The Entrepreneur approaches ELFM from a private economic perspective. While a profitable business is a primary concern, environmental and societal factors of a project are also important. Land recuperation and energy generation are seen as main drivers by Entrepreneurs, while uncertainties add to their hurdles. These include waste compositions, investment support, and regulations. The Entrepreneur can highly influence a specific ELFM project as they are usually part of a private business along the value-creation-chain of ELFM. Her or his overall impact on ELFM implantation can be considered moderate to high but depends on the interconnectedness with other stakeholders. Entrepreneurs present a willingness to take risks and a high knowledge base about ELFM processes. Because of the presumed environmental benefits of ELFM, they expect public financial support for ELFM implementation.

4.2.3 The Technology Enthusiast

Technology Enthusiasts approach ELFM from an innovative perspective. The development of new technologies is seen as the main driver of ELFM implementation. While the Technology Enthusiast clearly sees a need for private economic profitability of ELFM, her or his true motivations to engage lay in creating environmental benefits through technology. Combining thermal treatment of waste streams with carbon capture and storage (CCS) technology, for example. Technology Enthusiasts emphasize the potential for hydrogen production of ELFM and see a need for revising waste management regulations to emphasize the storage function of landfills. They take a long-term view on ELFM implementation and are willing to take financial risks. Since the Technology Enthusiast is usually, but not necessarily, engaged in ELFM projects along the value-creation-chain, she or he favors public financial support similarly to Entrepreneurs. Due to their engagement, Technology Enthusiasts provide a high knowledge base on ELFM processes, but often have difficulties understanding the needs of local communities or policymakers.

4.2.4 The Visionary

The Visionary approaches ELFM from a societal perspective. She or he believes that societal change is necessary for gaining environmental benefits. ELFM can function as a vehicle for this change, which is driven by technological development. The mitigation of future and long-term environmental burdens motivates a Visionary's engagement. Visionaries are usually part of a governmental institution and involved in policymaking. For them, strategic advantages through increased resource independence play a crucial role in ELFM. While their influence on specific ELFM projects is low, their impact on ELFM implementation is high. From a Visionary's point of view, ELFM could very well be implemented as a public activity, given the environmental and societal benefits are sufficient. She or he considers the general public rather than local communities. This emphasizes the importance of environmental risk mitigation for Visionaries. They have a high knowledge base about environmental and societal aspects of EFLM but lack knowledge of technical processes and project-specific needs.

4.2.5 The Skeptic

Skeptics approach ELFM mainly from an environmental perspective but are convinced ELFM needs to be feasible as a private economic activity to achieve brought implementation. Focusing on risks, A Skeptic tends to create a self-enforcing perspective and develop a rather risk-averse attitude. She or he expects ELFM implementation to take its time. Being part of a governmental or research institution, Skeptics see a need for investigating the implications of ELFM implementation and its relations with other industry sectors to add to their high and sometimes very specific knowledge-base. Their influence on a specific ELFM project can be moderate to high but overall impact on ELFM implementation is rather moderate to low.

5. DISCUSSION

While it is important to discover the different approaches of stakeholders, their perspectives must be put into context. The main concepts expressed are contrasted to former research findings on the Remo case. This comparative approach should provide new information explaining the justifications of beliefs and knowledge gaps across stakeholders. The first part of the discussion, Perceptions about the Remo landfill, is limited to a comparison with former research about the case. The second part, Implications for ELFM implementation, takes a more general view on ELFM and explains how implementation could differ when applying different perspectives. The last subsection, Implications for future research, gives an outlook about the direction of future ELFM assessment. The latter two subsections also explain how the stakeholder archetypes can be made applicable.

5.1 Perceptions about the Remo landfill

Looking at the waste composition at the Remo site, research indicates that beliefs by community members about toxic materials are not justified (cf. Quaghebeur et al. 2013). Danthurebandara et al. (2015a) even show that impacts from ELFM operations on human toxicity can be beneficial. However, in contrast to environmental burdens the impact category was insignificant (Danthurebandara et al., 2015b). Modern landfills are generally perceived as safe and even conditions of older landfills as being better than expected. But, as the waste composition is uncertain and can vary dramatically within one landfill site, sampling becomes either less effective or cost intensive (Quaghebeur et al., 2013). These circumstances, in combination with incomplete records and illegal dumping of waste, put beliefs about the safety of landfills generally into question. Nonetheless, at the beginning of the 20th century, about 80% of MSW consisted of ashes from residential heating and inert or easily degradable materials (Van Passel et al., 2013).

Land reclamation, material, and energy recuperation are considered to be the main revenue streams for ELFM operations at the Remo site. Especially community members perceived material recovery as a major driver for ELFM operations. This is questionable. The Remo landfill lays within a natural habitat where the land price is rather low. Van Passel et al. (2013) identify land reclamation to constitute a relatively low benefit and note that government incentives for renewable energy make up a major portion of the WtE revenue stream. They show the three most important impacts on ELFM's private economic performance are (i) WtE efficiency, (ii) electricity- and (iii) CO_2-price. This claim is supported by Danthurebandara et al. (2015b), who identify the plasma gasification process as a major economic impact and its efficiency as the main factor affecting the profitability of ELFM at the Remo site. This again shows the importance of changing market conditions, which all stakeholders perceived as one of the biggest challenges for ELFM implementation.

Danthurebandara et al. (2015a) also support the stakeholders' beliefs that high investment costs are a main barrier for implementation and identify investments in WtE technology as a major cost component. While technological development would push ELFM it could also hinder investments by raising uncertainty.

Some environmental benefits of ELFM have been assessed by Van Passel et al. (2013) and Danthurebandara et al. (2015a). Van Passel et al. (2013) conclude that benefits from a reduction in greenhouse gases through material recovery have the biggest impact. This contrasts with Danthurebandara et al. (2015a), who identify an environmental burden in the impact category Climate Change, and most benefits in the impact categories Fossil Depletion, Ionizing Radiation and Urban Land Occupation. The differences in GHG emissions is explained by distinct approaches: Van Passel et al. (2013) consider a longer methane recovery and purchasing materials and energy on the market for the do-nothing scenario (Danthurebandara et al. 2015a). Danthurebandara et al. (2015a) show all impact categories have beneficial effects but were not significant, other than the impact categories climate change and ozone depletion.

A topic mostly neglected by the stakeholders is biodiversity. Although impacts on biodiversity through ELFM are positive due to land reclamation, temporal burdens on biodiversity can occur during the time of operation (De Vocht et al., 2011). Overall, aiming at 75% open landscape after operations could lead to the restoration of 162 ha of Flemish heathland, representing 1.17%-1.75% in relative terms. Additionally, disturbance trough illumination, noise or transport can affect biodiversity negatively, however, covered WtE and WtM installations could help to minimize the risk. Impacts on the aquatic system are expected to be minimal, as the Remo site is situated above the groundwater level (De Vocht et al., 2011).

The belief that public involvement is one of the biggest societal challenges is well manifested within stakeholders. This is reasonable, regarding the Remo case, as public opposition has led to delays. Yet, this belief cannot be transferred to ELFM in general. The general perception of ELFM was described as positive even within opposing groups to the Remo site. Stakeholder involvement and communication were highlighted by several participants and therefore contradicts the belief about communication problems amongst stakeholders.

5.2 Implications for ELFM Implementation

Another societal challenge was not mentioned explicitly but can be derived implicitly from the interviews: Different stakeholders approach ELFM with different motivations. Should ELFM be implemented primarily as a clean-up activity or as a business activity? Depending on which point of view one takes, different implications come to light. As a clean-up activity, ELFM would be mostly done by governmental institutions and resource and energy recovery would have a cost-reducing objective. As a business activity, ELFM would be profit-driven, where conflicting goals can lead to trade-offs with its environmental performance. Of course, inner-dimensional trade-offs between environmental impact categories still have to be considered, even without motivations for profit maximization. A mixed approach could lead to cherry-picking by industrial actors and higher societal costs at the end, as cross-financing of less profitable projects becomes more difficult.

All stakeholders have a positive attitude towards ELFM. This is not very surprising considering their active involvement at the Remo site. Still, this attitude is also in line with the European strategy to transfer into a circular economy and reduce burdens from carbon dioxide (EC, 2015; UN, 2016). Even community members opposing the Remo case were not considered to be against ELFM in general, but opposition is rather linked to specific issues and ongoing landfilling operations (Internetgazet et al., 2018). The stakeholder archetypes can enhance the understanding between different ELFM practitioners. This can increase awareness about ELFM and help anticipate public opposition by integrating different perspectives. Policymakers can gain insights on important matters regarding ELFM implementation and avoid future conflicts without having to do time-consuming, and costly research on a project.

Another challenge for ELFM implementation lays in current and future regulations. Interestingly, landfills are in general perceived as temporary storage facilities by all stakeholders. This might be explained through their involvement in a specific ELFM case, and thus, a higher awareness for other perspectives. Technology providers preferring landfilling over incineration in contrast to the waste disposal hierarchy, supports the view of landfills as storage facilities, similarly to Van Passel et al. (2013). The development of ELFM puts this waste disposal hierarchy into question. As new technologies might emerge, higher benefits could be possible, when also landfilling current waste streams and processing, i.e. mining, them later on. In this context, landfill taxes can play a crucial role. In research these are mostly considered to be costs, taking a private economic perspective (Johansson et al., 2013; Winterstetter et al., 2015). The implied societal benefits (tax revenues) are usually not considered. Moreover, it is often unclear if these taxes have to payed or if exemption of landfill taxes would be granted, raising uncertainty about future outcomes. In Sweden and Austria, exemptions are possible but also depends on the composition and age of the redeposited waste (Hermann et al., 2014; Johansson et al., 2012). On the other hand, the exemption from taxes always implies a societal cost, that has to be considered. Hoogmartens et al., (2016) show, for example, that welfare maximization, through the combination of Enhanced Waste Management and optimal taxation, is possible. However, they focus on current waste streams and more research is needed.

5.3 Implications for Future Research

When considering implications for future ELFM research, especially from a societal perspective, it becomes evident that more work is needed. To derive implications for regulatory changes and to better understand the real potential of ELFM, it is important to take a holistic, industrial perspective into account. Estimating the resource potential of ELFM for Europe, for example, is a necessary next step, but not easy to achieve. Yet, it could help justify or deny public support and help design optimal monetary control and management tools to foster a sensible ELFM implementation. This research should be integrated to make interdimensional trade-offs visible.

Considering the private economic dimension of ELFM, the analysis shows that hidden private costs for stakeholder involvement, for example, have not been taken into account. Commonly, only operational and capital costs (e.g for transport, facilities or personnel costs) are assessed (Danthurebandara et al., 2015a; Frändegård et al., 2015; Kieckhäfer et al., 2017; van der Zee et al., 2004; Wolfsberger et al., 2016; Zhou et al., 2014). Additionally, time effects should be considered when building investment and cost models. While discounting is usually applied when assessing a project's net present value (NPV) (Hermann et al., 2016a; Van Passel et al., 2013; Winterstetter et al., 2018), delays through social resistance or permitting processes are not considered.

The main environmental benefits are believed to come from the mitigation of risk through waste removal. To put those beliefs into context, it is important to identify long-term risks of landfills, but this challenge still has to be taken on (Sauve and Van Acker, 2018). Waste composition, depending on factors like location, regulations or the time period of landfilling, plays a crucial role in determining these long-term environmental impacts (Quaghebeur et al., 2013). Institutions share this point of view, showing awareness for monitoring activities exceeding the obligatory 30 years aftercare period. Environmental impacts of ELFM operations are comparable to traditional landfilling. Since ELFM operations are expected to go on over a timeframe of 10 to 20 years, and environmental impacts of landfills would accumulate over time, the assumption can be made that there are environmental benefits from mitigating long-term environmental risks through ELFM. The extent of these benefits is still difficult to assess, making an economic evaluation of externalities for ELFM ambitious.

The societal dimension of sustainability is usually assessed through the monetization of environmental impacts, if at all (Damigos et al., 2015; Marella and Raga, 2014; Van Passel et al., 2013; Winterstetter et al., 2018, 2015). Fewer studies tackle societal impacts through non-monetary assessment (Hermann et al., 2016b; Pastre et al., 2018). Monetizing environmental impacts is problematic because impacts are chosen selectively and often do not represent a holistic picture. Non-monetary societal impacts are often left out, due to their subjectivity. If integrated, their validity is in question, specifically because of their subjective character. The developed archetypes can help to integrate different subjective approaches rather than creating a seeming objectivity through monetization. One option could be developing different weighing factors from the archetypes to integrate them into ELFM assessment methods. This way, the effect of different perspectives on societal impacts could be made visible and enhance the discussion on social burdens and benefits. The archetypes could be used in an educational context and help to understand inner- and interdimensional trade-offs better when assessing the sustainability of ELFM projects.

Regional differences should be taken into account when assessing ELFM. Damigos et al (2015) conduct a contingent valuation survey in Greece to determine and monetize stakeholder values. In contrast to the interviewees, the participants of the survey value job creation (70%) as their main incentive to engage in LFM operations only followed by environmental benefits (22.4%). Survey participants recognize water, soil, and air pollution as the biggest operational risks of landfilling, from which perceptions about perceived risks of ELFM operations can be derived. About 60% of survey participants valued WtE and WtM benefits as most important, whereas approximately 20% of participants valued the avoidance of environmental burdens, and equally landfill space reclamation, as very important (Damigos et al., 2015). These beliefs have to be further assessed. While it has been shown that WtE plays a crucial role in gaining private economic benefits from ELFM, WtM streams have proven less profitable (Van Passel et al. 2013; Danthurebandara et al. 2015a). Societal benefits and their monetization, however, need more scientific attention.

6. CONCLUSIONS

Landfills were perceived as temporary storage facilities and knowledge about ELFM was mostly well established. All stakeholders constitute a positive attitude towards ELFM, but motivations for engagement differ amongst stakeholders. Misconceptions exist about the main economic drivers for ELFM implementation, where industry and institutional actors identify land recuperation and communal actors material recovery as main drivers. Homogenously, stakeholders identified environmental benefits coming from the mitigation of risks through waste removal and avoidance of primary resource consumption. Stakeholder integration was perceived as the main societal challenge.

The fife stakeholder archetypes, namely the Engaged Citizen, the Entrepreneur, the Technology Enthusiast, the Visionary, and the Skeptic, outline the main perspectives to be taken on ELFM implementation. They convey major differences in approaching ELFM and new technological concepts alike and serve as a tool for ELFM practitioners and researchers, who seek a better understanding of the parties involved. Moreover, they can be used for educational purposes to enhance understanding of sustainability issues. They make inner- and interdimensional conflicts of sustainability visible and help understand the societal side of ELFM.

It is important to note that implementing ELFM at industrial scale and scope depends on its main purpose. If ELFM is primarily done as a business activity aiming for profitability, in contrast to a clean-up activity, different regulatory changes become necessary. A wide range of policy instruments including taxation, subsidies, public-private partnerships, investment support and more, have to be carefully analyzed and tested. This implies the need for new models in ELFM assessment integrating all dimensions of sustainability in a comprehensive and comparable manner.

Future research has to refine the private economic and environmental assessment, taking hidden costs and benefits and dynamic time effects into account. Special focus should be given to the societal dimension, which lacks a thorough assessment in ELFM research.

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ANNEX A

The appendix provides the developed interview guide providing the basis for the coding categories used to analyze the data.

Section 1: Approach to Landfills

Question 1
What is a landfill for you?
Follow up

- **Is it a waste management solution, a source of pollution, a resource reservoir or a land occupant?**
  - Why do you think this is the main function of a landfill?
  - How do you think this function is met?
- **What other functions does a landfill have?**
- **For the functions, you just described, what advantages and/or disadvantages can you see?**
- **While recycling becomes more and more important, why do you think landfills are still needed?**

Question 2
Can you, in general, describe what advantages and/or disadvantages having landfills comes with?
Follow up

- **Do landfills have an effect on your day-to-day life?**
- **Do you benefit from landfills? (Who benefits from landfills?)**
  - How? (get rid of waste, profits, protect society/environment, etc.)
- **What factors influence your perception of landfills that are uncertain?**
- **How safe do you think landfills are?**
  - Where do you think to lay unforeseeable risks of landfills?

If yes:

- **How are landfills managed to keep them safe? (How should landfills be managed to keep them safe?)**
- **What risks remain by landfilling waste?**
- **What should change to make landfills even safer?**

If no:

- **Why do you consider landfills unsafe?**
- **To whom are landfills unsafe?**
- **What exactly about landfills do you consider unsafe? (Management, processing, transport, after-(after) care, etc.)**
- **How could you manage landfills in a safer way?**

Question 3
When you think about the REMO site, do you have positive or negative associations?
Follow up

- **Can you explain where these associations come from?**
- **Specific to this landfill, what is its main function to you?**

- What experiences have you made with this landfill and/or its management?
- Are you satisfied with the current management of this landfill?
- What alternative options do you see for the future management of this landfill?
  - Which options would you prefer and why?
  - Which options would you avoid and why?

Section 2: Involvement in ELFM

Question 4
Are you familiar with the concept of LFM/ELFM?
Follow up
- Please describe your idea about LFM/ELFM to me.
- How did you learn about LFM/ELFM?
- How did you get involved with LFM/ELFM?

Question 5
Do you think LFM/ELFM should be done?
Follow up
If yes:
- How should LFM/ELFM be carried out?
- Who should be involved in such a project?
- Why should LFM/ELFM be done?
- Where should LFM/ELFM be done?

If no:
- Why not?
- Do you see risks in leaving a landfill untouched?
- Who do you think is/should be responsible for impacts after 30 years (the after-care period)?
  - Who should pay for it?
  - How should this issue be handled?
- Who do you think benefits from LFM/ELFM and why?
- Do you see LFM/ELFM as a recycling, mining, business, environmental, protective or risky activity?

Question 6
What projects about LFM/ELFM are you involved with?
Follow up
If any:
- What is your role in these projects?
- Why do you want to be part of this project? What motivates you?
- What impact has your involvement on your life/current situation?

If none:
- Why are you not involved?
- Would you like to get involved?
- How could you get involved?
- If you would get involved, what would your objective be?
- Why would that be your objective?

Section 3: Benefits of ELFM

Question 7
What are the main advantages/opportunities you see in LFM/ELFM projects?
Follow up
- For whom do you mostly see these advantages/opportunities?
  - Do you see mostly economic, environmental or societal opportunities?
- How could these opportunities be reached?
- Where do you see limits to these opportunities?
- What factors influence these advantages/opportunities that are uncertain to you?
  - Why do you consider these uncertainties?
  - What measures could be taken to reduce these uncertainties?

Question 8
According to you, which are the main environmental benefits of LFM/ELFM?
Follow up
- What types of environmental benefits exactly do you have in mind? (resource conservation, land use, groundwater safety, smell mitigation, pollution control, etc.)
- Where do you see these different benefits? (On which level? Global, national, regional, local?

- For whom do you see these benefits? Why?
- How could others also benefit from LFM/ELFM?
  - How could benefits be transferred to other levels?
- Are you sure, these benefits can be reached?
  - Why are you uncertain/certain about these benefits?
  - How could you make sure these benefits are reached?

Section 4: Risks of ELFM

Question 9
What main disadvantages/risks do you see with the realization of an LFM/ELFM project?
Follow up
- For whom do you mostly see these risks?
- Do you see mostly economic, environmental or societal risks/disadvantages?
- Why do you consider these to be risks/disadvantages?
- Why are you afraid of these risks?
- What would minimize these risks?
- What factors influence these disadvantages/risks that are uncertain to you?
  - Why do you consider these uncertainties?
  - What measures could be taken to reduce these uncertainties?

Question 10
According to you, which are the main negative environmental impacts/risks of LFM/ELFM projects?
Follow up
- Who is affected by these impacts?
- Who is responsible for these impacts?
- How could these impacts be avoided or limited?
- Are you sure, these impacts will occur?
  - Why are you uncertain/certain about these impacts?
- Who will pay for these impacts? How?
- Who should pay for these impacts? How?

Section 5: Challenges for ELFM

Question 11
According to you, which are the main challenges for the realization of LFM/ELFM projects?
Follow up
- Are these challenges mostly related to economic, environmental, regulatory, market-related or organizational matters?
- Why do you consider these to be the main challenges?
- How would you address these challenges?
  - In your opinion what are factors that influence the feasibility and performance of LFM/ELFM projects most?
  - Where do you see uncertainties in these challenges?
  - How could these uncertainties be minimized/controlled?

Question 12
What economic drivers and/or barriers can you identify?
Follow up
- How do these drivers/barriers affect (your) LFM/ELFM projects?
- How do these drivers/barriers work? Please explain the mechanisms.
  - Who is able to affect these mechanisms?
  - Where do uncertainties in these mechanisms remain?
- How could these drivers/barriers be emphasized/regulated/overcome?
- Where do you see economic limits to LFM/ELFM?

Question 13
What regulatory instruments do you know affecting LFM/ELFM projects?
Follow up
- What financial and regulatory instruments do you know driving/hindering development towards LFM/ELFM?
- How do these regulatory instruments work? Please explain the mechanism.
  - How do these mechanisms address uncertainty?
- What are the most important aspects?
- What regulations are in place to make LFM/ELFM safer/lower risks/profitable?
  - Why?
  - What aspects drive LFM/ELFM?
  - How?

- For whom?
- What regulations should be changed to promote LFM/ELFM?
  - What aspects hinder LFM/ELFM?
  - Why? How?

Question 14
Where do you see markets of the products/outcomes of LFM/ELFM?
Follow up
- In your opinion, is there a need for additional materials and/or energy from LFM/ELFM?
- Where do you see difficulties for the marketing products/outcomes from LFM/ELFM?
  - What are the uncertainties affecting these difficulties?
  - How could you manage these uncertainties?
- Who are purchasers of these products/outcomes?
- Who are competitors to these products/outcomes?
- Where do you see advantages to competitors?
- Where do you see disadvantages to competitors?

Question 15
What societal challenges do you expect/have you experienced in LFM/ELFM projects?
Follow up
- Why do you consider these societal challenges?
- What did you learn from your experience?
- How would you address these challenges?
- What instruments could/should be installed to communicate/educate about LFM/ELFM projects?
  - How do you communicate/educate about LFM/ELFM?
  - To whom do you communicate about LFM/ELFM?
  - With what purpose/intent do you communicate LFM/ELFM
  - How should different stakeholders be integrated into LFM/ELFM projects?

Section 6: The Role and responsibilities of Institutions and other ELFM Actors

Question 16
According to you, which are the most influential actors when it comes to the planning and realization of LFM/ELFM projects?
Follow up
- What do you think are these different actors' roles and responsibilities?
- What is your/your institution's role and responsibility in LFM/ELFM projects?
  - How do you put this role into practice?
  - Who is primarily affected by your role in LFM/ELFM projects?
- Where do you see the need for change in your role in LFM/ELFM projects?

Question 17
Who do you think is/should be responsible for regulating and/or communicating LFM/ELFM?
Follow up
- EU, Federal, Regional, Local?
- Who do you think is deciding right now if LFM/ELFM is done?
- What is their role in this process?
- How should their role be changed/differ from its current state to get better outcomes? For whom?
- Who should decide if LFM/ELFM is done?

Question 18
How do/does the authorities/your institution deal with uncertainties concerning LFM/ELFM projects?
Follow up
- What areas are mostly affected by uncertainties (Economics, environment or society?)
    - What regulatory instruments do you know handling uncertainties?
    - How do they work? Please explain the mechanism.
- What should be changed about them to get better effects?

Question 19
Are you happy with the role of institutions/authorities when it comes to LFM/ELFM?
Follow up
- What are they doing well?
- Where do you see the need for change?
- What regulations should be changed to make LFM/ELFM safe?
  - Why? How? What aspects?

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ASSESSING THE ECONOMIC POTENTIAL OF LANDFILL MINING: REVIEW AND RECOMMENDATIONS

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ABSTRACT

As landfill mining (LFM) gains public attention, systematic assessment of its economic potential is deemed necessary. The aim of this review is to critically analyze the usefulness and validity of previous economic assessments of LFM. Following the life cycle costing (LCC) framework, (i) the employed methods based on goal and scope, technical parameters and data inventory, and modelling choices were contrasted with respect to (ii) the synthesized main findings based on net profitability and economic performance drivers. Results showed that the selected studies (n=15) are mostly case study-specific and concluded that LFM has a weak economic potential, hinting at the importance of favorable market and regulation settings. However, several method issues are apparent as costs and revenues are accounted at different levels of aggregation, scope and scale—from process to sub-process level, from private to societal economics, and from laboratory to pilot-scale, respectively. Moreover, despite the inherent large uncertainties, more than half of the studies did not perform any uncertainty or sensitivity analyses posing validity issues. Consequently, this also limits the usefulness of results as individual case studies and as a collective, towards a generic understanding of LFM economics. Irrespective of case study-specific or generic aims, this review recommends that future assessments should be learning-oriented. That is, uncovering granular information about what builds up the net profitability of LFM, to be able to systematically determine promising paths for the development of cost-efficient projects.

1. INTRODUCTION

The shift from a linear to a circular economy has influenced the perception of landfills as final waste deposits. Apart from minimizing waste flows through circular design, production, and use (Ellen MacArthur Foundation, 2013), keeping resources in the loop also extends through considering landfills as anthropogenic stocks (Cossu and Williams, 2015; Johansson et al., 2012; Jones et al., 2013; Krook and Baas, 2013). The potential of extracting these previously deposited resources is increasingly gaining public attention (Financial Times, 2018; World Economic Forum, 2017) and is commonly referred to as landfill mining (LFM).

Although LFM has been in practice for nearly 70 years, the motivation for performing it has changed over time (Hogland et al., 2010). As a concept, it has gradually progressed from an initial focus on local landfill management issues and pollution risks, to an increasing emphasis also on the recovery of deposited materials and energy resources (Krook et al., 2012). The most recent concept of LFM even targets a zero-waste approach by including innovative resource recovery technologies, as well as extending the typical process chain (i.e. excavation, separation, and thermal treatment) with more downstream residue valorization processes (Danthurebandara et al., 2015a; Hernández Parrodi et al., 2018; Jones et al., 2013). Furthermore, the motivation for such projects has been suggested to go beyond traditional economic and environmental impacts by also considering revitalization of ecosystem services (e.g. land-use services) and broader sustainability perspectives (Burlakovs et al., 2017). Although these changes in the LFM concept try to capture a wider societal potential, there is also an inevitable increase in complexity when it comes to both its realization and sustainability consequences.

At present, however, the recovery of materials and energy resources from landfills remains at the niche level or at a laboratory to pilot scale level (Johansson et al., 2012). This gives a hint on the compelling challenges for realizing

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such projects. Beyond the technological challenges, the implementation of LFM is also subject to the complex web of political, organizational, environmental, and economic considerations (Hermann et al., 2016; Johansson et al., 2017; Krook et al., 2015; Van Der Zee et al., 2004), which is common to emerging concepts (Hekkert et al., 2007). In Europe, although LFM failed to be integrated into the recent amendment of the EU Landfill Directive (1999/31/EC), its implementation is neither prohibited (European Parliament, 2018). In fact, several LFM research projects are being funded (European ELFM Consortium, 2019), especially in the view of landfills as secondary sources for critical metals (Løvik et al., 2018). Moreover, going beyond research and envisioning a full-scale and widespread LFM implementation, development of sustainable projects should be assured to attract the support of various stakeholders (Hermann et al., 2014; Krook et al., 2018a; Van Der Zee et al., 2004).

To enable structured assessments of various systems (e.g. products, services, projects and policies), different systems analysis tools (Ahlroth et al., 2011; Finnveden and Moberg, 2005) have been widely used addressing separated or integrated sustainability aspects (Guinée, 2016; Heijungs et al., 2013). These assessments can serve multiple purposes (Finnveden and Moberg, 2005; ISO, 2006a; Swarr et al., 2011). A common objective of such studies is to obtain an accurate result on the net performance of certain systems to support decisions on capital investments or for marketing reasons. In contrast to such decision-oriented purposes, systems analysis tools can also be used to obtain a more in-depth understanding of what builds up the net performance of the system in question. Such learning-oriented purposes are often used to identify strategies and measures to further improve the performance of various systems through optimization and design development. These are particularly useful in guiding the development of emerging concepts through early assessments, or so-called ex-ante assessments (Cucurachi et al., 2018; Fleischer et al., 2005; Wender et al., 2014).

Although most decisions related to real-life projects rely on the economic potential (Martinez-Sanchez et al., 2015), studies accounting for environmental impacts are more common within the field of waste management (Laurent et al., 2014a, 2014b). When it comes to LFM, however, several economic assessments were done in recent years (Krook et al., 2018b). However, there is not yet any systematic synthesis of their main findings regarding the feasibility and challenges for the implementation of such projects. In addition, acknowledging that LFM is still an emerging concept with large practical knowledge deficits (e.g. lack of actual data, setting of best estimates, and upscaling), inherent large assessment uncertainties are expected and have to be properly addressed as pointed out in ex-ante assessments (Clavreul et al., 2012; Hellweg and Milà i Canals, 2014; Martinez-Sanchez et al., 2015). Thus, a methodological review of what uncertainties were accounted for and how they were subsequently handled is deemed necessary to reveal the quality of the main findings.

This review aims to critically analyze previous economic assessments of LFM in terms of the usefulness and validity of their provided results. In doing so, individual objectives and employed methods are considered as well as their collective contribution towards a generic understanding of the economic potential of LFM. Here, usefulness therefore both corresponds to the fulfilment of the intended objective of the assessment and the type of knowledge of relevance for LFM implementation that is addressed. The validity refers to whether the expected methodological rigor was followed according to certain standards (Swarr et al., 2011). Apart from that, different objectives of economic assessments require different methodological approaches, assuring validity also qualifies the real usefulness of provided results. That is, the results may have perceived usefulness as presented in the studies, but the corresponding validity may indicate otherwise, revealing their real usefulness. The specific research aims are (i) to review the methods in terms of goal and scope definition, key technical parameters and data inventory, and key modelling choices, and (ii) to synthesize main findings in terms of net performance and economic performance drivers. In the end, this review reflects on the key methodological shortcomings and provides a recommendation to improve the usefulness and validity of future economic assessments to support further LFM development and implementation.

2. METHODOLOGY

2.1 Search and selection of studies

The identification of studies dealing with the economic assessment of LFM was performed through a literature search using multidisciplinary science databases such as Scopus (1960-present) and Web of Science (1975-present) with restriction in publication date until 2017. To be able to account for all possible synonymous terms, the following search strings were used (i) for economic assessment: (economic* OR financial OR cost* OR benefit* OR expense*) AND (assessment OR analysis OR feasibility OR evaluation OR impact*); and (ii) for LFM: "landfill mining". It should be noted that this search may not be exhaustive of all LFM studies as there is also proprietary grey literature by private companies and consultancy firms. In addition, it was assumed that possible search terms such as landfill reclamation and landfill rehabilitation meant LFM without special emphasis on resource recovery, which was beyond the scope of this review.

The selection procedure had a particular focus on studies available as full papers (e.g. journal articles, conference proceedings, technical reports) with quantitative economic assessments covering the entire LFM process chain. It was done to acknowledge that LFM is composed of an array of processes and technologies and to allow for a balanced evaluation of the main findings and employed methods among the studies. In summary, a two-step studies search and selection procedure (Pinior et al., 2017) was used as illustrated in Figure 1.

The first step involved the search for studies in the databases (primary search) while the second step retrieved studies from the reference lists of the previously selected studies (secondary search). Duplicate studies from the two databases were excluded. To narrow down the identified studies from databases and reference lists, preliminary

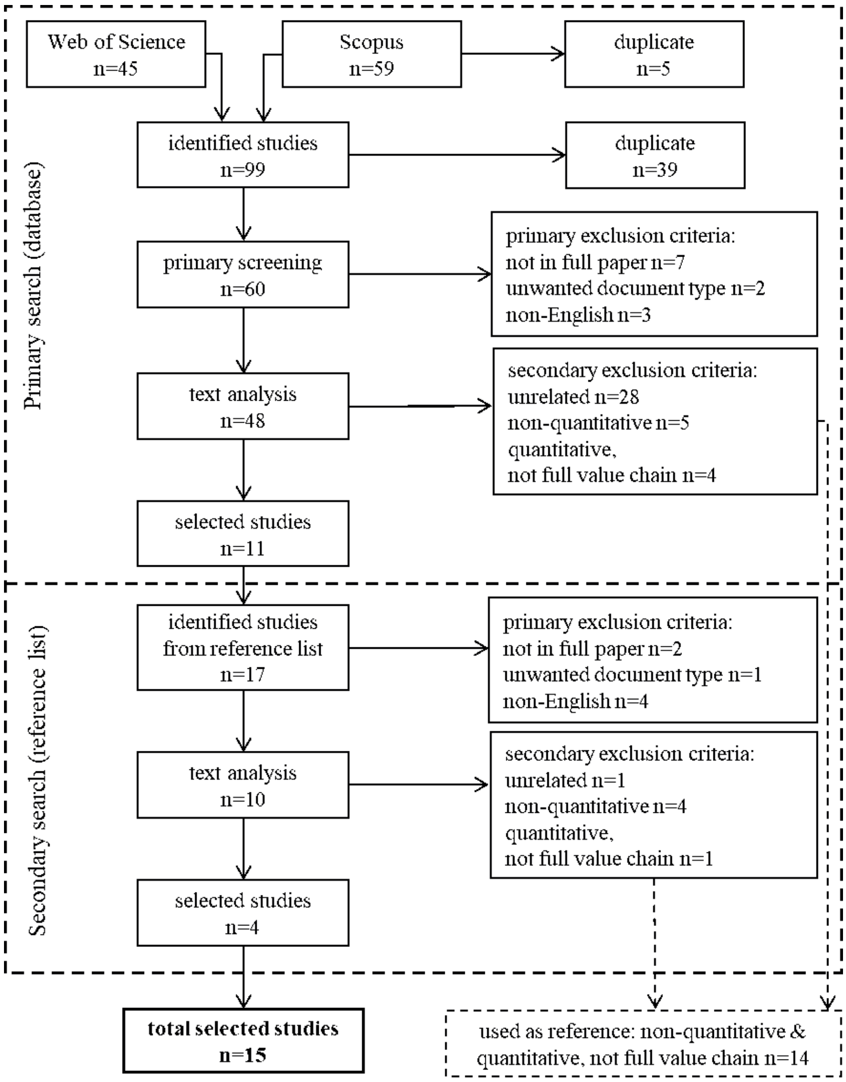

FIGURE 1: Flow diagram describing the literature search and selection procedure and the corresponding number of studies (n) for the review of economic assessment of landfill mining.

exclusion was done based on the content of title and abstract with the following criteria: (i) unavailability in full paper such as conference abstracts, (ii) unwanted document type such as science magazines and conference reviews, and (iii) written in non-English language. Subsequently, secondary exclusion was done based on the content of the full paper with the following criteria: (iv) being unrelated, the paper was not about LFM or did not include any economic assessment at all; (v) being non-quantitative, these papers were typically about economic frameworks suggesting cost and benefit items; and (vi) being quantitative, the paper did not consider the entire LFM process chain. The studies in the latter two exclusion criteria (v-vi) were not completely excluded but were instead used for further exemplification and elaboration in the discussion part.

2.2 Analytical review approach

The overall analytical review method was divided into three main parts (Figure 2). The first two parts focused on the critical review of (i) methods and (ii) main findings. These parts aided to contextualize the provided results, thereby offering a better understanding of the study objectives and limitations prior to the subsequent (iii) assessment of usefulness and validity of their provided results. Categories for usefulness were introduced here in terms of the four types of questions that the studies can address. These questions were based on two dimensions, that is the type of analysis (case study-specific or generic) and the type of application (decision-oriented or learning-oriented) that the studies intend to fulfill. In the end, these categories were also used to discuss what type of usefulness is common in the selected studies, as well as what type is relevant to support further LFM development and implementation.

2.2.1 Methods review

The methods review was based on several analytical criteria (Table 1) to determine the specific goals and corresponding methodological rigor of the studies. These criteria were selected and modified based on the main steps of the code of practice in life cycle costing (LCC by Swarr et

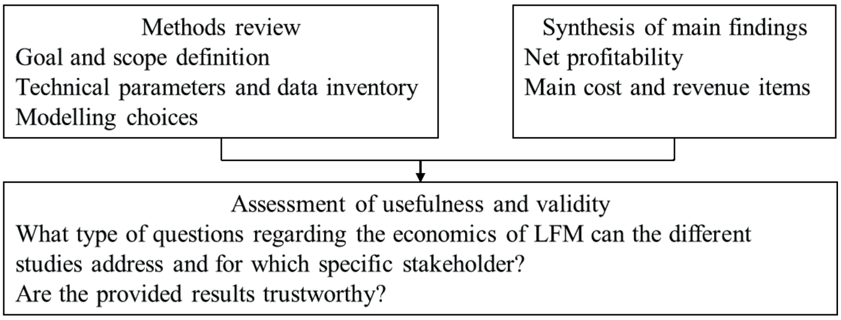

FIGURE 2: The overall analytical review approach.

al., 2011) and related reviews on systems analysis of waste management systems (Astrup et al., 2015; Laurent et al., 2014a, 2014b; Martinez-Sanchez et al., 2015). The main steps are goal and scope definition, technical parameters and data inventory, and modelling choices. By going through these main steps, the inherent uncertainties in assessing the economic performance of LFM were highlighted.

Goal and scope definition were analyzed to determine the type of analysis (case study-specific or generic) and the type of application (decision-oriented or learning-oriented) that the individual studies intended to fulfill, which corresponds to the perceived usefulness. The type of application refers to whether the main objective of the study was to obtain accurate results on the economic feasibility of LFM (decision-oriented) or if the emphasis was rather on assessing what factors build up such performance (learning-oriented), while the type of analysis instead refers to the explored settings for LFM and thus which scenarios were assessed in the studies. According to Laner et al. (2016), LFM could be realized in a wide range of different settings, and these variations can be classified at different levels such as site level (e.g. waste composition, landfill size, etc.), project level (e.g. technological and organizational set-up for separation, thermal treatment, and/or further residue valorization), and system level (e.g. surrounding policy and market conditions). Here, these different levels were used to categorize which scenario variations have been explored in the economic assessments of LFM, both for case specific and more generic studies. For a more comprehensive description of the assessed LFM scenarios, the corresponding geographical, technological and temporal scopes were also classified as well as the applied economic perspective (i.e. conventional LCC, environmental LCC and social LCC according to Swarr et al., 2011).

To investigate the technical specificity and corresponding data quality, descriptions of employed technical parameters and data inventories were analyzed for each LFM value chain process (i.e. separation, thermal treatment, and residue management), also including landfill settings and waste composition. The respective data sources were noted in terms of whether the studies used primary data, secondary data, mixed primary and secondary data, or were not specified at all. Specific modelling choices were analyzed in terms of the considered reference scenario (incumbent landfill management alternative instead of LFM), externalities (environmental and social), marketability and market prices of recovered resources, and economic indicator (direct or discounted cash flow analysis). In addition, the handling of uncertainties was also enumerated in terms of the type of employed uncertainty and sensitivity analyses. Parametric uncertainty analysis accounts for the uncertainties of input parameters (range of values instead of an absolute value per parameter), which gives additional information on the confidence level of the provided results. Sensitivity analysis, on the other hand, accounts for the robustness of results when input parameters are changed either one at a time as in local sensitivity analysis, or simultaneously as in global sensitivity analysis (Saltelli et al., 2008).

2.2.2 Synthesis of main findings

The main findings were assessed in terms of the reported net performance and main economic drivers. For net performance, apart from being net profitable or not, potentially profitable cases were noted if at least one of the considered scenarios generated a positive economic result. Main economic drivers referred to the cost and revenue items with the highest values. Cost items were noted as expenditures at each LFM process (i.e. excavation, separation, thermal treatment, transportation, and residue disposal), while revenue items were categorized into direct revenues from process outputs (e.g. material sales, energy sales and value of reclaimed land or landfill void space) and indirect revenues caused by avoided aftercare costs or governmental support (e.g. tax breaks or other policy instruments internalizing environmental externalities). Moreover, the criticalities related to the synthesis of main findings and the corresponding interpretation of subsequent results were also highlighted.

2.2.3 Assessment of usefulness and validity

From the goal and scope definition, the perceived usefulness of the reviewed studies was categorized based on the type of analysis (case study-specific or generic) and the type of application (decision-oriented or learning-oriented). Here, these two dimensions were taken further and used as an analytical framework for assessing both the usefulness and validity of the synthesized main findings. Usefulness was described through enumerating the type of questions the studies could answer, while validity was described through the specific methodological rigor focusing on the extent of scenario development and employed uncertainty

TABLE 1: Analysis criteria addressed in this review. The classification under each analysis criterion is listed and described (in *italics*) when deemed necessary. "Mixed" refers to either comparison or combination of preceding stated classification.

Analysis Criteria	Classification
Goal & scope definition	
Type of application	Decision-oriented, learning-oriented
Type of analysis	Case study-specific: single-subject assessment *(case study only)*, comparative assessment *(case study + scenarios of varying conditions at project and/or system level)* Generic: *(case study + scenarios of varying conditions at site, project and system levels)*
Geographical scope	Continent, country, region, multiple sites, single site
Technological scope	Separation: conventional, advanced, mixed *(depending on the number of secondary materials recovered)* Thermal treatment: incineration, plasma gasification, mixed, internal/external Residue management: re-landfill (internal/external), metal recovery, construction aggregates Reference scenario, *avoided costs if LFM is not performed*: do nothing, aftercare, aftercare with energy recovery
Temporal scope	Project duration *corresponds to total process capacity (Mg/yr)*
Economic perspective	Conventional LCC (C-LCC) *purely financial*, environmental LCC (E-LCC) *accounts environmental costs/savings*, social LCC (S-LCC) *accounts broader societal costs/benefits*
Technical parameters and data inventory	
Landfill settings and waste composition	Type: municipal solid waste (MSW), industrial waste (IW), mix MSW-IW, mixed Size: Small (<1 Mt), medium (1 to <10 Mt), large (>10 Mt) Composition: Material fraction, material fraction + chemical composition, not specified Data source: primary, secondary, mixed
Separation	*Separation efficiency* Data source: primary, secondary, mixed
Thermal treatment	*Energy efficiency* Data source: primary, secondary, mixed
Residue management	*Amount of secondary waste/intermediary materials produced* Data source: primary, secondary, mixed
Modelling choices	
Reference scenario	*Length of reference scenario implementation*
Externalities	*Valuation of cost/benefit items for E-LCC and S-LCC*
Marketability and market prices	Materials (ferrous metals, nonferrous metals, construction aggregates, RDF, valorized residues, etc.), energy (electricity, heat), land, landfill void space
Economic indicator	Direct cash flow, discounted cash flow *(accounts time-value of money i.e. lower value for future revenues and avoided costs)*
Uncertainty & sensitivity analysis	Parametric uncertainty analysis, local sensitivity analysis, global sensitivity analysis, mixed, none

and sensitivity analyses, apart from other possible general issues such as transparency in data inventories and modelling choices. This was done acknowledging the emerging character of LFM with inherent large uncertainties that must be handled.

Categories A (What is the economic outcome of a specific LFM project?) and B (How could the economic performance of a specific LFM project be improved?) cover a specific case in the perspective of landfill owner and/or project manager. While Categories C (What is the economic potential of large-scale implementation of LFM in a region?) and D (How could profitable LFM projects be developed through selection of sites, project set-ups, and policy and market conditions?) are much broader that cover wider regional scope and in the perspective of several stakeholders such as LFM contractors, investors, policymakers, and/or researchers. Methodologically, it follows that the scenario development for both Categories A and B are limited to variation at the project level (e.g. technological and organizational set-up for separation, thermal treatment, and/or further residue valorization). While for both Categories C and D, they also consider variation at the site level (e.g. waste composition, landfill size, etc.) and system level (e.g. surrounding policy and market conditions). Regarding the employed sensitivity analysis, local sensitivity analysis is proven to be inefficient in revealing the underlying interactions among the parameters, unlike the global sensitivity analysis (Ferretti et al., 2016; Saltelli and Annoni, 2010) that is particularly relevant for the learning-oriented type of application. Hence, local sensitivity analysis is at least expected for Categories A and C as they only intend to know the net performance, while global sensitivity analysis is expected for Categories B and D ("How" questions) as they are after the principles of performance. Lastly, to handle the inherent parametric uncertainties, parametric uncertainty analysis is expected for all Categories to properly account for the variation of data values and the extent of their effect to the spread of the study results.

This analytical framework helped in revealing the difference between the perceived usefulness (categorization of studies based on the stated aim) and the real usefulness with the corresponding validity of the provided results (assessment of methodological rigor based on the stated aim). For instance, issues on validity due to unsatisfactory methodological rigor directly led to problematic real usefulness. Proceeding discussion focused on the applications and limitations of the selected studies under each Category, and the corresponding recommendations for improvement in terms of economic assessment features and how it can methodologically be performed. Furthermore, synthesis discussion on how economic assessments can be used to facilitate the development of cost-efficient LFM projects and to guide future research prioritization for the LFM area as a whole were underscored.

3. RESULTS AND DISCUSSION

3.1 Goal and scope definition

In total, this review includes 15 studies (see Appendix A). Most of them involve specific LFM case studies with a decision-oriented type of economic analysis. The key objective of these studies is thus to assess the net performance of a specific LFM project, often by accounting for a limited number of scenario alternatives. Several studies (n=6), however, only assess the economic outcome of a single scenario for realizing a specific LFM project. The extent of assessed LFM processes and explored scenario alternatives in all of the reviewed studies are shown in Figure 3.

At the site level, variation in waste composition is seldom accounted for (n=1) in the explored scenarios, since most of the studies address a specific landfill. Different waste compositions are considered, such as municipal solid waste (MSW, n=6), industrial waste (IW, n=3) and mixed MSW and IW (MSW-IW, n=5). For the same reason, variation in system-level conditions (n=3) is also seldom investigated. The explored system-level variation is limited to inclusion/exclusion of re-landfilling tax, availability/unavailability of the market for recovered materials, and varying the required length of time of the reference scenario. The reference scenario is classified at both site and system level variation together with land resources. The former depends on both the regulatory requirements as well as the type of landfill waste composition (do nothing n=1, aftercare n=6, aftercare with energy recovery n=3), while the latter depends on both market conditions and the location of the landfill site (land recovery n=5, landfill void space recovery n=4, mixed n=4). Not all of the studies accounted for these two aspects, hence underestimating the economic performance of LFM by missing possible revenue items.

Variation at the project level is commonly investigated (n=8), and is done in different ways in individual studies. For the separation process, investigated technologies include a conventional separation process (n=5) that recovers metals and construction aggregates, or an advanced separation process (n=7) that additionally recovers combustibles (including refused derived fuel or RDF) and glass. A comparison between these separation technologies is also performed in the rest of the studies (n=3). For the thermal treatment process, investigated technologies include incineration (n=3), gasification (n=3), or the comparison between the two. Apart from these variations in technological set-up, variation in organizational set-up (n=3) is also investigated, which means the thermal treatment process is considered either internal or external to the LFM project. Similarly, variation in organizational set-up (n=5) is investigated for the disposal of separation residue, while the rest have individually considered either internal (n=6) or external (n=4) disposal. For thermal treatment residue, apart from the disposal in hazardous landfill (n=3), further valorization such as metal recovery from incineration bottom ash (n=1) and construction aggregates (n=2) from plasma gasification slag is also considered. It is notable that fewer studies investigated more downstream processes starting from thermal treatment, which is reflective of the emerging character of LFM.

Variation in scope is also observed geographically, technologically, and temporally. Regarding the geographical scope, a single landfill site (n=9) is typically considered, while some also covered a wider scope in terms of national (n=3) such as Scotland, Sweden, and Greece; regional (n=2) such as Flanders in Belgium and Styria in Austria; and also continental (n=1) such as entire Europe. It is notable that most of the selected studies come from Europe (n=12), and only a few come from Asia (n=2) and North America (n=1). The countries where these case studies are located are categorized as nations with high-income economies (World Bank, 2016), with more stringent standards expected for landfill management. This situates LFM to having a promising business case due to a favorable market (i.e. higher material prices) and aftercare obligations (i.e. higher avoided costs). Regarding the temporal scope, most studies considered the specific LFM project duration. As different case studies are considered, project duration depends on landfill settings, processing capacity, and length of landfill aftercare. Regarding the economic perspective, most of the studies were assessed based on conventional LCC (n=8), while the rest were based on environmental LCC (n=5) including green energy certificates and carbon emission trading, and social LCC (n=2) including health risk reduction and employment. This highlights that most of the studies were intended for LFM practitioners with a private economic view. Although a broader sustainability consideration has been suggested, this also implies additional complexity in terms of the valuation of external cost and revenue items (Burlakovs et al., 2017).

3.2 Key technical parameters and data inventory

3.2.1 Landfill settings and waste composition

In terms of landfill settings, studies consider varying landfill sizes, including small (<1 Mt, n=6), medium (1-10 Mt, n=5) and large (>10Mt, n=4) sites. The case studies are typically described in terms of the mass of landfill waste and seldom in terms of more specific information such as area, depth, and density. Without such information, the effect of excavation and internal transport logistics to the economic performance of LFM may be overlooked (Hogland et al., 2018; Hölzle, 2019). In terms of waste composition, it is typically presented by material fractions (e.g. metals, paper, wood, aggregates, etc.) and seldom by chemical composition. Consequently, it is difficult to qualify the material outputs as to whether they satisfy standard material quality requirements for the proceeding processes, may it be thermal treatment, material sales or even disposal.

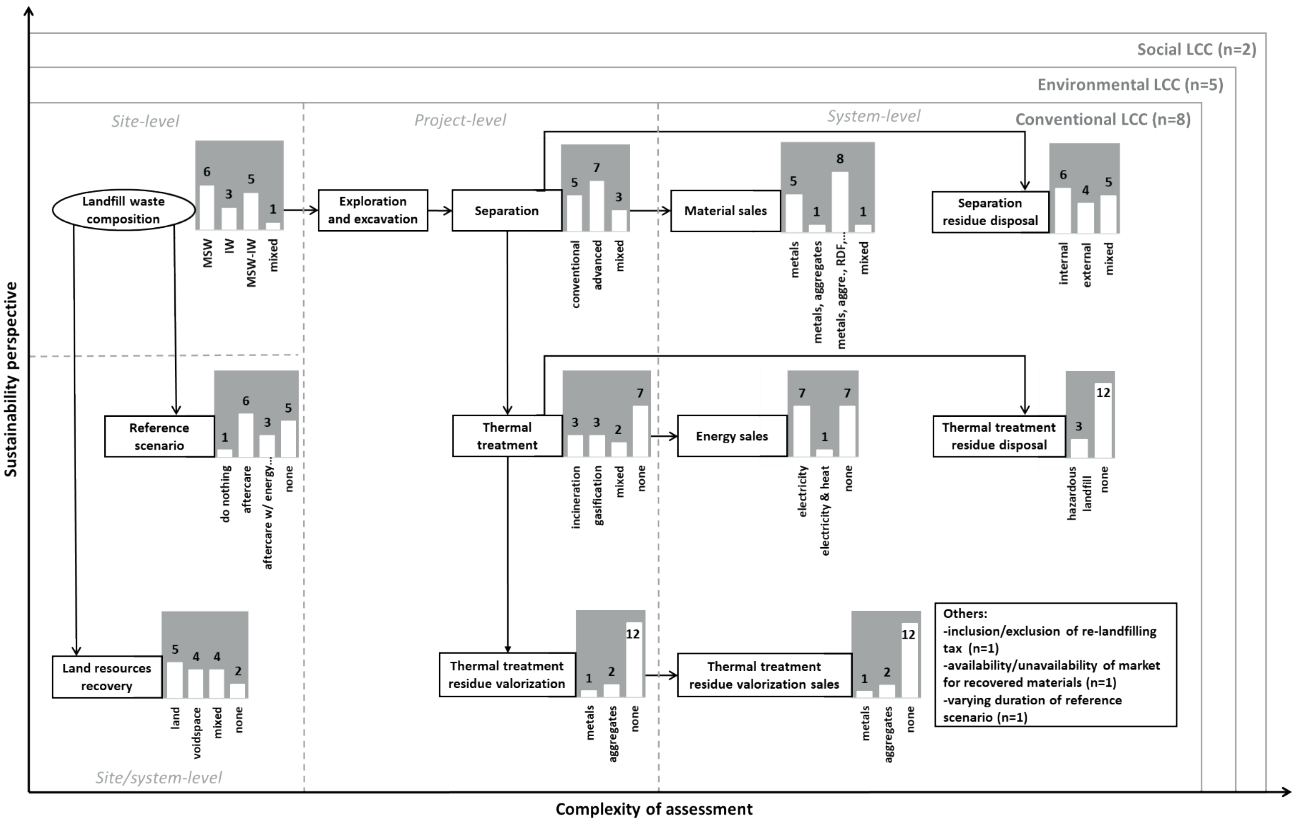

FIGURE 3: LFM processes included in the selected studies (n=15), with their respective variations categorized at different levels such as site, project and system (in dashed lines). The complexity of economic assessment is expected with the extended technological scope and broader sustainability perspective. “Mixed” refers to either comparison or combination of preceding stated classifications, while “none” refers to studies which excluded, or implicitly included, such processes.

Regarding data sources, most of the studies use primary sources (n=9), and the rest use secondary (n=4) and mixed sources (n=2). Only a few of the primary sources are based on full-scale excavation (n=2), and the rest (n=7) are based on logbooks, preliminary sampling campaigns, and pilot-scale excavation. The apparent use of primary sources corresponds to case-specific studies, while the use of secondary and mixed sources corresponds to either hypothetical case studies or studies with a wider geographical scope. The average of waste compositions from different landfills is used to represent continental (Van Vossen and Prent, 2011), national (Ford et al., 2013; Frändegård et al., 2015), and regional levels (Damigos et al., 2016; Danthurebandara et al., 2015b; Van Passel et al., 2013). Clearly, there is a large uncertainty to be accounted for, both within and among reported waste compositions (Hernandez Parrodi et al., 2018; Hogland et al., 2018; Hölzle, 2019).

3.2.2 Separation

Specifics of the separation process are typically presented through process flow diagrams. However, the corresponding separation efficiencies and the underlying machine specifics are seldom stated. Separation efficiencies are from 40% to 100% of the total waste composition, with most studies adopting the higher end. In addition, most of the primary sources are based on laboratory-scale separation. For hypothetical cases, secondary sources are often not closely related to the studied case but rather are obtained from industry estimates for fresh MSW processing. Similarly, secondary costs data are directly adopted from different geographical and temporal contexts. To assure representativeness to the case study of interest, these data have to be harmonized. Temporal cost harmonization can be done through a financial approach to remove the effect of inflation using indicators such as a gross domestic product deflator and consumer price index, while geographical cost harmonization can be achieved through purchasing power parity (World Bank, 2014).

3.2.3 Thermal treatment

Irrespective of the type of thermal treatment technology, energy efficiencies are reported from 25% to 30%, accounting for optimum performance. For this process, the considered RDF quality requirement in terms of input heating value is from 16 to 20 MJ/kg, which corresponds to high-quality input materials (Bosmans et al., 2013). Both of these specifics, however, are often based on secondary sources, either from existing pilot plants for plasma gasification or large-scale plants for incineration. Such plants use other process input materials such as fresh municipal solid waste that is not representative of landfill waste. For the secondary cost data, as previously stated, temporal and geographical cost harmonization is not performed.

3.2.4 Residue management

The amount of residue from the separation process is not clearly stated, despite the fact that about 40% to 80% of the total excavated waste ends up as residues (Hernandez Parrodi et al., 2018). A similar issue on material flow transparency is observed for the valorization of residue from thermal treatment. Specifically, separation efficiency and market quality requirements are seldom mentioned for the metal recovery and construction aggregates production from incineration and plasma gasification processes, respectively. In addition, information about the hazardous waste fraction is seldom noted that could significantly affect the total re-landfilling costs. Hazardous waste is significantly more expensive (100 to 200 Euro/ton) than its non-hazardous counterpart (3-100 Euro/ton) (Confederation of European Waste-to-Energy Plants, 2017).

3.3 Key modelling choices

3.3.1 Reference scenario

For the potential avoided costs, a reference scenario is stated acknowledging that there is an incumbent landfill management alternative instead of LFM. However, a significant number of studies (n=5) do not mention any reference scenario. However, for the ones that are mentioned, specific technical requirements and costs of aftercare vary widely depending on national or regional regulatory requirements. For example, landfill cover is commonly required but not in the Netherlands (Van Vossen and Prent, 2011) and Denmark (Rosendal, 2015), or none is required at all (do nothing) as in Sri Lanka (Danthurebandara et al., 2015b). Moreover, the model for leachate production and landfill gas emission is seldom specified, which directly affects the amount of emissions and consequent treatment costs. Also, the length of the aftercare period varies from 25 to 100 years, with 30 years as the most commonly used. This uncertainty is primarily due to the vague description in Article 12d of the EU Landfill Directive (1999/31/EC), which states that aftercare duration halts when "the competent authorities consider the landfill likely to cause a hazard to the environment".

3.3.2 Externalities

Some studies (n=5) internalize environmental benefits, which are limited to avoided climate impact in terms of carbon dioxide equivalent (CO_2 eq.). Different databases are used to quantify process-related environmental emissions (subsequently converted to CO_2 eq.), such as Bilan Carbone™ (Association Bilan Carbone, 2007), PROBAS (German Federal Environmental Agency, 2013), and EcoInvent v2.2 (Ecoinvent, 2010). According to the ILCD Handbook (European Commission-Joint Research Center, 2010), the selection of database must be based on completeness, representativeness and up-to-date datasets, however none of the studies justified such choices. Regarding the monetary valuation, CO_2 eq. savings are valuated differently showing wide variation in prices such as the hypothetical carbon tax (10 Euro/ton, Winterstetter et al., 2015), the social cost of carbon (20 Euro/ton, Tol, 2008), and the EU Emission Trading Scheme (40 Euro/ton, EU, 2007). Even wider variation is notable for the prices of incentives for renewable energy production such as the green certificate (108-117 Euro/MWh) in Belgium (Danthurebandara et al., 2015c; Van Passel et al., 2013) and the renewable obligation certificate (5-42 Euro/MWh) in Scotland (Ford et al., 2013).

3.3.3 Marketability and market prices

Most of the studies (n=13) assume the marketability of materials that they plan to recover and valorize. However, specific market quality requirements are seldom mentioned, and that all recovered and valorized materials are assumed to be saleable. There are also some studies that account for marketability and market price though preliminary discussions with potential buyers. Examples include plantation owners for the soil residues as fertilizers (Zhou et al., 2015) and construction companies for plasma gasification residue as construction aggregates (Danthurebandara et al., 2015c; Van Passel et al., 2013). However, there are also studies that have contradictory assumptions. For example, instead of the production of construction aggregates, Winterstetter et al. (2015) considered the re-landfilling of plasma gasification residue, arguing that such a valorization process has not gone beyond laboratory tests. Moreover, none of the studies considers the broader market dynamics of supply and demand upon the introduction of exhumed materials to the market competing with primary sourced materials and more high-quality secondary resources obtained from e.g. source separation programs.

3.3.4 Economic indicator

Studies perform either direct cash flow (n=7) or discounted cash flow (n=8) analysis. For the former, it follows that the studies consider small landfill size with high LFM processing capacity, leading to a project duration of about a year. For the latter, project duration is much longer, from 3 to 20 years, in which the time value of money has to be considered (Brealy et al., 2011). The discount rate varies from 3% to 15%, depending on if public or private financing is considered, respectively. In essence, the project duration and type of financing constitute a downplaying of the value of future revenues and avoided costs, in comparison to the initial investments accounting for higher risks.

3.3.5 Uncertainty/sensitivity analysis

From the previous sections, several possible variations are discussed along the LFM value chain processes (i.e. separation, thermal treatment and residue management), as well as in other aspects such as waste composition, externalities and some general assumptions (Table 2). These correspond to the uncertainties that occur in scenario building (scenario uncertainties) and data gathering (parameter uncertainties), which have to be properly addressed for all systems analyses, in general (Clavreul et al., 2012; Huijbregts et al., 2003).

Despite the abovementioned uncertainties, more than half of the studies (n=8) have not performed any parametric uncertainty or sensitivity analyses, of which the majority (n=6) have not considered even any scenario alternatives but instead just a single scenario for a specific LFM project. For the rest of the studies, sensitivity analysis is

TABLE 2: Overview of uncertainties in the economic assessment of landfill mining.

	General	Waste composition	Separation	Thermal treatment	Further valorization/ residue management	Externalities
Scenario uncertainties	• Inclusion/ exclusion of reference scenario	• Type of landfill waste inclusion/ exclusion of hazardous waste	• Technology choice (conventional to advanced technology) • Internal or external organizational arrangement • Marketability of secondary materials and energy (substitution: full, partial, no market)			• Inclusion/ exclusion of environmental and social costs and benefits/ revenues
Parameter uncertainties	• Origin of costs/ price data (where and when) • Amount of leachate and landfill gas • Discount rate	• Amount in terms of waste fraction or chemical composition	• Separation efficiencies • Material market prices	• Energy recovery efficiencies • Energy market prices	• Material market prices	• Values of environmental and social costs and benefits/revenues

more commonly performed, that is, either alone (n=2) or in combination with uncertainty analysis (n=5). About the same share of studies have performed either global sensitivity analysis (n=4) and local sensitivity analysis (n=3). This uncommon practice of performing uncertainty and sensitivity analyses indicates that the majority of the studies lack information on the robustness of their provided results, hence posing questionable usefulness and validity of fulfilling their intended objectives. Poor uncertainty management may lead to faulty decision support with missing risks information that is related to the net performance of LFM, as well as misunderstanding the principles of performance of LFM with the lack of systematic identification of its main economic drivers.

3.4 Synthesis of main findings and related criticalities

Despite the unique conditions and considerations of individual studies, it can be generalized that LFM is a challenging business venture based on the reported net economic performances. Only a few of the studies are profitable (n=2), while the rest are either not profitable (n=7) or potentially profitable (n=6). The net economic outcome of the studies ranges from a net deficit of -€112 to a net profit of +€67. Appendix A gives individual study results that are temporally (GDP deflator) and geographically (purchasing power parity) harmonized.

A main reason for reviewing the findings of previous case studies is to identify reoccurring conditions and settings of importance for the feasibility of a project, thereby contributing to the common knowledge building of a concept or strategy. However, when it comes to LFM, such a synthesis is difficult, due to a general lack of transparency regarding case-specific conditions, and different procedures for the way in which the projects and their different processes have been aggregated and modelled (Sections 3.2-3.3). In most of the case studies, for instance, the LFM value chain is depicted and modelled only in terms of main processes (e.g. excavation and sorting, waste-to-material and waste-to-energy), while the contributions from underlying factors in terms of the numerous parameters that build up each of these processes remain unknown, or at least not systematically accounted for. In addition, many of these processes and parameters are highly connected throughout the LFM process chain, and such interactions or combinational effects often have a significant impact on the economic performance of a project. This treatment of the process chain as a series of black boxes makes it difficult to develop any deeper understanding of what builds up the economy in the different LFM projects, and limits the identification of reoccurring performance drivers to some highly aggregated cost and revenue items, Figure 4.

For several methodological reasons discussed in previous sections, even the interpretation of this type of highly aggregated and superficial information about commonly reported LFM performance drivers should be done with caution. This is because each LFM project is uniquely designed in terms of its site, project set-up and system level conditions and, without a clear record of such settings (as in many of the reviewed case studies), conclusions about the general significance of a certain performance driver might be misleading. For instance, the different case studies involve different approaches to the treatment of combustibles exhumed from the landfills, and this has an overarching impact on the economy of the projects. In cases where such fuel is sent to external waste-to-energy plants, process-related (e.g. excavation and separation) and material flow-related (e.g. transportation and disposal) cash flows are often reported as main cost items. Although such costs remain important in projects involving internal thermal treatment of the extracted combustibles, capital investments and operational expenditures related to the (new) waste-to-energy plant then typically dominate the cost profile. In addition, revenues from energy sales only become applicable for such project set-ups in which the combustibles are thermally treated internally.

Also, when it comes to the reported performance drivers in terms of revenues, drawing conclusions of general relevance for the LFM area is somewhat difficult. For instance, virtually all of the case studies report revenues from recovered materials (which are almost exclusively metals) as an important revenue, while indirect benefits of a LFM project in terms of the value of reclaimed land and landfill void space, or avoided landfill aftercare costs, are less frequently identified as main drivers. However, this does not mean that such indirect benefits are not important for the economic outcome of a LFM project, but rather that the case studies often have involved landfills with no or low aftercare costs situated in locations with relatively low land values and needs for new landfill void space. This

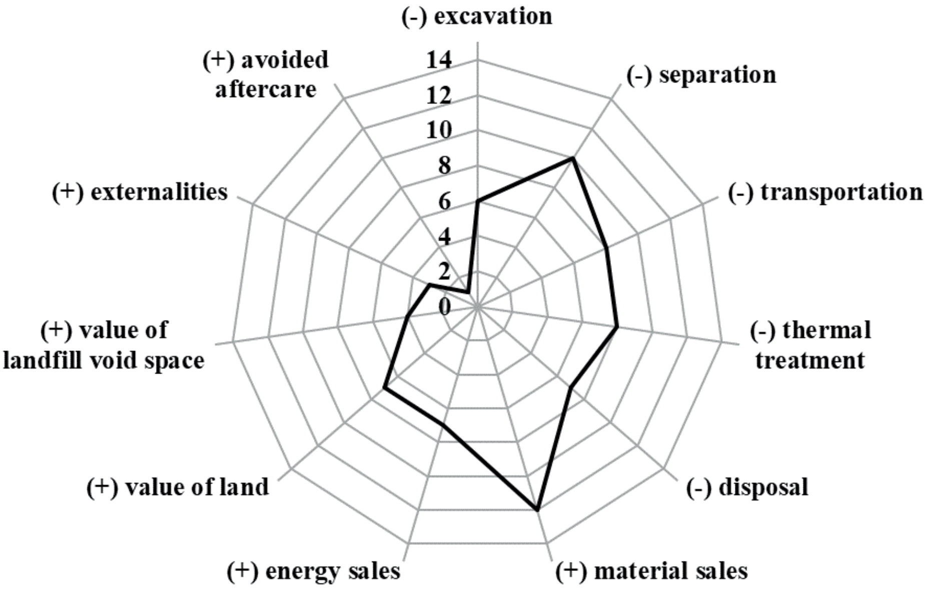

FIGURE 4: Reported economic performance drivers of LFM in the reviewed studies (n=15) in terms of the top three main cost (-) and revenue (+) items. However, due to the differences in the LFM cases and the economic assessment methods used, interpretation should be done with caution. See text for further explanation.

inability to address the importance of sitespecific conditions (e.g. material composition and aftercare needs) and other local settings (needs and values for land and landfill void space) for the economy of LFM projects is an inherent characteristic of the reviewed studies due to their focus on assessing only one case.

3.5 Usefulness and validity of selected studies and recommendations for future assessments

One main reason that most of the reviewed studies (n=9) only provide superficial knowledge on what builds up the economic performance of LFM is that they are decision-oriented, as shown in Figure 5. Thereby, they primarily aim to forecast the net outcome of conducting LFM in a certain landfill site (n=6, case study-specific, Category A) or within a wider geographical scope (n=3, generic, Category C). Both of these analyses aim to produce knowledge that is, indeed, essential for supporting investment decisions on both the project and regional levels (Finnveden and Moberg, 2005; Swarr et al., 2011). However, for emerging concepts such as LFM with a lack of real-life projects and records of accomplishment, the validity of the results obtained from such feasibility assessments can be questioned. For instance, current knowledge deficits about the different processes of the LFM value chain are typically addressed using secondary data from the sorting and recovery of other waste (e.g. fresh MSW) or experiences from small-scale (laboratory) tests (Section 3.2). Not only is the applicability of such data to the large-scale processing of landfilled waste unknown, but also most studies assume that the extracted materials and energy resources will be marketable (Section 3.3.5). Although such inherent knowledge deficits are inevitable for any emerging concept (Clavreul et al., 2012; Hellweg and Milà i Canals, 2014; Martinez-Sanchez et al., 2015), a major concern here is that most of the studies leave them unaccounted for (Section 3.3.5), and hence, their effect on the robustness of the results is unknown. Consequently, landfill owners, project managers, LFM contractors, investors and policy-makers are prone to making decisions based on results with large implicit, or even neglected, information on the economic risks. For instance, if the generally employed assumption that the extracted materials and energy sources will be readily accepted on existing markets is not true (Johansson et al., 2017), this will have significant implications for the economic feasibility of any LFM initiative.

In essence, we are not yet in a position to make this type of profitability claim regarding LFM, not on the project level (Category A) and certainly not on the regional scale (Category C). Before such assessments can be made with any trustworthiness, extensive and applied research is needed to address key issues such as what resources can be extracted from landfills, at what quality levels, and under what conditions they will be accepted on existing markets (Krook et al., 2019). In order to develop such knowledge, there is no alternative than to go from the often-seen laboratory studies to well-planned pilot studies in which the efficiency, capacities and performance of different separation, upcycling and recovery technologies are developed and monitored on a scale comparable to real-life projects. If any stakeholder wants even so to forecast the economic outcome of a specific project or estimate the economic potential of implementing LFM in a region, it is strongly recommended that this be done by employing existing scenario and parameter uncertainty analysis methods. As demonstrated by some studies of project assessments (Danthurebandara et al., 2015c; Frändegård et al., 2015; Van Passel et al., 2013; Winterstetter et al., 2015), such an analytical approach makes it possible to provide more fair feasibility claims. Instead of providing a single (but highly

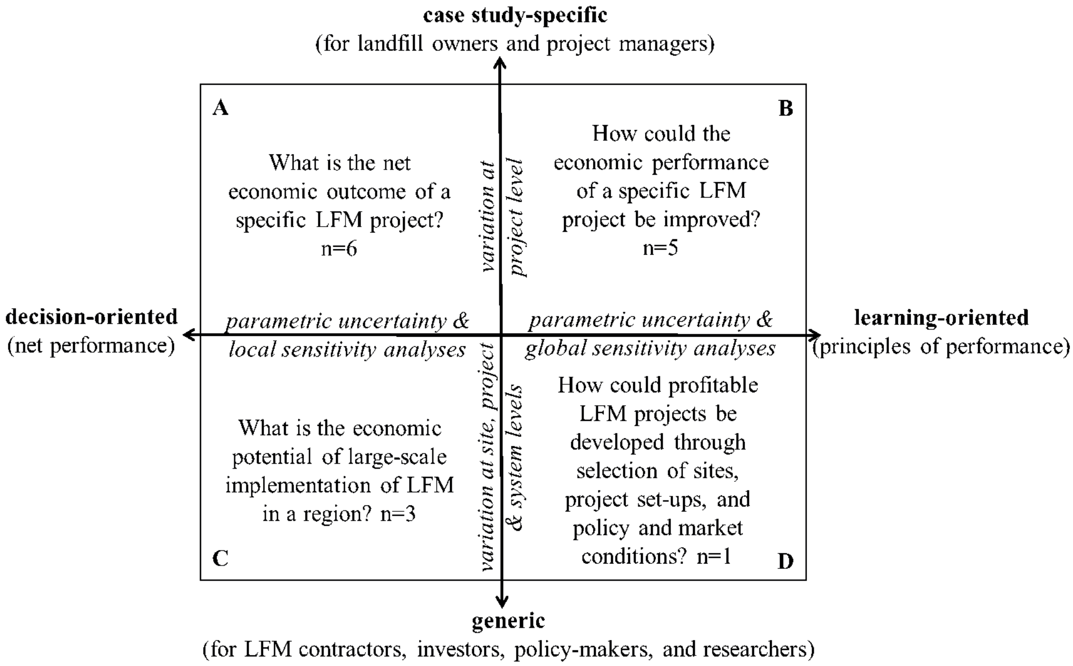

FIGURE 5: The categorization of the selected studies (n=15) in terms of their perceived usefulness. However, given the validity concerns of the results obtained, their usefulness is only partially, if not at all, fulfilled. See text for further explanation.

uncertain) value, it derives a wide range of plausible outcomes in which the implications of current knowledge deficits are explicit.

In order to guide LFM research and knowledge development towards key challenges and potential solutions for cost-efficiency, learning-oriented studies are necessary (Fleischer et al., 2005; Krook et al., 2019; Wender et al., 2014). Several features make such studies distinctively different from decisionoriented studies. To start with, learning-oriented assessments go beyond the intention to obtain an accurate estimate of the net economic outcome of a certain case, and rather aim to provide strategic guidance on how the economic performance can be improved, and to determine what type of knowledge is essential for developing such a project. This change in perspective has some major implications for how to design and execute economic assessments of LFM. In order to account for current empirical constraints and knowledge gaps, an explorative approach is needed (Voinov et al., 2016; Wender et al., 2014), in which multiple possibilities and scenarios are simultaneously assessed to scope in implications of different site-specific settings, choices of processing lines and technologies, and policy and market conditions. Another key characteristic of learning-oriented assessments is that the collection of data for different processes and parameters aims to cover the range of possible variation, both in terms of stochastic and epistemological uncertainties, rather than to obtain, as in many decision-oriented studies, a single (but highly uncertain) value. To handle such wide variations on both the scenario and parameter levels, the employment of systematic uncertainty and sensitivity analyses methods is key (Ferretti et al., 2016; Saltelli and Annoni, 2010). Not only do such methods make it possible to explicitly account for the uncertainties in the results, they also enable fine-grained assessments of the processes, parameters and interactions among them that jointly build up the net economic outcome of LFM.

Several of the reviewed studies can be categorized as learning-oriented in the sense that their main objective is to discover what builds up the net economic performance of conducting LFM in a specific landfill site (n=5, case study-specific, Category B), or within a wider geographical scope (n=1, generic, Category D). However, one major limitation of these studies is that they typically only involve a few scenarios in which some of the conditions and settings at the project set-up level are explored. For instance, technical options are limited to one set of separation and thermal treatment processes. Further, project organization is only considered as a certain process, which is either internal or external to the project (Section 3.1). In order to better scope in key challenges and potential solutions, it is necessary to consider a wider variety of options to technically and organizationally set up LFM projects. Moreover, possible variations and choices related to the landfill site and surrounding system levels are seldom explored. On the generic or regional level, the assessment of such variations is a necessity to identify which landfills are suitable for mining and how different policy and market environments influence the economics of such projects. However, even in case-specific assessments, an openness to different alternatives and conditions on these levels is useful, given the often early stage of development and thus huge knowledge deficits regarding such matters as the landfill composition, and the implications of various policies and market conditions. Such exploration of multiple scenarios can be done through the integration of existing knowledge from previous case studies and through participatory scenario development, in which a wider array of possibilities is co-created with different experts belonging to the different parts and processes of the LFM value chain (Voinov et al., 2016; Wender et al., 2014).

When handling uncertainties, several of the learning-oriented studies performed parametric uncertainty analysis. However, toonarrow ranges of variation are typically used that mainly cover natural or stochastic variations in the capacity and efficiency of conventional processes, while knowledge-related or epistemic uncertainties in the processing and recovery of previously landfilled waste are seldom addressed. For instance, the stochastic uncertainty related to the separation efficiency of processing fresh MSW is commonly accounted for in such case studies, but not the presumably much larger epistemic uncertainty related to the expected differences in process performance when the input is excavated LFM waste. A direct consequence of this in practice is that there is a risk that the importance of different processes and parameters for the economic performance of LFM will be underestimated. In addition, the quality of recovered materials (whether they have reached market quality standards) and their corresponding marketability (whether there is a market demand for such materials recovered from landfills) also entail huge epistemic uncertainties that are virtually never addressed. Such factors may nevertheless, have significant implications for the economic performance of LFM. Stochastic and epistemic uncertainties are typically addressed by collecting ranges of values and developing probability and possibility distributions together with the respective experts (Clavreul et al., 2013; Lacirignola et al., 2017).

When it comes to the analysis of critical conditions and factors for economic performance, most of the studies use local sensitivity analysis or one-at-a-time sensitivity analysis, in which parameters changes are accounted for individually, rather than simultaneously. Such a method, however, is unsystematic in revealing the important economic drivers, primarily because of its inability to address the interrelations among different processes and parameters (Ferretti et al., 2016; Saltelli and Annoni, 2010). The use of this sensitivity analysis method often leads to the processes and parameters downstream in the LFM value chain being identified as the most critical for the economic outcome, while the fact that their importance is rather a consequence of the realization and interactions with upstream parameters is missed. To take a very simple example, several of the studies conclude that (increasing) raw material prices are one of the most important drivers for increasing the material revenues from LFM, and the studies emphasize the need for exogenous changes or specific policy instruments to stimulate such development. However, the net material revenue that can be obtained is determined rather by the specific interrelations between the content of different materials in the landfill, the costs and efficiency of extracting them into well-defined and marketable material categories, and (to a significantly lesser extent) plausible variations in raw material prices. Uncovering such interrelations thus leads in a totally different direction, in which potential measures to improve the economic performance involve the selection of suitable and more high-grade landfills for mining, and the development of tailored processing and sorting lines, rather than calling for policy and market interventions that influence raw material prices. One way to systematically reveal such interrelations is by performing both first-order and higher-order variance-based global sensitivity analyses, in which both the direct and combinational economic effects of various conditions, settings and parameters are simultaneously assessed (Saltelli et al., 2010). In practice, extensive data collection must be carried out in order to achieve multiple scenario development. In addition, such analyses are mathematically demanding in terms of modelling design and execution. In the field of LFM, Laner et al. (2016) performed a learning-oriented study of the climate impact assessment of LFM in Europe. It employed multiple scenario development that accounted for variations occurring on the site, project and system levels, together with a global sensitivity analysis, which may be one of the bases for future economic assessment of LFM. In general, learning-oriented studies are expected to provide knowledge that can aid in developing a systematic overview of how different conditions, settings and parameters, as well as their interrelations, contribute to the net outcome. Consequently, the focus should be directed to where more learning is demanded. This will reveal what is potentially important and what is not, thereby facilitating priority-setting in terms of where investment into research and knowledge development should be directed.

LFM is an investment-intensive undertaking and strategic guidance for future projects is necessary. One fundamental question is which landfill site to prioritize to exemplify economically favorable projects. In this regard, generic and learning-oriented studies (Category D) can be used to determine strategic locations for future pilot-scale and (eventually) large-scale project implementations. To direct individual LFM projects in terms of technical and project organizational set-up, case-study specific and learning-oriented studies (Category B) can be used, showing complementarity of approaches. In this way, more practical knowledge and primary sourced data will become available, strengthening in this way the results of generic studies. With widely accepted conclusions that reveal the true economic potential of LFM, further development of favorable policy and market environments can be advised for more cost-efficient LFM projects in the future.

4. CONCLUSIONS

A total of 15 studies have been examined in this review, which quantitatively assessed the economic potential of LFM. The majority of the studies are case study-specific, with a decision-oriented type of application. This accounted for individual cases with quite varied LFM project descriptions, and considered scenarios classified at the site, project and system levels. Apparent scenario and parameter uncertainties were highlighted and acknowledged to be inherent to the emerging character of LFM, with the inevitable use of secondary data sources, or primary sources that are based on laboratory to pilot-scale tests. In this regard, transparent descriptions of goal and scope, data inventory and estimations, and model assumptions are called for. These are typical recommendations as stated in existing method guidelines (ISO, 2006a, 2006b; Swarr et al., 2011), but they remain unaddressed in the following LFM studies, and in most of the current systems analysis carried out in the field of waste management (Astrup et al., 2015; Laurent et al., 2014a, 2014b; Martinez-Sanchez et al., 2015). Apart from transparency, subsequent management of these uncertainties must be addressed to ensure the usefulness and validity of study results.

Moreover, this review highlights that economic assessments, obtained through a learning-oriented approach, can be used not just to obtain the net performance, but also to understand the principles of performance and improve current knowledge levels for future LFM project implementation. In dealing with LFM with large knowledge deficits, it is highly recommended that more extensive and applied research must be carried out. Such LFM initiatives can be guided by learning-oriented approach in terms of site selection (Which landfill site is suitable for mining?), project implementation (Which technological set up and project organizational set up are preferable?), and system setting (Which policy and market conditions are favorable?) towards the development of cost-efficient LFM projects. In this case, explorative scenario development can be used by accounting multiple variations at site, project, and system levels. Subsequently, the related uncertainties and their respective importance can be accounted for by performing parameter uncertainty analysis and global sensitivity analysis. Furthermore, with the broader scope of assessment and granular analysis of parameter importance, the overarching key potentials and challenges of LFM can systematically be identified. Hence, future LFM research prioritization can be guided. For instance, according to their relative importance, specific parameter improvements can be focused on, individually or in combination, such as better separation efficiency or energy conversion efficiency at the project level, or more favorable market standards and prices as well as lower taxes and gate fees at the system level. In consideration of LFM as an investment-intensive undertaking, such strategic guidance through a learning-oriented economic assessment can be beneficial in harnessing its economic potential even at an emerging phase of development.

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APPENDIX A

Overview of the empirical findings from the selected studies (n=15). The net result is presented as stated in each studies as well as the harmonized values for better cross-country comparison (literature value; harmonized value). Harmonization of values is done using GDP deflator (temporal harmonization) and purchasing power parity (geographical harmonization) conversion factors for 2018. For some studies (n=2), actual values are not explicitly stated that may be intended due to proprietary reasons, but at least, net profitability (positive or negative) can be inferred.

Selected studies (n=15)	Landfill characteristics waste type (W), size (S), and location (L)	Scale of excavation	Net result, €/ton of excavated waste	Economic performance drivers	
				Main costs	Main benefits/ revenues
Zanetti & Godio, 2006	W: monolandfill (foundry) S: medium (85 000 m^2) L: Crescentino landfill, Italy	pilot	not profitable (-3; -5)	treatment & re-landfilling (fines), fuel, amortization, transportation, labor	material sales (silica sands & iron powder), value of landfill void space (mentioned but not valuated)
Van Vossen & Prent, 2011	W: mix MSW-IW S: small (0.5 Mt) 60 European landfill sites) L: Europe	hypothetical	potentially profitable (-22 to +1.7; -27 to +2) complete material separation and sales	separation, transport on- & off-site, excavation, unforeseen costs	material sales (metals, plastics, CDW, stones, soil), value of landfill voidspace & land, reduction in process costs of materials, avoided aftercare costs
Ford et al., 2013	W: MSW S: medium (1.3 Mt) L: Scotland	hypothetical	potentially profitable (-91 to +33; -112 to +41) WtE int with energy recovery and sale of land	separation, excavation, re-landfilling, WtE int.	green certificate, value of land, electricity
Rosendal, 2015	W: monolandfill (shredder residue) S: small (0.3 Mt) L: Reno Djurs landfill, Denmark	full	potentially profitable (-22 to +20; -20 to +18) w/o re-landfilling tax, on-site separation, w/ tax refund, longer aftercare period (50-100 yrs)	re-landfilling, separation, WtE (incineration), transportation, excavation	material sales (metals), tax refund, financial provision refund, value of landfill voidspace, avoided aftercare (valuated but excluded in net result)
Van Passel et al., 2013	W: mix MSW-IW S: large (16 Mt, 182 Mt) L: REMO landfill and Flanders Region, Belgium	pilot	potentially profitable (-unspecified to +12; +15) societal benefit, sale of land	WtE (incineration), sorting & pre-treatment, contingency, excavation	electricity, material sales (shredder, metals, slag), value of land

Danthurebandara et al., 2015b	W: MSW S: medium (1 Mt, 50 000 m^2) L: open dumpsite, Sri Lanka	hypothetical	not profitable (-13 to -8; -16 to-10)	transportation, WtE, sorting, re-landfilling	electricity, material sales (metals, RDF, glass aggregates, glass), value of land
Danthurebandara et al., 2015c	W: mix MSW-IW S: large (16 Mt, 130 000 m^2) L: REMO landfill, Belgium	pilot	not profitable (-unspecified)	WtE (plasma gasification)	electricity, calorific value of RDF, green certificate
Frändegård et al., 2015	W: MSW S: small (0.1 Mt) L: hypothetical landfill, Sweden	hypothetical	potentially profitable, (-14 to +23; -15 to +25) 5% probability w/o re-landfilling tax, WtE int.	re-landfilling, WtE, separation, landfill reconstruction, transport	electricity & heat, material sales, value of land
Winterstetter et al., 2015	W: mix MSW-IW S: large (16 Mt, 130 000 m^2) L: REMO landfill, Belgium	pilot	not profitable (-19 to -12; -23 to -15)	WtE, separation, excavation & storage	electricity, material sales (metals), avoided aftercare
Wagner & Raymond, 2015	W: monolandfill (ashfill) S: large (725 700 Mt) L: Ecomaine landfill, USA	full	profitable (+49: +48)	separation, excavation, fuel, labor, maintenance	material sales (metals), value of landfill voidspace
Zhou et al., 2015	W: MSW S: small (0.5 Mt) L: Yingchun Landfill, China	pilot	profitable (+3 to +29; +7 to +67)	excavation, separation, transportation	electricity, value of land, recycling soil-like materials
Damigos et al., 2016	W: MSW S: small (0.4 Mt) L: Polygyros landfill, Greece	pilot	potentially profitable (-5.4 to +170; -9 to +269) socioeconomic costs & benefits	socioeconomic costs (harmful effects of excavation & processing, waste disposal, etc.), excavation, separation	socioeconomic benefits (direct employment, minimization of contamination, etc.), material sales (plastic, metals)
Wolfsberger et al., 2016	W: MSW S: small (0.7 Mt) L: Ave. Sanitary Landfill, Austria	pilot	not profitable (-40; -48)	re-landfilling (incl. transport), separation	material sales (metals, aggregates)
Hermann et al., 2016	W: MSW S: small (0.7 Mt) L: Ave. Sanitary Landfill, Austria	pilot	not profitable (-39 to -12; -47 to -14)	re-landfilling separation, excavation	value of landfill voidspace, material sales (metals)
Kieckhäfer et al., 2017	W: MSW S: medium (2.6 Mt, 270 000 m^2) L: Pohlsche Heide Landfill, Germany	pilot	not profitable (-62 to -35; -76 to -43)	WtE (waste incineration & RDF incineration plant)	value of land & landfill voidspace, material sales (metal)

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INTEGRATION OF RESOURCE RECOVERY INTO CURRENT WASTE MANAGEMENT THROUGH (ENHANCED) LANDFILL MINING

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ABSTRACT

Europe has somewhere between 150,000 and 500,000 landfill sites, with an estimated 90% of them being "non-sanitary" landfills, predating the EU Landfill Directive of 1999/31/EC. These older landfills tend to be filled with municipal solid waste and often lack any environmental protection technology. "Doing nothing", state-of-the-art aftercare or remediating them depends largely on technical, societal and economic conditions which vary between countries. Beside "doing nothing" and landfill aftercare, there are different scenarios in landfill mining, from re-landfilling the waste into "sanitary landfills" to seizing the opportunity for a combined resource-recovery and remediation strategy. This review article addresses present and future issues and potential opportunities for landfill mining as an embedded strategy in current waste management systems through a multi-disciplinary approach. In particular, three general landfill mining strategies are addressed with varying extents of resource recovery. These are discussed in relation to the main targets of landfill mining: (i) reduction of the landfill volume (technical), (ii) reduction of risks and impacts (environmental) and (iii) increase in resource recovery and overall profitability (economic). Geophysical methods could be used to determine the characteristics of the landfilled waste and subsurface structures without the need of an invasive exploration, which could greatly reduce exploration costs and time, as well as be useful to develop a procedure to either discard or select the most appropriate sites for (E)LFM. Material and energy recovery from landfilled waste can be achieved through mechanical processing coupled with thermochemical valorization technologies and residues upcycling techniques. Gasification could enable the upcycling of residues after thermal treatment into a new range of eco-friendly construction materials based on inorganic polymers and glass-ceramics. The multi-criteria assessment is directly influenced by waste- and technology related factors, which together with site-specific conditions, market and regulatory aspects, influence the environmental, economic and societal impacts of (E)LFM projects.

1. INTRODUCTION

From the very beginning of the development of human settlements and the accumulation of residues discarded by their inhabitants, certain places, known today as landfills, have been created for the disposal of waste. Prior to the 1950s, those sites were mostly wild dumpsites in which environmental, health and safety implications were not taken into account, making them critical sources of pollution and posing a threat to the environment (Meegoda et al., 2016).

Research carried out over the last decades as well as growing public awareness have led to modern guidelines and regulations (Meegoda et al., 2016), with an increasing

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tendency for the implementation of an integrated waste management system into a circular economy. However, the role of the landfill in a modern waste management system as an ultimate sink for contaminants is still valid (Brunner, 2004). Nowadays it is well known that landfilled waste undergoes several degradation processes during a long period of time, and with negative environmental implications (Belevi & Baccini, 1989; Bozkurt et al., 1999; Martensson et al., 1999). Leaching of heavy metals and other toxic compounds to soil, surface- and groundwater, gas emissions, such as SO_2, CH_4, CO_2, surface runoff, windblown litter and dust, and proliferation of birds, vermin and insects are among the most common negative environmental and health effects of waste landfilling (Höxter, 2001), if landfill gas and leachate are not properly managed.

Contemporary landfills, known as "sanitary landfills" (Figure 1) are engineered disposal sites designed to minimize adverse environmental and health impacts, while higher safety measures are imposed and the storage of waste is enhanced (e.g. waste compaction and conformation) (EU Landfill Directive 1999/31/EC). Before depositing waste in a sanitary landfill, the place destined to become such a disposal site is carefully selected and its base is covered by a number of protection barriers (e.g. a layer of compacted clay, asphalt and/or synthetic liners), which prevent the infiltration of leachate directly into the ground and groundwater bodies. Additionally, a drainage system is placed at the bottom of the landfill basin, where a network of pipes collects the leachate generated by the disposed waste and transports it to further treatment. Waste disposed of in a landfill is also covered with a low permeability top layer to prevent infiltration of rain water, migration of gaseous emissions, windblown waste, and presence of harmful fauna. An additional pipe network is employed to collect gaseous emissions (i.e. biogas) produced during the biological decomposition of waste (Meegoda et al., 2016). Gaseous emissions, leachate, and groundwater quality in the area of the landfill site are continuously monitored in order to detect problems and, ideally, ensure that no damage is done by the landfilling of waste (Chian & DeWalle, 1976; Meegoda et al., 2016).

However, aftercare activities (i.e. emissions monitoring and treatment and infrastructure maintenance) related to landfill sites need to be carried out over a long period of time, since the potential emissions from landfills can have significant impacts to human health and the environment for decades or even centuries (Laner et al., 2012). Over this long time period, the space used by the landfill remains occupied and unavailable for certain uses, e.g. due to insufficient geotechnical stability, which leads to paramount costs and blocked capital. Moreover, control and protection barriers in sanitary landfills may eventually fail and, alike in non-sanitary landfills or wild dumpsites, liquid, gaseous and solid emissions can be released into the environment (Laner et al., 2011b; Pivato, 2011).

Despite being an apparently low cost and relatively simple waste management disposal method, the role of waste landfilling in a circular economy model is more restricted to that of an ultimate sink of contaminants. If landfill aftercare is not conducted adequately, even contemporary landfills might represent an environmental and health hazard.

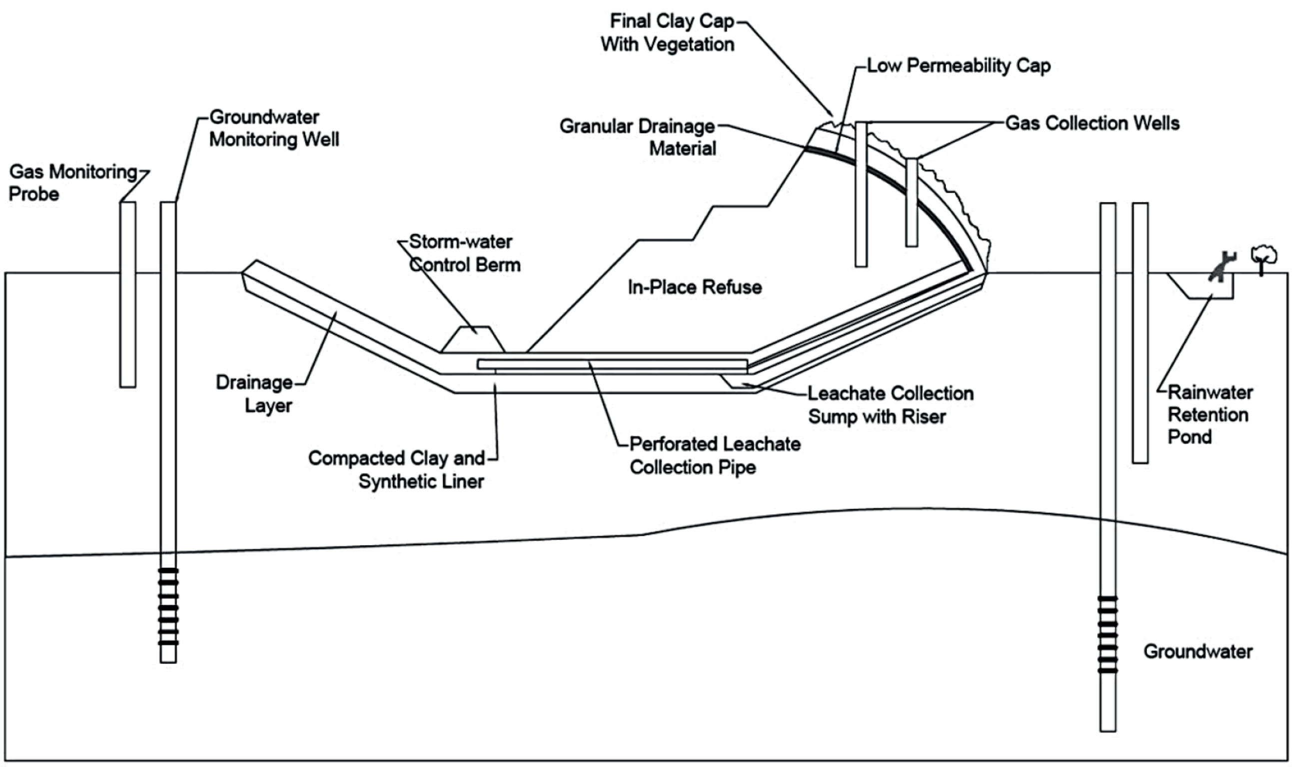

FIGURE 1: Cross-section of a contemporary sanitary landfill (Meegoda et al., 2016).

2. LANDFILL MINING

The extraction of waste from disposal sites for the recovery of certain materials is far from being a novel and unprecedented concept. It is, in fact, a relatively well-known and widespread practice that has been carried out in many countries over the last six decades, which is known as landfill mining (LFM). There are several definitions of LFM; the first one was introduced by Cossu et al., 1996, in which LFM was defined as "the excavation and treatment of waste from an active or inactive landfill for one or more of the following purposes: conservation of landfill space, reduction in landfill area, elimination of a potential contamination source, mitigation of an existing contamination source, energy recovery from excavated waste, reuse of recovered materials, reduction in waste management system costs and site re-development". As defined by Krook et al., 2012, LFM is "a process for extracting materials or other solid natural resources from waste materials that previously have been disposed of by burying them in the ground". More than half a century has passed from the beginning of LFM projects (Savage et al., 1993) and the drivers for LFM have spanned from regaining landfill capacity to recovering valuable materials, such as organic matter for soil improvement purposes, refuse derived fuel (RDF) and metals (Hogland, 2002; Prechthai et al., 2008; Savage et al., 1993; Shual, 1958; Van der Zee et al., 2004). As confirmed by Krook et al., 2012, "landfill mining has primarily been seen as a way to solve traditional management issues related to landfills such as lack of landfill space and local pollution concerns. Although most initiatives have involved some recovery of deposited resources, mainly cover-soil and in some cases waste fuel, recycling efforts have often been largely secondary".

In general terms, LFM projects have focused on expanding landfill lifetime and consolidating landfill area to facilitate the closure and remediation of those sites (Cha et al., 1997; Dickinson, 1995; Krogmann & Qu, 1997; Spencer, 1990). The recovery of land and materials represent important drivers for LFM, together with the potential to reduce surface-, groundwater and soil contamination by remediating the landfill (Marella & Raga, 2014). This could also contribute to the reduction of aftercare costs and other pollution-related costs. Although the mentioned factors represent important drivers for LFM, the excavation and material valorization processes could also lead to additional costs and impacts (Hermann et al., 2016). Moreover, LFM has faced great and growing challenges over time, many of them led by low amount and quality of high-value materials present in landfill sites, high costs for its implementation and increasingly stringent regulation in the waste management sector, as well as raising standards in the production industry (Krook et al., 2012).

2.1 Scenarios of LFM

Within the framework of the "EU Training Network for Resource Recovery through Enhanced Landfill Mining – NEW-MINE" (NEW-MINE), LFM has been classified in four scenarios, namely "Do-Nothing", "Classic remediation with relandfill", "Classic landfill mining with RDF state-of-the-art (co-)incineration" and "NEW-MINE" scenarios. The routes followed by each scenario, as well as the processes included in those routes are schematized in Figure 2.

2.1.1 "Do-nothing" scenario

As previously mentioned, old landfill sites or wild dumpsites have very few or no protection measures at all to prevent environmental and health damages that contemporary sanitary landfills normally have. Moreover, the containment system of sanitary landfills is likely to fail over time. Therefore, the "Do-nothing" scenario should not really be an option, since it turns a blind eye to the critical risks posed by those sites and leaves the problem unsolved indefinitely. Some basic and relatively inexpensive improvements that can be implemented in those sites could be (Höxter, 2001):

- Definition of dumping areas
- Waste delivery control and documentation
- Volume reduction of disposed waste by means of waste conformation and compaction
- Aerobic pre-treatment of waste to reduce methane emission
- Installation of biogas collection system
- Installation of groundwater wells for monitoring
- Installation of leachate collection system (if possible)

2.1.2 "Classic remediation with relandfill" scenario

This scenario envisages the extraction of waste from wild dumpsites and old landfills, or problematic sanitary landfill sites, in order to place the excavated waste in a more appropriate disposal site, such as a new or contemporary sanitary landfill. For example, this can be the case due to the need to fulfill modern regulatory requirements and conventional solutions are not able to improve environmental conditions or remediate the problem (Höxter, 2001; Jones et al., 2013, 2018; Van Passel et al., 2013). According to the Flemish Public Waste Agency in Belgium, the costs for landfill remediation for the EU-28 could be as high as 100 billion to 1 trillion euros. This approach is also addressed in the initiative "Closing Dumpsites" of the International Solid Waste Association (ISWA), as the costs for re-landfilling are still far below from those of all alternatives, because the costs for processing might exceed the revenues from potentially recyclable fractions (Winterstetter et al., 2015).

2.1.3 "Classic LFM with RDF state-of-the-art (co-)incineration" scenario

The classic LFM approach is looking not only to remediate the landfill site, but also to minimize remediation costs through the valorization of landfill waste materials. This approach has been largely applied in previous LFM projects, since it also aims to decrease the amount of waste to be re-landfilled; valorizing waste through the separation of materials with high calorific value, such as paper, plastics, textiles and wood, among others, for thermal valorization, and recyclable materials, such as metals and glass, among others, for material valorization. The thermal valorization is carried out mainly by the production of RDF, which is used in (co-)incineration plants to recover energy in the form of

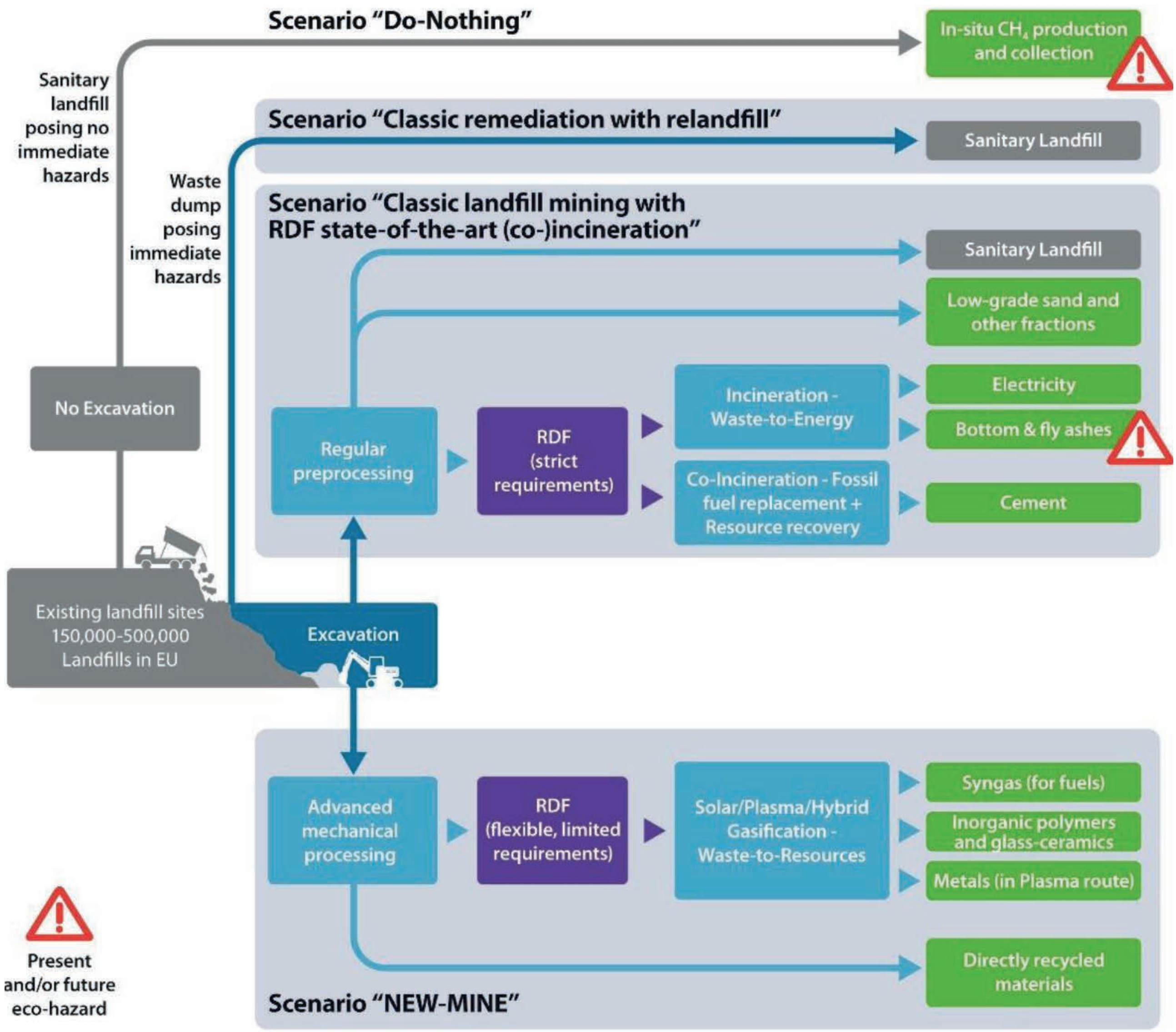

FIGURE 2: LFM scenarios (source: https://new-mine.eu).

heat and electricity (Jones et al., 2013). However, as nowadays prices for RDF are commonly negative, this scenario results less economical than the "Classic remediation with relandfill" scenario.

2.1.4 Enhanced landfill mining or "NEW-MINE" scenario

The need for a common framework to address LFM issues, technological development and further research on this subject has pushed scientists to develop a holistic concept called enhanced landfill mining (ELFM), which is, as defined by Jones et al., 2010, "the safe conditioning, excavation and integrated valorization of (historic and/or future) landfilled waste streams as both materials (Waste-to-Material, WtM) and energy (Waste-to-Energy, WtE), using innovative transformation technologies and respecting the most stringent social and ecological criteria", and has been under development by the Flemish ELFM Consortium since 2008 (Jones et al., 2013). To this end, the European Union´s Horizon 2020 research and innovation programme within the framework of the Marie Skłodowska-Curie actions has funded the NEW-MINE project, which is referred to as "NEW-MINE" scenario in this review article. In the "NEW-MINE" scenario the technological innovation follows a value-chain approach, from advanced landfill exploration, mechanical processing, thermochemical conversion and upcycling, while the multi-criteria assessment methods compare combined resource-recovery/remediation ELFM methods with the previous scenarios: "Do-nothing", "Classic remediation with relandfill" and "Classic LFM with RDF state-of-the-art (co-)incineration". The ELFM concept or "NEW-MINE" scenario is currently under development and the main goal is to insert LFM in a circular economy context, where most of the residues are upcycled and, therefore, minimized.

2.2 Stages in LFM

2.2.1 Site exploration

The material composition and physicochemical properties of the waste disposed of in a landfill site are the preliminary and most important information to be gathered in LFM in order to assess the economic, technical and environmental feasibility of the project. However, it is not rare that there are no records about the type or location of the waste deposited in a landfill. Hence, in the best case, LFM projects need to resort to invasive exploration by means of bore sampling or small scale excavations throughout the whole landfill site; in other cases the available disposal records are used to determine the composition and characteristics of the waste, while in the worst case, no previous analysis is done at all (Hernández Parrodi et al., 2018a). In the case of invasive exploration, the excavated waste samples are classified according to material type and particle size, which are used to determine the amount of material that might be valorized and estimate the remediation costs of the whole site (Bhatnagar et al., 2017; Cha et al., 1997; García López et al., 2019; Hernández Parrodi et al., 2018b; Hogland, 2002). However, certain fractions which might be valorized from fresh waste, may not be valorized from landfilled waste due to degradation and contamination processes in the landfill body.

From an ELFM perspective, geophysical methods, such as the ones used for underground water or petroleum exploration, could be used to determine the material characteristics in a rough manner without the need of an invasive exploration, as well as to identify the most interesting area, in terms of depth, water content and presence of certain materials, before carrying out the extraction of landfilled waste. The characterization of landfill subsurface structures using non-destructive and rapid approaches could greatly reduce the exploration costs (Bobe et al., 2018) and be useful to develop a procedure to either discard or select the most appropriate sites for LFM, according to specific criteria. For example, Figure 3 depicts the characterization of the subsurface structures of a landfill, as well as their electric and dielectric properties. These could allow the identification of the type of material to be expected according to the depth and extent of the landfill, as well as the potential presence of metallic materials and water.

2.2.2 Excavation and material processing

After the exploration of the site and a positive assessment of the feasibility of the site in question for LFM, the excavation of the landfilled waste takes place. This is normally done by using bulldozers to remove the top cover layers and excavators to dig out the landfilled waste. The excavated waste is usually loaded in trucks and transported to the processing plant.

Relatively simple technologies have been employed to process the excavated landfilled material, as for example trommel sieving, magnetic separation, and density classification, which in some cases have shown marginal performance in producing marketable recyclables (Krook

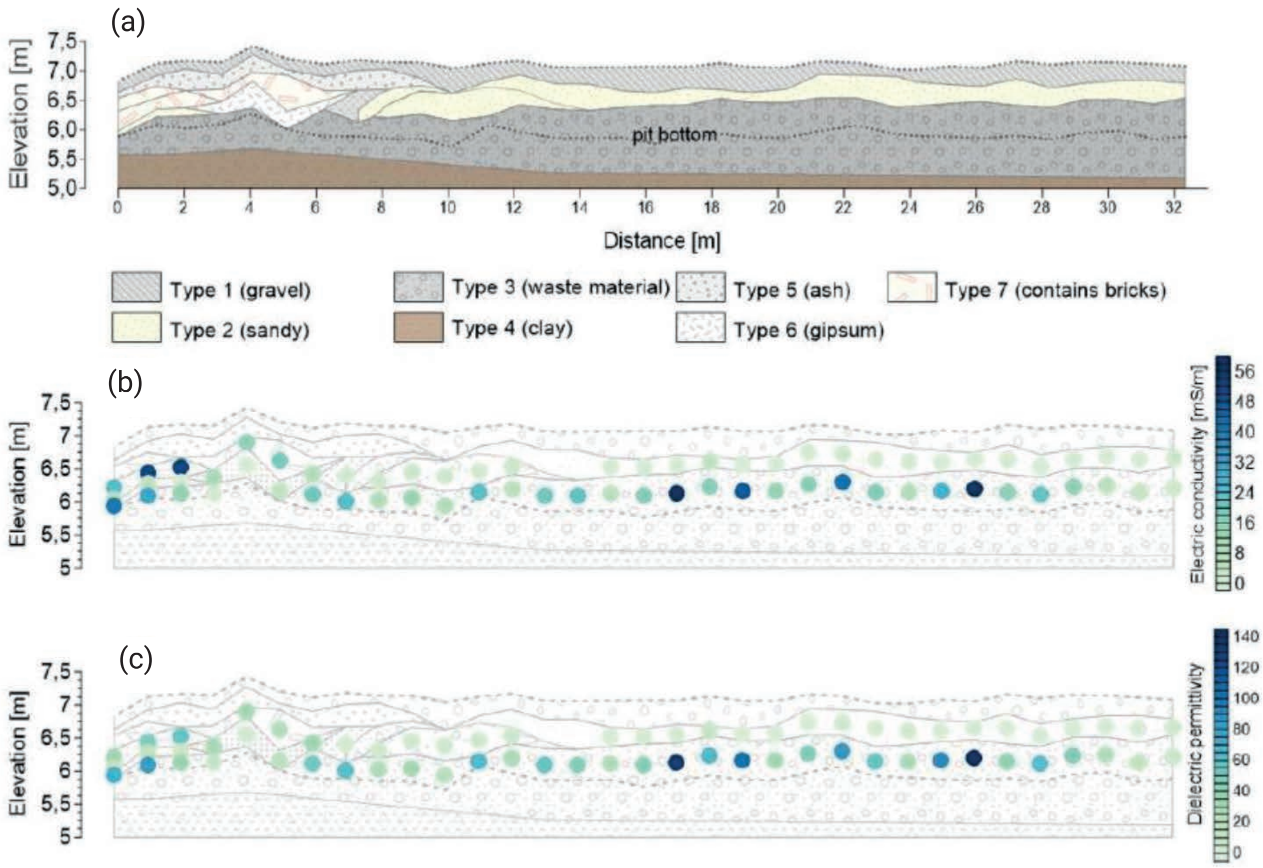

FIGURE 3: Schematic representation of the (a) profile description indicating the main types of material discriminated and (b) profile measurements of electric conductivity and (c) dielectric permittivity (Bobe et al., 2018).

et al., 2012). Moreover, LFM has faced great and growing challenges over time, many of them led by low amount and quality of high-value materials present in landfill sites, high costs for its implementation and increasingly stringent regulation in the waste management sector, as well as raising standards in the production industry (Krook et al., 2012).

Besides the traditional techniques used in traditional LFM projects, new equipment is nowadays being tested with promising results. Ballistic separators can separate landfill waste into three different fractions, namely three-dimensional (3D) and two-dimensional (2D) materials, and an under-screen fraction. This technology can be used to pre-process the landfilled waste directly after excavation and precondition the material for further mechanical processing (García López et al., 2019). Further processing, such as drying, particle size reduction equipment, particle size classification, ferrous and non-ferrous metal separators, density separation methods and sensor-based sorting, could be employed in order to sort the landfilled waste into different material outputs (Hernández Parrodi et al., 2019b; Küppers et al., 2019), such as:

- High calorific value materials (e.g. plastics, wood, textiles, paper and cardboard, among others)
- Ferrous metals (e.g. iron and steel)
- Non-ferrous metals (e.g. Cu, Al, Zn, Pb, Ni, among others)
- Inert materials (e.g. glass, ceramics, and concrete, among others)
- Residual fraction (i.e. normally fraction with finest particle size)

Some of these output flows could be used to recover materials through recycling (e.g. glass, inert materials, ferrous and non-ferrous metals) and to produce an alternative fuel (i.e. high calorific value materials), while a certain amount of the residual fraction might need re-landfilling or further processing (Hernández Parrodi et al., 2018b). Comprehensive studies on the resource potential of LFM materials can be found in Wolfsberger et al., 2015, García López et al., 2019, and Hernández Parrodi et al., 2019a.

In general, there are two main strategies to valorize waste in the current waste management system. The first, known as waste-to-material (WtM), targets recycling of waste, such as plastics, metals and minerals, to replace primary raw materials. The second one, known as waste-to-energy (WtE), aims to valorize waste materials with a high calorific value in (co-)incineration plants to produce thermal and electrical energy.

3. WASTE-TO-MATERIAL

3.1 Metals

Ferrous and non-ferrous metals are considered the most valuable resource extracted from landfills. According to Winterstetter et al., 2015, and Van Vossen & Prent, 2011, those contribute the most to the revenues from LFM. The technology for recycling and upcycling metal scrap is nowadays the most developed compared to that for other waste fractions, such as inert materials or plastics. For this reason, finding a market for metal scraps coming out from landfilled waste is not considered to be a critical issue. Nevertheless, the quality of the recovered metals from landfilled waste does play a relevant role in the extent of their recyclability and marketability, and, hence, is to be taken into account while assessing the recovery potential and economic feasibility in (E)LFM projects.

A detailed study on the quality assessment of the non-ferrous metals recovered from a Belgian landfill can be found in Lucas et al., 2019.

3.2 Inert materials

According to previous investigations, most of the excavated waste in LFM projects corresponds to fine fractions (Hernández Parrodi et al. 2018a). Fine fractions are mostly composed of a mixture of degraded organic matter and weathered inert materials, which, if adequately separated, might be used to produce recycled construction aggregates (e.g. construction sand). The use of inert materials recovered from waste as construction aggregates is regulated by Article 6 of the EU Waste Framework Directive (2008/98/EC), which states a series of criteria to be complied with in order to recycle such recovered materials. However, additional criteria that depend on local legislation might be enforced as well (EU-Report 26769 EN, 2014).

3.3 Plastics and other materials

The recovery of plastic materials from LFM for recycling purposes might be possible; nonetheless, the high degradation state in which these materials are recovered, and their degree of contamination with impurities and surface defilements represent a relevant obstacle to follow the WtM route (Wolfsberger et al., 2015). Therefore, plastics recovered from LFM may result more suitable for the production of RDF, which can be used in WtE co-incineration plants (Bhatnagar et al., 2017).

Other materials, such as organic matter, wood, textile, leather, paper and cardboard cannot be recycled directly to replace primary raw materials due to their level of degradation and contamination, and poor quality (Quaghebeur et al., 2013; Spooren et al., 2013; Wolfsberger et al., 2015) (Table 1).

4. WASTE-TO-ENERGY

Carbonaceous material sorted from landfilled waste, which cannot be recycled directly, can be valorized into energy. Three main thermal treatment technologies have been developed in order to recover energy from municipal solid waste (MSW) and industrial waste (IW): incineration, pyrolysis and gasification (Yan et al., 2016; Kalogirou, 2018).

4.1 Incineration

Incineration is the most widespread and mature WtE technology to dispose of MSW and IW (the combustible solid waste volume can be reduced up to 90%) and, simultaneously, produce electricity and district heating. This process can accept waste without any pre-treatment and in a wide range of compositions and is, therefore, very robust and versatile and relatively simple. Complete combustion

TABLE 1: Overview of the situation with respect to the utilization of waste-derived aggregates in some selected EU Member States (EU-Report 26769 EN, 2014).

Member State	Regulation of the use of waste aggregates?	Criteria on total content?	Criteria on leaching?	Type(s) of leaching tests required?
Austria	Guidelines	Yes	Yes	EN 12457-4 (L/S=10 l/kg)
Belgium	Yes, in the Flemish region	Yes	Yes	CEN/TS 14405 (L/S=10 l/kg)
Czech Republic	Based on Landfill legislation *	Yes	Yes	EN 12457-4 (L/S=10 l/kg)
Denmark	Yes	Yes	Yes	EN 12457-1
Finland	Yes	Yes	Yes	CEN/TS 14405; EN 12457-3 (L/S=10 l/kg)
France	Yes	Yes	Yes	EN 12457-2 and 4**
Germany	Guidelines (new regulation in preparation)	Yes	Yes	EN 12457-2 and DIN 19528 (new legislation)
Hungary	Some	No	Yes	Unknown
Italy	Yes	No	Yes	EN 12457-2 (L/S=10 l/kg)
The Netherlands	Yes	Yes	Yes	CEN/TS 14405 (L/S=10 l/kg)
Poland	No	No	No	Unknown
Portugal	Some guidelines	No	No	Unknown
Slovakia	No	No	No	Unknown
Spain	Yes, regional	No	Yes	EN 12457-4 and DIN 38414-S4
Sweden	Guidelines, case by case	Yes	Yes	CEN/TS 14405 (L/S=10 l/kg)

** Considering adopting Austrian guidelines*
*** For compliance testing (CEN/TS 14405 for basic characterization)*

of the waste is achieved in a controlled oxidizing environment, usually with excess air, at temperatures that can vary from 800°C to 1200°C, typically in the range 800-900°C. The carbonaceous solid waste undergoes four consecutive stages: (i) evaporation of the moisture content, (ii) release of volatile hydrocarbons/charcoal formation, (iii) combustion of the volatiles and (iv) combustion of the residual charcoal. The combustion chamber is commonly a moving grate furnace, but also different configurations like fluidized bed and rotary kiln are used. Most of the calorific content of the waste is transferred to flue gases in form of sensible heat and delivered to a downstream power block. Flue gases include products of incomplete combustions (e.g. carbon monoxide, alkenes, organic acids, soot, etc.), particulate matter (usually inorganic salts or oxides mixed with incomplete combustion compounds), acidic gases (HCl, SO_2, SO_3, NO_x, etc.), heavy metals and dioxins. Modern flue gas cleaning systems benefit of a wide range of air pollution control techniques that allow to comply with the strictest admissible environmental emissions limits before discharging into the atmosphere. The residues from waste incineration, in the form of bottom ash and air pollution control residues (APCR), are then treated and recycled, e.g. paving roads, or landfilled (Gleis et al., 2001). The overview of a typical incineration plant is shown in Figure 4.

4.2 Pyrolysis

Pyrolysis and gasification are sub-processes of incineration, but they can also be entirely distinct technologies. Pyrolysis of carbonaceous solid waste consist of its thermal decomposition, typically in the temperature range 300-850°C, in an inert/reducing environment, i.e. with no addition of oxygen. Due to the complex composition of MSW, a multitude of reactions, mainly endothermic, occur simultaneously in the reactor and its output is divided into gaseous products and a solid residue (the so-called char). The first consists mainly of CO and H_2, but includes also CH_4 and other volatile organic compounds (VOCs). A fraction of the latter is condensated and results into liquid products: oils, waxes and tars. The solid residue is a mixture of coke and a non-combustible inorganic fraction and, although it can be further processed to release the energy content of the organic part, typically there is low demand for it. The reactor configuration (fixed bed, fluidized bed, screw kiln, rotary drum, etc.) is selected based on the operation mode (batch, semi-continuous or continuous) and on the method used to deliver the heat, usually transferred indirectly, i.e. by thermal conduction. Although pyrolysis allows to reduce the volume of the waste and simultaneously recover energy from it, due to the low energy outputs obtained in the case of MSW, its industrial scale application is very limited.

4.3 Gasification

In the gasification process, carbonaceous waste reacts with a gasification agent (oxygen and/or steam) at temperature that vary from 700 to 1600°C in partially oxidizing conditions (absence or substoichiometric presence of oxygen) to produce a fuel gas called syngas, which is a mixture of CO and H_2. A significant advantage of gasification is that syngas can be combusted at higher temperatures than those achievable with the original fuel or even in fuel cells, so that the thermodynamic efficiency of the downstream power cycle is enhanced. In addition, problematic chemical elements (e.g. chloride and potassium) can be separated from the syngas, allowing the production of clean combustion flue gases. Furthermore, syngas

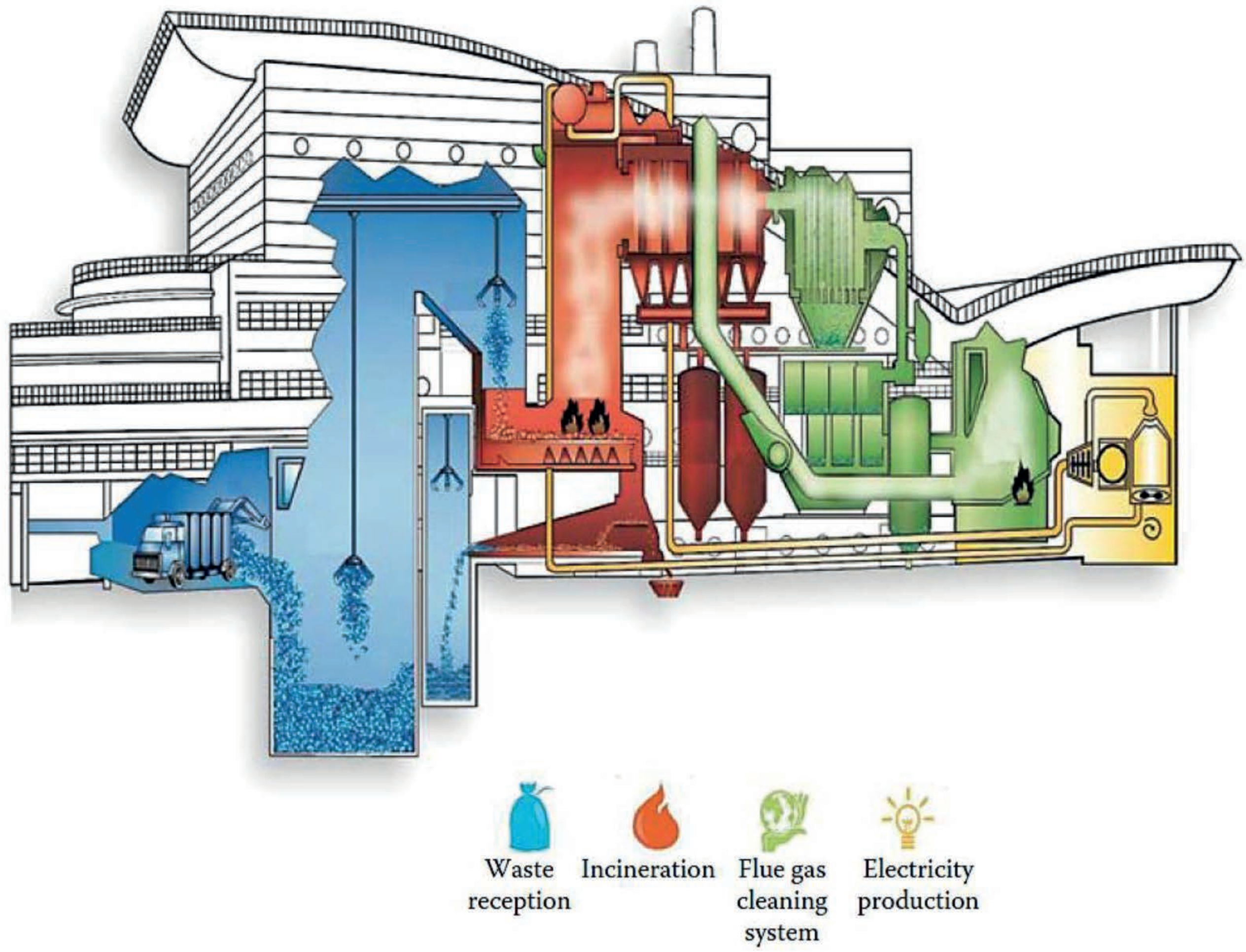

FIGURE 4: Scheme of a MSW incineration plant for power generation (SEVEDE, 2007).

can be stored and used in internal combustion engines or converted into high-purity hydrogen or synthetic fuels. The other products from gasification are a solid inert residue (ash) and tar, which is usually cracked into smaller hydrocarbon molecules in a downstream reactor placed after the gasifier in order to avoid deposit and blockage of the piping. Types of gasifiers include fixed bed, fluidized bed, rotary kiln and other less common configurations. The gasification process consists of several chemical and physical phenomena occurring in series-parallel with each other and, generally, the feedstock is subject to 4 stages (not necessarily carried out in the same reaction chamber): (i) evaporation of the moisture content, (ii) release of volatile hydrocarbons/formation of char (mixture of fixed carbon and inert), (iii) reduction (with steam or hydrogen) of a portion of the fixed carbon with release of hydrocarbons and simultaneous oxidation (with oxygen or steam) of the other fraction of the fixed carbon and some hydrocarbons, (iv) melting of the inorganic residues. The heat released from the exothermic partial oxidation of the waste compensates for the heat absorbed by the endothermic reactions (decomposition and reduction) and for the latent heat of fusion of the inorganic residues. The melting of the ash generates a vitreous inert material that, instead of being landfilled, could be exploited by the construction industry. Considering the potential benefits (e.g. production of a storable and clean energy carrier, significant waste volume reduction and efficient pollution control), gasification has become the most attractive integrated solution for both waste treatment and energy recovery.

4.4 Other emerging gasification technologies

In the "autothermal" gasification part of the waste is oxidized instead of being converted into syngas, in order to drive the chemical and physical phenomena that absorb heat. In the "allothermal" gasification the extraction of the energy content of the waste is maximized by preheating it with an auxiliary energy source: external combustion, electrical energy, solar energy, etc. (Fabry et al., 2013; Sanlisoy et al., 2016; Piatkowski et al., 2011a; Loutzenhiser et al., 2017).

4.4.1 Plasma gasification

Hot plasma, an ionized gas formed by using electrical energy, can contribute to sustain the high-temperature transformations occurring in the gasification process. The maximum temperature achievable by combustion is around 3000 K (for the acetylene-oxygen mixture) while plasma can go up to 15000 K. The higher temperature attained allow to break down nearly all compounds into their elemental form and significantly accelerate the reaction rates, resulting in a higher waste conversion and cleaner products. Different process configurations are possible depending on where the plasma is injected: in the upper part of the reactor close to the waste inlet, at the bottom close to the molten bath, or in a separate chamber to treat the gases leaving the reactor. The higher the presence of hot plasma, the larger the fraction of the waste which is saved from being oxidized, so that it can be converted into syngas. However, an extensive use of plasma implies a significant electricity consumption that, besides being a potential indirect cause of CO_2 emissions, may exceed the energy content of the produced syngas, resulting in a low or even negative net power production of the process.

4.4.2 Solar gasification

By concentrating the diluted sunlight over a small area with the aid of mirrors, it is possible to obtain a dense beam of solar radiative energy that can heat up the carbonaceous waste to the high temperatures necessary for its gasification (Steinfeld et al., 2001). The solar energy input saves part of the feedstock from being burned, so that the process has the potential to be free of combustion by-products and yield a higher syngas output with respect to conventional gasification. Solar gasifiers, which have been demonstrated capable to operate at high temperatures (>1400 K), can be classified as (i) directly irradiated, where the solid waste is directly exposed to the concentrated radiation, or (ii) indirectly irradiated, where the concentrated solar beam heats up the reactor wall or a heat transfer fluid. Directly irradiated reactors offer efficient heat transfer, but need a transparent window that has to be carefully designed to withstand pressure fluctuations and prevent deposition of particles or condensable compounds on it. Indirectly irradiated reactors eliminate the necessity for a window at the expense of a less efficient heat transfer. The possible reactor configurations can be: packed bed (see Figure 5a), fluidized bed, entrained flow and vortex flow (see Figure 5b), among others. Solar gasification has the potential to be operated also during off-sun periods (i.e. night-time or cloudy days) with the integration of a thermal energy storage unit that can accumulate high-temperature (>900°C) solar heat during the day and release it when required (Ströhle et al., 2017; Gigantino et al., 2019).

4.5 Recycling residues from WtE plants

Treating MSW, IW or landfill waste in WtE plants does not completely solve the problem of waste disposal. Incineration, pyrolysis and gasification can reduce the volume of waste to a great extent, but there are still solid outputs, which can contain significant amounts of pollutants, such as chlorine, dioxins and heavy metals (Chimenos et al., 1999; Gleis et al., 2001; Sorlini et al, 2017).

Several EU states have adopted a critical attitude toward generation and disposal of waste. It is customary for EU countries, such as Belgium, Finland and the Netherlands to treat and recycle MSW bottom ash from WtE plants as aggregates for construction or road-paving (Kahle et al., 2015; Lynn et al., 2017). However, bottom ash does not always fulfill the requirements in terms of content and leaching of pollutants, so that countries such as Austria and Switzerland are still landfilling the ashes. APCR, such as fly ash or boiler ash, are also some of the outputs from WtE plants, which are considered as hazardous waste and need to be disposed of in special landfills, after metal recovery by the FLUWA process. In this regard, gasification and other novel waste treatment technologies could enable the upcycling of the residues after thermal treatment into a new range of eco-friendly construction materials based on inorganic polymers and glass-ceramics. Novel techniques allow to transform inorganic residue from WtE plants into thermal and acoustic isolation materials, such as traditional bricks and tiles (Kriskova et al., 2015; Rabelo Monich et al., 2018; Rincon Romero et al., 2018). An example of such upcycled materials is shown in Figure 6.

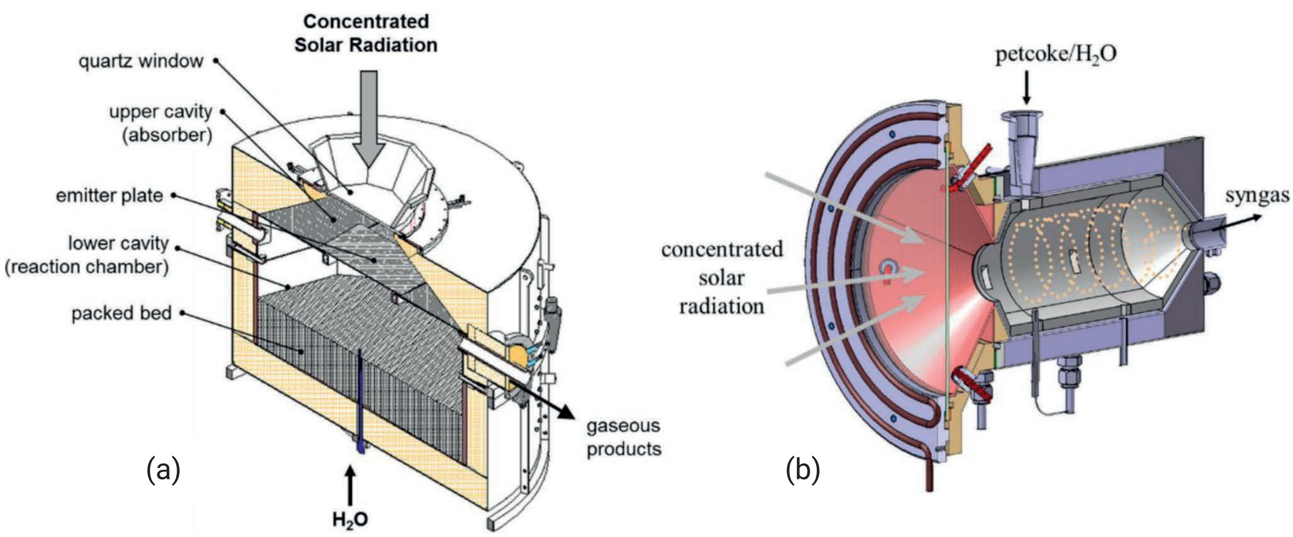

FIGURE 5: Scheme of (a) an indirectly irradiated packed-bed gasifier and of (b) a directly irradiated vortex flow gasifier (Piatkowski et al. 2011b, Z'Graggen et al. 2008).

FIGURE 6: Upcycling of vitrified landfill waste into construction materials (Machiels et al., 2017).

However, the final sink of contaminants in this approach is still unclear.

5. MULTI-CRITERIA ASSESSMENT IN (E)LFM

During the past decade, multi-criteria assessments (MCAs) of LFM projects have become of growing interest to academia, industry, and policymakers. MCAs of LFM aim to account for impacts and risks from an environmental, economic, and societal perspective. Results of such assessments support LFM stakeholders in the decision-making process among the previously described LFM scenarios, i.e. landfill remediation, classical LFM, and ELFM. As described in the previous parts of this study, these scenarios do not refer to a fixed process chain or technologies, but rather show the evolution of a concept to display the major differences and implications of those scenarios.

Most MCAs of LFM projects address environmental impacts and economic feasibility, while societal impacts are typically not addressed (Krook et al., 2018). Environmental impacts are commonly derived from life cycle and risk assessment (Danthurebandara, 2015; Frändegård et al., 2013a, 2013b; Gusca et al., 2015; Jain et al., 2014; Laner et al., 2016), whereas private economic costs and benefits are often determined by the means of the net present value (NPV) (Frändegård et al., 2015; Hermann et al., 2016b; Kieckhäfer et al., 2017; Laner et al., 2019; Winterstetter et al., 2015). Since societal impacts are of a diverse and often complex nature, including welfare changes as well as health risks, no common assessment method for LFM projects exists (Einhäupl et al. 2019a). Moreover, societal impacts are often strongly related to environmental and economic ones, as health risks are usually a consequence of environmental impacts and taxes are accounted for as a private economic cost, but also represent a societal income. Hence, it is often difficult to define clear boundaries between the different dimensions of sustainability in (E)LFM, especially when considering causal relations between different impacts.

Figure 7 displays the sustainability concept as the basis for a holistic multi-criteria assessment in LFM research. The lack of a common and integrated framework, which considers all three sustainability dimensions, shows the complexity of MCAs for LFM projects. While efforts have been made in literature (Hermann et al., 2016a; Pastre et al., 2018) to assess the feasibility of LFM in a holistic manner, challenges, like emerging technologies or long-term impacts of existing landfills, remain with respect to the estimation of the extent of these environmental, economic and societal factors influencing the feasibility of LFM due to varying contextual conditions and related stakeholder perceptions of drivers and barriers.

5.1 From landfill mining to enhanced landfill mining: a conceptual and technological evolution of LFM drivers and barriers

To address the previously mentioned challenges considering future MCA method development and modeling, this study provides a simple synthesis of critical factors that drive or hinder LFM projects as a result of previous sustainability assessments (Table 2). This synthesis discusses motivational drivers and barriers of LFM projects and contrasts the critical factors of the three different LFM scenarios. As previously mentioned, these scenarios represent the evolution of the concept of LFM, which led to changes in perceptions of potential drivers and barriers. For example, while environmental concerns usually drive remediation projects, increasing urbanization and growing resource scarcity have made the reclamation of land and materials important drivers for LFM, still including the potential to reduce surface-, groundwater and soil contamination by excavating the landfill (Marella & Raga, 2014). In

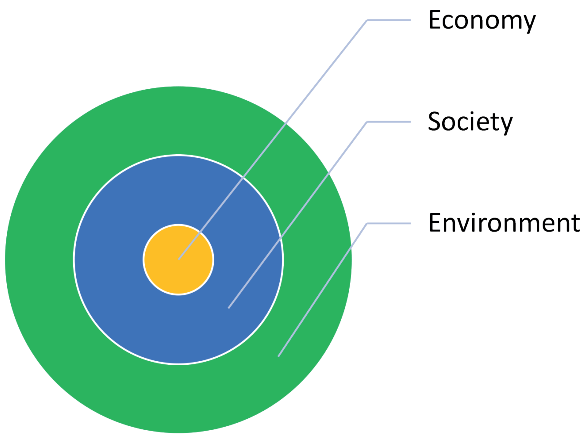

FIGURE 7: The sustainability concept as the basis for multi-criteria assessment in LFM research.

addition, LFM and ELFM have been linked to the avoidance of potential environmental hazard that is due to flooding risk brought by climate change (Laner et al., 2009a; Wille 2018). This could also contribute to the reduction of aftercare and other pollution-related costs. Moreover, technological development could potentially lead to the further valorization of currently re-landfilled waste streams, or their use in lower value applications, leading to the more integrated approach that characterizes the concept of ELFM. Nevertheless, while the mentioned factors represent important drivers for LFM, the excavation and material valorization processes could also lead to additional costs and impacts on every level of sustainability (Hermann et al., 2016a; Marella et al., 2014; Pastre et al., 2018).

The concept of LFM was first introduced in 1953 in Israel (Calderón Márquez et al., 2019; Krook et al., 2012). The aim of that LFM project was to recover materials as fertilizer, while also recovering ferrous and non-ferrous metals (Calderón Márquez et al., 2019). Since then, around 112 projects worldwide have been studied with objectives ranging from environmental protection, over avoidance of closure and post-closure care, the extension of landfill lifetime, land reclamation, to resource recovery, among others (Calderón Márquez et al., 2019). Drivers for the different landfill mining projects have been exhaustively summarized in the study by Calderón Márquez et al., 2019, and are discussed here in light of the evolution of the LFM concept, with particular focus on European projects.

Environmental protection has indeed been the most important driver in LFM (Calderón Márquez et al., 2019; Danthurebandara, 2015; Gusca et al., 2015; Laner et al., 2016; Marella et al., 2014). The need to reduce soil, surface and groundwater contamination has led to an increased interest in remediation strategies. However, given the high costs of remediation processes, new concepts and technologies were developed with the aim to recover valuable resources from landfills, such as combustibles and metals, to compensate the costs and recover materials (Jones et al., 2013; Krook et al., 2012). The concepts of LFM and ELFM, therefore, developed to further increase the resource recovery potential. As mentioned in Van Passel et al., 2013, apart from technological development (technology push), also regulatory- and market- related factors (regulatory push and market pull) determine the economic, societal and environmental performance of LFM projects. These regulatory push factors include legislative changes due to public and environmental pressures, urban development, subsidy schemes or strategic resource independence, among others. On the other hand, market pull factors include increase in material prices and resource competition or rising land prices that can help to facilitate the excavation and processing of the formally buried waste. Market-related barriers for LFM implementation can include quality standards for secondary raw materials, processing capacities of waste incinerators (Johansson et al., 2017b), for example, or a lack of investments due to awareness gaps among stakeholders (Einhäupl et al., 2018). Regulatory barriers mostly derive from legal uncertainty, since it is often unclear if gate fees or taxes that could hinder a project's implementation have to be paid or not (Johansson et al., 2017a). Moreover, public opposition due to environmental uncertainties and the risk of disamenities, i.e. dust, odor, noise, and traffic, can also hinder LFM projects.

During the past years and throughout the studies worldwide, drivers for the mentioned LFM projects have since varied, based on time or local and regional requirements for land-use and landfill void space. Moreover, new landfill regulations, such as the Landfill Directive 1999/EC/31 and similar global regulations on waste disposal, increased the interest in LFM projects to reduce risks of contamination and the related costs, as well as to comply with closure and post-closure requirements (Calderón Márquez et al., 2019; Laner et al., 2016). Today, landfills predating the 1999 Landfill Directive are commonly referred to as "dumpsites". As most dumpsites are lacking up-to-date environmental protection measures, classic remediation has usually motivated the excavation of such landfills.

While regulation can push project implementation, as well as hinder it, it is important to take a closer look at the regulatory situation of (E)LFM today. In general, the so-called EU Landfill Directive defines the legal framework for the design, management and closure of landfills (Council Directive, 1999). The so-called EU Waste Directive defines the regulations for waste treatment and safety issues when treating hazardous waste materials, for example (Council Directive, 2008). Since the request for an ELFM amendment to the Landfill Directive was rejected by the EU Commission in 2018, no specific regulations for (E)LFM exist (Jones et al., 2018), adding to the legal uncertainty. However, according to the EU Commission and a legal report from Austria, no current regulations prohibit (E)LFM operations, even at an industrial scale and scope (Eisenberger, 2015; Jones et al., 2018). Nonetheless, the lack of overarching European legislation leaves member states with a variety of options to deal with (E)LFM and gives little room to address common challenges for its implementation (Einhäupl et a., 2019b).

Table 2 summarizes the drivers for LFM projects and their evolution from the remediation concepts to enhanced landfill mining. As shown in Table 2, the goal of ELFM compared to LFM is to maximize the recovery of resources while complying with all other objectives (environmental protection, societal benefits, etc.). Hence, to minimize the re-disposal of excavated fractions is an important focus of ELFM projects.

5.2 Critical factors of LFM projects

Increasing environmental and societal pressures and higher resource recovery targets have led to increasing challenges related to the waste processing technologies and, therefore, to the quality and quantity of the materials and energy recuperation. Economic, environmental and societal assessments of LFM and ELFM projects have highlighted the influence of critical factors in the feasibility and potential benefits of such projects. For example, while potential economic benefits could mainly derive from land reclamation, and material and energy valorization (Hermann et al., 2016b; Krook & Baas, 2013), potential costs are related to the excavation and processing of the different waste fractions (Hermann et al., 2016b). These costs

TABLE 2: Drivers and barriers throughout the evolution of the LFM concepts.

	Landfill remediation	Landfill mining	Criteria on leaching?
Drivers	• Environmental protection (remediation)	• Environmental protection (remediation) and • risk mitigation • Legislative changes • Cost reduction through resource recovery • Extension of useful landfill lifetime • Mitigation of closure and post-closure after-care • Urban development • Flooding risk	LFM drivers with the addition of: • Resource recovery (maximization) • Innovative landfill management concept: integrated valorization routes • Minimization of re-landfilling of waste • Resource independence • Increasing resource scarcity
Barriers	• Remediation costs	• Low market prices for primary and secondary raw materials • Relatively high processing costs	• Public opposition • Quality standards • Legal uncertainty • Taxes and fees • Technological challenges

are highly dependent on the waste composition and quality, as well as technology choices. Implied environmental impacts again can cause potential societal impacts that can lead to public opposition, for example.

In general, technology availabilities and efficiencies for processing landfilled waste are still uncertain and under current investigation. These technological uncertainties make it difficult to assess LFM projects, as potentially related costs, risks and impacts could outweigh the potential benefits (Hermann et al., 2016b; Krook et al., 2012).

In the relevant literature, critical factors are often discussed based on their influence on the economic, environmental or societal performance of the LFM projects (Danthurebandara, 2015; Frändegård et al., 2013b; Gusca et al., 2015; Hermann et al., 2014; Laner et al., 2016). However, most factors have implications for two or all sustainability dimensions. For example, environmental protection, which addresses the need to minimize soil, surface and groundwater contamination, has mostly been discussed as an environmental driver. Nonetheless, this has economic and societal implications in the reduction of pollution-related costs or of health risks to local communities. The recovery of materials and their marketability has potential environmental and economic benefits from the avoided production of primary materials, and the revenues from the secondary raw materials. From a societal perspective processing and recycling of waste could lead to job creation, avoid post-closure costs and risks, or increase property values. Given the interconnection of the factors between the three sustainability dimensions, a few studies have divided the critical factors affecting the performance and feasibility of LFM projects according to the level of influence (Laner et al., 2016; Winterstetter, 2018; Winterstetter et al., 2018). In particular, site-, project-, and system-level factors have been identified, which are summarized in Table 3.

At a site-level, waste composition is one of the main critical factors in LFM projects and multi-criteria assessments, and strictly related to the specific landfill. Waste composition influences the emission potential of the landfill and, therefore, the environmental impacts, pollution related costs and remediation requirements. It also greatly affects the resource recovery potential of LFM and the valorization routes suitable for the specific case (García López et al., 2019; Hernández Parrodi et al., 2018b; Quaghebeur et al., 2013). Moreover, disamenities like dust and odor are partly dependent on the waste composition. Quality and quantity of materials define the choice of technologies and their efficiencies, which have proven to be critical factors in previous economic and environmental studies of LFM (Danthurebandara, 2015; Frändegård et al., 2013a; Gusca et al., 2015; Laner et al., 2016). In environmental and economic assessments, given the importance of environmental protection as one of the main drivers, the reference case is also of great importance. It represents the importance of leaving the landfill as it is, with environmental, societal and economic consequences. These are related to the landfill emission potential which can last for centuries after landfill closure (Doka et al., 2005; Laner et al., 2009b), and which are respectively related to the waste composition and its degradability, as well as landfill design and management (Laner, 2011a).

At a project-level, technology choices and their efficiencies greatly affect the performance of LFM. In particular, technology choices for WtM and WtE, combined with the background energy system and the waste quality, could be decisive in potential applications of LFM (Frändegård et al., 2013a; Gusca et al., 2015; Laner et al., 2016). In fact, the overall aim is to outweigh the costs and impacts related to the reference case with the processing and recovery of resources. Excavation, separation and sorting technologies could have high environmental and economic impacts, also in relation to the quantity and quality of the materials recovered. On-site and off-site processing options also affect the performance, as transportation distances have been identified as critical factors (Frändegård et al., 2013a; Gusca et al., 2015). Given the high amounts of waste that LFM projects could address, logistics also becomes an important factor. Storage and processing capacities and equipment are also crucial for the design of valorization routes, the quality and quantity of recoverable materials (Kieckhäfer et al., 2017). In addition, the choice of project motivation, i.e. the main driver, in terms of land recovery or void space recovery has critical influence on the performance of an LFM project. Land recovery means external re-landfilling of fine fractions, while void space recovery means internal re-landfilling. With the large share of fine fractions in the landfill waste composition (Hernández Parrodi et al., 2018a), its subsequent choice of management is particularly important (Laner et al., 2019).

System-level factors are those which cannot be ad-

TABLE 3: Selection of critical factors for (E)LFM implementation at the level of impact.

Critical factors		
Site-specific	**Project-level**	**System-level**
• Waste composition: quantity and quality of resources for recovery (3rd) • Reference case (1st)	• Technology choices and efficiencies: - Excavation, separation, and sorting (mobile, stationary and advanced, etc.): - WtE treatment: type and efficiency, energy carriers • Quality of the materials recovered and marketability • Logistics • Energy and processing costs • Investment and operating costs, costs for external treatment and disposal • Land or void space recovery • Avoided impacts	• Background energy system (2nd) • Primary material production system • Transportation: requirements and distances • Financial effects • Materials and energy prices • Legal, institutional, organizational, and societal structures: - Policy support - Community engagement

dressed for each case specifically, but that influence the environmental, economic and societal performances as they refer to country-specific, European-, or even global structures. These include legal, institutional, organizational and societal structures. In particular, the background energy system influences the environmental impacts based on the energy production mix of the country of implementation of the project. Materials and energy prices, and their variation in time affect the revenues of materials and energy recovery, also in relation to primary raw materials. The marketability of valorized WtE residues is also uncertain, as different studies have assumed different prices to no market at all (Danthurebandara et al., 2015; Winterstetter et al., 2015). Similarly, the value of recovered land and landfill void space depends on the existing market conditions, but also influenced by site-specific factors like location: whether a landfill is situated nearby a residential, industrial or natural areas (Marella et al., 2014; Van Passel et al., 2013).

Societal aspects are therefore becoming of increasing interest in the multi-criteria assessment of LFM, and have been mentioned throughout the relevant literature. When assessed, however, commonly interviews and ranking systems are used (Hermann et al., 2016b; Pastre et al., 2018) as well as monetization techniques (Marella et al., 2014; Winterstetter et al., 2018). Consequently, considerable subjectivity resonates with the assessment of societal factors, and various societal effects become entangled. This again leaves decision-makers having to deal with major uncertainties when evaluating societal risks and benefits, such as safety issues, health implications through groundwater contamination (Krook et al., 2012), disamenities (Einhäupl et al., 2018), welfare changes (Damigos et al., 2016) and so on. Before evaluating these risks and benefits it is important to carefully analyze societal drivers and barriers for LFM implementation to better understand the origins and mechanisms behind the impacts.

When integrating economic, environmental and societal impacts and perspectives into LFM multi-criteria assessment, several issues have to be addressed. LFM multi-criteria assessment has to deal with intra- and inter-dimensional trade-offs and conflicts. For example, taxes for re-landfilling excavated waste is a private economic cost and could, thus, hinder a project's implementation. On the other hand, these tax revenues are also a societal benefit. Moreover, different stakeholders are affected by various societal and environmental impacts. Emissions like particular matter coming from LFM operations affect neighboring communities, for example, while avoided impacts are often manifested in other locations or at a global level. To deal with these issues more research is needed. Specifically, the most important influencing factors on societal risks and benefits have to be identified and their interaction with private economic and environmental aspects analyzed.

6. CONCLUSIONS

This review article has extensively highlighted the current scenarios for landfill management and potential scenarios for the combined valorisation of waste, as both materials (WtM) and energy (WtE).

Landfills represent a source of hazards to human health and the environment. Moreover, the long-term potential emissions of landfills and the risk of failure of the containment systems, increase the need for aftercare activities and their related costs. In a context of lack of land surface and primary resources, landfills also represent a source of feedstock that could be recovered to answer the increasing demand for raw materials. LFM aims at addressing the potential to recover waste from landfills, while reducing the long-term impacts of landfills by remediating the sites.

Different scenarios can be considered which address LFM to different extents. Overall, the choice of scenario depends on technical, as well as economic, environmental and societal aspects. New technologies are under research to increase the recovery potential of waste materials, such as MSW and excavated waste from landfills. One of the main factors that influences the technical feasibility and efficiency of the recovery processes, is the quality of landfilled waste, since its heterogeneity, agglomeration, degradation and contamination could hinder the potential for material and energy recovery. Therefore, material composition and physicochemical properties of the waste disposed of in a landfill site are some of the preliminary and most important information to be gathered in order to assess the economic, technical and environmental feasibility of the project. Geophysical methods could be used to determine the subsurface structures and landfilled waste characteristics in a rough manner without the need of an invasive exploration, as well as to identify the most interesting area, in terms of depth, water content and presence of certain materials, before carrying out the extraction of landfilled waste. This could greatly reduce the exploration costs and be useful to develop a procedure to either discard or select the most appropriate sites for (E)LFM.

The recovery of material and energy from landfilled waste can be achieved through the implementation of relatively simple separation methods, such as particle size classification, ferrous and non-ferrous metal separation, density classification and sensor-based sorting, coupled with thermochemical valorization technologies and residues upcycling techniques. Gasification, among other novel waste treatment technologies, could enable the upcycling of the residues after thermal treatment into a new range of eco-friendly construction materials based on inorganic polymers and glass-ceramics, which allow to transform inorganic residue from WtE plants into thermal and acoustic isolation materials, such as traditional bricks and tiles.

As for the technical aspects, the multi-criteria assessment of ELFM is also influenced by the same waste- and technology related factors. These, together with site-specific conditions, market and regulatory aspects, influence the environmental, economic and societal impacts of this kind of projects. Intra- and interdimensional conflicts should be identified and taken into account for a broader assessment. The most influencing factors need to be considered at different levels to cover landfill emissions and societal impacts (site-level), include technology choices (project-level) and take into account the regulatory context (system-level) and background system.

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LANDFILL MINING APPLICATIONS IN THE LOMBARDY REGION

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ABSTRACT

Landfill Mining is still today a rare activity in Italy, certainly due to the high costs of realization, the difficulty of finding outlets for the waste extracted and its quality. An important contribution is given by unclear administrative procedure in the absence of a regulatory framework that contemplates this case. However, something is moving: in the last year in the Lombardy Region two interventions have been approved and implemented and a third is at the Environmental Impact Assessment procedure. All of them are aimed at the recovery of areas and have in common a limited extension and depth, an absence of environmental contamination and focus on a type of waste easy to recover/dispose of.

1. INTRODUCTION

Landfill is the last option in the list of priority that the European Commission defined for waste management; the landfilled products in fact cannot be used and can cause pollution, and the area is definitely compromised. In Europe it is estimated that there are more than 500.000 landfills, closed or in operation (source Eurelco - European Enhanced landfill mining consortium, 2018), that could be a potential risk for the environment and human health and contain a large quantity of metals, minerals or fuels to recover.

The application of Landfill Mining (LFM) could facilitate a different use of the site and could convert waste to materials to make new products or to generate energy.

Here is a summary of goals that can be achieved with a LFM project:

- reduction or zeroing of environmental impact of old landfills;
- material recovery, with an economic return;
- energy production from waste with a high calorific value and that cannot be recovered as material;
- availability of the landfill area for different use;
- availability of the landfill volume for non-recoverable waste.

LFM is perfectly compliant with the principles of Circular Economy, because it allows the material recovery and the utilisation of the site for other purposes. However, Landfill Mining has, till now, few applications in Italy, why? The first reason is probably the high cost of intervention, especially in case of contamination of ground or water, compared to the value of recovered materials. The high cost is due to the difficulties of operating it safely in the landfill area and the lack of available technologies to recover the material from the landfilled waste.

An important role is also played by unclear regulations for this sort of activities and uncertainty on the permitting procedures that discourage investments. The European and Italian laws do not regulate this kind of activity, that can be considered as an activity of soil remediation or as waste management, so it is difficult to define the correct administrative procedures to allow it.

In the Italian legislation, the definition of LFM doesn't exist yet, even if we have already some examples. In 2009 Lombardy Region defined LFM as an "innovative" activity «finalised to recover materials and/or areas and to reuse landfills only for residual waste resulting from recovery activities". Taking advantage of the submission of four landfill mining projects, Lombardy Region created a specific framework for the assessment of LFM projects that considers different cases.

In our opinion LFM should be authorised under the waste management legislation rather than under the soil remediation, even in case of contamination of environmental matrices.

(every time the landfill has been approved was under national law n. 152/2006 – Testo Unico Ambientale – and n. 36/2003 - Landfills).

2. EXAMPLES OF APPLICATION OF LFM IN THE LOMBARDY REGION

In the last few years in the Lombardy Region four experiences of LFM application have been proposed; this shows

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the interest of the waste management market in LFM and its opportunities.

Two out of four are not approved yet and concern waste from steelwork, the others two are completed and a lot of information on the way of realization is available.

All these examples have in common the goal of recovering the area devoted to the landfill however, only in the case of aggregate waste they have a plan for the material recovery.

A description of Lombardian LFM application is in the following.

2.1 Mella 2000

This is a case of an old landfill, realised at the end of ’70 and closed in 1997, near a watercourse, in an industrial area; the landfilled waste is inert residual from steel production. Landfill has not waterproofing system, nor a system for collecting and removal leachate and biogas. Investigation did not show contamination of environmental matrices.

The goal of the LFM activity is recovering the area to realise a new building (comparto D) to complete the commercial district “Mella 2000”. Steel waste will end up to another landfill, construction and demolition waste will be used – when possible - for the re-profiling of the area. The propriety of the area is private. The landfill area is of about 26.400 m^2 and the removal project excludes a part of 900 m2, where a high voltage power line is located.

The campaign plan, following drilling in August 2008, November 2011 and May 2013, denotes the presence of waste belonging to the category of steelworks, in some cases mixed with fragments of bricks and iron. Rare mud horizons have been detected in some cores. In any case, the results of the analyses carried out on the waste samples in the points investigated have found that the material deposited there is definitely to be classified in the steelworks family. Waste are landfilled from a minimum depth of about 4 meters to a maximum depth of 8 meters (Figure 1).

With regard to the delimitation of removal operations, it has been taken into account that the area of intervention is characterized by multiple presences of sub-service lines (traced high voltage power line Terna; high-pressure meth-

FIGURE 1: The “Mella 2000” project *.

ane gas pipeline SNAM; agricultural irrigation channel. The Terna line is not susceptible to translation, as it has recently been laid, and given the complexity and delicacy of the work. The presence of such infrastructure obliges compliance with a range of 3 m from the axis of the cables, within which no work can be performed. In addition, excavations or any supporting bulkheads must be maintained at a proper distance to ensure the safety and stability of the infrastructure, profiling the ground with stable escarpments.

For this reason, the removal project, on the east side, excludes the portion occupied by Terna and its respect band. On the other hand, the SNAM pipeline, which is also burdened by a respect band of 3 m from the axis of the pipeline, is expected to be translated. To relocate the methane gas pipeline, it is convenient to locate the latter alongside the Terna power grid, thus allowing the overlap of the respective respect bands and minimizing the dimensions. The hypothesized solution guarantees, in fact, to maximize the extent of the waste removal intervention area (actual excavation area is equal to approximately 24.500 sqm). The solution proposed in the project makes it possible to use the basement wall, supporting the underground surfaces of the future property that will be realized on the sector, thus profiling the ground with stable escarpment. The agricultural irrigation channel will be placed underground. The boundary of waste removal along the east side is therefore identified with the basement wall of the future property that will be built on the area, while, along the north side, the limit is dictated by the existing road, and by the foot grading along the west and south sides.

On the portions not affected by the removal, a capping will be made with the same characteristics for the safety of the escarpment of the landfill front.

This capping will therefore be made by a 30 cm thick layer of clay and a 30 cm draining layer (as a replacement for the layer of vegetable soil equal to 30 cm). Prior to the removal of waste, the entire area will be divided into two lots: lot 1 will coincide with the west portion of the area and Lot 2 with the eastern portion of the area. The removal of waste will begin from Lot 1, the steps that will be carried out are listed below:

- characterization of the material (one per mesh);
- waste removal and loading on vehicles without intermediate storage;
- sending to a suitable disposal center;
- certification of the excavation bottom plan.

Once the waste removal is completed in Lot 1, it will be carried out in Batch 2, in the same manner as the first batch.

In consideration that waste will be sent to disposal in another landfill, according to Directive 2008/98/EC, table 1, annex 1, we decided to assign the operation "D13 - Blending or mixing prior to submission to any of the operations numbered D1 to D12"; this determined the need of an environmental impact assessment (EIA). The realisation of the mall also needed an EIA, so the two assessments were considered in an overall procedure that it is not already completed (Figure 2, Table 1).

2.2 Railroad Brescia - Verona

At the moment this project is not formalized yet, but it will be before the end of 2019.

Available data is limited but this is an interesting case because the LFM activity is key to the realisation of a high-speed railroad and it includes only the removal of landfilled waste along the railway line (about 50.000 m^3). The dump was in operation from 1987 to 2009, regularly authorised. LFM will concern more or less 50% of the total volume of landfill. The portions not affected by the removal will be

FIGURE 2: Rendering of the future configuration.

TABLE 1: Characteristics of the landfill mining activity.

Amount of removed waste	150.000 m³	Partial removal
EWC European waste code	100202	Untreated slag
Recovery Operations	D13	Blending or mixing prior to submission to any of the operations numbered D1 to D12
Days of work	110	0.5
Treated amount per day	2570 t	43.6

TABLE 2: Characteristics of the landfill mining activity.

Amount of removed waste	50.000 m³	Partial removal
EWC European waste code	100202	Untreated slag
Recovery Operations	D13	Blending or mixing prior to submission to any of the operations numbered D1 to D12

made safe with a barrier of containment poles, 15 m deep and 120 m long.

Landfilled waste is slag from steelworks, mixed with other waste, so the recovery cost is too high according to the benefit coming from the sale of products. Furthermore, it is difficult to find a market of this kind of waste; so, it will be sent to another dump.

Environmental investigations carried out in 2014-15 analysed groundwater, upstream and downstream, and soil down and up to waste.

The surveys showed light contamination of soil (Pb, Zn, Cr, hydrocarbons) and a new campaign will be organised in the next few months.

The Ministry of Transport is responsible to authorise the permit of the railroad, while the LFM activity is a Lombardy Region's responsibility (Table 2).

2.3 Municipality of Sermide

Landfill is located far from urban area, near a municipal waste collecting plant. The Municipality is the owner of the plant and of the landfill and needs to increase the waste collecting area (Figure 3 and Figure 4).

In operation since 1996 to 2002, the landfill was divided in two parts: 7000 m^3 of inert waste have been landfilled in the part A, and 740 m^3 in the part B. The maximum thickness is 4.5 m, there is no waterproofing system, nor a system for collecting and removal leachate and biogas. In 2015 an environmental investigation analysed ground water, upstream and downstream, and ground down and up to waste.

Investigation did not show contamination of environmental matrices and brought out that waste was for 80% gravel and crushed rocks with 20% of plastics and inert waste (construction and demolition waste).

The project includes a shredder mobile plant: after the separation of different waste, plastic and construction and demolition waste will be sent to recovery plants, gravel and crushed rocks will be shredded and used in site for the re-profiling of the area or sent to recovery plants (Figure 5).

2.3.1 Project phases

There are 4 sampling points to be performed on the ground below the mounds, 3 of which are below Cumulus A and one below Cumulus B. Three stages of work (phases) will be made for pile A and 1 for pile B. For each stage, the work will be organized with the following steps:

- excavation of the material;
- visual verification of the presence of foreign coarse fractions of a non-inert nature (plastics, wood, glass, metals, etc.); if found, these will be removed manually

FIGURE 3: View of the dump and the plant.

FIGURE 4: Landfill (in orange) and municipal collecting plant (in blue)

FIGURE 5: The landfill area before LFM intervention.

by the operators or with the assistance of a supplied mechanical tools (excavator bucket); this waste will then be identified and managed under a temporary storage regime, with storage in heaps and delivered to authorized facilities for subsequent recovery/disposal;

- storage waiting for an optimal quantity for processing;
- mechanical treatment by crushing in a plant with a mobile crusher;
- sampling of the material obtained from the treatment for compliance verification (this check will be performed every 3,000 m^3 of material produced);
- excavation of trench in the natural soil at the bottom, up to a depth of 1.5 meters below ground level and collection of 1 representative sample to be analyzed;
- once the ground has been declared free of contamination and the material leaving the treatment has been deemed compliant, the material will be laid out in the area of the lot considered. Excavation and internal handling of the material will be carried out by mechanical shovel/hydraulic excavator.

The shredder mobile plant will perform the volumetric reduction of large brown elements and it will allow the homogenization between the different types of inert materials and the separation of the pieces of iron. The machine is equipped with a powder control system which uses nozzle nebulizers which are positioned at the mouth and at the discharge of the mill; these nebulizers deposit very fine water particles on the material. From the process described above, a recycled material is obtained, free of iron parts and other non-inert fractions, of granulometry 0-40 mm: this recycled material will then have to be verified to ascertain its qualification of end of waste. As there is a lack of information about its nature and origin, landfilled waste will be classified with EWC 17 09 04, mixed construction and demolition waste; in any case it is of solid, non-pulverulent, inert waste (presumably concrete, bricks, tiles, stones, plasters, mortars, and the like) not odorous and non-putrescible with possible amounts of other materials such as wood, plastic, glass, rubber, etc. (Figure 6, 7 and 8).

Recovery Operations is R5 "Recycling/reclamation of other inorganic materials" for 13.800 t of non-hazardous waste (Table 3).

2.4 Erba

This case considers a dump for construction and demolition waste, for a total volume of 38.000 m^3, managed by a public administration in the '90, regularly authorised by Lombardy Region and monitored by Provincia di Como. The project includes the transformation of an area of 57.000 m^2 in industrial/commercial buildings (Figure 9 and Figure 10).

In 2012-2014 the private owner of the area executed an environmental investigation ; the investigation concerned an area used as a landfill (23.000 m^2) and another green portion next to it (35.000 m^2). The project includes the transformation of the entire zone in industrial/commercial buildings.

The green area has never been used, the other portion was a gravel pit in the '60, a motocross track from 1970 to 1989, a landfill from 1993 to 1996. The landfill area is in banking, on the contrary the other portion is in the depression; this means that a morphologic rearrangement is needed.

FIGURE 6: The landfill area before LFM intervention.

FIGURE 7: The shredder and the different grounds sorted.

FIGURE 8: The area at the end of LFM

The Investigations which ended in 2014, concerned:

- cartographic, documental and historical survey;
- geophysics survey, that showed 2 gas pipelines and an old water pipeline;
- hydrogeological survey;
- plano-altimetric test;
- geognostic survey.

This led to a conceptual model of the area.

The Investigations also brought to the verification of the characteristics and quality of:

- the groundwater, downstream and upstream;
- ground at the bottom and around the landfill;
- waste and capping.

No contamination of environmental matrices was found (Figure 11).

The perimeter of the area of intervention is about 24.000 m^2, the volume of landfilled waste 38.000 m^3 with a height, from the ground level, of 4,3 m maximum and 2,8

TABLE 3: Characteristics of the landfill mining activity.

Amount of waste	7730 m^3; 13.800 t	Complete removal
EWC European waste code	170904	Mixed construction and demolition waste other than those mentioned in 17 09 01, 17 09 02 and 17 09 03
Recovery Operations	R5	Recycling/reclamation of other inorganic materials
Days of work	75	0.5
Treated amount per day	940 m^3; 1680 t	43.6
Estimated recovered material	9000 t	

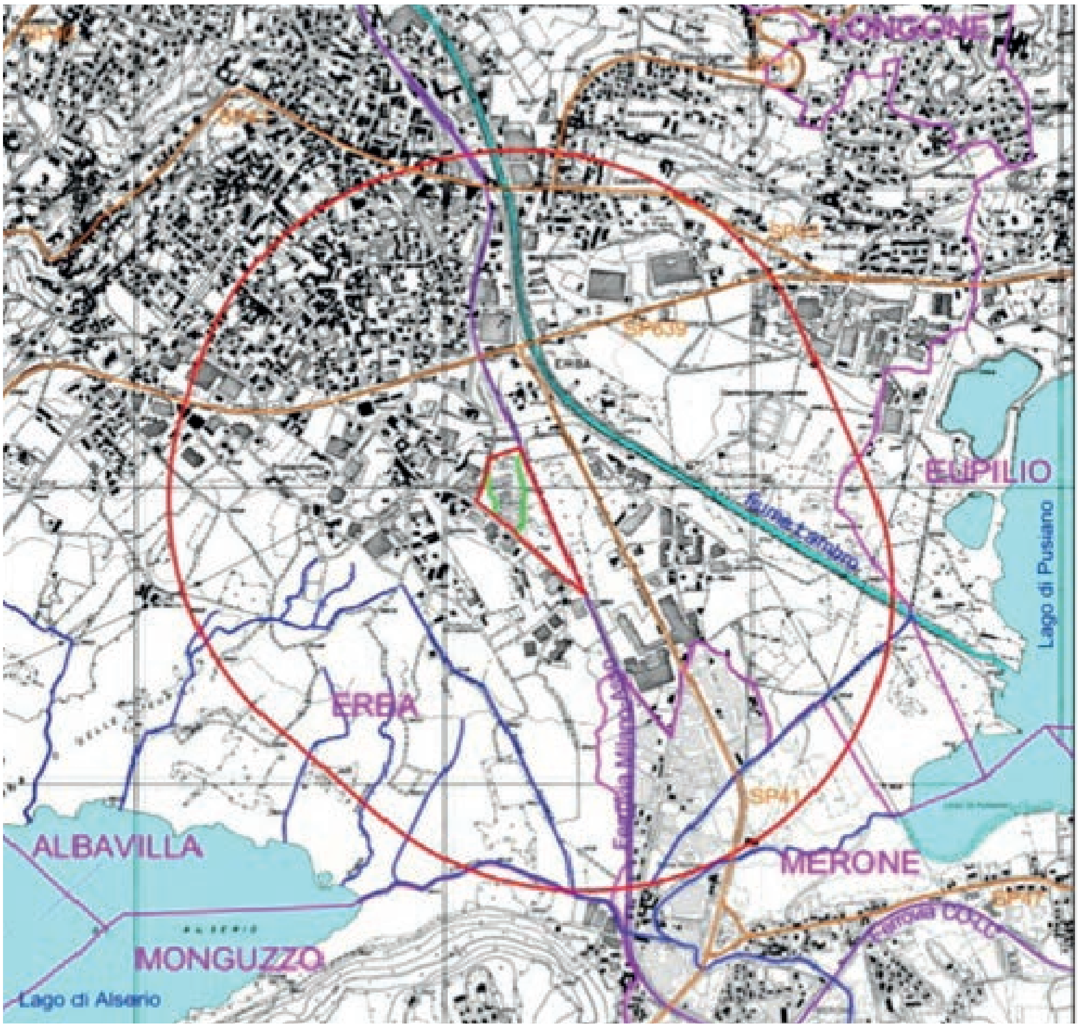

FIGURE 9: Localisation of intervention (in green the landfill).

FIGURE 10: Rendering of the future configuration.

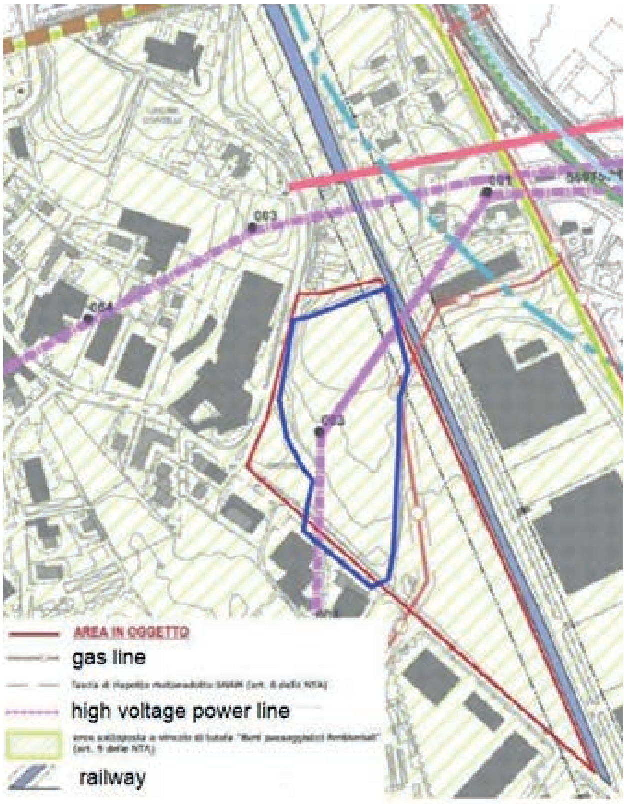

FIGURE 11: The area of intervention (in red), the landfill (in blue) and the facilities.

of average. The minimum quota of waste is at 267,00 m a.s.l. and the groundwater is at average 265,5 m a.s.l., with a seasonal fluctuation of +/- 3.5 m. Before the disposal of waste, the ground has been regularised and the slice of depression have been covered with 30 cm of clay.

The capping is realised in 30 cm of clay and 50/70 cm of topsoil.

Characteristic of landfilled waste found in management documents were verified in site: the total amount of waste is construction and demolition materials.

The project includes a shredder mobile plant to decrease dimensions of waste and leads to the use of materials for the re-profiling of the area.

The Recovery Operation is R5 “Recycling/reclamation of other inorganic materials” for 30.000 t of non-hazardous waste, the amount needed for the morphological reconfiguration; 8.000 t will be sent off-site.

The treatment of 38.000 t needs an environmental impact assessment, which is responsibility of the Lombardy Region (Table 4).

The analysis of available time and space brought to the solution of dividing the area of intervention in portions; waste recovered by the shredder will be used for filling depressed areas. During these activities the mobile plants will move in 3 different positions.

2.4.1 Project phases

- removal of the cover layer (50/70 cm of cultivated soil and 30 cm of clay) of the landfill and reuse of a part for filling portions of natural soil area. Before the excavation operations start, a chemical – physical analysis will be performed by taking samples of both the culture soil and the clay layer; this analysis is aimed at providing the characteristics of these materials for a proper delivery to a different site for a portion and for the reutilisation project of the remaining portion . The recovered material will be used for the morphological reconfiguration of the entire area of transformation, starting from the depressed zones, leveling the entire area from the existing road surface.

Once the installation of the construction site and the displacement of the site portions of the “natural soil” area, FASE 0 will be completed with the realization of a new piezometer “PZM-2018” and contextual sealing of the piezometer “PZM-1994” at the centre of the body of the landfill (which dates back to 1994): such actions are needed because subsequent landing activities will necessarily result in the demolition of the existing PZM function. The piezometer made during the Preliminary Environmental Survey further south in 2013 (PZV-2013) will instead provide representative samples of the first groundwater downstream of the perimeter of the landfill, the monitoring will be on a monthly basis for the entire duration of the recovery activities;

- removal and recovery of the inert material that constitutes the body of the landfill and reuse of the recycled aggregate obtained to fill portions of natural soil area. Before the on-site recovery treatment, chemical-physical analyses of waste will be carried out, and at the end of the mechanical recovery process (screening and crushing), the material will be further analyzed in batches from a product and chemical-physical point of view in order to qualify the recycled product aggregate that, once certified, will be re-used to continue

TABLE 4: Characteristics of the landfill mining activity.

Amount of waste	38.000 m^3; 68.400 t	Complete removal
EWC European waste code	170904	Mixed construction and demolition waste other than those mentioned in 17 09 01, 17 09 02 and 17 09 03
Recovery Operations	R5	Recycling/reclamation of other inorganic materials
Days of work	63	0.5
Treated amount per day	1250 m^3; 2250 t maximum	43.6
475 m^3; 855 t average		
Estimated recovered material	30.000 m^3	

morphological reconfiguration of the entire area of intervention. Recycled inert aggregates, will be obtained from the on-site processing operations, this will need to be compliant with Italian law (Annex C4 of the Circular UL/2005/5205, which is related to the production of a "recycled aggregate for the realization of environmental recoveries and fills"). Especially a leaching test will verify the grain size and the composition of the material, its fractions and its chemical compatibility;
- chemical and physical analysis on natural soil under the landfill and start of the morphological reconfiguration of the ex-landfill portion by re-using the recycled aggregate obtained from the recovery cycle. An Environmental survey of soils below ex-landfill (bottom check) is necessary in order to confirm non-contamination, thus certifying the recovery of the area which was until then a former landfill of inerts. In the "natural soil" area there is a gas pipeline owned by SNAM's gas network; the pipe trunk, placed at a depth of about 1.5 m from the campaign plan, is contained and protected by a concrete artifact that allows operation of the area with these constraints: the possibility of work being undertaken at no less than 2 m distance from piles and fixed installations.

The morphological reconfiguration project planned to achieve a level of - 50 cm from the finished floor; this will be achieved using 1/3 of the recovered materials (about 10,000 mc of the 30,000 mc produced), some of this material will also be used for the construction of part of the road in the new viability design. The project will continue with the spread and compaction of 30/40 cm layers of the aggregate recycled with the use of mechanical shovel to be followed by rolling operations and the testing of the compaction with daily density tests and load tests with plate at the end of each layer. Based on the results of these tests and on the basis of the size and geological records of the material, an assessment will verify the need to improve the material's mechanical properties of the soil by mixing the soil with binders (lime and/or cement) - Figure 12.

3. CONCLUSIONS

The experiences described in this article show an interest versus LFM in case of interventions characterised by:

- limited extension and volumes;
- non-hazardous waste;
- possibility to convert landfill area in other uses.

The interest for the recovering of these areas is in part due to the limitation of using virgin soil established by Lombardy Region in 2014 (regional law n. 31); this added an extra cost for edification in agricultural area and a reduction for interventions in degraded areas.

Sometimes – as in the example of point 2.2 - LFM is not a choice, but it is a need to complete a project (the construction of new infrastructure or new urban complex). In this situation the strategic value of infrastructure is so high to justify the cost of LFM intervention, even in case of contamination of environmental matrices.

The recovery of materials is, at the moment, limited by the cost, which is too high compared to the value of landfilled waste.

These costs include:

- surveys to define the contest (characteristic and quality of groundwater, soil, waste), often difficult and expensive also because of the old age of landfills;
- activities for ensuring safe excavation;
- treatment (mechanical, chemical, thermal,..);
- disposal of residual waste.

In Italy there is also an additional problem created by the difficulty for the authorities to certificate end of waste (EoW) on a 'case by case' basis without national criteria for evaluation; these have been in the Italian Government's programme since 2006 but they have not been finalised yet except for a few specific kind of waste and processes.

This creates a distrust for EoW with a resulting difficulty in finding a market.

FIGURE 12: The area at the end of LFM.

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RESEARCH TO INDUSTRY AND INDUSTRY TO RESEARCH

RETHINKING THE ENVIRONMENTAL CRITERIUM FOR THE END-OF-WASTE STATUS OF INERT WASTE

Facilitating waste recycling is crucial to promote materials circularity and reach the objectives established by the EU Directive on Circular Economy (European Parliament and European Council, 2018).

This is particularly important for the management of non-hazardous inert waste, which should be aimed at the recovery of bulk mineral resources, recently defined as "recovered aggregates", to be used as secondary raw material by the civil engineering sector. Hence, the term "recovered aggregates" includes both recycled and manufactured aggregates as defined by EN 12620 and by EN 13242. To understand the scale of the issue, the 2016 major mineral waste (i.e. non-hazardous waste from construction and demolition, foundry slags and residues from the incineration of municipal solid waste) production was estimated in almost 1.6 billion tons within the EU28 group, corresponding to a pro-capita share of about 3 tons per EU citizen and per year (Eurostat 2017 and 2019).

In this context, adopting sustainable End-of-Waste (EoW) criteria, which legally establish when recovered materials cease to be considered as waste and obtain the status of marketable products, is a fundamental step to allow circular reuse of recovered aggregates while reducing the amount of inert waste to be landfilled.

Within the five generic requirements laid down by the European Waste Framework Directive (European Parliament and European Council, 2008a), the compliance assessment of the so-called environmental criterium ("Will the use of the material lead to overall adverse environmental or human-health impacts?") still represents a regulatory issue potentially hindering the stakeholders (both producers and controllers) involved in the waste recovery sector.

As a general trend, EU Member States regulations establish that environmental impacts of recovered material must be assessed through the comparison of results from chemical analyses, performed on the recovered material itself or on water extracts obtained through conventional leaching tests (e.g. standards from the EN 12457 series), with fixed concentration limits (CL). In Italy, this approach was originally introduced by a ministerial decree regulating the so-called "simplified procedures" for defined categories of waste recovery processes (Decreto Ministeriale 5-02-1998). This decree regulated the achievement of EoW status for recovered materials according to proven compliance with a list of fixed CL for several chemical-physical parameters, which must be measured in a waste eluate derived through a leaching test performed according to the EN 12457-2. The same approach (with identical CL) is included in the Italian EoW regulation laid down specifically for the so-called "bituminous conglomerates", recovered from asphalt waste (Decreto Ministeriale 28-03-2018).

Lately, several issues have been highlighted by the involved parts on the high degree of protection characterizing the aforementioned approach. The use of a set of conservative CL reflected the approach used by the regulator to set simplified waste recovery procedures. In fact, these latter do not require full and ordinary authorization processes but just simple communication to the intended authority of the undergoing waste treatment activity.

Besides, in these last years, questions such as the definition of more realistic CL in the field of aggregate applications, the necessity of defining approaches able to provide information on the real bioavailability of all the chemicals of the tested material, etc. have been raised to better clarify this technical and regulatory framework.

In this context, a scientific working group was established in Italia, aimed at proposing a protocol to update current technical procedures approaches, involving the Veneto technical round-table on circular economy applied to infrastructures, and independent experts. After several meetings, preliminary results were presented during the technical seminary "Construction and demolition waste" held on 27-02-2019 in Padua and the workshop on "Environmental criteria for the achievement of end-of-waste status for inert waste" held on 6-11-2019 in Rimini, during the "Ecomondo", the green technology expo.

In this document, the scientific committee wants to summarize the principles underlying the protocol, addressing the issue of environmental assessment of EoW criteria for inert waste.

- The working group addresses the need to provide a scientifically-sound protocol, which will allow the involved stakeholders (producers and controllers) to verify the compliance of environmental criteria to obtain EoW status for the so-called "recovered aggregates", derived from inert waste treatment. Both analytical protocol and concentration limits will be proposed according to the professional experience of the involved stakeholders. The aim is to update the current Italian CL and procedures (Decreto Ministeriale 5-02-1998), too conservative for certain parameters (e.g. sulfates and chemical oxygen demand).
- According to the Waste Framework Directive 98/2008/EC (European Parliament and European Council, 2008a), both the protocol and the EoW criteria will be

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based on the general principles of technical feasibility, economic viability and environmental protection.

- The protocol should be detailed enough to be fully adopted as a Ministerial Decree or as an authorization case-by-case for waste treatment plant operations. For this reason, the final document must define both environmental criteria to be met to obtain the EoW status and technical instructions, laying down precise requirements for sampling, transport, storage and sample preparation, together with technical guidelines for required analyses and criteria for analytical results interpretation.
- Both protocol and criteria will be developed specifically for recovered aggregates when applied as unbound materials, i.e. not applied in bound-like applications containing binders (e.g. hydraulically bound materials). When recovered aggregates are used in bound application, the protocol should be further developed considering i) the role of specific binder used, ii) alternative leaching test (i.e. monolithic leaching test) and iii) relative specific CL considering the peculiar features of this kind of applications.
- According to the general requirement regulating EoW criteria, the proposed protocol should allow the assessment of the overall environmental impact, due to solid and leachable fractions of materials constituents and the related different exposure pathways. For this reason, the protocol must include chemical analyses (and eventually ecotoxicological) both on solid samples and on its water extract (derived according to EN 12457-2, or EN 12457-4). The established CL for solid samples will be related to a limited set of parameters, considered significative for this kind of recovered materials.
- The proposed protocol will include chemical-physical relevant analyses on solid samples and leachates, with the possibility to conduct ecotoxicological tests, whether the results from chemical characterization does not comply with the CL necessary to obtain EoW status. In this latter case, results derived from bioassays must prevail, in accordance with the regulation on ecotoxicity classification of waste (Regulation 997/2017/EC, European Council. 2017).
- Ecotoxicological assessment will be based on the following test battery, according to the proposal of Pandard and Römbke, 2013: terrestrial bacteria (Arthrobacter globiformis), plants (Brassica rapa) and terrestrial invertebrate decomposers (Eisena fetida) for solid samples; aquatic bacteria (Vibrio Fischeri), freshwater algae (Pseudokirchneriella subcapita), freshwater crustaceans (Daphnia magna) for water extracts. The experimental results expressed as effect concentration (EC50) will be assessed though the comparison with proposed ecotoxicological concentration limits (EC50,L).
- The proposed methodology will be based, for each analytical phase, on technical guidelines currently established for the characterization of waste material. In fact, the material not showing compliance with all the EoW criteria will remain a waste. Procedures aimed at labeling and classification of products and substances will be evaluated if they are not less precautionary than the waste legislation.
- Consequently, the analytical procedures laid down for ecotoxicological EoW assessment are independent from European regulation establishing criteria for products classification and labeling (CLP 2008/1272/EC, European Parliament and European Council. 2008b), in accordance with Waste Framework Directive 2008/98/EC amended by Directive 2008/851/EC (European Parliament and European Council, 2008, European Parliament and European Council, 2018). Indeed, article 6 of the WFD currently states that EoW criteria must be met prior to the application of the regulation on substances and products. The ecotoxicological classification methods of the CLP can lead to a less cautious evaluation if compared to the results derived from the analytical methods used for the ecotoxicological characterization of the waste HP 14 according to the approach described in Hennebert (2018) and Pivato (2019). As an extreme consequence, a recovered aggregate considered compliant with EoW criteria established in accordance with the CLP methods, could at the same time be classified as hazardous waste according to HP 14 classification method.
- Interpretation criteria for experimental results (chemical-physical characterization of solid samples and water extracts and eventual ecotoxicological characterization) are listed in following Table 1.

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TABLE 1: Proposed environmental criteria for obtaining the "End-of-waste" status for recovered aggregates. C = Concentration measured, specifically for each required parameter, in solid samples and/or water extracts. C = Limit concentration, established for each parameter as required by the chemical characterization of solid samples and water extracts. EC50 = Concentration expected to produce 50% of the effect measured in each considered bioassay. EC50,L = Limit concentration producing 50% of the effect measured in each considered bioassay.

Chemical characterization of solid sample	Chemical characterization of water extract	Ecotox characterization of solid sample	Ecotox characterization of water extract	OUTCOME
Positive (C < CL)	Positive (C < CL)	Not necessary (-)	Not necessary (-)	Compliant with EoW environmental criteria
Negative (C ≥ CL)	Positive (C < CL)	Positive (EC50 ≥ EC50,L)	Not necessary (-)	Compliant with EoW environmental criteria
Negative (C ≥ CL)	Positive (C < CL)	Negative (EC50 < EC50,L)	Not necessary	Waste status
Positive (C < CL)	Negative (C ≥ CL)	Not necessary (-)	Positive (EC50 ≥ EC50,L)	Compliant with EoW environmental criteria
Positive (C < CL)	Negative (C ≥ CL)	Not necessary (-)	Negative (EC50 < EC50,L)	Waste status
Negative (C ≥ CL)	Negative (C ≥ CL)	Positive (EC50 < EC50,L)	Positive (EC50 ≥ EC50,L)	Compliant with EoW environmental criteria
Negative (C ≥ CL)	Negative (C ≥ CL)	Negative (EC50 ≤ EC50,L)	Positive (EC50 ≥ EC50,L)	Waste status
Negative (C ≥ CL)	Negative (C ≥ CL)	Positive (EC50 ≥ EC50,L)	Negative (EC50 < EC50,L)	Waste status
Negative (C ≥ CL)	Negative (C ≥ CL)	Negative (EC50 < EC50,L)	Negative (EC50 < EC50,L)	Waste status

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CISA

NEW PROJECTS

	Name	**BIO-PLASTICS EUROPE**
	Partners	Hamburg University of Applied Sciences (HAW) - Germany
	Funding scheme	Horizon 2020
	Project duration	Four years (2019 – 2023)
	Principal Investigators	Hamburg University of Applied Sciences (HAW) - Germany. Research institutions, universities and companies from 12 EU countries (Austria, Belgium, Estonia, Finland, France, Germany, Italy, Lithuania, Poland, Sweden, Spain and the UK) and Malaysia.
	Website	https://bioplasticseurope.eu

BIO-PLASTICS EUROPE

BIO-PLASTICS EUROPE is a project funded by Horizon 2020, an EU Research and Innovation Program aiming to ensure Europe's international competitiveness. The main objective of the project is "The development of sustainable strategies and solutions for bio-based plastic products, as well as the development of approaches focused on circular innovation for the whole bio-plastics system. These may be deployed to support policy-making, innovation and technology transfer". With a budget of around 8.4 million Euro, the 4-year project started in October 2019.

BIO-PLASTICS EUROPE is coordinated by Hamburg University of Applied Sciences (HAW) in Germany. Research institutions, universities and companies from 12 EU countries (Austria, Belgium, Estonia, Finland, France, Germany, Italy, Lithuania, Poland, Sweden, Spain and the UK) and Malaysia form the core members of the project. The network partners include companies and NGOs such as TetraPak, Unilever and the Ellen MacArthur Foundation. In addition, more than 15 network cities such as Hamburg (Germany), Manchester (UK) and Aveiro (Portugal) are in the loop, showing interest to implement solutions at sub-national level.

The project tasks cover critical elements over the entire lifecycle of a bioplastic product. They include ethical considerations, design and production of bioplastic prototypes, laboratory and field tests, evaluation of the potential impacts of bioplastics on the current waste management system, safety and environmental assessments, policy assessment as well as business model development.

Moreover, the project aims to foster strategic networking across Europe to enable fast dissemination of ideas, solutions, and leverage synergies. The results would enable bioplastic value chains to become more circular, resource efficient with a decreased carbon footprint.

For further and more detailed information please contact the project coordinator (HAW Hamburg, Faculty Life Sciences, Franziska Wolf, Managing Director "European School of Sustainability Science and Research" (ESSSR), franziska.wolf@haw-hamburg.de).

A webpage is under construction (https://bioplasticseurope.eu/).

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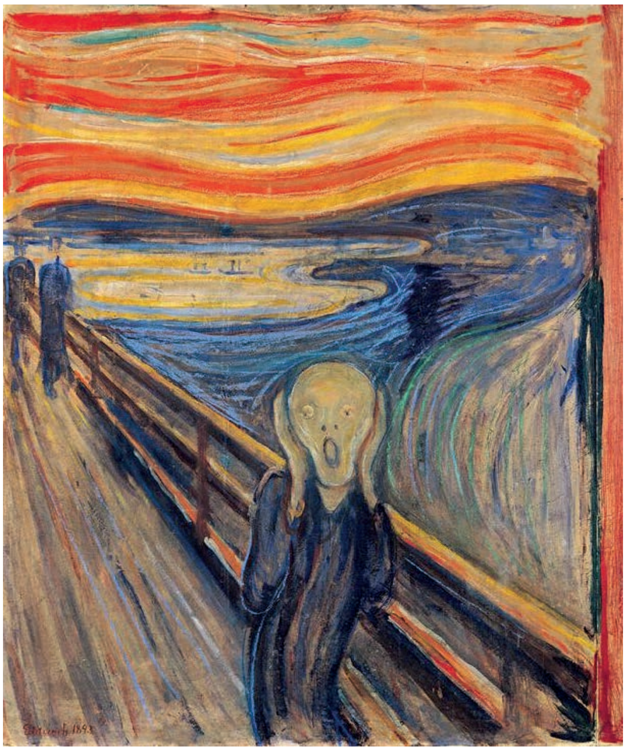
DETRITUS & ART / A personal point of view on Environment and Art

by Rainer Stegmann

Artists seldom provide an interpretation of their own work; they leave this to the observer. Indeed, each of us will have our own individual view of a specific piece of art, seeing different contents and experiencing a range of feelings and emotions. Bearing this in mind, I created this page where you will find regularly a selected work of art from different epochs and I express my thoughts on what the work conveys to me personally. The interpretation should be related specifically to the environment and what kind of message I deduct from it: it may be to preserve the beauty and integrity of nature, to prevent destruction, issue warnings, etc. Any comments or suggestions regarding this column should be addressed to stegmann@tuhh.de



EDVARD MUNCH / THE SCREAM - 1983, National Gallery and Munch Museum, Oslo, Norway

There are different versions with this theme: four paintings and one black and white lithography. I selected this amazing colourful version because for me it is especially expressive. There is a person screaming, pressing the hands over the ears. A bold head with open mouth and torn eyes intensifies the scream. Obviously we cannot hear the scream but in my view he made the scream visible by using this winding lines as if the screams sets the environment - the outer world - in motion. The red sky strengthens this impression by its swinging lines and intensive colours. It looks to me as if the land is forced back by the in streaming water. The expression of the face shows fear, which is the reason for this intensive screaming. The winding lines of the person and the landscape flow into each other making the person part of nature; both are in turmoil.

An obvious relation to our situation on earth today is fear about the future of our planet, the imbalance of the natural environment, the lack of sustainable living and acting. The person shouts this fear out into the world.

The bridge is straight, no movement; the two persons in the background on the bridge are walking up right and do not seem to be influenced by the scream, the environment in uproar. We may interpret this that these people do not or do not want to hear the scream, they may not see the problematic or better alarming situation of the environment. For me this wonderful painting depicts our situation on earth today as there are many people that see the global change and others are ignoring it. But may be we can also believe in another interpretation, where we see the two persons going straight forward in order to act, as a kind of reaction to the scream. Let us be optimistic and believe in the latter interpretation and see the two persons as symbols of hope.

Edvard Munch was a famous Norwegian Painter and Graphic Artist. He was born in Loten, Norway on 12.12.1863 and died in the age of 80 in Oslo, Norway. He was a forerunner of Expressionism and is rated today as one of the main expressionists worldwide. He was beyond others influenced by Vincent van Gogh, Paul Gauguin and Henri Toulouse Lautrec. He produced more than 1700 paintings and many graphics. "The Scream" is one of his most famous paintings.

CASPAR DAVID FRIEDRICH
WANDERER ABOVE THE SEA OF FOG

In the next issue I will present a famous painting by Caspar David Friedrich (Wanderer über dem Nebelmeer, painted around 1817).

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