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Detritus – Multidisciplinary Journal for Waste Resources and Residues – is aimed at extending the "waste" concept by opening up the field to other waste-related disciplines (e.g. earth science, applied microbiology, environmental science, architecture, art, law, etc.) welcoming strategic, review and opinion papers.
Detritus is an official journal of IWWG (International Waste Working Group), a non-profit organisation established in 2002 to serve as a forum for the scientific and professional community and to respond to a need for the international promotion and dissemination of new developments in the waste management industry.

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Editorial

LANDFILLING OR BIKING?

The first day of the Solid Waste Management course I teach at the University of Padua, Italy, I usually ask the students a simple question:

"In your opinion, which is the best system of waste management?"

Their immediate reply, in chorus and with very few exceptions, is: "Recycling!"

I then go on to the second question.

"In your opinion, which is the best means of transport?"

After a moment of bewilderment due to this abrupt jump from one topic to another, no response is forthcoming until the first brave person breaks the ice: "It depends!" I ask for an explanation and immediately receive a clear, logical justification. Everyone agrees that it is reasonable to think that to go from Paris to Beijing by car or by train may be nostalgically romantic, or at least adventurous, but it is much more practical to fly there. On the contrary, it would, to say the least, be totally absurd, or even grotesque, to think of catching a plane to go to buy the bread!

We further developed the discussion by mentioning how, for reasons of practicality, we rarely use the same means of transport all the time. We walk to the garage to get the car. We use the car to get to the airport. Here we board a plane and, on leaving the plane we may indeed catch a train or take a boat, or ….

I then ask my third question:

"Why do you reply «it depends» when referring to means of transport, taking into account your requirements at the time and the context in which you are placed, whilst giving such a decisive answer when focusing on waste management?"

I then go on to ask a fourth question: "Why did it not occur to you to combine the different methods of waste treatment and disposal?" Indeed, in the same way that means of transport are largely diverse, the field of waste management affords a series of well-differentiated options, as follows:

- minimization of waste generation;
- recovery and recirculation of material resources present in the wastes;
- combustion of waste with the main aim of reducing waste volumes;
- landfilling of residual wastes in order to close the material loop.

Of course, these diverse options are applied using different and increasing levels of technology, advanced and not so advanced, sophisticated or less sophisticated, efficient and less efficient, both reliable and less reliable. It is however undeniable that, in the same way as transportation, and particularly in view of the knowledge we possess, these solutions must be combined and integrated. On analyzing the disposal techniques adopted worldwide, it is clear that countries characterized by a high population density (e.g. Japan, Singapore, Denmark, Germany, etc.) benefit enormously from the use of incineration combined with intense programs for the recovery of material resources (sorting, recycling, biological treatment) and with landfilling of residual wastes (Cossu, 2009). At the same time, countries with a low population density prefer to combine waste recycling with landfilling (e.g. Canada, United States, etc.).

Why then do my students fail to implement the obvious concepts that we rationally apply to the transportation of people and goods to the systems of waste management?

The response obtained from my students is indeed aligned with the response of the majority of the population (Figure 1): "because incinerators and landfills pollute the environment and endanger our health!".

We are well aware that all waste management technologies, ranging from composting to landfill, and from mechanical treatment to incineration, are characterized by the emission of contaminants that should be, wherever possible, prevented and rigorously monitored. Even the recovery and recirculation of waste materials results in the accumulation of a series of contaminants contained in the materials, and are certainly not devoid of negative environmental impacts.

However, on taking a closer look, the various means of transport do not fare too well in the field of emission of contaminants and environmental risks.

Over recent decades the use of planes has risen exponentially thanks to the diffusion of low-cost airlines, but this form of transport is characterized by high emissions of noise, greenhouse gases, (NO_x and CO_2), and risks linked to cosmic radiations.

Maritime transport likewise produces both atmospheric pollution through the emission of carbon dioxide, sulfur dioxide, nitrogen oxides and particulate due to the use of poorly refined fuel oils, and water pollution through leakages of oils, solid wastes and liquids. An increase in the number of cruise ships may indeed upset the equilibrium in delicate and sensitive ecosystems (Artic, Antarctic, coral reefs, lagoons, etc.).

Even trains, the means of transport operationally linked to production of the lowest amounts of greenhouse gases,

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FIGURE 1: An example of the reaction of the population to waste management issues. (Courtesy of Vertigogen, from Flickr).

on taking into account all the infrastructures associated with operations (tracks, stations, etc.), prove to be even less competitive than planes in a detailed life cycle analysis (Chester and Horvat, 2009). There is little point in referring to cars, being the most highly targeted method of transport from both an environmental and political viewpoint.

On considering the risks involved, a particularly alarming situation is manifested for all methods of transport. Unfortunately, planes at times may crash, ships may sink, trains may derail and road accidents destroy cars, bicycles and pedestrians ….

Of course the available technologies are in a position to strongly attenuate the pollution and risks lined to means of transport, but the same is true also for waste management systems.

Why therefore do my students, and the public opinion in general, provide such different answers?

It is because methods of transport are not bombarded by defamatory campaigns comprising aggressive misinformation and by political manipulation which, on the contrary, have massively impinged on the field of waste management on a communicative and regulatory level.

It is easy to find how landfilling and incineration, without any evidence or scientific documentation, have been associated to cancer (gastrointestinal, esophageal, lung, stomach, colon, rectal …), birth defects and reproductive disorders (low birth weight, fetal and infant mortality, spontaneous abortion, malformations…) and other problems (cardiovascular diseases, respiratory symptoms, asthma, reduced lung function, irritation of the skin, nose and eyes, gastrointestinal problems, fatigue, headaches, psychological problems, allergies …).

This fake news has created dangerous prejudices and scaremongering that has given rise to paradoxical solutions and diseconomies. The example of several Italian cities where waste management has been rather improperly associated with the systems of transportation is well known! Indeed, numerous wastes produced in cities where the local administrations had rejected incineration and landfills in favor of recycling, have actually been loaded onto trains, ships and lorries to be disposed of abroad, after a long journey. How? By means of incineration and landfilling!

Hurrah! Let's get ready to go biking to Beijing!!!

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ANALYSIS OF SEWAGE SLUDGE THERMAL TREATMENT METHODS IN THE CONTEXT OF CIRCULAR ECONOMY

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ABSTRACT

As of now, the most common applications of sewage sludge treatment and disposal methods globally are in agriculture and deposition in landfills. In particular, landfill disposal causes problems associated with environmental pollution, as well as problems caused by the loss of the chance to recover energy and nutrients out of the sewage sludge. The critical content of hazardous substances in the sewage sludge makes its use in agriculture as fertilizer questionable. Thermal treatment methods offer a solution to these problems because energy can be recovered and used, some hazardous materials can be destroyed or removed, and valuable nutrients such as phosphorus can be utilized in the generated products or recovered from these products. In a first step, the objective and scope of the study and especially the important characteristics of the circular economy when considering sewage sludge treatment possibilities is described. Based on these characteristics for the three investigated thermal treatment methods – incineration, gasification and pyrolysis – a comparative analysis on the basis of a suggested set of criteria (1. cost, 2. energy efficiency, 3. nutrient recovery, 4. product market value and 5. flexibility) was carried out. In the result, incineration of sewage sludge performs best in terms of treatment costs, energy efficiency, nutrient recovery, and flexibility concerning feedstock dry matter content. Pyrolysis performs best in terms of market value of the generated products and flexibility regarding plant size.

1. INTRODUCTION

Due to the rapid development of the pharmaceutical industry and the increasing consumption of emerging contaminants such as personal care products, the composition of pollutants contained in municipal wastewater has changed over the last years. The composition now includes organic compounds, oil products, suspended particles, heavy metals, pathogenic substances and chemical contaminants. At the same time, conventional wastewater treatment methods have remained unchanged for decades in many cases. Mechanical and biological treatment methods are most often used for cleaning municipal wastewater, since they are most effective for removing the above-mentioned contaminants. The conventional municipal wastewater treatment scheme leads to the generation of a large amount of paste-like waste — so-called wastewater sludge or sewage sludge.

Statistical data collected all over the world shows that in general the amount of sewage sludge strongly depends on the total population of a country as the specific sewage sludge production per person per year is more or less stable at the level of 25 kg of dry substance (UN-HABITAT, 2008). Thus, the expanding population in a number of developing countries such as China and India results in yearly increases in sewage sludge volume. At the same time, higher-income countries may continuously improve infrastructure and wastewater treatment technologies, producing increasingly larger masses of sewage sludge per country. Urbanization processes that occur in both developing and developed countries lead to increases in the volumes of water used in municipal economies; these processes also result in a rise in total sewage sludge production.

Sewage sludge is a large-tonnage waste with some specific characteristics caused by its high water content, which hampers its final disposal. Sewage sludge contains considerable amounts of nutrients, such as nitrogen and phosphorus, but also a number of harmful contaminants, e.g. inorganic pollutants: toxic heavy metals such as lead, mercury, cadmium, copper and uranium; as well as organic toxins such as dioxin, polychlorinated biphenyls, perfluorinated surfactants, pharmaceutical residues, pathogens and others (Erikson et al., 2007) that limit its application as a fertilizer. Sewage sludge has high water content, and

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the moisture it contains is presented in several different forms, including free water, pore water and colloid moisture (bound water) (Deng et al., 2011). The latter in particular makes sewage sludge difficult to dewater. Dry solid content of sewage sludge after a wastewater treatment plant is only 1-3%, then the thickening process enlarges this parameter to 3-5%, and mechanical dewatering results in a maximum of 20-30% dry solid content. Only with the use of thermal drying can sewage sludge dry solid content be increased to approximately 92% (Stasta et al., 2006; Uggetti et al., 2009).

Until now, the most commonly used sewage sludge treatment and disposal methods globally are its use in agriculture after the thickening process, production and use of compost after mechanical dewatering, and deposition in landfills (Stasta et al., 2006). Even in some European countries, landfilling of sewage sludge is still the primary method of its disposal (see Figure 1). The disposal of sewage sludge in landfills has a number of significant drawbacks, including the necessity for sizeable lands; negative aesthetic impact; pollution of soil, surface and groundwater; emissions of heavy metals into the environment; potential risk of negative impact due to the presence of pathogenic microflora; and loss of energy and material waste potential — the latter mainly because of the irreparable loss of valuable nutrients. Thus, the problems associated with the landfilling of sewage sludge can be brought together into two groups — firstly, problems associated with environmental pollution, and secondly, problems caused by the lost potential to recover energy and/or nutrients out of the sewage sludge.

According to the integrated waste management hierarchy presented in the Directive 2008/98/EC on waste (the so-called Waste Framework Directive), landfilling is the least preferable means of waste treatment (Directive 2008/98/EC, 2008). If it is impossible to prevent generation of a certain type of waste at all, it is advisable to ensure reuse of the waste, followed by the use of its material or energy potential. An alternative to landfilling can be the use of sewage sludge in agriculture as a fertilizer, or for the restoration of disturbed lands, either pre-dried or composted. This ensures recovery of nutrients contained in sewage sludge to the natural metabolic cycles but does not solve the problem of environmental pollution by harmful substances. Some of these contaminants can be removed by means of thermal treatment methods that allow not only significant reduction of the negative environmental impact of untreated sewage sludge but also utilization of its energy potential (Houillon and Jolliet, 2005).

In recent years both landfilling of sewage sludge and its use in agriculture are being significantly reduced in EU member states due to the restrictions of respective pieces of EU legislation (Lundin et al., 2004; Smol et al., 2015; Valderrama et al., 2013).

Consequently, scientific research aimed at evaluating efficiency and searching for the optimal technological parameters for processes of sewage sludge thermal treatment have become in demand. However, at present, the use of thermal methods for the processing of sewage sludge is practiced only in a few countries around the world (Raheem et al., 2018; Pavlik et al., 2016).

The reasons for limited application of thermal treatment methods and such a wide use of sewage sludge landfilling are primarily economic. In this paper, an attempt has been made to prove the efficiency of the application of sewage sludge thermal treatment methods on a broader scale than an individual economic entity or a waste recycling plant. To do so, the authors have analyzed three wide-

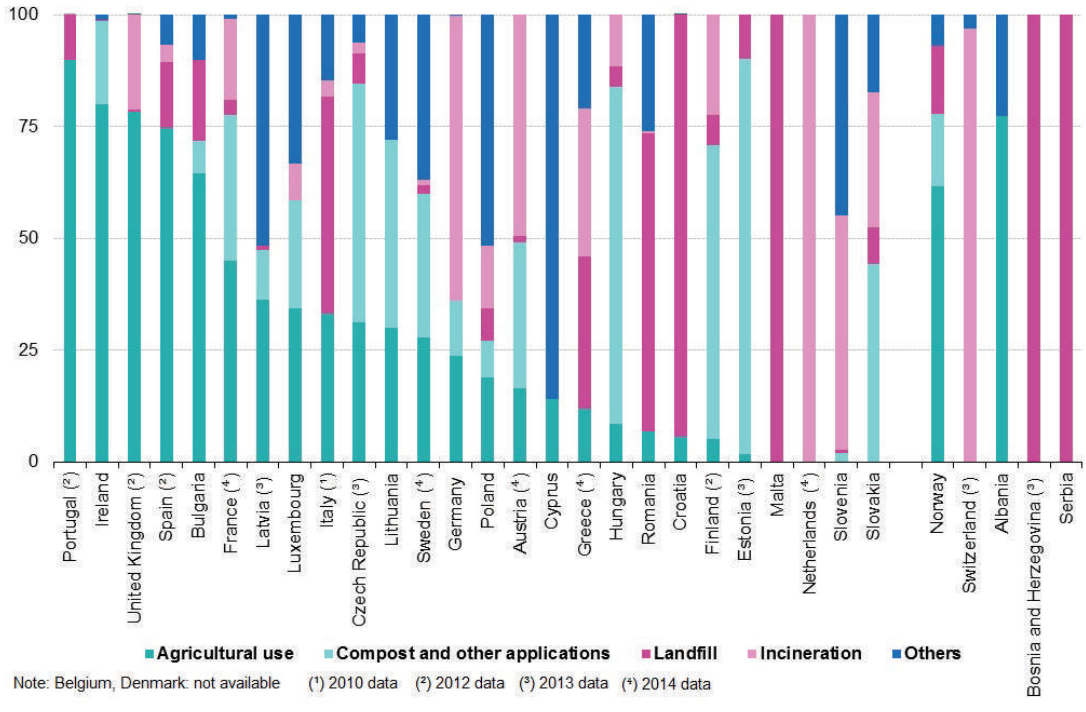

FIGURE 1: Municipal sewage sludge disposal by type of treatment in Europe, 2015 (Eurostat, 2015).

ly known sewage sludge thermal utilization methods in the circular economy context, which implies maximum extended retention of product added value and exclusion of waste generation where possible.

2. METHODOLOGICAL APPROACH

2.1 The objective and the scope of the study

The main objective of the study was to evaluate the efficiency of the use of thermal sewage sludge treatment methods within the framework of a circular economy. The subject matter of the study is the three most widely used thermal waste treatment methods, namely incineration, gasification and pyrolysis. The scope of the study performed was the comparative assessment of these three methods applied for sewage sludge treatment on the basis of the concept of a circular economy.

In order to achieve the main objective, the following tasks were formulated:

- To examine the concept of circular economy in order to define the place and the role of waste management within it;
- To analyze the model of circular economy, defining its main characteristics and differences from the conventional "linear" economy model;
- To specify the main characteristics of a circular economy with regard to the subject matter of the study and to draw up a set of evaluation criteria for the assessment of sewage sludge thermal treatment methods in the context of circular economy;
- To carry out a comparative analysis of the considered sewage sludge thermal treatment methods on the basis of the suggested set of criteria and to select the most advantageous method.

2.2 The concept of circular economy and its main characteristics

The circular economy concept was chosen as a basis of analysis since it is one of the priority concepts of economic development underlying the current European policy in the field of environmental protection. To ensure the transition to a circular economy, the European Commission has developed an Action Plan for the Circular Economy, in which four key action areas have been defined (European Commission, 2015). In the case of sewage sludge, the study should be focused only on two key areas of this action plan: waste management and secondary raw materials.

Waste management and the recovery of secondary raw materials play a central role in a circular economy. As it is specifically indicated in the action plan, the EU waste hierarchy should be applied so that the options that deliver the best environmental outcome are encouraged. Biological materials are to be returned to the natural metabolic cycles after the necessary pre-treatment, such as composting or digestion. The waste that cannot be prevented or recycled is to be used for the recovery of its energy potential, which is considered preferable to landfilling. The introduction of secondary raw materials into the economy is considered a positive factor that extends the security of supply. This would mean fewer risks connected to exposure to volatile raw material prices, and also fewer risks connected to unstable supply because of sudden natural disasters or changes in geopolitical situations. Nutrients are indicated as an especially important category of the secondary raw materials produced out of waste (Ellen McArthur Foundation, 2013).

When considering the transition to a circular economy concept in general, the application of systems thinking is extremely important. Transformation of waste into secondary raw materials ensures reorganization of linear material flows of a conventional economy into circular flows, where waste generation is excluded. Systems thinking should be taken into account when evaluating the efficiency of different sewage sludge treatment methods.

Another important feature of a circular economy arises from one of the main subjects it addresses, namely waste, which can contain harmful or hazardous pollutants. In the case of sewage sludge, there is a risk associated with possible negative impacts to the environment by pathogenic substances, endocrine disrupters, heavy metals and the accumulation of heavy metals in living organisms.

Summarizing the conducted study of circular economy and its basic characteristics, keeping in mind peculiar properties of the considered type of waste, sewage sludge, the following features demonstrating the circular nature of the given sector of economic activity can be singled out:

- Exclusion of waste disposal in landfills;
- Exclusion of pollutant emissions into the environment;
- Reuse/recycling/energy recovery out of the waste;
- Reduced input of primary natural resources, fossil fuels and electricity in comparison with the traditional model of the economy;
- Application of systems thinking, when at individual stages of a product life cycle, different enterprises, service providers or even allied industries are involved;
- Exclusion of accumulation of hazardous substances in the environment and living organisms.

The selected characteristics of a circular economy will be considered in further detail in the analytical part of the paper. An overview of the highlighted important aspects of the circular economy and some known examples of their implementation with sewage sludge (SS) as an object of treatment, as well as the associated risks, are presented in Figure 2.

2.3 Criterion model for the assessment of sewage sludge thermal treatment methods

A preliminary rough examination of sewage sludge treatment methods, performed above, shows that thermal treatment methods meet a number of requirements for the handling of waste within the framework of a circular economy. These methods allow energy to be recovered from waste, natural resources (such as land) to be saved, and valuable by-products to be gained for use as raw materials or marketable products in various industries (Cieślik and Konieczka, 2017; Vadenbo et al., 2014). In order to arrange and specify this information, it was necessary to carry out

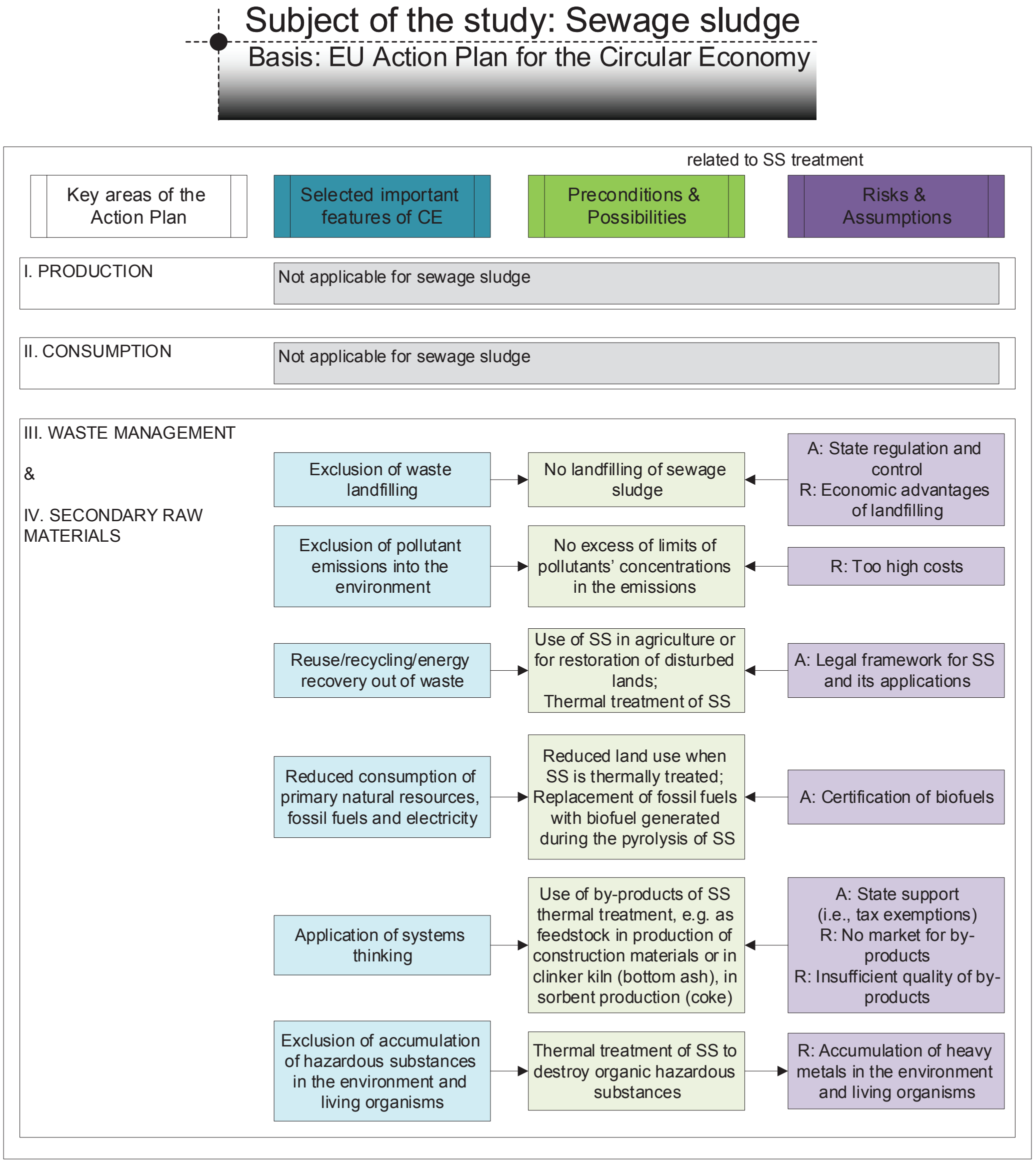

FIGURE 2: Important characteristics of a circular economy when considering sewage sludge treatment possibilities. Abbreviations: A: assumptions; R: risks; SS: sewage sludge.

a comparative analysis of thermal sewage sludge treatment methods in the context of a circular economy.

The comparative analysis was carried out on the basis of a criterion model. The subject of analysis was three thermal methods of sewage sludge treatment — incineration, gasification and pyrolysis.

When developing the criteria, a number of important aspects of circular economy were taken into account, such as the minimum possible energy input in waste treatment processes (energy efficiency), the maximum retention of value in the economy, removal of pollutants from the material cycle, and the possibility for the recovery of nutrients contained in the waste (Alvarenga et al., 2015; Li et al., 2017). In addition, since sewage sludge thermal treatment methods are considered more costly and more technically complex than the disposal of sewage sludge at landfills or its application in agriculture, the criteria also concern the cost-benefits and technical feasibility of using a particular method. The purpose of the analysis was to identify the sewage sludge thermal treatment method which was the most preferable in the context of a circular economy, and also the most economically and technically feasible.

The following five criteria were developed for carrying out the comparative analysis:

1. Costs of waste treatment using the selected method;
2. Energy efficiency of the processing scheme;
3. Recovery of nutrients;
4. Market value of products/by-products;
5. Flexibility when applying the method.

When performing the analysis, assessments for individual criteria were given in the ranges of the corresponding units. For example, costs were assessed in euros per ton of treated dry material per year, energy balance in megawatts (MW). After the comparative analysis was performed, a rating of the three thermal treatment methods was assigned on a three-level scale “Best - Average - Worst”.

3. COMPARATIVE ANALYSIS OF THERMAL SEWAGE SLUDGE TREATMENT METHODS

3.1 Costs of waste treatment using the selected method

Costs of waste treatment using a particular thermal method differ markedly from country to country. Moreover, when comparing the expenditure for waste treatment with the use of different methods, it is necessary to take into account a number of technical aspects, such as:

- The type of equipment used (facilities designed specifically for the processing of sewage sludge, or standard equipment used in other industries);
- Treatment of the certain type of waste exclusively or co-processing of sewage sludge and other wastes such as municipal solid waste.

When this study was conducted, the literature data on the results of thermal treatment of sewage sludge only (no co-processing with other types of waste) with the use of the equipment specifically designed for this type of waste was analyzed.

3.1.1 Incineration

The following financial numbers for incinerating sewage sludge include the thermal drying process at the incineration plant as well as the flue gas cleaning to reach European emissions limits (usually based on dry sorption and a bag filter).

The biggest known fluidized bed incineration plant for sewage sludge in Hong Kong has 4 lines with a capacity of 45,000 Mg/a dry substance (ds) (for 6.2 million people), and the smallest plant has one line with 2,000 Mg/a dry substance (for 70,000 people).

(Frank and Schröder, 2014) as well as (Glatzer and Friedrich, 2015) report the following financial figures to fluidised bed incineration plants with the capacities of 35,000, 4,000 and 2,000 Mg dry substance/a. The plant with the capacity of 35,000 Mg produces electricity and heat by a steam boiler and a steam turbine with a generator. The smaller plants produce heat only by a thermal oil boiler because the generation of electricity in smaller plants is very inefficient from an economic point of view.

As shown in Table 1, the biggest plant (35,000 Mg ds/a for approx. 1.2 million people) is, with 157 euros/Mg treated dry substance, much more economically efficient than smaller plants (2,000 Mg or 4,000 Mg ds/a for 70,000-140,000 people) with about 500 euros/Mg ds.

3.1.2 Gasification

No comparable data on the specific costs of gasification were found for this paper. However, a general overview of scientific literature allowed the authors to conclude that the specific costs for gasification of sewage sludge significantly exceed the specific costs of incineration and pyrolysis (Mills, 2015; EPA, 2012). This is due to both the high cost of equipment and the complexity of maintaining the gasification process itself.

3.1.3 Pyrolysis

When thermally treating sewage sludge with the use of pyrolysis technology, a significant downscaling is possible. (CORDIS, 2017) gives the data on a small-scale pyrolysis plant with the capacity of a maximum 200 Mg of sewage sludge per year. The system thermally dries the dewatered sewage sludge and uses pyrolysis technology to convert it into biochar and gas by-products. To reduce the energy consumption, the energy contained in the sewage sludge is recovered and reused. The pyrolysis syngas is burned in a gas engine, generating heat and electricity that is then used in the system. The waste heat from the gas engine is reused in the dryer. The estimated average costs of the sewage sludge treatment is 400-650 euros/Mg treated dry substance.

The analysis of another study with a larger installation for the pyrolysis of sewage sludge with a capacity of 155.7 kg ds/h, assuming that the operating time of the plant is 7,000 h/a (hence the annual capacity is approx. 1,000 Mg ds), shows that, depending on the selected mode (the most costly one is when, after the pyrolysis reactor, the electricity is produced in an externally fired micro gas turbine and organic ranking cycle motor), the maximum specific costs of sewage sludge treatment are 183 euros/Mg ds (Morgano et al., 2016).

It should be emphasized that in order to carry out a comprehensive estimation of expenditures, in addition to the above mentioned technical aspects, a number of other factors that significantly affect the cost of treatment using the selected thermal method should be taken into account, including:

- Costs of land acquisition;
- Operation scale (smaller scale can significantly increase the costs, as shown above);
- Legal requirements for flue gas treatment and pollutants quotas;
- Requirements and possibilities for treatment and dis-

TABLE 1: Economic comparison of different plant sizes for stationary fluidised bed incineration (Glatzer A. and Fiedrich M., 2015; Franck J. and Schroeder L., 2014).

Capacity/throughput	35,000 Mg ds/a	4,000 Mg ds/a	2,000 Mg ds/a
Capital costs	35 million Euro	12 million Euro	6.6 million Euro
Running costs	5.5 million Euro/a	1 million	1 million Euro/a
Specific costs *	157 Euro/Mg ds	487 Euro/Mg ds	510 Euro/Mg ds

** Specific costs include all the costs related to the certain plant, i.e. capital costs and running costs, calculated per ton of dry substance.*

posal/recovery of ash residues;
- Efficiency of energy recovery, and the revenue received for the heat/electricity produced;
- Taxes levied on emissions and subsidies received for thermal treatment (if any);
- Logistics and availability of infrastructure required for the delivery of waste;
- Insurance, administration and personnel costs (IPPC, 2006).

3.2 Energy efficiency of the processing scheme

3.2.1 Incineration

During the incineration process, combustible material is oxidized and energy is released as thermal energy in the flue gas. Only small amounts of energy (approx. 5%) are lost by radiation over the surface of the incinerator and the boiler as well as heat in the filtered ashes. In bigger fluidized bed incinerators approx. 80% of the energy of the flue gas is recovered by a boiler, and the generated steam is used to produce electricity (electrical efficiency up to 20%) and heat for the drying of the sewage sludge and for district heating (heat efficiency up to 60%) (ÖWAV, 2013; Brunner P. and Rechenberger H., 2015). In smaller plants, the generation of steam and the production of electricity are economically inefficient, so only heat is recovered from the flue gas by a thermal oil boiler for the drying of sewage sludge and the generation of some district heating. Table 2 shows energetic data for different plant sizes.

The plants with capacities of 35,000 and 4,000 Mg ds/a produce electricity of 1.4 MW (approx. 11,000 MWh/a) and 0.12 MW (900 MWh/a), respectively. Only the one with the capacity of 35,000 Mg ds/a finally exports, after its own consumption, approx. 0.4 MW (approx. 5,000 MWh/a). The small plant produces no electricity but uses at least the produced energy to cover the heat for drying the sewage sludge.

3.2.2 Gasification

During the gasification process organic substances are converted to syngas and the inert material to ash. Gasification is carried out with substoichiometric oxygen and in some cases also with additional external heat. The energy for the thermal drying of mechanically dewatered sewage sludge (75-90% of dry substance is necessary for the gasification process) is realized by the utilization of the produced syngas. In Germany two plants for the gasification of sewage sludge are in operation. One plant has a capacity of approx. 2,000 Mg ds/a and the other one approx. 4,500 Mg ds/a.

Both plants are based on a two stage gasification process where sewage sludge (dry content 85-95%) is conveyed from the silo, together with limestone, to a thermolysis screw feeder. The products, thermolysis gas as well as carbon and ash generated during thermolysis, are conveyed to a fluidized-bed gasifier. In the second stage of gasification, the carbon is converted under substoichiometric conditions into gas, and the long-chain molecules (so-called tar) of the thermolysis gas are cracked. The produced syngas is, after treatment (gas cleaning), used to run a gas engine to generate combined heat and power (Figure 3). Surplus gas can be used in a combustion chamber to generate heat (Sülzle Kopf, 2017).

Based on information from the manufacturer, for a plant with 4,500 Mg ds/a (2.3 MW), electricity of 4,200 MWh/a (0.6 MW) and heat of 5,500 MWh/a (0.8 MW) can be produced. Operators of wastewater treatment plants (gasifiers are connected to the WWTP) only give information that with the recovered energy, the plants' own consumption for sewage sludge drying and operation of the gasifier can almost be covered. The gasification process, finally, has an almost equal energy balance for the thermal treatment of anaerobically stabilized and mechanically dewatered sewage sludge. No surplus energy can be exported from the process.

3.2.3 Pyrolysis

Pyrolysis of sewage sludge is in most cases connected to a final combustion of the pyrolysis gas in an afterburner chamber. In theory it is also possible to treat the pyrolysis gas to run gas engines or gas turbines. Another option is to run the pyrolysis process so as to produce more oil instead of gas. In some applications the produced char out of the pyrolysis step is further used as a product; in other cases the char is incinerated in a second step to directly recover the residual energy, and an inert ash is the final product (EISENMANN, 2017).

In Europe only a small number of plants for sewage sludge pyrolysis are in operation, with two different processes. The first needs an input of sewage sludge with a dry substance of at least 65% and performs the pyrolysis at 600°C. The produced bio-char is removed as a product and the pyrolysis gas is incinerated in a combustion chamber at 1,100°C (Figure 4). The energy of the produced flue gas is used to heat the pyrolysis reactor and to dry the mechanically dewatered sewage sludge to 65% dry substance (Greenlife, 2017).

From 500 kW sewage sludge input, the entire plant (ca-

TABLE 2: Energy balance of different plant sizes for stationary fluidized bed incineration (Franck J., 2015; Franck J. and Schroeder L., 2014; Franck J and Schroeder L, 2015; Glatzer A. and Friedrich M., 2015).

Capacity/throughput	35,000 Mg ds/a	4,000 Mg ds/a	2,000 Mg ds/a
Thermal input	14.5 MW	1.9 MW	0.95 MW
Energy for drying	7 MW	0.7 MW	0.43 MW
Produced electricity	1.4 MW	-	-
Electricity consumption	1 MW	0.11 MW	0.07 MW
Delivered electricity	0.4 MW	-	-

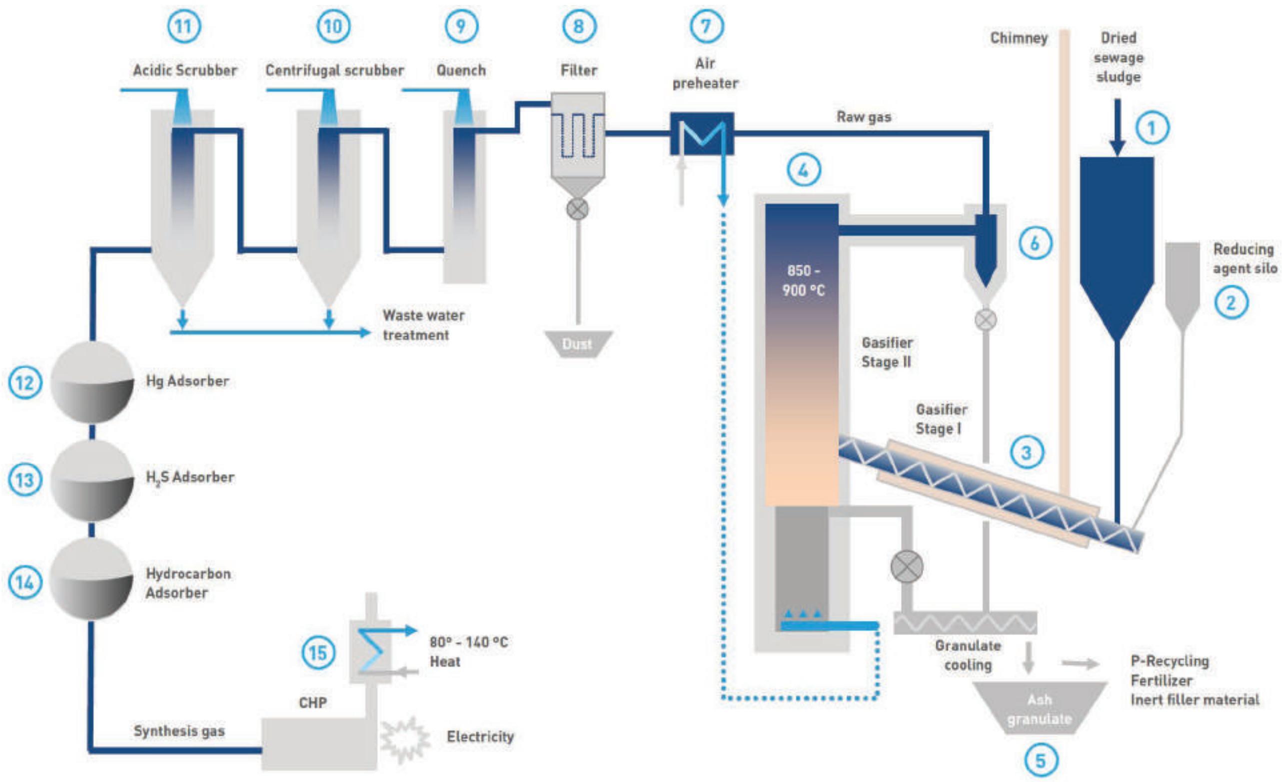

FIGURE 3: KOPF SynGas utilisation process (Sülzle Kopf, 2017).

pacity of 1,000 Mg/a ds) produces about 150 kW flue gas heat that is used for the heating of the pyrolysis chamber and drying of sewage sludge. The plant is more or less energy self-sufficient but needs an electrical input of approx. 7.5 kW (an amount which seems to be very small and might concern only the pyrolysis reactor). The product of the process is roughly 500 Mg/a bio-char (Greenlife, 2017).

The second process is based on the pyrolysis of 90% dry substance sewage sludge at 300-350°C and the combustion of the produced coke at 650°C. The produced flue gas of the combustion is used to heat the pyrolysis and is then after-burned with the pyrolysis gas at 900°C. The energy of the produced flue gas of the afterburner chamber is transferred by a thermal oil heat exchanger to the drying of the dewatered sewage sludge (Figure 5). The final product out of the process is inert ash (EISENMANN, 2017).

The plant has a capacity of approx. 4,000 Mg/a ds (approx. 2,000 kW). Small amounts of external fuel in the form of natural gas (260 kW) are used to ensure a flue gas temperature that heats the pyrolysis and the afterburner chamber. After energy losses, approx. 2,000 kW of flue gas is produced to dry the dewatered sewage sludge (approx. 21% ds) to 90% ds. For this thermal drying, approx. 1,000 kW of additional external thermal energy is necessary (Neumann and Tittesz, 2011; EISENMANN, 2017). No information about the plant's own consumption of electricity is available; based on the size of the plant, 100 kW is estimated.

Finally, it can be stated that the energy balance of the pyrolysis of sewage sludge is negative. The overall process, from the drying of the mechanically dewatered sewage sludge to the final treatment of the flue gases, needs more energy than can be recovered out of the process.

3.3 Recovery of nutrients

3.3.1 Incineration

During the combustion process, organic pollutants, endocrine disruptors and pathogens contained in sewage sludge are destroyed, and volatile heavy metals such as quicksilver are transferred to the flue gas. The nutrients, especially phosphorus, are not transferred in the form of gaseous compounds to the flue gas and are kept in the incinerator ash. In fluidized bed incinerators most of the generated ashes have a very small particle size and are transported with the flue gas flow. Commonly, these fly ashes are removed by a filter (usually an electric precipitator) (Wiechmann et al., 2013; Outotec, 2016; Infraserf, 2018).

The content of phosphorus in these ashes is up to 20% by mass as P_2O_5 (the amount of phosphorous in this substance is 43.6% by mass, so finally up to 8.6% by mass as P in the ash) and is therefore much higher than in sewage sludge (Adam et al., 2009). The phosphorus contents obtainable in the sewage sludge ash are about 50 to 70% less than the phosphate contents of crude phosphate fertilizer or triple superphosphate, which is still high (Waida et al., 2010). However, during the combustion process the phosphorus is transferred into low-solubility mineral phases with low plant availability. It is therefore required to transfer the phosphate by means of a suitable thermochemical reaction to a plant-available form (Drissen, 2012).

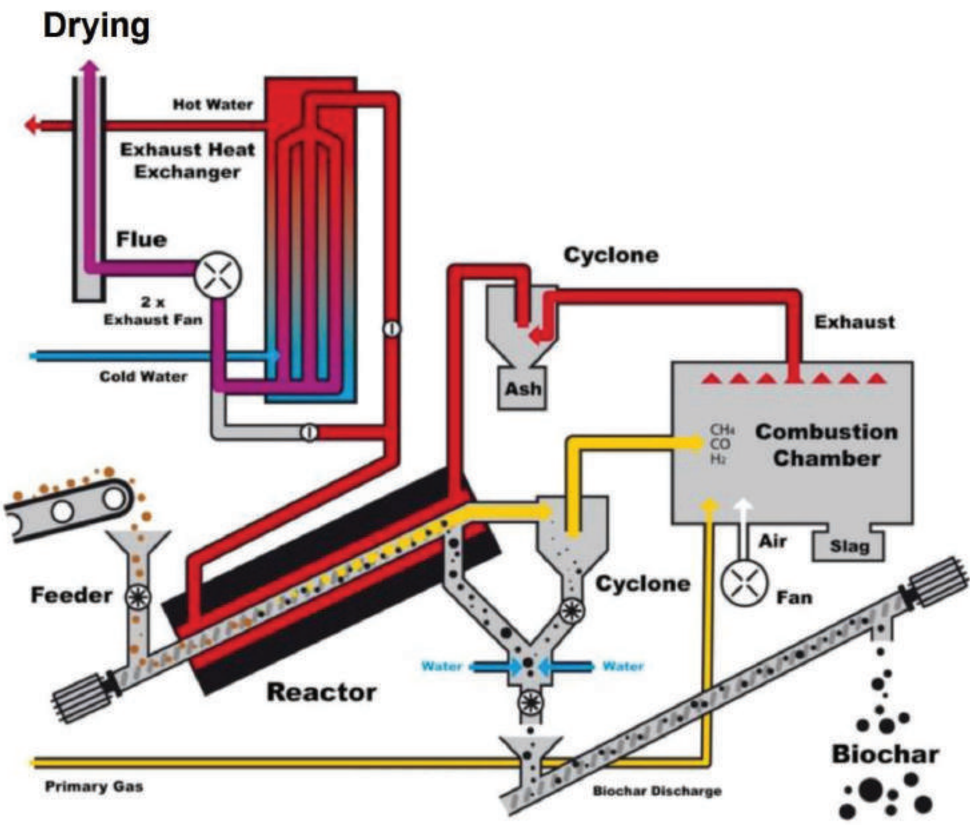

FIGURE 4: Pyrolysis process with production of bio-char (Greenlife, 2017).

After technical preparation (e.g. PASCH (Pinnekamp et al., 2010), RecoPhos (Weigand et al., 2013), EcoPhos (DeRuiter, 2014), LEACHPHOS (Buehler and Schlumberger, 2014), MEPHREC (Gruener and Reinmoeller, 2016) or Ash-Dec (Nowak et al., 2011), theoretically over 90% of phosphorus that is contained in sewage sludge can be recovered (Egle et al., 2016). Assuming 90% of the phosphorus contained in the wastewater in the sewage sludge is eliminated, a recycling potential of up to 80% of the phosphorus load in the inlet of the sewage treatment plant can be achieved.

3.3.2 Gasification

After sewage sludge gasification, phosphorus can also potentially be recovered from the ash (Gorazda et al., 2018). Recovery of nitrogen is not possible as it is diluted in the form of N_2 in produced syngas (Winkler, 2012). At Kopf gasification plant (Balingen, Germany), it is reported that phosphorus out of the mineral granulate, produced from the slag, can be recovered (EPA, 2012).

3.3.3 Pyrolysis

With the application of the pyrolysis process it is possible to save nutrients by converting sewage sludge into a carbon-phosphorus fertilizer. A pyrolysis plant with a capacity of 4,000 Mg/a of dewatered sewage sludge (25% ds) produces some 500 Mg/a of biochar (Greenlife, 2017). The most interesting property of the biochar is its resistance to biological and physical degradation when incorporated into soils, thus conferring other properties (e.g., nutrient and water retention, microbial activation, liming, and others) that improve soil functions over time periods from decades to centuries. Thus, biochar can be used in soil conditioning.

Depending on the nature of the sewage sludge feedstock, the nitrogen and calcium content in pyrolysis biochar allow for its consideration as a potential fertilizer. The investigations also show high phosphate content of biochar, which exceeds the minimum EU standard for phosphate fertilizers (CORDIS, 2017). However, the heavy metal content of biochar can limit its agricultural application depending on the national legal requirements.

3.4 Market value of products/by-products

3.4.1 Incineration

During the incineration process, sewage sludge is combusted and the energy is recovered as heat and electricity. These "products" are first of all used to cover the incineration plant's own consumption; the residuary amounts of energy can be sold. The fluidised bed material containing some small amount of ash has no market value but can be used in road construction for example, or to produce other construction materials. The residues from the flue gas cleaning can be cost-intensively disposed of or can also be cost-intensively stabilized and further used in the construction industry, e.g. for cement production, in the manufacture of building ceramics, or as a substitute for sand or cement in road construction (Smol et al., 2015).

Currently, the recovery of phosphorus out of fly ashes is even more expensive than the production of phosphorus out of phosphorus ores; the former is therefore not economically feasible. At the same time, a price increase is ex-

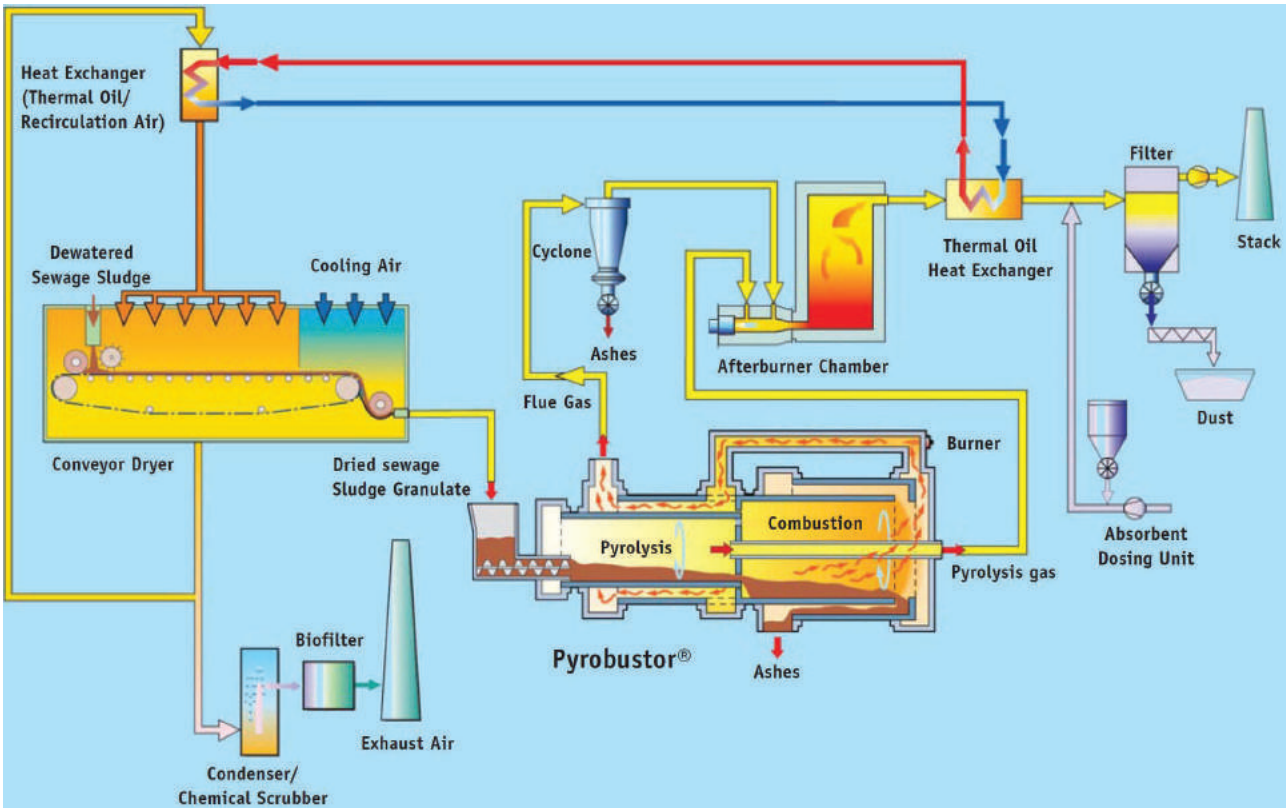

FIGURE 5: Pyrolysis process by Pyrobustor (EISENMANN, 2017).

pected for phosphorus from natural deposits in the future and a reduction of costs for phosphorus recovery (thanks to learning effects and up-scaling) out of sewage sludge (Cieślik and Konieczka, 2017; Guedes et al., 2014; Tan and Lagerkvist, 2011). This makes the ashes already valuable enough to landfill the material on separate sites in order to get it back when phosphorus recovery becomes economically feasible.

3.4.2 Gasification

The main useful product of sewage sludge gasification is syngas. Syngas can be directly used for the necessary thermal drying of sewage sludge to a dry substance between 75 and 90% (closed coupled gasification), or to generate electricity and heat in a gas turbine or a gas engine (two stage gasification). For utilization in gas turbines and gas engines, the syngas has to be treated (cleaned); for gas turbines the syngas also needs a minimum heating value of approx. 17.0 MJ/Nm^3 and needs to be pressurized. Usually, as compared to other types of fuel such as biogas or natural gas, the energy content of syngas is three to five times lower (Winkler, 2012).

A study in which a comparative analysis of the efficiency of sewage sludge gasification and pyrolysis was conducted shows that when the pyrolysis process is operated at relatively high temperatures (>800°C), the obtained proportion of combustible gases (78%) is even higher than the proportion after the gasifier (42%). It was also determined that, relative to gasification, the pyrolysis process produces larger concentrations of gases with higher combustion enthalpies and also includes a small quantity of ethane. The combination of these factors means that a syngas produced in the pyrolysis process has twice as much calorific value as the one after the gasifier (Mills, 2015).

Slag from a high temperature gasifier is non-leachable, non-hazardous and typically suitable for use in construction materials. For example, at Kopf gasification plant in Balingen, Germany, mineral granulate produced out of slag is used for asphalt and construction materials (EPA, 2012).

3.4.3 Pyrolysis

During pyrolysis, as a result of the heating of sewage sludge in an anoxic environment, a chemical decomposition of the organic component of sewage sludge takes place to form solid char containing pyrolytic carbon and mineral components. In addition, pyrolysis gases are generated, some of which are capable of condensing and forming a liquid fraction, while the remaining part is a combustible gas. Gaseous non-condensable pyrolysis products can be used in the thermal desorption process itself as a fuel to maintain the required temperature and the auto-thermal process. The condensed pyrolysis gas is an oily water-miscible liquid and is a fraction of hydrocarbons with a boiling point of 200-440°C. It is known that hydrocarbons with a boiling point up to 400°C (C12-C22) are used as diesel or boiler fuel. This liquid phase does not contain toxic heavy metals, such as lead or mercury; the content of aromatic compounds (benzene, toluene, ethylbenzene, n-xylene and o-xylene) does not exceed their concentrations in the fuel oil. The calorific value of a fuel containing hydrocarbons with a boiling point of 200-400°C is 35,000-40,000 kJ/kg. Pyrolysis processes are typically operated to maximize the bio-oil yield. The energy potential of such bio-oil from condensed pyrolysis gases can be used for drying sewage sludge or for creating the necessary temperature for pyrolysis, which leads to a significant reduction in operating costs. Alternatively, this bio-oil can be upgraded to transportation fuel. The oil, once refined, can be stored and transported (Cao, 2011; Kim, 2008).

Solid char formed during pyrolysis is half composed of pyrolytic carbon. The significant content of pyrolytic carbon in solid char and the determination of the total porosity by moisture capacity allow one to assume that it has sorption properties. High hydrophobicity and sufficient sorbent oil capacity of the solid char mean it can be recommended as a sorbent for extracting oil and petroleum products from water and absorbing oil spilled on solid surfaces during accidental spills of petroleum products (Spinosa et al., 2011; Khodyashev et al., 2009).

An alternative application of the biochar generated out of sewage sludge pyrolysis is apparent from the phosphate content of the material, which is well above the minimum EU standard for phosphate fertilizers. Potential biochar fertilizer values are to be exploited also with regard to its nitrogen and calcium content, depending on the exact nature of the feedstock sewage sludge (CORDIS, 2017; Spinosa et al., 2011). The limitation of the agronomic properties of the biochar can be related to the heavy metal content of the product and should be thoroughly investigated in each particular case.

3.5 Flexibility when applying the method

3.5.1 Incineration

Incineration of sewage sludge in a fluidized bed is only possible under the precondition of a minimum heating value of 3,000-4,000 MJ/Mg wet substance (depending on the pre-heating temperature of the primary air) for reaching an auto-thermal combustion. Depending on the sewage sludge quality (pre-digested or not, degree of stabilisation, content of organics and carbon), the mechanically dewatered sewage sludge of usually 20-35% dry substance content has to be thermally dried to a minimum of 40% dry substance. If the incineration plant is situated near a sewage treatment plant or if the delivered sewage sludge is just mechanically dried, a thermal drying step has to be included in the treatment process. This thermal drying can be achieved with some extra effort by the energy recovered out of the incineration process (Franck J., 2015; Franck J. and Schroeder L., 2014; Franck J. and Schroeder L., 2015; Glatzer A. and Friedrich M., 2015).

A big disadvantage of the incineration process is low flexibility in terms of downscaling. The smallest plants have capacities of approx. 2,000 Mg ds/a (for approx. 70,000 people) with very high specific costs of more than 500 euros/Mg ds.

3.5.2 Gasification

There are only a small number of plants for sewage sludge mono-gasification worldwide because of a technically highly demanding procedure, with high down times, which is not very efficient in terms of energy or costs. The syngas produced during the sewage sludge gasification contains combustible components; it enables the use of the syngas as a feedstock (through some reforming processes), or as a fuel. The composition of the main combustible components of the syngas (H_2 and CO) that defines its lower heating value (LHV) depends on the amount of the air supplied to the reactor. The optimum value of the air ratio (λ) equal to 0.18 leads to LHV taking its maximum value and thus favors gasification resulting in combustible gases, rather than the case of complete combustion with an air supply that mainly produces CO_2 (Werle, 2016).

Depending on the technology used (fixed bed updraft or downdraft gasifier, fluidized bed gasifier), the dry matter content of the feedstock for gasification should be 45-50%, more than 80-85% correspondingly (Winkler, 2012). Usually, water content in the sewage sludge treated with the gasifier should be between 10-20%.

Both undigested and digested sewage sludge can be treated using the gasification method, but undigested sludge is preferable as it results in higher energy content of the produced syngas (Winkler, 2012).

The scale of a gasification plant strongly depends on the type of the gasifier used and the ranges from small scales of 5 kW-20 MW for downdraft fixed bed gasifiers up to 100 MW for circulating fluidized bed gasifiers (EPA, 2012). Economic feasibility of a gasification plant depends electricity tariffs among other factors; some calculation shows that a gasification plant becomes economically feasible at a plant capacity of about 0.093 m^3/s (raw sewage flows) (Lumley et al., 2014).

3.5.3 Pyrolysis

During the pyrolysis of sewage sludge, (depending on feedstock, pyrolysis temperature, pressure, retention time and additives, and different amounts of gases) liquids and solids are produced. For pyrolysis, the mechanically dewatered sludge has to be thermally dried to a dry substance between 65 and 90%.

Pyrolysis technology applied to sewage sludge is more flexible in terms of downscaling than incineration or gasification. Pyrolysis plants can be compact and used to treat and dispose of the sewage sludge produced in small municipalities (with fewer than 10,000 inhabitants). The sludge load for such a compact plant can be up to 200 Mg ds/a with specific cost of 400-650 euros/Mg ds (CORDIS, 2017).

4. RESULTS AND DISCUSSION

A comparative analysis of three se wage sludge thermal treatment methods in the context of a circular economy has been carried out using a criterion model consisting of five criteria. To make the final evaluation, the results of the analysis were aggregated in Table 3.

The costs of treatment in the cases of incineration and gasification of sewage sludge strongly depend on the scale of the plant; downscaling makes sense only if pyrolysis technology is applied. If it is possible to use a large

TABLE 3: Results of the comparative analysis of sewage sludge thermal treatment methods on the basis of the criterion model.

Criteria	1. Cost of treatment		2. Energy efficiency			3. Nutrient recovery	4. Product market value			5. Flexibility		
	Plant capacity, Mg DS/a	Specific costs, euros/Mg DS	Plant capacity, Mg DS/a	Produced electricity, MW	Delivered electricity, MW	-	Plant's own consump-tion	Products with no market value	Valuable products	Feed-stock dry matter content	Down-scaling	Other consider-ations
Incineration	2,000	510	2,000	N/a*	N/a	Potentially up to 80% of phosphorus load in the sewage sludge can be recovered from the fly ashes	Heat and electricity	Use of ash in road construc-tion	Electricity	Approx. 40%	Limited	-
	4,000	487	4,000	N/a	N/a							
	35,000	157	35,000	4.4	0.4							
Gasification	No data available; the most expensive method according to the general overview of literature		4,500	0.6	N/a (and also 0.8 MW of heat is pro-duced)	Examples of phosphorus recovery from the slag; potentially possible phosphorus recovery from the ash	Heat and electricity	Use of slag for asphalt and con-struction materials	Syngas	80-90%	Hardly possible	Techni-cally com-plicated process
Pyrolysis	200	400-650	1,000	N/a	-0.0075	Application of biochar for soil condi-tioning or as a fertilizer	Heat and electricity	Sulfur	Bio-oil; Char for the pro-duction of sorbent and/or as soil con-ditioner; Syngas	65-90%	Possible	-
		183	4,000	N/a	-0.1 (and also -1.0 MW of thermal energy)							

** N/a: not applicable*

capacity plant (e.g., in a large city with a population of more than a million people), incineration is the most preferable technology in terms of costs. If it is necessary to use a plant with a small capacity, pyrolysis is the most appropriate technology for thermal treatment of sewage sludge.

When processing sewage sludge using each of the three technologies considered, some heat and electricity are generated; it can be utilized in the treatment process itself or delivered to external consumers. The data obtained when comparing the technologies on the basis of the energy balance criterion made it possible to identify a technology generating excess electricity that can be put on the electricity market. Pyrolysis of sewage sludge in this respect has negative characteristics: heat and electricity produced are utilized completely in the plant for drying sewage sludge and maintaining the auto-thermal process. Additionally, some extra energy input is needed to maintain the overall process from the drying of mechanically dewatered sewage sludge to the final treatment of flue gases. The energy generated during the gasification of sewage sludge is also fully utilized in the plant itself, and no surplus energy can be exported from the process. Export of a certain amount of electricity is only possible with the use of sewage sludge incineration, provided that the plant has a sufficiently high capacity. The smaller incineration plants produce no electricity but can at least cover the heat needed for drying sewage sludge with the produced energy.

All three sewage sludge thermal treatment technologies considered are quite inefficient in terms of the possibility for nutrient recovery. The most promising method under this evaluation criterion is incineration, where fly ash can be used as a source of phosphorus, subject to the appearance of appropriately efficient and economically feasible technologies for phosphorus recovery out of the ash. The coke produced during the pyrolysis of sewage sludge can be used directly as a fertilizer or as a soil conditioner, as long as the level of heavy metals in the coke is acceptable. There are some data on the extraction of phosphorus out of gasification slag (Gorazda, 2018).

When evaluating the criterion of the market value of products and by-products of sewage sludge thermal treatment methods, three types of such products were considered: products used in the treatment process itself, environmentally safe products with zero market value and products with market value. For all the three technologies analyzed, the first type of products includes heat and electricity, which are used in the thermal treatment process itself and thus allow savings on primary energy resources. If sewage sludge is incinerated at a high-capacity plant, it may create some surplus electricity that can be put on the market and thus become a market product. The main marketable product of gasification of sewage sludge is syngas. Its calorific value is low in comparison with other types of fuel; also, syngas has to be treated (cleaned) and pressurized in order to be used in gas turbines and gas engines. In the pyrolysis of sewage sludge, syngas is also generated. In addition, the condensable gas fraction has high caloric value and can be used as a boiler fuel or fuel oil. Biochar produced during sewage sludge pyrolysis shows the presence of sorption properties, which, taking into account its hydrophobicity makes it possible to use biochar as a sorbent in the liquidation of oil spills. The coke can also be used as a fertilizer or soil conditioner due to the high content and plant availability of phosphorus, nitrogen and calcium.

The main preconditions that determine the application of a particular sewage sludge thermal treatment method are the moisture content in the feedstock and the possibility of using a low-capacity plant. Incineration is less demanding on high contents of dry substance in the sewage sludge than gasification and pyrolysis. The gasification method has not been widely used in the world due to the technical complexity of maintaining the process. The downscaling capabilities are limited when using incineration and gasification methods; the pyrolysis method can be applied at a level of cost comparable with that of low capacity installations.

Ultimately, the decision on the preferability of a certain method of thermal sewage sludge treatment depends on which of the presented criteria are of the highest importance for the decision maker. If all the criteria are equally relevant (e.g., there is a state circular economy development program, which includes subsidies for the implementation of appropriate technologies), the incineration method is actually the most optimal, taking into account the principles of the circular economy.

5. CONCLUSIONS

The feasibility of using thermal methods for the treatment of large-tonnage waste generated in the process of traditional municipal wastewater treatment, namely sewage sludge, has been analyzed in the context of a circular economy. The concept of circular economy has been studied, and its main characteristics with regard to sewage sludge treatment have been identified, including the exclusion of waste landfilling, pollutant emission and hazardous substance accumulation, reuse/recycle/energy recovery from waste, reduced input of primary natural resources, and application of systems thinking.

Comparative analysis of three sewage sludge thermal treatment methods (incineration, gasification and pyrolysis) has been performed on the basis of the developed criterion model. The analysis has shown that the most advantageous method for the criteria of cost of treatment, energy efficiency, nutrient recovery and flexibility in terms of feedstock dry matter content is incineration, whereas the most preferable method for the criteria of product market value and flexibility in terms of downscaling is pyrolysis. This makes incineration the most preferable sewage sludge thermal treatment method in the context of a circular economy within the framework of the developed criterion model, based on the assumption that all the criteria are of equal importance.

The results of this study could be used for establishing an effective sewage sludge management system at a regional or state level. In further studies, the authors plan to carry out a quantitative comparison of material flows and to develop corresponding diagrams for each of the three considered thermal treatment technologies. In addition, it is planned to assess the costs of sewage sludge treatment using thermal treatment methods taking into account the revenues from the sale of generated market products and by-products, as well as the costs incurred when disposing sewage sludge at landfills.

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POTENTIAL IMPACTS OF THE EU CIRCULAR ECONOMY PACKAGE ON THE UTILIZATION OF SECONDARY RESOURCES

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ABSTRACT

The dependency of the European Union on the imports of primary raw materials was one of the major drivers for the release of the circular economy strategy, in which resource input and waste, emission, and energy leakage are minimised by closing material loops. As part of the shift towards a circular economy, proposals introducing new waste-management targets regarding reuse, recycling and landfilling have been made. In this study the potential impact of these new targets for packaging waste (PW) and Municipal Solid Waste (MSW) on the EU´s supply of four raw materials, namely Iron & Steel, Aluminium, Plastics and Paper & Board, has been assessed. Thereto the method of material flow analysis has been applied in order to evaluate current and potential future flows of secondary raw material. The results of the investigations indicate that for Iron & Steel and Paper & Board already today about 50% of the EU´s production is made out of secondary raw materials. For Aluminium (36%) and Plastics (10%) this share however is significantly lower. Implementing higher recycling targets according to circular economy package would increase the domestic supply of secondary materials between 0.6% (Iron & Steel) and 70% (Plastics). Since today significant amounts of recyclables (equivalent to more than 10% of the total domestic raw material consumption) are already not utilized by the European industry but exported, it is highly questionable whether these additional quantities of recyclables derived from post-consumer waste will substitute primary raw materials in the EU. Quality constraints of the industry as well as production capacities for secondary raw materials in place might limit the domestic utilization of recyclables liberated by the circular economy. Hence, additional policy measures (e.g. targets for secondary production) seem to be necessary to enhance the rate of secondary production within the European Union.

1. INTRODUCTION

In December 2015 the European Commission has released an action plan for a circular economy (COM / 2015 / 0614 Circular Economy Package CEP). This action plans demands a “transition to a more circular economy, where the value of products, materials and resources is maintained in the economy for as long as possible, and the generation of waste is minimized, which is seen as an essential contribution to the EU’s efforts to develop a sustainable, low carbon, resource efficient and competitive economy.” Within this action plan for a circular economy waste management plays a central role. The released proposal contains quantitative targets for the reduction and recycling of wastes. By the year 2030, for instance, 65% of the municipal waste and moreover 75% of packaging waste generated have to be recycled or prepared for reuse, while landfilling of all wastes should be reduced to 10% in each member state. Moreover separately collected wastes are completely banned from landfilling. All these measures are believed to contribute to the development of a sustainable and resource-efficient economy in Europe. This requires that the additionally generated secondary resources are utilized by European industries to a greater extent. At present large quantities of waste derived raw materials (including waste paper, scarp metals, or waste Plastics) are exported out of the European Union, suggesting an existing surplus of secondary resources.

Hence, the aim of the present paper is to analyze the current flows of secondary resources for selected commodities in the EU and to predict their future quantities in case that the Circular Economy Package, and in particular its recycling targets for packaging waste and MSW are fully implemented.

Besides a quantitative analysis, also qualitative aspects

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of secondary resources and their impacts on the recovery/export will be discussed. The investigations focus on the following commodities: Iron & Steel, aluminum, Plastics, and Paper & Board.

2. MATERIALS AND METHODS

2.1 Material Flow Analysis of Status quo

In a first step, a material flow analysis MFA (Brunner and Rechberger, 2004) for each commodity has been conducted. Thereby a systematic assessment of all materials flows in the European Union has been accomplished. In particular, the following data have been collected for Paper & Board, Plastics, Iron & Steel, and aluminum for the EU-28:

- Domestic production and consumption of raw materials (indicated as raw material and finished goods in Figure 1);
- Net imports or exports of commodities via semis, finished products, End of Life products (mostly vehicles) and wastes (indicated as net exports of semis and finished goods, net export of recyclables and export of EoL products in Figure 1);
- Waste production divided into MSW, other wastes, production waste and processing or internal waste (indicated as internal scrap, production waste/new scrap, MSW and other wastes in Figure 1);
- Material losses during production (indicated as material losses in Figure 1);
- Net stock increase within the EU (indicated as stock increase of the consumption process in Figure 1)
- Final disposal paths or recycling of waste (indicated as waste to incineration, waste to landfill and recyclables/scarp in Figure 1).

In general, the MFA was conducted at a rather superficial (highly aggregated) level, meaning a low level of detail. The MFA model used for the analysis of the commodity flows is present in Figure 1. It basically consists of four processes, namely Raw Material Production, Manufacturing & Trade, Consumption, and Waste Management. The spatial system boundary is the European Union (EU-28), whereby primary raw materials (nonetheless whether they are extracted within or outside the EU) are considered as imports into the system. On the other hand waste (e.g. waste Plastics, waste Paper & Board) thermally utilized or landfilled are accounted for as exports, although the respective processes (e.g. waste-to-energy plants, landfills) are located within the European Union. For the temporal system boundary one "average" year representative for the period 2014 to 2016 was chosen. This was mainly necessary due to the fact that not all data were available for same year.

The "commodities flows" considered include primary raw materials demanded, raw material produced, material losses during raw material production, the net import of semis and finished products, finished good, production or manufacturing waste, Municipal Solid Waste (MSW), all other wastes, net exports of recyclables and end of life products (vehicles and electronic waste), as well as wastes landfilled or incinerated.

2.2 Assessment of future material flows

In a second step the management of MSW and packaging waste with respect to the four commodities has been investigated in detail. This was done in order to evaluate the status quo and to assess the potential impact of the Circular Economy Package (namely increased recycling targets for packaging waste and MSW) on the overall domestic supply with secondary resources. For simplicity reasons it was assumed that recycling targets for packaging wastes are also applicable to the alike materials present in MSW.

For the additionally provided secondary resources two "extreme" scenarios are basically possible: On one hand, all additional quantities of recyclables and scrap might be utilized within the European Union. On the other hand the amount of recyclables/scrap liberated by the implementation of a circular economy is exported and thus not utilized by European industry. Based on historical data about supply, utilization and export of recyclables it has been dis-

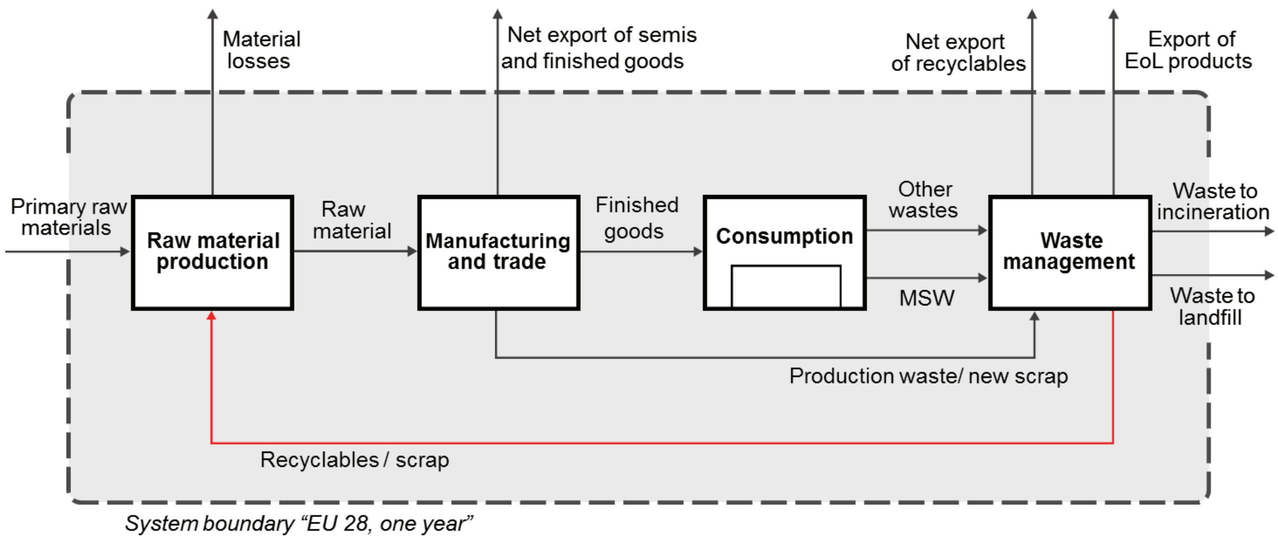

FIGURE 1: Model used to analyse the material flows of Iron & Steel, Aluminium, Plastics and Paper & Board within the EU-28.

cussed which of the two options (domestic utilization or export of recyclables) is likely to prevail.

2.3 Evaluation of Circularity

Based on the results of the MFA it is possible to assess the current and potential future share of primary and secondary production for the different commodities. In order to quantify these shares the following indicators are calculated:

- Share of secondary production (SSP)

$$SSP = \frac{purchased\ scrap(recyables)\ utilized}{amount\ of\ sellable\ commodity\ produced}$$

- Share of secondary "resource losses" due export of waste and end of life products (SSL)

$$SSL = \frac{net\ export\ of\ scrap(recyables) + export\ of\ end\ of\ life\ products}{amount\ of\ sellable\ commodity\ produced}$$

- Share of secondary production for the final domestic consumption (SSC)

$$SSC = \frac{scrap(recyables)\ utilized}{amount\ of\ commodity\ domestically\ consumed}$$

In addition current and potential future reductions of greenhouse gas emissions due to the utilization of secondary raw materials are calculated. Thereto Life Cycle Assessment LCA data (after Ecoinvent & Turner et al., 2015) are used. In practice, the following specific greenhouse gas emission factors (expressed in kg CO_2 equivalents per kg of commodity related to the primary and secondary production of the four commodities are applied (see Table 1).

2.4 Data collection

A wide range of different data sources was needed to establish the material budgets for Iron & Steel, aluminum, Plastics and Paper & Board of the EU-28.

For the Paper & Board, data were mainly derived from the Confederation of European Paper Industry, which publishes annual statistics about production of pulp and paper and therefore utilized raw materials (CEPI, 2016). In addition, the data of the CEWPI statistics have been complemented and crosschecked with MFA figures recently published on paper recycling in the EU (Pivnenko et al., 2016).

For Plastics most data used for the budget (e.g. production, consumption, recycling, waste to landfill, waste to incineration) have been obtained from Plastics Europe (Plastics Europe, 2016). Information about the export of recyclable Plastics was retrieved from a study recently conducted by Verlis (2014). Furthermore, Plastics exported via end of life products was assessed using data about the composition (Plastics content) of vehicles and estimates about the (official and unofficial) exports of end of life vehicles. For the latter a total number of approximately 5 million passenger cars was assumed according to Oeko-Institut e.V. (2016). The assignment of waste Plastics to MSW and other wastes was accomplished in accordance to information provided by Van Eygen et al. (2017).

For European flows of aluminum (including data about production, consumption, scrap generation), comprehensive data sets for the last years were provided by the European Aluminium Association (2016). Exports of Al scrap was obtained from the UN Comtrade database and exports via end of life products were estimated in analogy to plastic flows. Al flows through MSW were assessed using data about production statistics (Al used in packaging) and information provided by Buchner et al. (2015) and Warrings and Fellner (2017).

Information about Iron & Steel production and consumption as well as data about scrap generation were obtained from the European Steel Association Eurofer (Eurofer, 2017). In addition the generation of internal (home) and production scrap was estimated using data (ratio between steel production and scrap generation) provided by Ghenda and Lüngen (2013) and Wang et al. (2007). Iron & Steel exported via end of life products (mainly due to end of life vehicles) were estimated in analogy to alike exports of Plastics or aluminum. For the estimation of Iron & Steel present in MSW, data of waste sorting analyses from different countries (Germany, Denmark, Austria, Croatia and Sweden) were used.

3. RESULTS AND DISCUSSIONS

3.1 Present Recycling Rates for Iron & Steel, Aluminium, Plastics and Paper & Board

The results of the material flow analyses reveal that for Iron & Steel (SSP=46.4%) and Paper & Board (SSP=52.4%) secondary production is equally important as primary production. For aluminum, almost 37% of the sellable production originates from scrap. In contrast, only about 10% of the Plastics produced in the EU are made out of secondary raw materials. In addition, it can be assumed that these secondary Plastics only partly substitute primary Plastics. This assumption is based on the fact that secondary Plastics produced in the EU contain to a large extent mixed-polymer re-granulates, which are used for garden benches, roof tiles or other products, which would usually not be made out of Plastics.

A significant share (between 42 and 47%) of the final consumption of commodities (except for Paper & Board) contributes to an increase of anthropogenic material stocks in the EU. The annual stock growth amounts in absolute figures to 55,000 kt for Iron & Steel, to 4,700 kt for aluminum and 22,000 kt for Plastics (equal to 110 kg/cap/y for Iron & Steel, 9.4 kg kg/cap/y for Aluminium and 44 kg/cap/y for Plastics - see Figure 2 to 5). This observation demonstrates that even in a highly developed econo-

TABLE 1: Greenhouse gas emissions factors used for the four commodities (based on Turner et al., 2015).

Greenhouse gas emissions [kg CO_2,eq/kg]	Iron & Steel	Aluminum	Plastics (mixture of PE, PP & PET)	Paper& board
Primary production	2.3	9.2	1.8	0.68
Secondary production	0.7	1.1	0.3	0.56

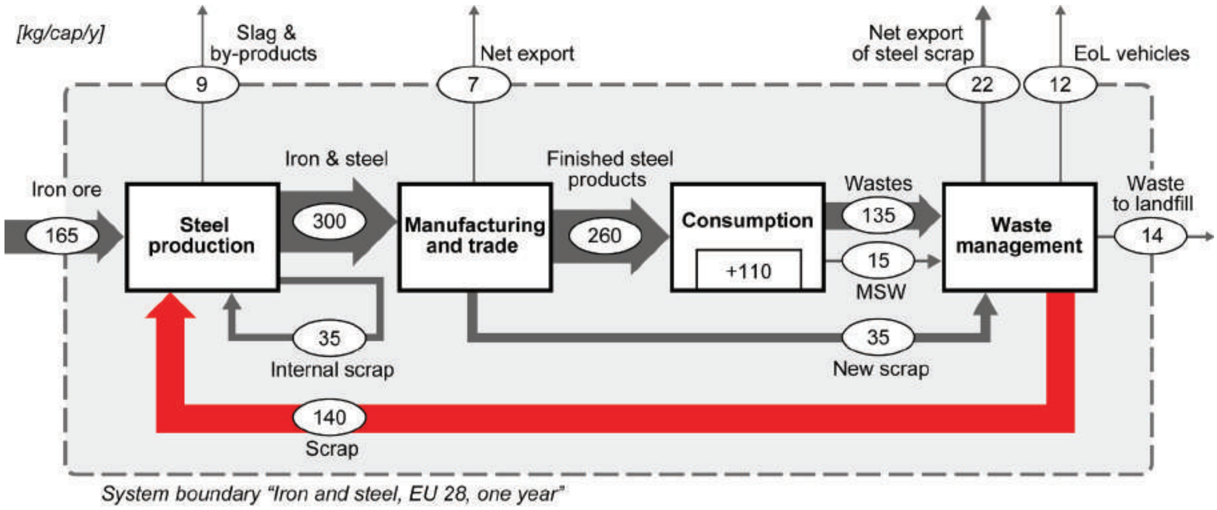

FIGURE 2: Annual Iron & Steel flows in the EU-28 including the stock increase in the consumption process (data given in kg/cap/y), since the data of the different sources have not been reconciled, the flows of some processes are slightly unbalanced.

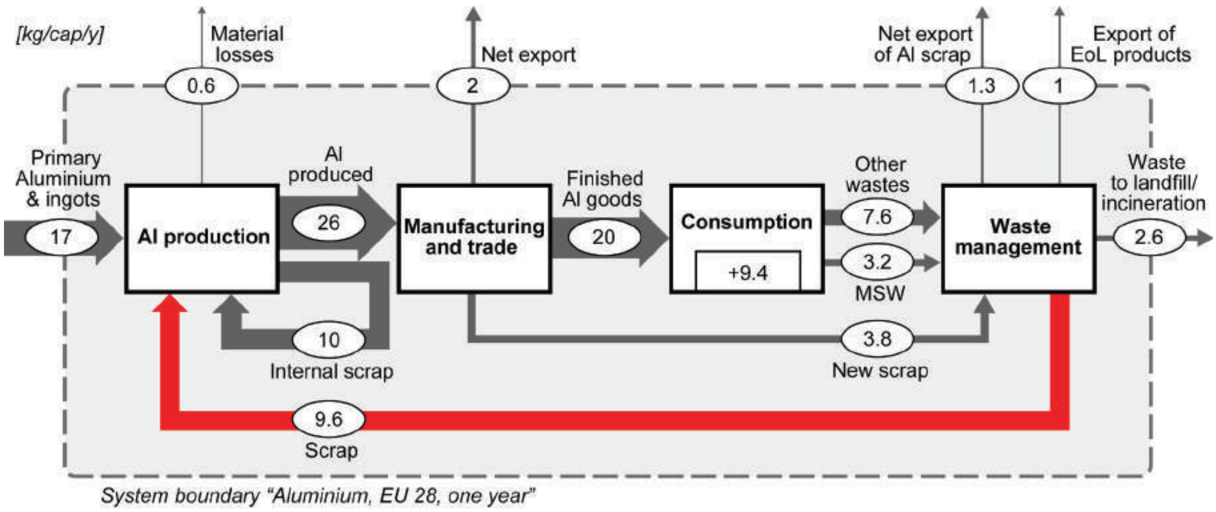

FIGURE 3: Annual Aluminium flows in the EU-28 including the stock increase in the consumption process (data given in kg/cap/y); since the data of the different sources have not been reconciled, the flows of some processes are slightly unbalanced.

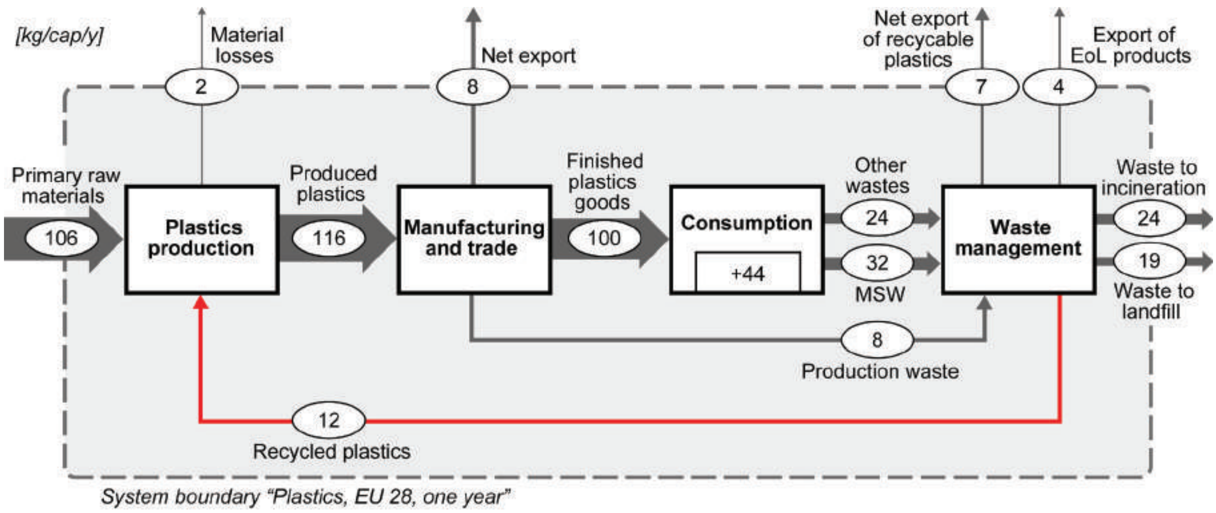

FIGURE 4: Annual plastic flows in the EU-28 including the stock increase in the consumption process (data given in kg/cap/y); since the data of the different sources have not been reconciled, the flows of some processes are slightly unbalanced.

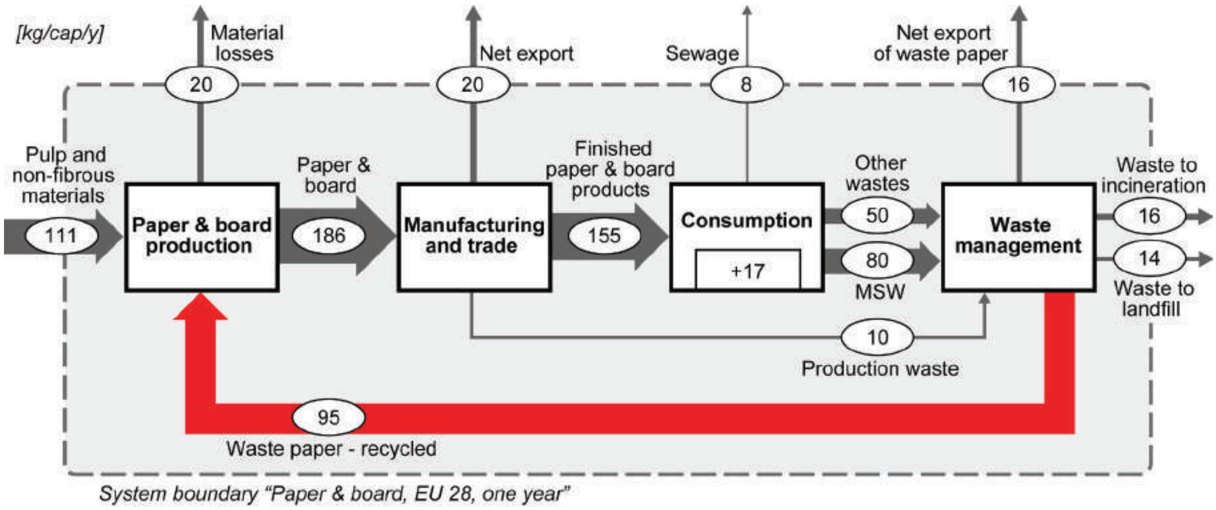

FIGURE 5: Annual Paper & Board flows in the EU-28; including the stock increase in the consumption process (data given in kg/cap/y); since the data of the different sources have not been reconciled, the flows of some processes are slightly unbalanced.

my like the European Union societies´ material flows are far from being balanced (input > output of materials), thereby limiting the overall potential of a circular economy to substitute primary resources.

Furthermore, for all four commodities a significant export of recyclables/scrap is observable. In case that besides the exports of recyclables also "hidden" material exports via end of life products are accounted for, the total "resource" losses (SSL) for the European industry amounts to approximately 10% for all four commodities. This implies that terminating exports of recyclables/scrap and end of life products could potentially increase the share of secondary production SSP in the EU by almost 10% (absolute). This is based on "highly questionable" assumptions, such as that there are no quality constraints for recycling and production facilities and capacities in place could handle this increased quantity of recyclables and scrap, respectively.

Referred to the final domestic consumption of the four commodities, the current share of supply via secondary resources SSC amounts to 54% for Iron & Steel, 48% for aluminum, 12% for Plastics and 61% for Paper & Board.

With respect to greenhouse gas emissions the current substitution rate of primary resources by utilizing waste derived materials results in total savings of 165 Mt/a (330 kg/cap/y), which represents about 3.7% of EU´s current greenhouse gas emissions.

3.2 Potential Impact of the Circular Economy Package (for MSW & packaging waste) on the Recycling Rates of Iron & Steel, Aluminium, Plastics and Paper & Board

Based on the current "commodity flows", the potential impact of implementing the circular economy package for MSW and packaging waste was assessed. Thereto it was assumed that the recycling targets proposed are met and the thereby additionally recovered recyclables / scrap (in comparison to the status quo) are fed into the European recycling market.

In Table 2 the current management of packaging waste and MSW with respect to the four commodities is summarized. It is obvious that for Iron & Steel and as well as for Paper & Board current recycling rates are already close to target values proposed by the Circular Economy Package

TABLE 2: Quantities of recyclables derived from packaging waste and MSW (current status versus fulfilment of the Circular Economy targets for packaging waste and MSW [1]).

	Unit	Iron & Steel	Aluminum	Plastics	Paper & Board
Total packaging waste & non packaging present in MSW	[kg/cap/y]	15	3.2	44	125
Current recycling rates (packaging & non packaging) [2]	[%]	77.5%	47%	23%	83%
Current quantities of recyclables derived from packaging waste and MSW	[kg/cap/y]	12	1,5	11	104
Recycling targets (according to the Circular Economy Package CEP) [3]	[%]	85%	85%	55%	85%
Additional quantities of recyclables (implementation of CEP)	[kg/cap/y]	**1.1**	**1.2**	**13**	**2.5**

[1] *For simplicity reasons it was assumed that recycling targets for packaging wastes are also applicable to the alike materials present in MSW.*
[2] *Based on EUROSTAT data about recycling rates for packaging waste by material and considering recycling rates for non-packaging material present in MSW.*
[3] *Until 2030.*

(85%), whereas for Aluminium (47% - 85%) and Plastics (30% - 55%) substantial improvements would be necessary to the met the targets. It needs to be noted that for simplicity reasons it was assumed that recycling targets proposed for packaging waste are likewise applied for non-packaging materials present in MSW.

Based on the necessary improvements (increase) of recycling rates and the quantities of materials present in packaging waste and MSW, the impact of the proposed circular economy in terms of additional quantities of recyclables / metal scrap as well as potential savings of greenhouse gas emissions were calculated.

3.2.1 Iron & Steel

For Iron & Steel the achievement of a recycling target of 85% would imply an additional amount of 1.1 kg of scrap per capita and year (or 550 kt/a for the EU-28). This represent about 0.6% of EU´s total ferrous metal scrap generation. If the total amount of additional scrap would be utilized by the European steel industry (no export of this additional scrap), the share of secondary production (SSP) would marginally increase by 0.36% to 47%. In case that also home scarp is considered, the share of secondary production for Iron & Steel would reach almost 55% (see Figure 6) In terms of greenhouse gas emissions this increased scrap supply / utilization would results in reductions of less than 2 kg CO_2eq/cap/y, neglecting the efforts for the collection / separation and sorting of the scrap.

3.2.2 Aluminum

For Aluminium an achievement of the proposed recycling target of 85% would translate in an additional scrap quantity of 1.2 kg/cap/y (or in total 600 kt/a). This would increase the current quantities of Al scrap available (9.6 kg/cap/y) by 11%. In case that all the additional quantities are utilized by European Al smelters, the share of secondary production (SSP) would theoretically increase by almost 5% to 41.5%. In terms of greenhouse gas emissions this would be associated with reductions of about 10 CO_2,eq/cap/y. However, it is highly questionable whether such raise in Al scrap supply could be absorbed by the European manufactures. It is more likely that these quantities are at least to some extent exported for recycling, as it was observable for iron scrap (see Figure 6). There higher shares of scrap in relation to the production volume, expressed as potential share of secondary production (light blue bars in Figure 6), did not translate into higher secondary production (dark blue bars in Figure 6). The “additional” quantities of scrap were not utilized by the European steel makers and hence exported.

3.2.3 Plastics

For Plastics alike to Aluminium current recycling rates (23%) would almost need to double to reach the target value of 55% proposed by the circular economy package. Such an increase would generate an additional quantity of recyclable Plastics of almost 13 kg/cap/y (6,500 kt/a). In comparison to the status quo (19 kg/cap/y), this would rise the recyclable quantities by more than 65%. Considering that already today a significant share of Plastics is exported for recycling (7 kg/cap/y), it might be questioned if huge amounts of additional quantities of recyclables will be utilized within the European Union. In general, higher collection and sorting rates will most likely lower the quality of the recyclable Plastics obtained, which further challenges the production of high quality re-granulates able to substitute primary polymers. It is more likely that the collected materials are down-cycling (e.g. mixed polymer re-granu-

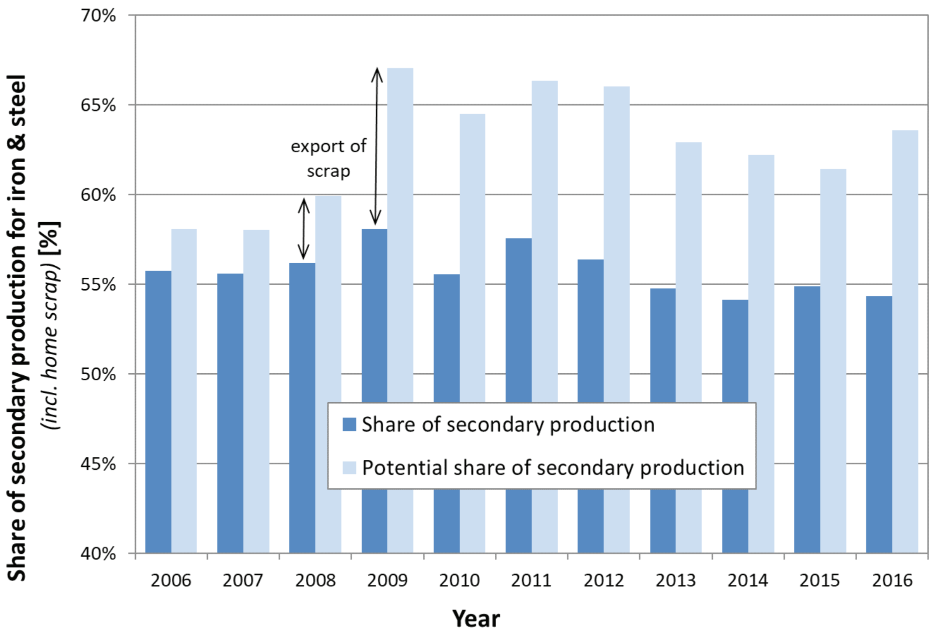

FIGURE 6: Share of secondary production for Iron & Steel (incl. home scrap) within the European Union for the period 2006 to 2016. The significant increase of the potential share of secondary production between 2008 and 2009 results from the reduction of steel production in the EU during this time (shutdown of steel production plants).

late) or exported. Hence, the potential of greenhouse gas reduction of 20 kg CO_2,eq/cap/y theoretically possible through higher recycling rates (55%) for packaging Plastics and other Plastics present in MSW will at best only partly be become effective. The "real" emission reduction strongly depends on the substitution achieved by secondary Plastics.

3.2.4 Paper & Board

Current recycling rates for Paper & Board (83%) are already very close to the proposed target values (85%). The improvements required would thus translate into an additional quantity of recyclable waste Paper & Board of only 2.5 kg/cap/y (equals 1,250 kt/a). This is about 2% of the waste paper quantity generated at present in the EU-28. Assuming that the total additional quantity of waste Paper & Board is domestically utilized, the share of secondary production would increase to 53.8% (+1.3%).

Considering the recent changes in secondary production (the share rose constantly from 48% in 2005 to 52.5% in 2015 – (see Figure 7), this moderate increase seems to be manageable for the European Paper & Board industry. Quality constrains for Paper & Board recycling – as highlighted by Pivnenko et al. (2016) – however, might limit the utilization of additional waste paper quantities. The total reduction potential for greenhouse gas emissions by increasing paper recycling rates to 85% is well below 0.5 kg CO_2,eq/cap/y.

4. CONCLUSIONS

Although the analysis was conducted on a rather superficial level (by utilizing and combining highly aggregated data), the results clearly demonstrate that space for improvement in terms of secondary resources utilization in countries with rather highly developed waste management systems in place, like the European Union, is limited and might only slightly increase the quantities of waste derived resources. In case that the ambitious Circular Economy targets for packaging waste are met and also applied to alike non-packaging materials present in MSW, the additional quantities of recyclables available amount to 550 kt/a for iron and steel (1.1 kg/cap/y), 600 kt/a for aluminum (1.2 kg/cap/y), 6,500 kt/a for Plastics (13 kg/cap/y) and 1,250 kt/a for Paper & Board (2.5 kg/cap/y). In case that these secondary materials could substitute the same amount of primary raw materials, annual greenhouse gas emissions of the European industry would be reduced by about 17 Mt CO_2,eq (equals 33 kg CO_2,eq/cap/y), which is less than 0.4% of EU´s overall greenhouse gas emissions of 4,500 Mt CO_2,eq/a. To which extent other environmental impacts are affected has not been investigated in the present study. This however is subject of ongoing investigations of the authors.

As already today significant amounts of secondary raw materials are not utilized by the European industry but exported for recycling, it is doubtful if the sole prescription of higher recycling targets will result in an increased share of secondary raw material production in the EU. Moreover, since the additional quantities of recyclables (scrap) derived from post-consumer waste are most likely of lower quality than recyclables currently recovered, the risk of down-cycling and thus limited substitution of primary raw materials and also the risk of increased export of recyclables is evident. The trend for the latter is already observable as prices of scrap and recyclables exported from the EU tend to be significantly lower than the ones of imports (Brunner, 2017; Buchner et al., 2015).

Hence, in order to ensure that higher recycling targets will translate into a higher share of secondary production and thus a reduced dependency of European industry from primary resources, additional policy measures (e.g. quota for the share of secondary raw materials utilized) might be required. Furthermore, capacities for the recycling (utiliza-

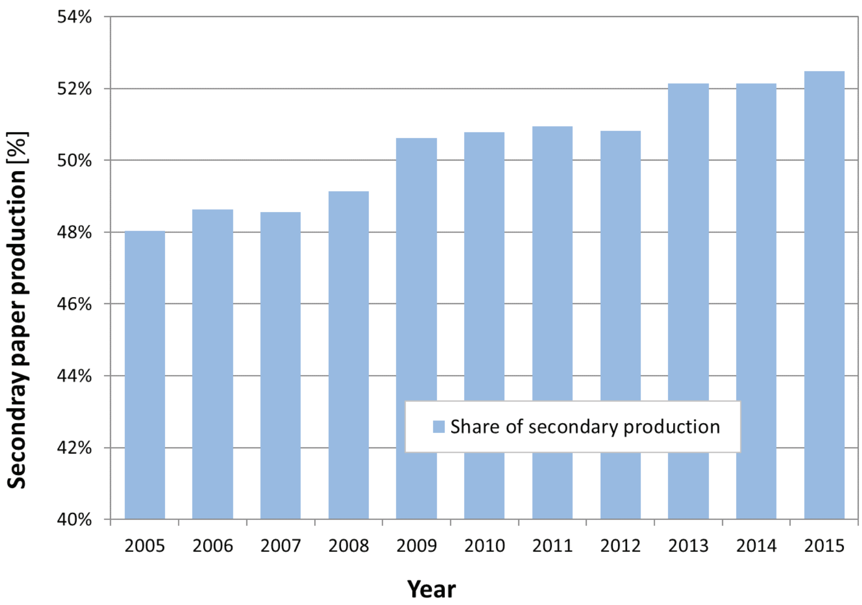

FIGURE 6: Development of secondary production for Paper & Board within the European Union during the period 2005 to 2015.

tion) of the additional quantities of secondary raw materials need to be provided, which might represent a challenge, in particular for countries of lower economic development (e.g. Bulgaria, Romania).

The present study analyzed the potential impacts of the circular economy package on the supply and utilization of secondary resources in the EU focusing on material flows only. Impacts on the economics of the commodities, which largely drive the market of secondary resources, have not been considered yet. They should be subject of future investigations.

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RESOURCES FROM RECYCLING AND URBAN MINING: LIMITS AND PROSPECTS

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ABSTRACT
Direct and indirect effects (DIERec) of the recovery of secondary resources are in the range of 500 million tons per year in Germany; energy savings are 1.4 million TJ. These savings are between 10 and 20% of the total. The effects of materials recovery exceed those of energy recovery by far except for secondary plastic material, where DIERec from energy recovery is higher by factor of 2.7. Untapped potential for the recovery of secondary resources exists in the fine fraction of bottom ash from municipal solid waste incineration, mainly Cu and precious metals, and in urban mining.

1. INTRODUCTION

Global resource consumption increased by a factor of 8 within the 20th century; the rise for metal ores was even by a factor of 18.8 (Krausmann et al., 2009). A reversal of this trend is not in sight. In Germany, direct material input (DMI, biotic, and abiotic resources) was 1,667 million tons in 2013 (Statistisches Bundesamt, 2017). Domestic extraction of raw materials (without unused extraction) was 1,060 million tons; roughly 20% of this mass accounted for by lignite for the generation of electricity. 607 million tons were imported. Extraction of gases (to balance with the output) was 1,084 million tons. On the output side are the exports (370 million tons), the dissipative losses (35 million tons), and the emissions to the atmosphere (845 million tons). The remaining 755 million tons are net addition to stock (713 million tons, e.g. in products, buildings, and infrastructure (Tanikawa & Hashimoto, 2009)) and waste to landfills (42 million tons). Net addition to stock would be zero only in a perfect circular economy with constant population and constant affluence.

DMI does not account for resources used abroad in upstream chains expended for the production of the imported materials and goods. Converted to Raw Materials Equivalents, the figure for imports would be higher by a factor of roughly 3 (calculation for the year 2011: factor 2.74) (Buyny & Lauber, 2009). DMI in raw material equivalents would be higher by 60%.

The DMI would be even higher if only primary resources were used. Fortunately, the fraction of secondary raw materials in total resource consumption is considerable. The effect of the use of secondary raw materials can be estimated with the newly defined parameter DIERec (direct and indirect effects of recovery) (Steger et al., 2018). DIERec are the total savings of resources caused by using secondary raw materials, including all upstream chains on a global perspective. The direct effects of recovery (DERec) are, in contrast, limited to the domestic scale (Wagner et al., 2012). The DERec of a material flow is the additional amount that would be necessary to generate the substituted raw materials and energy in the absence of recycling and energy recovery, without considering upstream chains of processing and production abroad.

Calculation of DIERec is based on Cumulative Resource Demand (CRD), which considers all flows of raw material (including material flows associated with energy production, but without unused extraction, water, or air) back to the source. For metallic raw materials, the reciprocal metal content in an ore can be used as a proxy for CRD. For example, the copper content in today's ores is about $5x10^{-3}$ kg/kg_{ore}. CRD_{Cu} is therefore 200 kg/kg (Simon & Holm, 2017). CRD resembles the concept of Total Material Requirement (TMR), i.e., the total mass of primary material from nature. TMR includes both materials used for further processing and hidden flows that have not been further used, but have an environmental impact, such as overburden and extraction waste (Halada et al., 2001).

The use of secondary raw materials also saves energy.

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The amount can be calculated by comparing the Cumulative Energy Demand (CED, measured in terms of primary energy) of the process routes with and without secondary raw materials, ΔCED. The concept of CED is described in detail elsewhere (Verein Deutscher Ingenieure, 2012).

2. EFFECTS OF RECOVERY IN GERMANY

The direct and indirect effects of recovery (DIERec) were estimated for Germany in a recent study (Steger et al., 2018). Selected materials categories were 1.) metals, 2.) polymers, 3.) construction and mineral materials, and 4.) biomass- or fossil fuel-based materials. For this purpose, statistical data were retrieved from trade associations (e.g. Wirtschaftsvereinigung Metalle) and the German Federal Statistical Office (destatis) on primary and secondary resource use in Germany and abroad. Total material usage for these materials was 648 million tons in 2013; material flow from secondary production was 173 million tons (sums of column 1 and 2 in Table 1). The values for CRD were calculated by a method described in detail elsewhere (Saurat & Ritthoff, 2013). The values for CRD in Table 1 are weighted mean net values for the respective material. For example, CRD for gold from primary production is 835,622 kg/kg. In secondary production, the value is 1,710 for the process of refinement and 0.06 kg/kg in precious metals separating works.

For the category of polymers (i.e., PE, PS, PET, PVC and mixed plastics), DIERec was calculated to 3,366,000 tons for materials recovery processes, i.e., substitution of primary plastic, wood, concrete, or heavy oil in the steel industry. Compared with other material flows, this figure is quite low; however, another 8,943,000 tons can be accounted for by DIERec from energy recovery using plastic waste. Here the assumption was made that 60% of the waste plastic was treated in municipal solid waste incineration plants and 40% in refuse-derived fuel plants. CRD for electricity (German electricity mix) is 0.175 kg/kg and 0.18 kg/kg for heat. The data for the individual plastic types differ substantially. For PET, the value for DIERec from material recovery was 1,440,000 tons (more than 40 % of the total) versus 348,000 tons from energy recovery, whereas the values DIERec for PP are 262,000 tons for material recovery and 2,254,000 tons energy recovery, respectively.

DIERec from materials recovery of food waste is negative because the substitution effects for nutrients (1,096,000 tons) are outweighed by the expenses for collection and treatment (1,877,000 tons). In total, a positive value results together with the DIERec from energy recovery. As mentioned above, CRD in Table 1 are weighted net values. The low (net) value for wood (0.7 kg/kg) is composed of CRD for substitution of primary wood (2.77 kg/kg) and the respective expenses incurred by collection, transport etc.

The environmental impact of resource use can be related directly to the Cumulative Energy Demand (CED) (Huijbregts et al., 2010). In the study by Steger et al. (Steger et al., 2018), the energy savings ΔCED resulting from the use of secondary resources was also calculated. The results are listed in Table 2. All values for ΔCED are related to materials and energy recovery. Energy savings from the materials recovery processes of plastic waste account for 110,346 TJ only and were even negative for food waste, at -3,384 TJ.

Using secondary resources for Zn production requires more energy than in a complete primary production. This is visible in the negative ΔCED value of -6,924 TJ. The reason is the energy-consuming treatment of Zn-containing residues, e.g. in the Waeltz process or in hydrometallurgical processes. The case of recycling of Zn from flue gas cleaning residues was discussed in detail in the work of Fellner et al. (Fellner et al., 2015). It was shown that recovery of Zn is economic only under opportune circumstances such as extremely high Zn concentrations. However, any heavy metal recovery process at least avoids landfilling of otherwise problematic residues (Gehrmann et al., 2017).

The Direct and Indirect Effects of Recovery (DIERec) in Germany were in the range of 500 million tons in the year 2013, more than 95% of this as a result of materials recovery, in contrast to energy recovery. This means that the raw material input of Germany (i.e., the DMI in terms of raw material equivalents (Buyny & Lauber, 2009)) would be higher by approx. 18%.

3. WASTE MANAGEMENT OPTIONS

The management of municipal solid waste (MSW) is an important part of the recovery of resources. In 2013, some 50 million tons of MSW were collected and delivered to different treatment processes. Landfilling of untreated waste stopped in Germany in 2005. Which treatment process is the best in view of resource recovery aspects and environmental compatibility is the subject of ongoing discussions. A variety of evaluation methods for waste treatment processes exist (Gehrmann et al., 2017). In a recent publication, four different waste management options - 1) disposal on landfills, 2) treatment in mechanical-biological treatment (MBT) plants, 3) treatment in municipal solid waste incineration (MSWI) plants, and 4) separate collection and individual treatment – were compared with regard to the potential of the reduction of greenhouse gas emissions (Wünsch & Simon, 2018). The worst option regarding greenhouse gas emissions was option 1), with net emissions of 0.239 tons $CO_{2,\,eq}$ per ton of waste. Maximum reduction was achieved by option 4.), with -0.129 tons $CO_{2,\,eq}$ per ton of waste. This is in good agreement with the results from the DIERec study (Steger et al., 2018). Apart from the data for metals and large-volume waste streams such as recycled construction waste, DIERec for paper is among the highest values of all (27.1 million tons). This is possible through the separated collection of paper waste and subsequent treatment in the paper industry. Equally simple is materials recovery in the case of glass (DIERec 3.8 million tons, ΔCED 17,045 TJ). In contrast, separate collection of mixed plastic waste does not lead to a large effect in materials recovery; here energy recovery prevails. Options 2) (MBT) and 3) (MSWI) had reductions of -0.015 and -0.039 tons of $CO_{2,\,eq}$ per ton of waste, respectively.

A continuous source of secondary metals, which is exceptionally visible in the public eye, is MSWI bottom ash

TABLE 1: Material flow from secondary production in Germany in 2013. Cumulative Resource Demand (CRD) can be used to calculate the Direct and Indirect Effects of Recovery (DIERec). All data are derived from (Steger et al., 2018).

	Total material usage (tons)	Material flow from second. production (tons)	CRD (kg/kg)	DIERec from material recovery (10^3 tons)	DIERec from energy recovery (10^3 tons)
Metals					
Steel, iron	42,645,000	23,031,900	6.1	139,545	-
Al	2,980,000	877,000	21	18,546	-
Cu	1,389,000	673,000	194	130,292	-
Stainless steel	1,091,000	412,000	55	22,723	-
Zn	652,000	242,000	7.2	1,746	-
Pb	367,000	134,000	29	3,872	-
Sn	24,230	6,000	1,243	7,459	-
Ag	4,135	479	8,985	4,304	-
Pt	21	9.5	389,250	3,114	-
Pd	21	8.0	47,263	449	-
Au	40	7.6	835,000	6,346	-
Polymers	19,800,000	6,233,000	2.0	3,366	8,943
Construction & mineral materials					
Recycl. constr. mat.	497,000,000	52,700,000	1.0	55,066	-
Asphalt granulate	41,000,000	11,500,000	1.1	12,137	-
Steel slags		10,540,000	1.5	15,858	-
Power plant residues		8,547,000	1.1	9,493	-
Glass cullet	3,900,000	2,470,000	1.5	3,811	-
Biomass & fossil fuel-based materials					
Paper	19,982,000	16,489,000	1.9	27,121	3,421
Food waste, compost		26,331,000	0.02	-782	1,413
Wood	16,937,000	11,230,000	0.7	1,431	6,710
Textiles		1,083,000	1.8	1,698	270
Tires	550,000	442,000	2.2	722	232

(BA). Around 5 million tons of MSWI BA were generated per year in Germany. The incineration itself serves as a concentrating and cleaning process for metals. However, due to the almost exclusive wet extraction out of the furnace chamber, the various metals are integrated in a heterogeneous and unstable matrix. The metal recovery is therefore still a challenge in terms of both recovery rate and purity (Holm & Simon, 2017). With state-of-the-art treatment trains in Germany, around 7.7% of ferrous metals and 1.3% of nonferrous metals can be recovered from MSWI BA (Kuchta & Enzner, 2015).

Today's focus in the recovery of nonferrous metals from municipal solid waste lies on Al and Cu alloys, which can be easily separated from a waste stream using eddy current separators. In 2013, 42,000 tons were recovered from MSWI BA, another 10,000 tons came from MBT. Together with aluminum scrap from other sources and post-consumer waste, 597,000 tons of secondary Al were produced in Germany in 2103. However, the complete material flow of aluminum in Germany was almost 3 million (2,980,000) tons. DIERec for the use of secondary Al is 18.5 million tons; ΔCED is 166 PJ (see tables). The reason for the high value of DIERec is the energy consumption of primary Al production, mainly of the fused-salt electrolysis step, which is not needed in Al scrap smelting. Material flows associated with energy production are included in DIERec. The conservation of natural resources is less relevant, because bauxite, the raw material for Al production, is a ubiquitous resource. This is different for copper. Production of secondary Cu is advantageous due to lower energy consumption and the conservation of ores, which have displayed a decreasing metal content over the last decades (Simon & Holm, 2017).

Whereas the recovery rate for ferrous metals from MSWI BA is appropriate, the rate for nonferrous metals could be distinctively raised by carrying out innovative treatment trains and taking into account the fine fraction < 2 mm. Particularly, the fraction of 1 to 2 mm contains an appreciable amount of nonferrous metals, so the recovery rate may be doubled in the future (Holm & Simon, 2017). Whereas in coarser grain sizes, mainly Al, Cu, and brass occur, the fine fraction holds also precious metals (Holm et al., 2017; Morf et al., 2013; Muchova et al., 2009). Unpublished operation data (Gronholz, 2017) regarding refined heavy fractions of nonferrous metals (generated out of the grain size distribution 2 to 18 mm) confirm the smelters'

TABLE 2: Material flow from secondary production in Germany in 2013. Total ΔCED can be calculated from the specific Cumulative Energy Demand. All data are derived from (Steger et al., 2018).

	Material flow from second. production (tons)	Specific CED (MJ/kg)	Total energy savings (ΔCED) (TJ)
Metals			
Steel, iron	23,031,900	5.7	130,622
Al	877,000	189	166,031
Cu	673,000	53	35,935
Stainless steel	412,000	56	23,243
Zn	242,000	-28.6	-6,924*
Pb	134,000	13	1,759
Sn	6,000	318	1,905
Ag	479	8,232	3,943
Pt	9.5	358,526	3,406
Pd	8.0	82,000	656
Au	7.6	238,026	1,809
Polymers	6,233,000	34.9	217,682
Construction & mineral materials			
Recycled constr. mat.	52,700,000	0.1	3,302
Asphalt granulate	11,500,000	1.5	17,045
Steel slags	10,540,000	2.3	24,129
Power plant residues	8,547,000	0.8	6,878
Glass cullet	2,470,000	9.0	22,293
Biomass- & fossil fuel-based materials			
Paper	16,489,000	32.1	529,671
Food waste, compost	26,331,000	0.5	14,344
Wood	11,230,000	12.5	140,563
Textiles	1,083,000	22.6	24,447
Tires	442,000	40.9	18,082

** A negative value indicates an increase in energy consumption instead a saving.*

routine revenue for silver and gold (24 batches of 21.1 to 24.8 tons, i.e. more than 500 tons). Although the share of Cu in these batches averaged 64%, the revenue for silver (2,672 ppm, variance 11%) and gold (90 ppm, variance 31.4%) together exceeded the revenue for Cu (Gronholz, 2017). Assuming realistic shares of around 70% for Al and 20% for Cu in the nonferrous metal fraction (optimistically ~2.5% of the whole MSWI BA), the contribution of secondary production to the material flow in Germany would be 87,500 and 25,000 tons per year for Al and Cu, respectively (Allegrini et al., 2014; Holm & Simon, 2017).

4. URBAN MINING

The largest output fraction in the German material balance is the net addition to stock, i.e., 755 million tons in 2013. Most of this are construction materials, which are used partly for long-lasting buildings and infrastructure. Other components of the stock are vehicles and consumer goods. For Vienna, it was found that the copper stock (178 kg per capita) is distributed to 45% to buildings, 35% to infrastructure, 8% to vehicles, and the remaining 12% to consumer goods (Kral et al., 2014). Thus, for copper, the building sector is the largest part of the urban mine.

For Vienna, a saturation was found for copper with an annual increase of 2% only, differing from Taipei with its increase of 26% per year to a today's much smaller copper stock of 28 kg per capita. Cities, in particular, therefore display a large reservoir for resources. In the future, this reservoir could be reused instead of geogenic ores; this is named urban mining. Baccini and Brunner define urban mining as the exploration and exploitation of material stocks in urban systems for anthropogenic activities (Baccini & Brunner, 2012). They estimate the stock at 300-400 tons per capita (in cities). However, little is known about the exact quantity and quality. Also, separation and recovery methods in urban mining are still underdeveloped. Recovery of metals in urban mining is easier than recovery of minerals. In demolition projects, steel and copper can be separated in high purity, whereas the recovery of the mineral fraction is less straightforward and often ends up in downcycling. The same is true of the recovery of polymers from long-lasting products. End-of-life plastic is usually used as a fuel instead of a raw material or feedstock for polymer production, as shown in chapter 2.

5. CONCLUSIONS

The direct and indirect effects of recovery in Germany are in the range of 500 million tons per year when resources used abroad in upstream supply chains are considered. Without the material flow from secondary production, the raw material input would be higher by almost 20%. Energy savings from the use of secondary resources (1.4 million TJ in sum) are around 10% with regard to primary energy consumption in Germany. These figures are impressive; however, they show that a circular economy with no or only small additional input from primary resources is far off, even in mature industrialized economies like Germany with a fully developed recycling infrastructure. Resource consumption is still related to economic wealth. According to the well-known IPAT equation (I=P x A x T), the environmental impact of resource consumption (I) is the product of population (P), affluence (A), and a technology term (T=1/efficiency) (Ehrlich & Holdren, 1971). As long as P and A increase, the only way to reduce environmental impact is technology that is more efficient. For the management of waste and an approach to urban mining, this is equivalent to an increase in separation efficiency. To close the loop, improved separation efficiency is needed in waste management and, more importantly due to its huge volume, in the construction sector (Maletz et al., 2018).

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LANDFILL MINING: A CASE STUDY REGARDING SAMPLING, PROCESSING AND CHARACTERIZATION OF EXCAVATED WASTE FROM AN AUSTRIAN LANDFILL

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New mine

ABSTRACT

The following case study belongs to the New-Mine project and the objective of the project is to develop a new "Enhanced Landfill Mining" (ELFM) scenario for a combined resource-recovery and remediation strategy. This strategy could reduce future remediation costs and reclaim valuable land while simultaneously unlocking valuable resources. In the past, insufficiently reliable data about the composition of landfills, overestimation of the quality of excavated material and poor product marketing of the possible recyclables have resulted in a bad reputation for landfill-mining projects. The ongoing research in the NEW-MINE project shall show that there are possibilities to create valuable outputs from landfills with enhanced treatment processes, such as a better distribution of the different mechanical processes. To create mechanical routes to recover valuable materials from old landfills, it is important to characterize the material, creating a basis for the research. The objective of this case study, executed from November 2016 until June 2017 at the landfill site in Halbenrain (Austria), is to study the efficiency of different sorting technologies with old landfill material. The excavated material was transported and used as feedstock in a configured state-of-the-art mechanical-biological treatment (MBT) plant located next to the landfill. During the mechanical processing, metals and high-calorific fractions were sorted out from the input flow. As a result of the mechanical processing, approx. 3% of the ferrous metals were recovered, approx. 20% of potential RDF (pRDF) was separated and could have been energetically recovered, and approx. 74% belonged to the finer fraction (< 40 mm). Each sample from the sampling campaign was sieved to obtain the particle size distribution. Via manual sorting, the material was classified into plastics, wood, paper, textile, inerts, Fe metal, NF metals, glass/ceramic and residuals. In addition, the moisture (wt%), the ash content (wt%), the calorific value (MJ/kg) and the concentration of heavy metals (%) of the finer fraction (<40 mm) were analysed. The aim of this study is to assess the possibilities of different mechanical processes with landfill mining (LFM) material and to gain information about the characterization of five material flows derived from the mechanical treatment, together with the mass balance of the MBT. Although every landfill has its own characteristics, the results obtained from this case study can help to understand the general potential, contribute to develop methodologies for characterization of old landfill material and identify problematic fields that require further research.

1. INTRODUCTION

As the world population increases, the generation of municipal solid waste (MSW) is increasing, and landfills continue to be filled with recyclables that could be used otherwise as raw materials or for energy recovery. The situation is even more critical if we look back in time. Before the European Directive 1999/31/CE was implemented and defined different categories of waste, MSW could have been mixed and buried without treatment/sorting according to local legislation. Moreover, insufficient disposal charges for landfilling did not prevent the negative impacts of landfills, either.

In the 1970s, there was a period of rapidly increasing raw material prices and rising concern about finite natural resources. Several studies forecasted serious shortages by the end of the century. Recycling of household waste was considered a partial solution to the problem. There

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was also increasing awareness regarding the negative impact of simple dumping without appropriate barrier systems or pretreatment prior to landfilling. Since then, many technological advances have been developed to produce refused-derived fuels (RDF) and to separate recyclable materials from residual MSW (Ferranti et al. 1985). However, despite the efforts in the last 40 years to improve the situation, landfilling is still the most common method of organized waste disposal in Europe, according to Eurostat.

In 2014, Europe treated 2319 mill t of MSW by six treatment operations, defined in the Waste Framework Directive 75/442/EEC: 41% disposal on land, 36% recovery (other than energy recovery - except backfilling), 10% recovery (backfilling), 7% land treatment/release into water, 5% energy recovery, and 1% incineration (Eurostat, 2014). Figure 1 shows four of six management systems for MSW in Europe by country.

Currently, Europe accumulates between 150.000 and 500.000 old landfill sites, of which approximately 10% meet the EU Landfill Directive requirements and are considered sanitary landfills. In most cases, non-sanitary landfills lack the required environmental protection technologies and will eventually demand costly remediations to avoid future problems (NEW-MINE, 2016). Due to the existence of those non-sanitary and sanitary landfills, possible valuable resources are being lost and concurrently the environment and human health damaged. Therefore, remediation strategies for existing landfills are fundamental in the direction to preserve resources, environment and human health. A further argument to recover the valuable resources is the crisis that concerns the economic situation and the energetic matrix, which is mainly based on fossil fuels and water energy, where prices for energy and secondary resources are increasing steadily.

The composition inside a landfill generally depends on different parameters, such as waste regulations and legislation, differences in the waste management systems, recycling systems, standard of living and the society and culture of the setting (Quaghebeur et al., 2013). Proper investigations of each site, including the operation history, waste type dumped, dimensions of the landfill, topography and physical-chemical analyses, are necessary to make a careful feasibility analysis about the material potential inside the landfill (Salerni, 1995). Apart from considering the material potential, a critical factor to take in consideration before starting an ELFM project is the quality of the materials to recover and the market price, which varies over time and region.

The present work belongs to the New-Mine project, supported by the European Commission since September 2016, in collaboration with another landfill mining project of FCC at the landfill site of Halbenrain (Austria). The scope of the project is to transform a large fraction of old excavated LFM into higher-added-value products. The project is designed to combine a remediation strategy with the recovery of resources, as seen in Figure 2.

The purpose of this study is to provide foundational knowledge of the composition and characteristics of excavated material from a specific MSW and Industrial landfill, which is important for the sizing of mechanical sorting. Moreover, this paper aims to assess the possibilities of different mechanical processes with landfill mining (LFM) material and to gain information about the characterization of five material flows derived from the mechanical treatment, together with the mass balance of the MBT. The novelty of this research is the biological treatment (drying stage) prior to mechanical treatment, in addition to the use of a complete MBT plant, which differs from other studies in which mobile machinery is applied.

Although every landfill has its own characteristics, the results obtained from this case study can help to understand general potentials, contribute to develop methodologies for characterization of old landfill material and identify problematic fields that require further research.

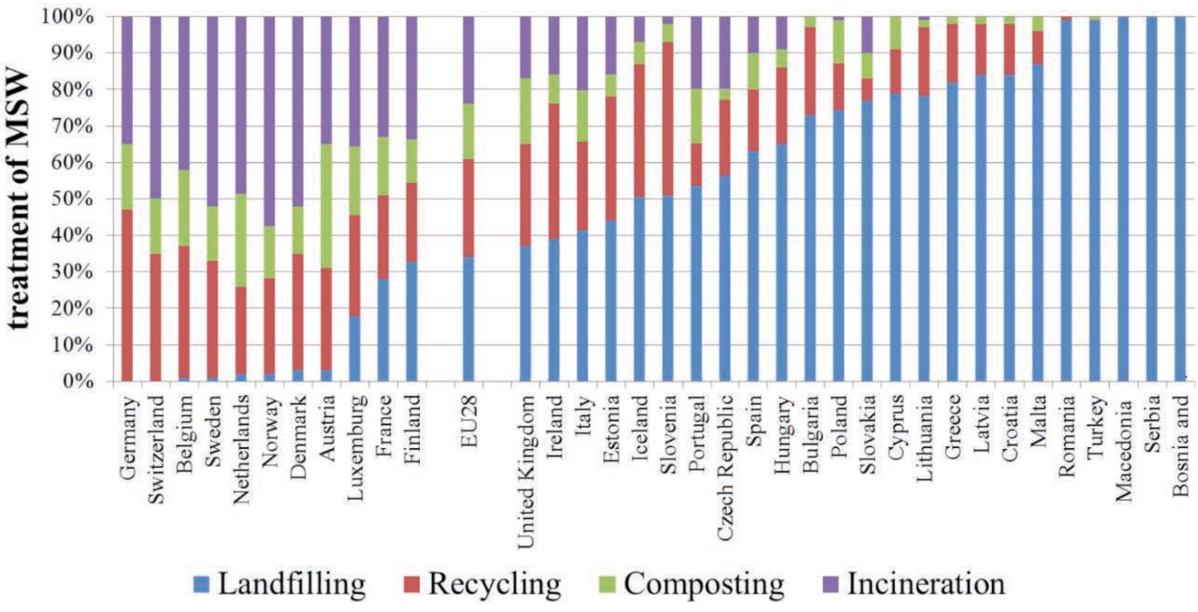

FIGURE 1: Relevance of the main MSW management systems in the EU-28 in 2012 (Source: Eurostat, 2014).

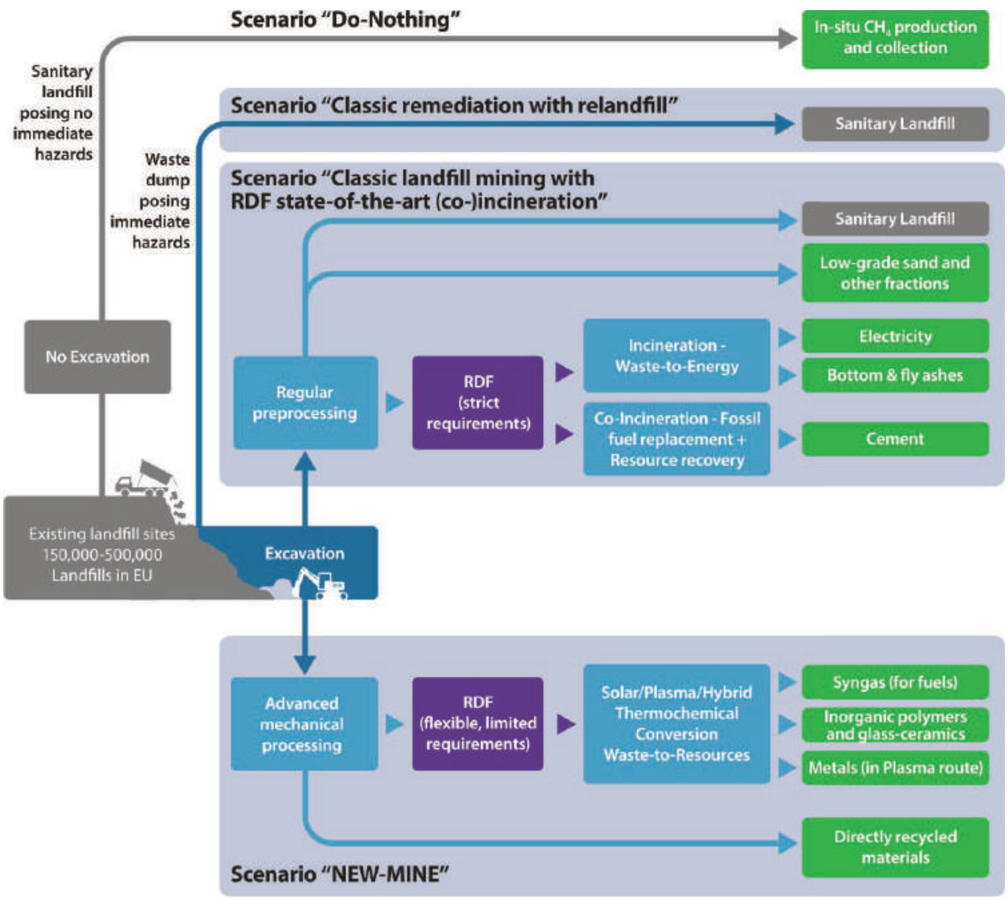

FIGURE 2: Comparison of different scenarios for the EU's landfills, Do-Nothing (only acceptable for well-monitored sanitary landfills), Classic Remediation (where the materials are excavated and re-landfilled), and Classic Landfill Mining coupled with (co)incineration, and the NEW-MINE, ELFM Scenario (Source: EURELCO www.new-mine.eu).

2. MATERIAL AND METHODS

2.1 Site description

The landfill site in Halbenrain belongs to FCC Halbenrain Waste Treatment Centre and is located 75 km south-east of Graz (Austria). The landfill was established in 1978 and received MSW and industrial waste. Currently, the examined area is in the post-closure phase, and it has an extension of 16 ha, with a total volume of 2.4 million m^3 of waste. The site has developed over the years to include a waste disposal facility with leachate treatment, conversion of landfill gas into electricity, composting, sorting and mechanical-biological waste treatment.

2.2 Excavation and processing at the site

In June 2016, FCC initiated a landfill-mining project on site with the aim of recovering metals disposed between 1997 and 1999. Based on records about the landfill composition, eight areas of interest were estimated to contain a relatively high percentage of recyclable materials, being metals of great interest for ELFM. The material examined during the case study was extracted of the projected area marked in red (Figure 3), approx. 20x20x10 m, at a depth of 6 m.

The mining activity included the following steps: 1) excavation (Figure 4), 2) transportation to a Mechanical and Biological Treatment (MBT) plant, 3) biological treatment and 4) mechanical treatment with potential Refused Derived Fuels (RDF) separation and metal recovery.

During the case study, two batches (batch 1: 220 t, batch 2: 280 t) were excavated, treated and characterized.

Normally, MBT plants stand at the beginning of an efficient waste treatment process. By using a selective treatment process, unsorted waste can be separated into different fractions, which then can undergo further treatment (e.g., potential RDF) or be used for material recovery (metals). The design of these processes must be adapted to national regulations and market situations to be successful.

After the excavation, the LFM material was sent to an already-existing and configured MBT plant on site, which is used to treat fresh household waste. As a first step of the MBT process, the material was dried by aerobic activity (rotting boxes for 3-4 weeks). During this treatment, a loss of water of approx. 10 wt% and a possible organic matter reduction were achieved. Once the material was stabilized, it was sent to the full-scale mechanical process.

The semi-dry material was fed to a single shaft shredder that reduced the particle size of the material to 250 mm. The shredder also transformed the input material to a uniform size and eliminated overlengths that could interfere with the successive process. Afterwards, an overbelt magnet separated the large pieces of ferrous metals. After

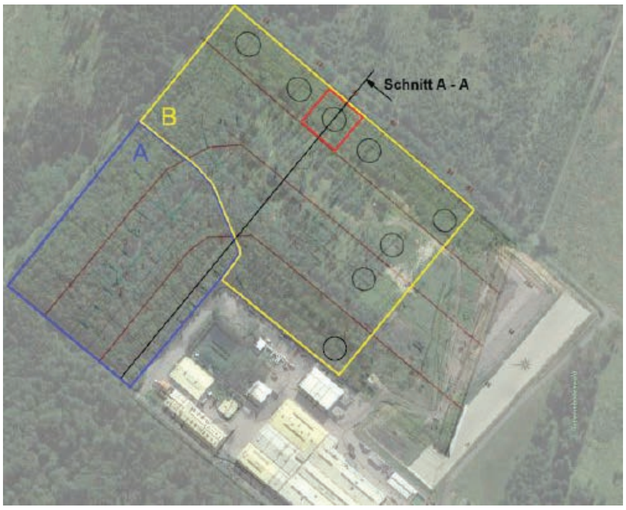

FIGURE 3: Overview of landfill site Halbenrain; A" filled 1979-1990; "B" filled after 1990; black circles mark especially interesting areas with higher amount of metals; projected area marked in red.

FIGURE 4: Excavation of LFM material from late 1990s at the landfill site in Halbenrain (Austria).

the magnet, the first sampling point (SP1) was established; see Figure 5.

Due to the plant design, the input material (prior the magnetic separator) was not accessible as a sampling point, and the Fe-metal concentration (wt%) sorted before the SP1 was calculated by measuring the weight of the pile after processing the batch of excavated waste. After passing the magnetic separator, the material was screened with circular vibratory screens with a mesh size of 60 mm (S1). The screen enabled the enrichment of metals and high calorific fractions – referred as potential RDF in the following – in the coarse and to concentrate the biogenic material in the underflow.

The material flows of the 250-60 mm and <60 mm fractions were further processed and characterized:

1. The overflow, 250-60 mm, was sorted with an additional overbelt magnetic separator (MS2) and screened at a diameter of 200 mm (S2). The fines from the 200-mm screen (fraction: 200-60 mm) were treated with a zigzag windsifter to separate the light and heavy fractions of the flow. The coarse fraction, 250-200 mm, was directly balled, together with the light fraction of the windsifter.
2. The underflow, <60 mm, was stored for further treatment. The results are not reported in the present publication.

2.3 Sampling campaign

The selection of a method for representative sampling is one of the most difficult decisions regarding waste flow analysis due to the material's heterogeneity. Normally, the material composition inside the landfill is variable, depending on the digging point chosen for the excavation. The sampling campaign was designed in order to obtain reliable information about different techniques of waste segregation (Figure 6).

In this study, the material was organized by batches, and during the processing of the whole batch, several samples were taken at different times. The sampling points, labelled SPx in Figure 5, were directly conveyor belt discharges. The number (n) of the single samples in each SPx was based on the German directive LAGA PN 98 - procedures for physical, chemical and biological testing in connection with the recovery/disposal of waste. LAGA PN 98 defines that the number of samples depends on the total quantity (m3) of the material flow (see Appendix A: Tables). The mass of the sample depends on the maximum diameter, Dmax, of the particle size (Formula 1) according to LAGA PN 78.

$$\text{Single sample in kg} = D_{max}\,[\text{mm}] \text{ x } 0{,}06 \text{ kg} \qquad (1)$$

During the sampling campaign, two batches of ~230 t/batch of semi-dried landfill material were characterized.

2.4 Mass balance

The mass balance of the MBT plant in Halbenrain was calculated based on the batches that fed the MBT plant. The weight of both input and outputs were measured.

2.5 Characterization of LFM material

The outcome of the characterization helps to study the potential of landfills for raw materials. In addition, it provides data about the effect of MBT technology with old LFM waste. It has to be noted that the facility used during this case study was not designed for treating old landfilled material. Therefore, the choice of the technology in the process was not perfectly optimized for the treatment, and the results of the sorting should be interpreted with care.

The quantity and quality of the LFM material after each mechanical process were determined based on the particle size distributions in each material flow, classifying each sample by hand (Table 1) and analysing physical and chemical characteristics of the fine fractions (<40 mm).

2.5.1 Bulk density

The bulk density of each material flow at SP1, SP2, SP3,

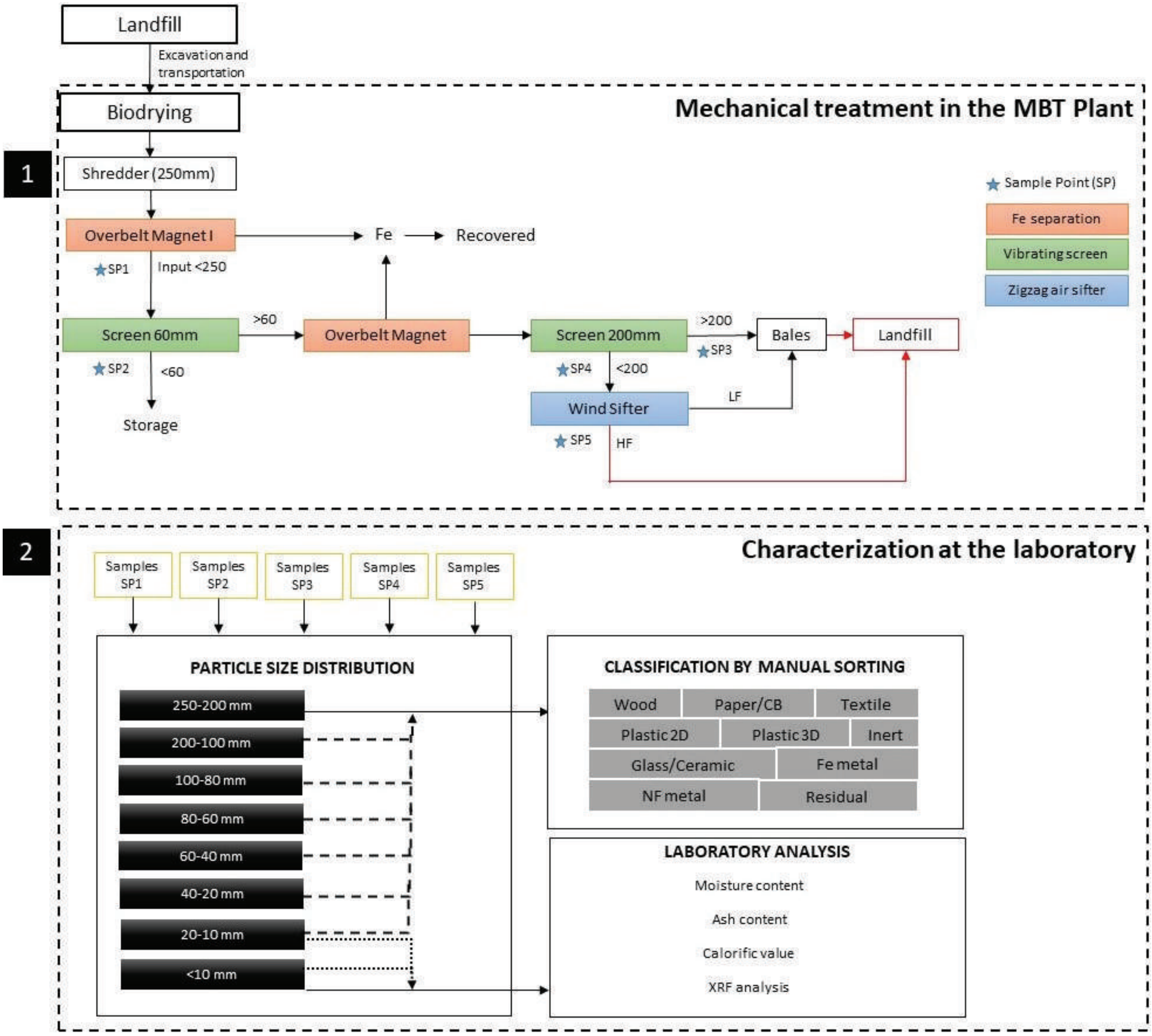

FIGURE 5: Methodology for the characterization of the LFM material in Halbenrain (Part 1: flowchart of the MBT process at the site with sampling points (SPx); Part 2: characterization of the smaller fractions < 40 mm in the laboratory).

FIGURE 6: Input material after a shredder and an overbelt magnet, SP1 (left) and fines from the first screen 60 mm, SP2 (right).

SP4 and SP5 was computed by taking the mean of several filling recipients of 90 L directly from the flow and measuring its weight.

2.5.2 Particle size distribution analysis

Each composite sample from each sampling point was individually sieved for 90 seconds using drum sieves with the following mesh sizes: 200, 100, 80, 60, 40, 20, and 10 mm. The particle size distribution helps to determine in which fractions of the flow desired materials are concentrated.

2.5.3 Manual sorting

The outcome of the drum sieve were eight fractions: >200 mm, 200-100 mm, 100-80 mm, 80-60 mm, 60-40 mm, 40-20 mm, 20-10 mm and <10 mm. Subsequently, all fractions >10 mm were sorted manually and classified into 10 categories (Table 1).

2.6 Characterization at the laboratory: physical-chemical analysis

The fine fractions <40 mm (40-20 mm; 20-10 mm; <10 mm) were reduced in mass based on the German standard given in the previous chapter 2.3 and delivered to the RWTH Aachen University. Further analysis was performed with the objective to estimate the waste-to-energy characteristics (moisture content, ash content, calorific value and heavy metal concentration; Jani et al. 2016). To be able to analyse the last three parameters, the particle of the samples had to be mechanically reduced to <2 mm, and each sample (<40 mm) was separated into 3 subcategories: light fraction (LF), heavy fraction or rest (HF) and metals.

2.6.1 Moisture content

The moisture content of waste is closely related to the amount of organic matter, and it differs with the habits of the population. In the EU and the USA, it ranges from 20-30%, whereas values in China are from 30-60% due to the higher content of kitchen garbage (Jani et al., 2016).

Looking at existing landfills, different factors affect the moisture content, e.g., the type, composition and properties of the waste, climatic conditions, landfill operating system and soil cover layer (Hull et al., 2005). The moisture content is important when considering the recycling of waste to produce energy through biological and/or thermal treatment (Brunner and Rechberger, 2015), in addition to for sorting the material during the mechanical pretreatment.

The standard DIN EN 14346:2007 "Characterization of waste – Calculation of dry matter by determination of dry residue or water content" suggests to dry the samples at 105 °C. However, volatile fractions would also evaporate at this temperature, making certain plastic particles melt, thus resulting in a less precise analysis. Therefore, the moisture content was determined by drying the samples of each material flow in a ventilated oven at 75°C until reaching a constant temperature.

TABLE 1: Classification by categories.

Category	Material
Wood	All types of wood
Paper	Paper/cardboard/composite carton
Textile	All types of textile
Plastic 2D	Aluminum package/bags (transparent/white/colored)
Plastic 3D	PP/PET/PET Oil/PEAD/PEBD/PVC/PS/Others
Fe metals	Iron
NF metals	Copper/Aluminum can/steel
Inerts	Mineral fraction (stones)
Glass	Colorless glass/green glass/brown glass/others
Residual	Sanitary material, rubber, foam, silicone, melted plastics, sandpaper, electronic plates, hazards, undefined

2.6.2 Particle size reduction (pretreatment)

After the drying process, a reduction of the particle size of three fractions (40-20 mm, 20-10 mm and <10 mm) was necessary to analyse the calorific value and organic content and to determine the heavy metals. Each sample was classified with an air sifter into a light fraction (LF) and a heavy fraction (HF). The metals contained in the sample were sorted previously using a magnet to avoid damage caused by further processing machines. In general, the LF had larger particle sizes than the remaining inerts in the flow. By sieving, using a mesh size of 2 mm, the LF could be freed of the majority of the inerts. The HF was crushed with a hammer mill and afterwards grinded in a disk mill. In the case of LF with a particle size >2 mm, a cryogenic comminution method using liquid nitrogen was used to reduce in size flexible materials such as 2D plastics. The result of comminution was a powder or flakes with sizes <2 mm for the LF and <1 mm for the HF.

This reduction (pretreatment) is needed to provide relatively homogenic material fractions in comparison to the initial material for further tests (ash content, calorific value and XRF analysis). The mass of each fraction (LF, HF/rest and metals) was measured to consider the share of each, in wt%.

2.6.3 Ash content

The ash content/organic content was calculated using 1 g of sample in each fraction, 40-20 mm, 20-10 mm and <10 mm (from SP1), and analysed by using a muffle furnace according to DIN EN 14775.

The volatile compounds calculated as the difference between the initial weight in the test and the weight of the remaining solids after the incineration is an indicator of the organic matter content.

2.6.4 Net calorific value

The calorific value was determined for the same samples as for the organic content. In this case, a bomb calorimeter was used according to the standard DIN 51900. The test consisted of complete combustion with oxygen of approx. 0.5 g of a dried sample in a bomb with a pressure of 40 bars. The heat transmitted to the surrounding water was measured and the net calorific value calculated accordingly.

2.6.5 XRF analysis

The content of heavy metals was determined with a

Niton™ XL3t X-ray fluorescence (XRF) spectrometer. For the analysis, the samples of each fraction (< 10 mm, 10-20 mm and 20-40 mm) of the sampling points (SP1-SP5) were analysed eight times. Afterwards, the mean values for all samples and the standard deviation were computed.

3. RESULTS AND DISCUSSION

3.1 Mass balance of the MBT plant with LFM material

In total, 2786 t of excavated material were treated mechanically, and 3% of Fe metal was recovered. The rest of the output flows with possible recoverable materials were landfilled again. Figure 7 depicts the mass balance of the MBT plant.

3.2 Characterization of the material flows

3.2.1 Bulk density

The bulk density depends on both the density and arrangement (compaction) of the particles in the flow. For instance, the bulk density in the input material flow (SP1), with a particle size 250-0 mm, was 0.25 t/m^3 (see Table 2), whereas higher-bulk-density material was found in the fine fraction (<60 mm) of the 60-mm screen, with 0.62 t/m^3. The coarses "250-200 mm" had a lower bulk density (0.05 t/m^3) compared to the fines "200-60 mm" (0.18 t/m^3). This result can be explained with the enrichment of light material types such as foils in the coarses and an increased amount of e.g., inerts and wood in the fines. These shifts in the composition can be explained as a result of the effects of the screening has on the material flow.

TABLE 2: Mean values of the bulk density at each sampling point (SP1-SP5) of the MBT plant.

Sampling point	Bulk density t/m^3
SP1	0,25
SP2	0,62
SP3	0,05
SP4	0,18
SP5	0,27

3.2.2 Composition of the material flows

Input flow of the mechanical treatment (<250 mm) (SP1). The composition of the input flow is decisive for the rest of the flows in the plant, since this is the raw material that shall be classified, sorted by density, magnetism, induction, etc. Figure 8 shows the average composition (wt%), where the largest proportion is the fine fraction (<10 mm), with approx. 50% of the total mass. The fines (<10 mm) are mainly soil, glass shards and mineralized organic matter. In chapter 2.3., "Physical-chemical analysis", one finds a detailed characterization of this fraction in the input flow (SP1). The content of ferrous metals (Fe), as presented here, must be considered with care, since it is only representative for the material flow after the first magnetic separation unit (overbelt magnet).

It must be considered that the weight of most defilements (fine particles) remains on the manual sorted fractions (wood, paper, plastics, etc.). These defilements not only have the effect of changing the mass balance but can

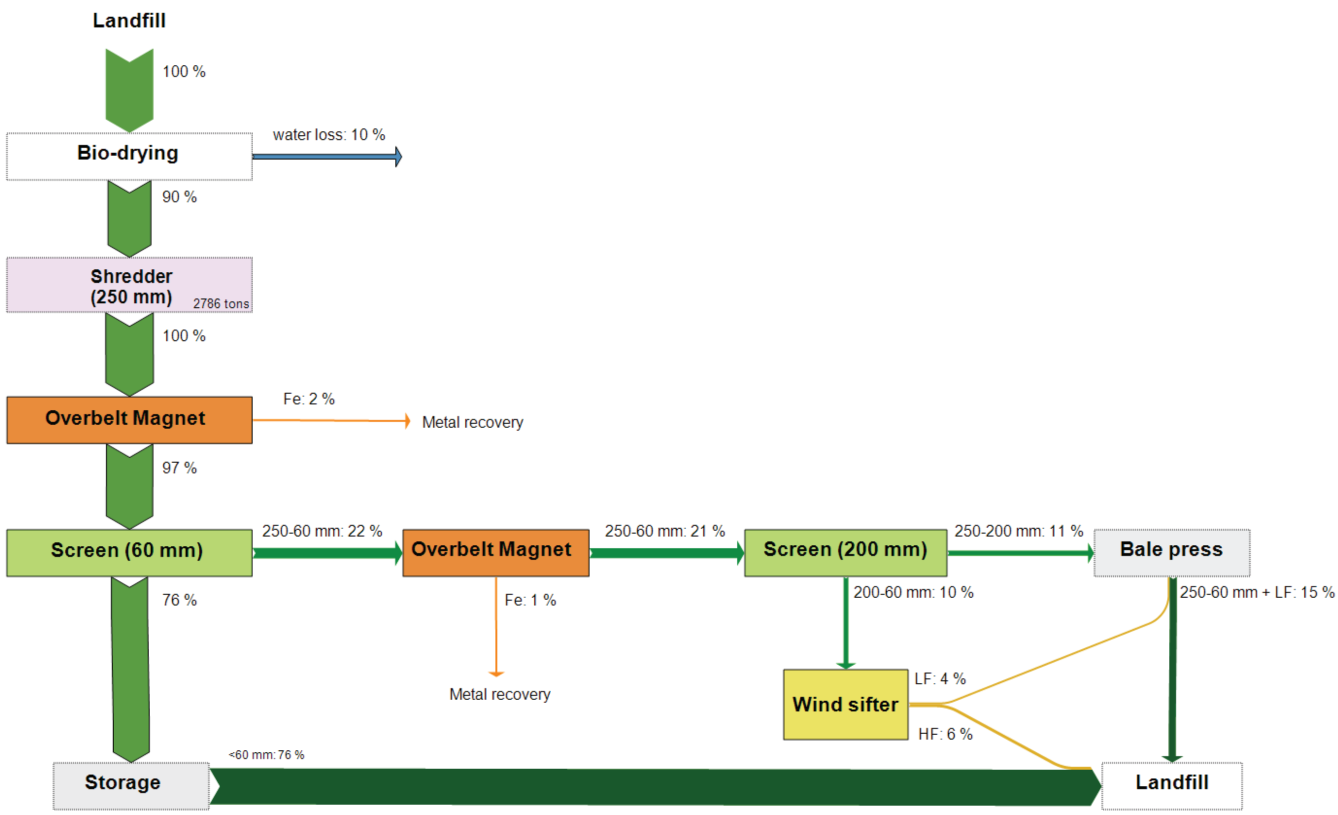

FIGURE 7: Mass balance of the MBT in Halbenrain (Austria).

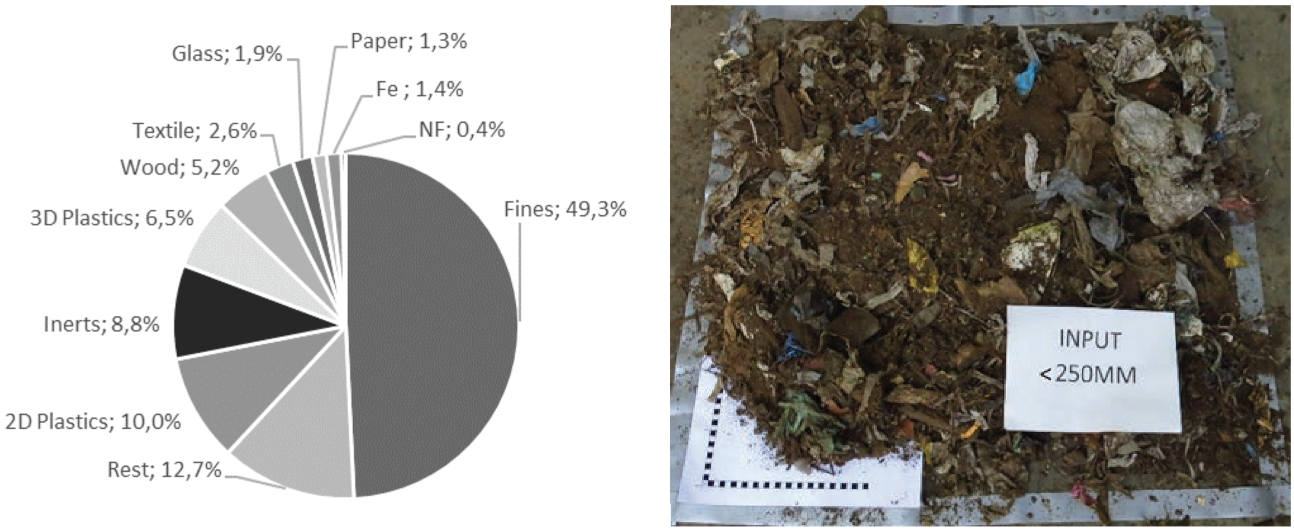

FIGURE 8: Average composition (wt%) of the material supplied in the MBT plant during the sampling campaign after a shredder and a magnet separator, SP1.

also reduce the heating value. The influence of the defilements attached to coarse particles is larger for flat particles (e.g., 2D plastic) since the surface is bigger in comparison to the total weight of a particle than it is for round or cubic objects.

The results from the input composition cannot be used for comparison with earlier studies in a reliable manner since the proceeding in each investigation varies. In this manner, the results are not properly comparable due to differences in properties and pre-processing, e.g., moisture content, analytical approach, mechanical and/or biological treatment. Once a common methodology is used, a proper comparison can be possible. However, the amount of fines (<10 mm) found in Halbenrain, 49%, is comparable to those found in Kuopio, Finland, with 50-54% (Kaartinen et al., 2013); Lower Austria, Austria, with 47% (Wolfsberger et al., 2015); and Remo, Belgium, with 44±12% (Quaghebeur et al., 2013). Regarding the amount of plastics (2D and 3D plastics), Halbenrain accounts for 16.5%, whereas the amount in Kuopio, Kudjape, Lower Austria and Remo are 23%, 22.4% (Bhatnagar et al., 2017), 18%, and 17±10%, respectively.

The variability of the results from 12 samples taken in the same sampling point (SP1) can be observed in Figure 9.

The absolute fluctuation in fines (<10 mm) is greater than in the remaining smaller categories, e.g., Fe, NFe, and glass. In fines (<10 mm), there is a 13% absolute variation, whereas the ferrous content fluctuates 4%, but its relative fluctuation is greater than the one of the fines (<10 mm), resulting in a bigger impact on the amount of product that can be generated from it. In this manner, for example, the potential revenues would rather be affected by ferrous fluctuations since this material type is initially more marketable than the fines (<10 mm).

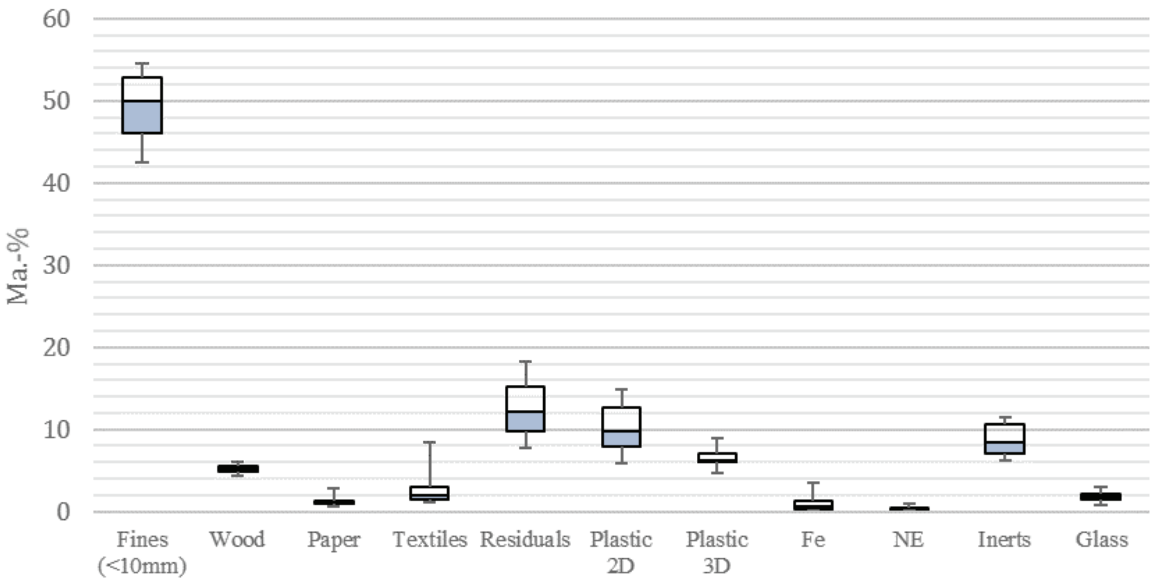

FIGURE 9: Range of the share of different material groups in the input flow of both batches (n=12).

Figure 10 shows the composition of the input material and other sampling points after each mechanical step, %wt semidry basis. See Appendix A for more information.

Output flow of the screen 60 mm - fine fraction (< 60 mm) (SP2). The fines from the 60-mm screen mainly consist of <10 mm fines, which account for approx. 71%; see Figure 10. Impurities in the flow are also found, including plastic particles (6%), glass (3%) and metals (1%). The content of glass in this material flow is greater than in the rest of the examined flows. The screening efficiency turned out to be lower than expected, resulting in an increased amount of fines in the coarses flow. This reduced efficiency can be ascribed to agglomeration of fines.

Output flow of the 200-mm screen - coarse fraction (250-200 mm) (SP3). The coarses of the 60-mm screen (250-60 mm) were sorted by a second magnet separator and sieved with a 200-mm screen. The amounts of fines in the flow are reduced, but there is still a remaining portion of 7.1%, which indicates that screening efficiency and the quality of the material. In the next chapter, regarding the particle size distribution, the efficiency of the screen can be assessed. This flow is characterized by its light fraction of 2D plastics (28.9%), followed by 3D plastics (21.2%), residuals (16.6%) and textiles (13.7%). This is an example that demonstrates the necessity of cleaning the material flow of impurities (mineral fraction) and concentrating the potential RDF (plastics, wood, textiles, and paper) via screening.

Output flow of the 200-mm screen - fine fraction (200 -60 mm) (SP4). The fine fraction from the 200-mm screen (200-60 mm) consists of a large share of inerts (20.2%), which could have been part of the covering layer of the landfill. MSW rarely consists of that many medium-sized stones. Apart from inerts, 3D and 2D plastics account for the biggest share in the composition of the flow. Once again, a significant amount of fines (<10 mm), 8.9%, was found. The screening efficiency could have been improved by using a bigger screen or a mesh size with a bigger opening size surface. The order of the 200- and 60-mm screens should have been switched, performing coarse screening first and afterwards using the 60-mm screen. In this case, the mechanical process was already configured prior the landfill mining project.

Output flow of the windsifter I - heavy fraction (HF 200 -60 mm) (SP5). The input material of the windsifter had a particle size of 60-200 mm. The major categories inside the heavy fraction flow are inerts (34.5%), wood (19.6%) and 3D-plastics (24.1%). The LF flow had an enrichment of 2D plastics, paper, textile and fines, which were almost removed from the heavy fraction flow. In addition, a big share of metals was found in the HF flow in comparison to the rest of the material flows. Hypothetically, a magnetic and an eddy current separator, after the windsifter in the heavy fraction, could have recovered 3.2% of Fe metals and 1.7% of NF metals. Instead, these valuable metals were returned to the landfill.

3.2.3 Particle size distribution of the material flows

The results for the particle size distribution demonstrate the efficiency of the screens and other sorting aggregates according to the grain sizes: <10 mm, 10-20 mm, 20-40 mm, 40-60 mm, 60-80 mm, 80-100 mm, 100-200 mm and 200-250 mm. Figure 11 provides an overview of percentage of the total mass by particle and the cumulative screening throughput in the input material flow <250 mm (SP1).

Input material flow (SP1): The composition of the coarses (>40 mm), Table 3, consists of mainly potential RDF (pRDF) materials, which are 2D and 3D plastics, textiles and wood with supposedly high calorific value, whereas the opposite is true for inerts, metals and glass. In addition, the category "residuals" contains a high concentration of combustibles, such as nappies, which could also be val-

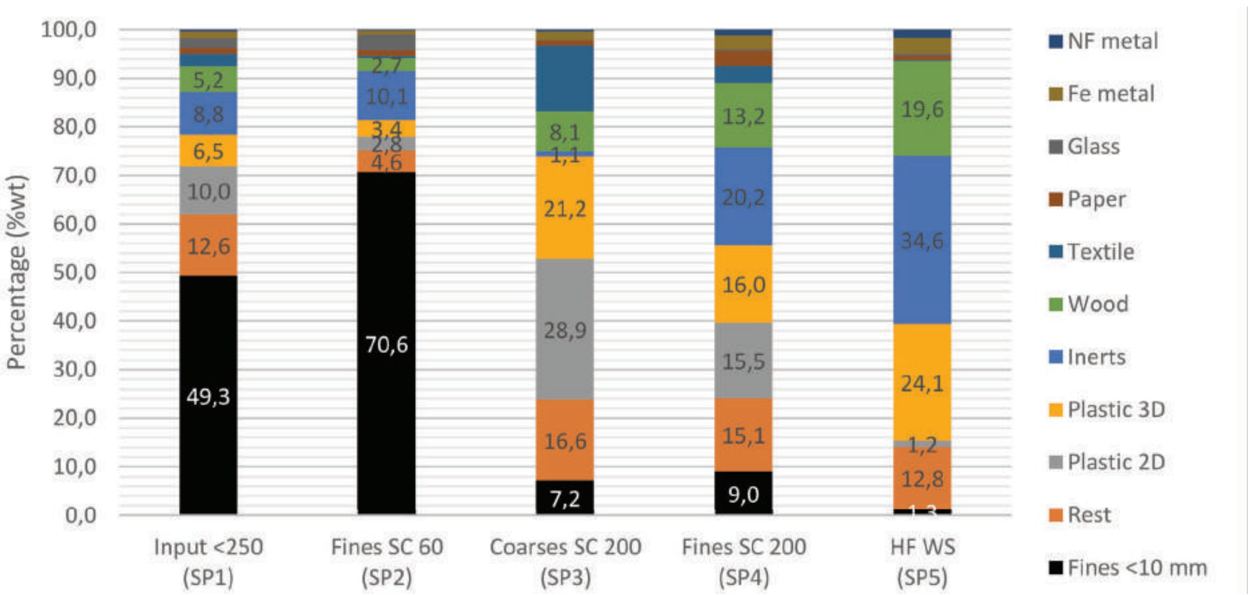

FIGURE 10: Composition of the input material (SP1) and output flows after each mechanical process (SP2, SP3, SP4, SP5).

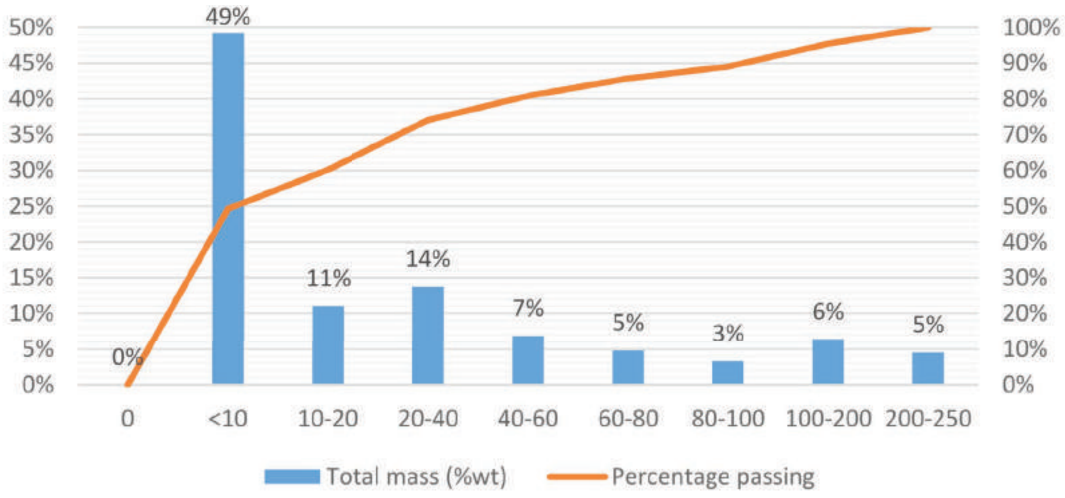

FIGURE 11: Particle size distribution of the input flow (SP1).

orized energetically. The amounts of NF and Fe in the input flow remain relatively low.

The fine fraction (<40 mm) in SP1 represents approx. 74 wt% of the entire input mass flow, being 60% the share of <20 mm and 49% of <10 mm. Similar amounts of fines have been found in previous characterizations of fine fraction mined. For example, the fine fractions (<20 mm) of two MSW landfills in Finland, Lohja and Kuopio, were on average 45%±7% and 58%±11%, respectively (Mönkäre, T. J. et al. 2015), similar to U.S. landfill reclamation projects: at least 50% and 46% of the excavated material were <25 mm in a landfill in New Jersey and one in Delaware, respectively (Hull et al. 2005; Miller et al. 1991), and in a landfill in Pennsylvania and one in Florida, approx. 41% and 60% were <20 mm (Forster 1994; Von Stein et al. 1993).

Table 4 presents the pRDF materials fractions in different particle size ranges. In the case of the coarse fraction, >80 mm, there is a bigger share of plastics and textiles. Wood and paper have a greater share in the size range of 20-100 mm than in the rest of fractions.

Initially, the composition of the LFM material is not very positive in terms of finding large quantities of recyclables, Table 5, but via biological and mechanical treatment, this complex material can be partly cleaned from impurities (particles <40 mm), and possible desirable materials, e.g., pRDF, metals and soil can be sorted out from the flow for further treatment processes, e.g., thermal valorization, fines treatment, and metal recycling.

As seen from the results of the mechanical processing, the coarse flow of the 200-mm screen (250-200 mm) is an example of enrichment of pRDF. The amount of pRDF in the particle size fraction 100-200 mm of the input material is 7.3 wt%, whereas in the coarses of the 200-mm screens, in the same particle size class, is 28.5 wt%. The treatment of these would reduce the landfill volume that is occupied. Another point of the sieving is to classify the material by

TABLE 3: Comparison of the composition of the particle size classes in the input material flow (SP1), wt%.

Particle size (mm)	<10	10-20	20-40	40-60	60-80	80-100	100-200	200-250
RDF	0	28	43	55	60	64	63	88
Fe	0	2	4	4	1	3	1	2
NF	0	0	1	2	1	2	0	0
Inerts	0	22	23	23	18	8	8	0
Glass	0	9	6	1	1	0	0	0
Residual	0	38	24	16	18	22	28	10
Fines	100	0	0	0	0	0	0	0

TABLE 4: High calorific materials by particle sizes in the input material flow (SP1), wt%.

Particle size (mm)	<10	10-20	20-40	40-60	60-80	80-100	100-200	200-250
Wood	0	9	13	13	15	10	7	0
Paper	0	2	4	4	3	3	1	0
Textile	0	0	1	2	4	7	7	26
Plastic 2D	0	10	13	19	21	25	32	43
Plastic 3D	0	7	12	17	17	20	16	19

TABLE 5: Total mass of the input material flow and percentage by categories in the input material after a magnetic separator, wt%.

Categories	Coarse fraction 250 - 40 mm	Fine fraction <40 mm
Wood	2	3
Paper	1	1
Textile	2	0
2D Plastics	7	3
3D Plastics	5	2
Fe metals*	1*	1*
NFe metals	0	0
Inerts	3	6
Glass	0	2
Rest	5	8
Fines (<10 mm)	0	49
Total	26	74

** Fe metals sum a total of 3% in the initial material feedstock. The amount of Fe-metals in this table is reduced due to the influence of an over-belt magnetic separator prior the sampling point SP1.*

size and achieve a higher efficiency in the following mechanical treatment. For example, according to Pretz et al. 2010, for an effective windsifter treatment, the ratio between the maximum and minimum particle sizes should not exceed 3:1. Even though the grain size range of 60-200 mm used in the MBT did not match the recommendations, still, the air sifting process was satisfactory. The windsifter helped to reduce the impurities contained in the fraction with a size from 200-60 mm, from a share of 8.9 wt% of fines (<10 mm) to 1.3 wt%. Furthermore, the windsifter concentrates the heavy fraction, which commonly consists of 3D plastics, inerts and metals. The composition of the HF material flow of the windsifter contains 87 wt% of coarses (200-60 mm), where 45 wt% are pRDF and 37 wt% are inerts, in the form of stones. It is also important to highlight the amount of residuals, between 200-60 mm, in the same material flow, which has a share of 13 wt%.

Based on this information, particle size distributions of the LFM material and the results of this study can be used as guidance for mechanical treatment. Further chemical analyses are mandatory to determine whether the quality of the excavated material, e.g., fines and inerts, fits the standards for its recuperation.

See Appendix A-B for more results from each sampling point.

3.3 Physical-chemical analysis

Hull et al. pointed out in his study that waste fractions that can absorb moisture such as fines, paper, cardboard, wood and textiles had much higher moisture contents than fractions that cannot absorb water. However, it has to be considered that the size of the fines is another factor that influences the moisture content As can be observed from Table 6, <10 mm has a higher moisture content, 32 wt%, than 10-20 mm and 20-40 mm, 17 wt% and 20%, respectively, due to capillary forces on particles <10 mm.

Even if the results of this chapter are focused on the fines, moisture of samples containing all fractions could be almost equal. The mean moisture content of fines in previous investigations ranges from 16 to 43% (Hull et. al 2005).

The calorific value varies of the samples analysed from 7.2 to 11.9 MJ/kg, depending on the share of organic content (<10 mm: 34.6 wt%, 10-20 mm: 32.3 wt%, 20-40 mm: 40.0 wt%).

The percentage of heavy metals contained in the fine fractions, reported in Table 7, are not below the Austrian limits, at least, to use the fine fraction for compost. In previous characterizations of landfilled material characterizations (Hull et al. 2005), selected trace metals also exceeded soil background levels and recommended levels when applying sewage biosolids to agricultural land.

There is an increase of the heavy metal content according to the diminution of the particle; < 10 mm has a higher concentration of heavy metals than 20-40 mm. Further analysis is mandatory to estimate the potential of this fraction for construction material due to the amount of impurities (metals, glass shards and plastics).

4. CONCLUSIONS

The results from the investigations in Halbenrain landfill show that almost 90 t of ferrous metals could be recovered from 2785 t of mined landfill waste (approx. 3%). However, even combined with the profit from nonferrous metals, the profit would be insufficient to make such a landfill mining project feasible today. This fact is further influenced negatively by the fact that the defilements on plastics that could e.g., be used for thermal valorization

TABLE 6: Mean values and standard deviation (in brackets) of the moisture content, organic content and calorific value of the input material in fine fractions (<10 mm; 10-20 mm: 20-40 mm).

Input (mm)	Moisture (wt% semidry basis)	Composition (wt% dm)			Organic content (wt% dm)			Calorific value (MJ/kg dm *)			
		LF	HF	Metal	LF	HF	Metal**	LF	HF	Metal**	Total
<10	32	7	91	2	78.0 (0.02)	32.0 (0.02)	0.0	25.9 (0.01)	6.0 (0.02)	0.0	7.2
10-20	17	18	77	5	81.0 (0.01)	23.0 (0.00)	0.0	25.0 (0.88)	5.1 (0.01)	0.0	8.5
20-40	20	23	73	4	84.0 (0.01)	29.0 (0.07)	0.0	32.5 (0.56)	6.1 (0.21)	0.0	11.9

** dm: dry matter*
*** It is assumed the amount of organic content of the metals is 0 wt% dm; therefore, its calorific value is also 0 MJ/kg dm*

TABLE 7: Mean and standard deviation (in brackets) of the heavy metals contained in the fine fractions: <10 mm, 10-20 mm, 20-40 mm, (%, dry basis).

Metals	Fraction			Limit values * (%) [Austria, 2018]
%	<10 mm	10-20 mm	20-40 mm	
Pb	0.155 (0.098)	0.112 (0.00)	0.087 (0.000)	0,020
As	0.007 (0.000)	0.007 (0.00)	0.002 (0.000)	-
Zn	0.708 (0.104)	0.490 (0.00)	0.297 (0.000)	0,180
Ni	0.046 (0.002)	0.025 (0.00)	0.026 (0.000)	0,010
Cr	0.220 (0.004)	0.187 (0.00)	0.093 (0.000)	0,025
Cd	0.003 (0.000)	0.003 (0.00)	0.004 (0.000)	<0,001
Cu	0.659 (0.366)	0.395 (0.010)	0.324 (0.020)	0,050

** General requirements for waste-compost, quality class B, in Austria (Bundesrecht konsolidiert: Gesamte Rechtsvorschrift für Kompostverordnung, Fassung vom 05.03.2018, BGBl. II Nr. 292/2001)*

reduce the heating value and increase the ash content of such pRDF.

Furthermore, comparisons with previous landfill mining studies could be performed, even though such comparisons are made difficult by differences in landfill composition, preprocessing and the analytical approaches in such projects. However, certain similarities among this and former studies can be noted, such as similar amounts of fines (50 wt%) and plastics (17 wt%). In particular, the amounts of fines, 74 wt% < 40 mm and 50 wt% < 10 mm, represent the biggest share of the excavated material, as Hernández Parrodi et al., 2017 also indicated in his study where this fraction can be as high as 40-80 wt% in various landfills around the world, and they need further investigation in order to reduce the financial burden they are today.

Moreover, fluctuations in the material compositions can be highly problematic. On one hand, these fluctuations can influence the load on different machines, potentially reducing the performance of the whole plant. On the other hand, such fluctuations, even if they might seem relatively small, can reduce the marketable fractions. Even if ferrous or non-ferrous products only make up a small amount of the total LFM material, when this amount varies between 1 and 5 wt%, the amount of marketable product can also fluctuate approximately 500%. Another problem with the mechanical process is the moisture of the material: if the material is not dry or semi-dried, as in this case for biological treatment, the quality of the sorting is lower, the amount of impurities adhered to the surface increases and the mass balance of each category would not be representative for the real composition (e.g., soil-type paper absorbs large amounts of water).

Chemical analysis reveals the presence of heavy metals in specific fractions. Depending on the further treatment of the different output fractions of such an LFM project, the limit values must be considered. Depending on the country LFM material recovery should be conducted according to the applicable limit values, which can constitute a problematic hurdle and determine whether such a project can be viable.

Another inconvenience of landfill mining is the presence of hazardous materials, in addition to the contaminated soil, which drastically reduces the potential of the landfill as a source of resources. However, this inconvenience should not stop landfill mining activities, since the environmental issue that landfills entail cannot be ignored, and remediation strategies are necessary to avoid future costs.

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APPENDIX A: Tables

TABLE i: Minimum number of individual/composite samples according to LAGA PN 98.

Name sampling point	Volume m³	Theoretically			Sampling campaign		
		Individual samples n	Composite samples n	Sample weight Kg	Individual samples n	Composite samples n	Sample weight Kg
SP1	400	32	8	15	18 *	6 *	21
SP2	30	8	2	15	8	2	12 *
SP3	30	8	2	15	8	2	15
SP4	30	8	2	12	8	2	18
SP5	30	8	2	12	8	2	21

** For technical reasons the number and the amount of the samples were not possible to reach in accordance to the guideline*

TABLE ii: Composition by sampling point (%wt semidry basis).

Sampling point	Input <250 (SP1)	Fines SC 60 (SP2)	Coarses SC 200 (SP3)	Fines SC 200 (SP4)	HF WS (SP5)
Fines <10 mm	49,3	70,6	7,2	9	1,3
Rest	12,6	4,6	16,6	15,1	12,8
Plastic 2D	10	2,8	28,9	15,5	1,2
Plastic 3D	6,5	3,4	21,2	16	24,1
Inerts	8,8	10,1	1,1	20,2	34,6
Wood	5,2	2,7	8,1	13,2	19,6
Textile	2,6	0,4	13,7	3,5	0,2
Paper	1,3	1,2	0,9	3,2	0,9
Glass	1,9	3,3	0,1	0,3	0,4
Fe metal	1,4	0,7	1,8	2,9	3,2
NF metal	0,4	0,2	0,4	1,1	1,7

TABLE iii: Comparison of the composition by particle sizes in the fine fraction of the screen 60mm (SP2), wt%.

Particle size (mm)	<10	10-20	20-40	40-60
Fines <10mm	100,0%	0,0%	0,0%	0,0%
RDF	0,0%	28,1%	37,1%	47,3%
NF	0,0%	0,2%	1,1%	1,3%
Fe	0,0%	2,2%	1,9%	3,6%
Inerts	0,0%	37,6%	32,9%	30,5%
Glass	0,0%	16,1%	11,0%	1,3%
Residual	0,0%	15,8%	15,9%	16,1%

TABLE iv: High calorific materials by particle sizes in the fine fraction of the screen 60mm, wt%.

Particle size (mm)	<10	10-20	20-40	40-60
Wood	0,0%	7,2%	9,8%	11,6%
Paper	0,0%	4,5%	3,4%	3,8%
Textile	0,0%	0,6%	1,2%	3,2%
Plastic 2D	0,0%	6,5%	9,9%	14,6%
Plastic 3D	0,0%	9,3%	12,8%	14,1%

TABLE v: Comparison of the composition by particle sizes in the output flow (>200mm) of the screen 200, wt%.

Particle size (mm)	<10	10-20	20-40	40-60	60-80	80-100	100-200	200-250
Fines <10mm	100,0%	0,0%	0,0%	0,0%	0,0%	0,0%	0,0%	0,0%
RDF	0,0%	75,5%	71,4%	69,5%	73,9%	71,1%	79,6%	81,4%
NF	0,0%	0,0%	1,6%	1,2%	1,1%	1,8%	0,4%	0,0%
Fe	0,0%	0,0%	1,5%	0,6%	0,9%	0,1%	1,4%	3,4%
Inerts	0,0%	2,7%	3,1%	11,8%	0,6%	4,7%	0,0%	0,0%
Glass	0,0%	2,9%	1,5%	0,0%	0,0%	0,0%	0,0%	0,0%
Residual	0,0%	18,9%	21,0%	17,0%	23,5%	22,2%	18,6%	15,2%

TABLE vi: High calorific materials by particle sizes in the output flow (>200mm) of the screen 200 mm, wt%.

Particle size (mm)	<10	10-20	20-40	40-60	60-80	80-100	100-200	200-250
Wood	0,0%	31,6%	15,5%	26,1%	17,3%	14,4%	10,5%	0,8%
Paper	0,0%	9,1%	8,1%	3,7%	2,1%	1,2%	0,5%	0,0%
Textile	0,0%	7,1%	6,2%	2,4%	5,3%	6,5%	8,7%	26,9%
Plastic 2D	0,0%	16,8%	16,4%	18,5%	23,5%	23,4%	35,5%	32,9%
Plastic 3D	0,0%	10,9%	25,1%	18,8%	25,6%	25,5%	24,4%	20,8%

TABLE vii: Comparison of the composition by particle sizes in the output flow (<200 mm) of the screen 200, wt%.

Particle size (mm)	<10	10-20	20-40	40-60	60-80	80-100	100-200
Fines <10mm	100,0%	0,0%	0,0%	0,0%	0,0%	0,0%	0,0%
RDF	0,0%	54,5%	64,2%	63,2%	39,2%	48,5%	64,5%
NF	0,0%	1,9%	0,1%	1,7%	1,9%	0,2%	1,5%
Fe	0,0%	2,8%	2,6%	1,3%	0,2%	0,9%	5,7%
Inerts	0,0%	7,4%	5,0%	16,4%	39,4%	35,2%	12,8%
Glass	0,0%	3,5%	1,4%	1,6%	0,3%	0,0%	0,0%
Residual	0,0%	29,8%	26,8%	15,8%	18,9%	15,2%	15,5%

TABLE viii: High calorific materials by particle sizes in the output flow (<200 mm) of the screen 200, wt%.

Particle size (mm)	<10	10-20	20-40	40-60	60-80	80-100	100-200
Wood	0,0%	11,7%	24,5%	25,2%	10,5%	15,4%	12,9%
Paper	0,0%	20,8%	11,6%	6,0%	3,0%	2,7%	2,4%
Textile	0,0%	1,7%	1,3%	1,4%	2,0%	1,3%	6,4%
Plastic 2D	0,0%	11,2%	11,3%	12,2%	10,6%	15,0%	21,6%
Plastic 3D	0,0%	9,2%	15,5%	18,4%	13,1%	14,1%	21,2%

TABLE ix: Comparison of the composition by particle in the heavy fraction flow (HF) of the windsifter I, wt%.

Particle size (mm)	<10	10-20	20-40	40-60	60-80	80-100	100-200
Fines <10mm	100,0%	0,0%	0,0%	0,0%	0,0%	0,0%	0,0%
RDF	0,0%	35,1%	45,6%	58,5%	38,4%	41,7%	55,2%
NF	0,0%	3,1%	0,0%	0,9%	1,8%	0,9%	2,8%
Fe	0,0%	10,2%	9,5%	4,2%	4,8%	2,8%	1,3%
Inerts	0,0%	19,0%	7,6%	28,4%	45,1%	41,5%	24,3%
Glass	0,0%	4,7%	2,8%	1,4%	0,6%	0,0%	0,0%
Residual	0,0%	27,8%	34,5%	6,6%	9,4%	13,0%	16,4%

TABLE x: Comparison of the composition by particle in the heavy fraction flow (HF) of the windsifter I, wt%.

Particle size (mm)	<10	10-20	20-40	40-60	60-80	80-100	100-200
Wood	0,0%	8,2%	18,3%	28,7%	17,5%	20,0%	19,6%
Paper	0,0%	5,2%	4,3%	0,7%	0,5%	1,0%	0,7%
Textile	0,0%	0,7%	0,3%	0,0%	0,1%	0,7%	0,0%
Plastic 2D	0,0%	4,6%	2,2%	1,5%	0,4%	1,7%	1,1%
Plastic 3D	0,0%	16,4%	20,5%	27,6%	19,9%	18,3%	33,8%

APPENDIX B: Figures

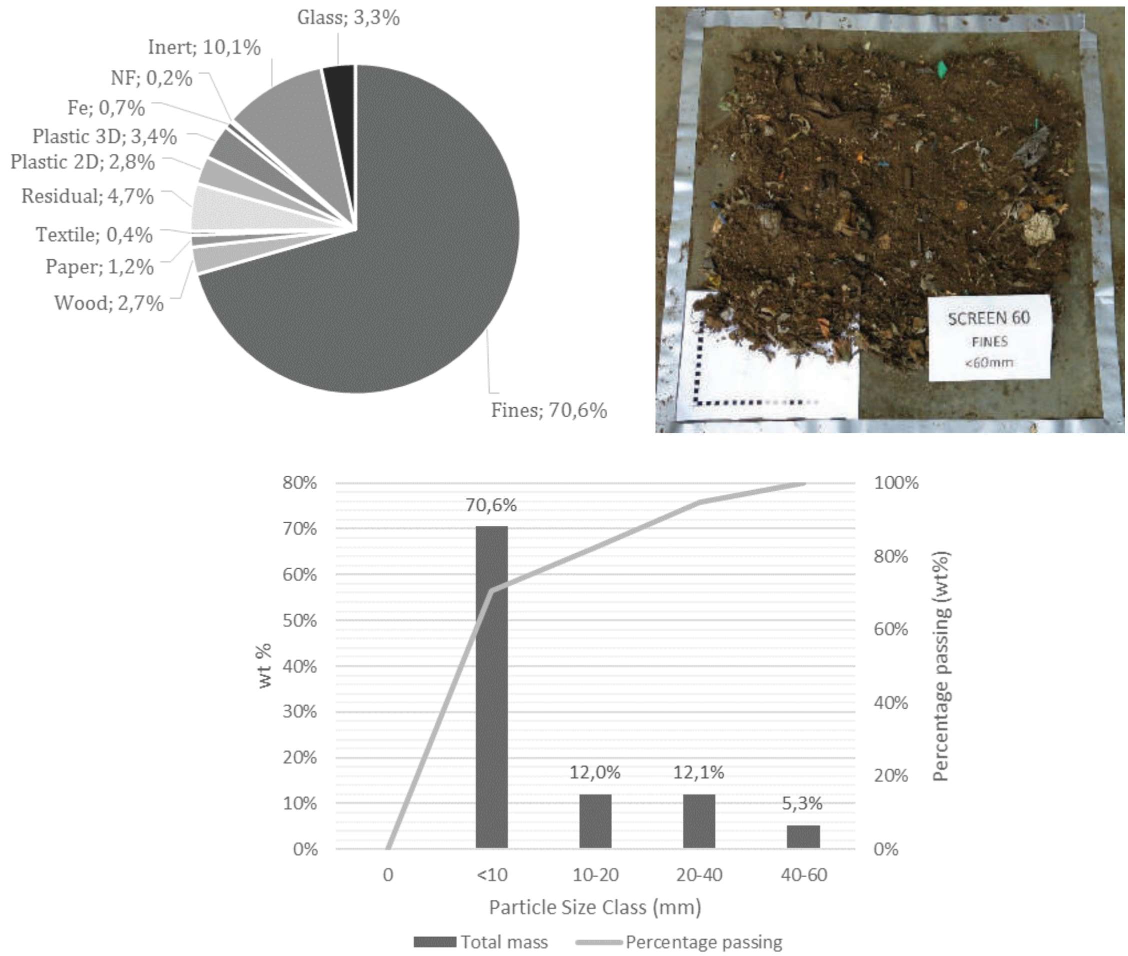

FIGURE i: Average composition of the fine fraction of the screen 60mm, SP2, (up) and its particle size distribution (down).

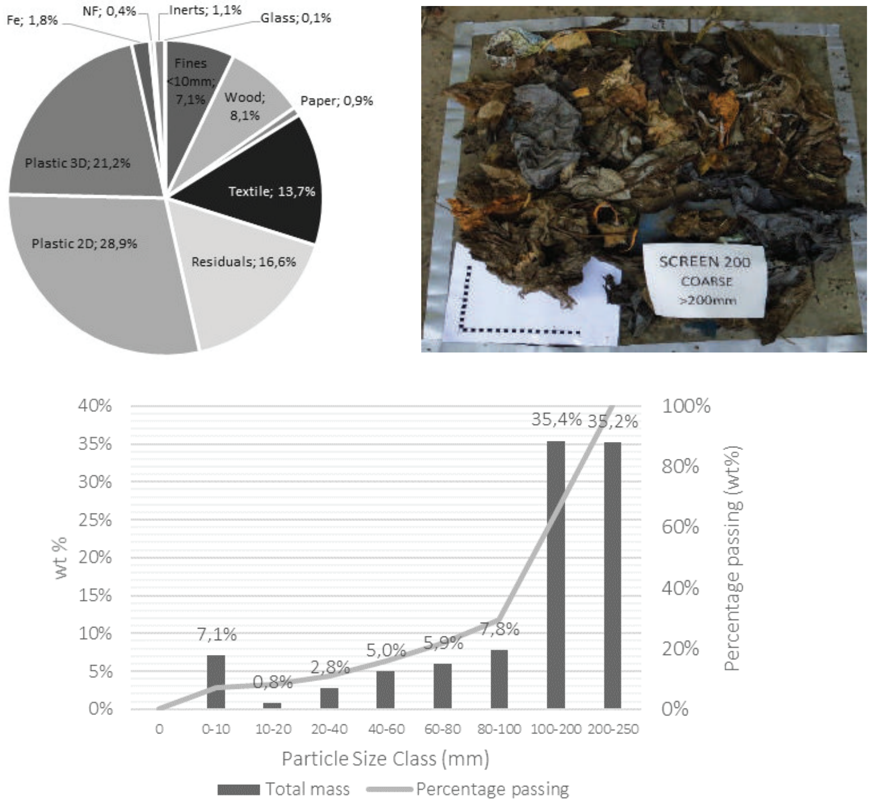

FIGURE ii: Average composition of the coarse fraction (250-200 mm) of the screen 200, SP3, (up) and its particle size distribution (down).

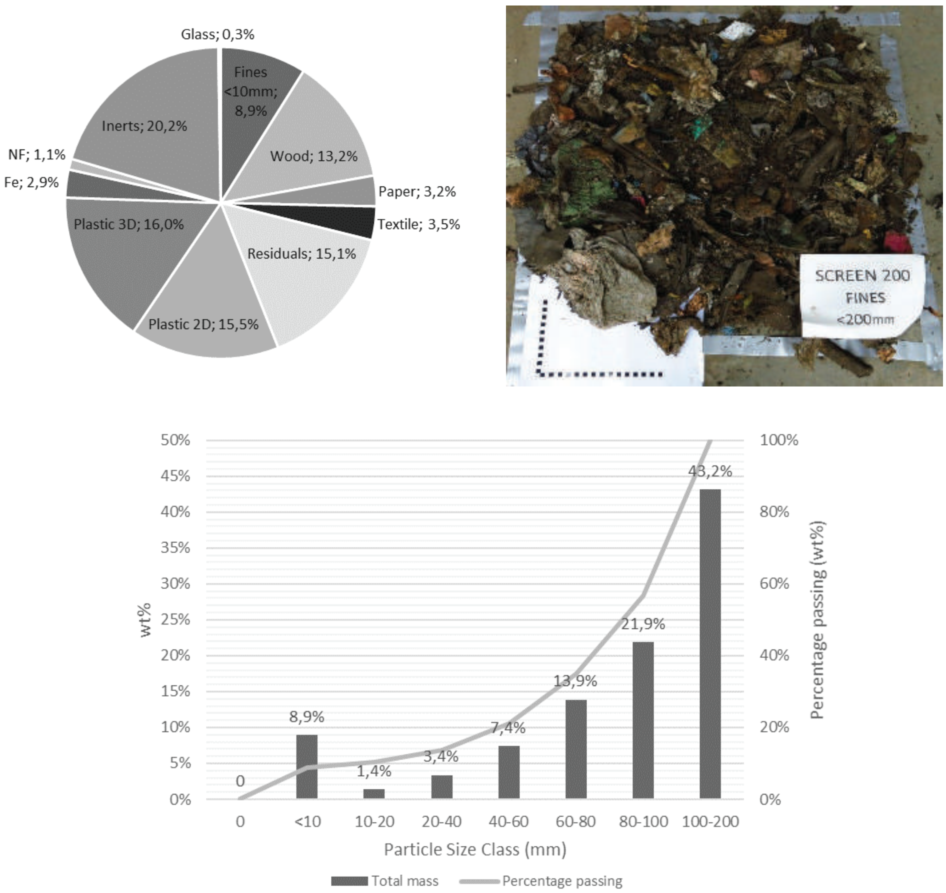

FIGURE iii: Average composition of the fine fraction (<200 mm) of the screen 200, SP4, (up) and its particle size distribution (down).

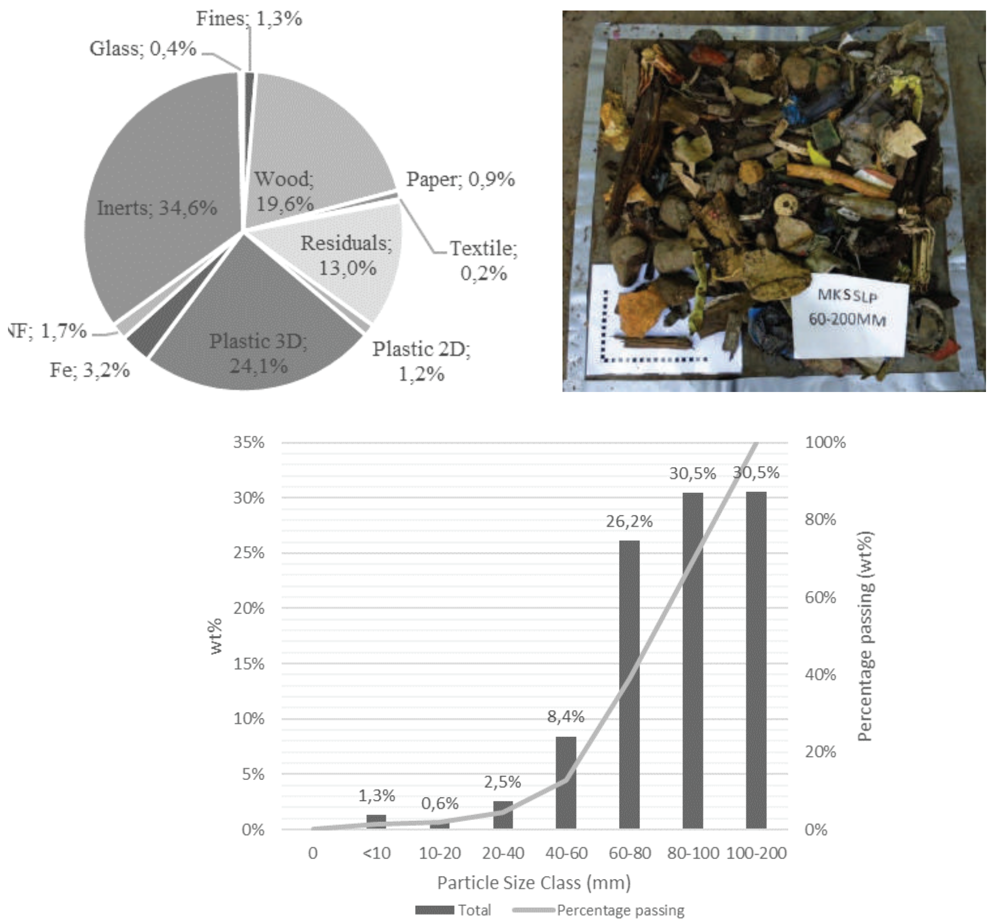

FIGURE iv: Average composition of the heavy fraction (HF) windsifter, SP5, (up) and particle size distribution (down).

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CHARACTERIZATION OF FINE FRACTIONS FROM LANDFILL MINING: A REVIEW OF PREVIOUS INVESTIGATIONS

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Fines

ABSTRACT

Several landfill mining (LFM) studies have been carried out in recent years all around the world. From these studies qualitative and quantitative information regarding the composition and characteristics of the different fractions excavated from landfills has been obtained. This information comprises data from various landfill sites around the globe from which useful correlations for future LFM projects can be identified. Of particular interest to this paper is the information regarding the fine fractions, which represent to this day a crucial obstacle in the implementation of LFM and enhanced landfill mining (ELFM). The fine fractions make up a considerable portion of the total amount of waste disposed of in landfills. Depending on the particle size chosen as upper limit to define the fines fraction, the portion of this fraction can be as high as 40-80 wt.% of the total excavated waste. These fractions consist of decomposed organic substances, e.g. humic substances, partly weathered mineral waste, e.g. sand, brick fragments, concrete, but also of fine metal particles, especially non-ferrous metals, and still a significant amount of plastics, paper and other calorific fractions. However, although calorific fractions might be used for energy recovery and inorganic fractions for material (especially metal) recovery, current LFM studies are discarding the fine fraction due to lacking or too expensive processing routes. Therefore, it is of critical interest to LFM and ELFM projects to reduce the particle size down to which the excavated material can be processed. This paper, which was elaborated within the framework of the EU Training Network for Resource Recovery through Enhanced Landfill Mining – NEW-MINE, aims to review the obtained data from different LFM studies from municipal solid waste (MSW) landfills, concerning the fines fraction, in order to identify key aspects to be taken into consideration while designing the processing approach in future LFM and ELFM investigations.

1. INTRODUCTION

Since its commencement, in 1953 at the Hirya landfill in Israel (Savage, Golueke, & Von Stein, 1993), the focus of LFM has been evolving, incorporating different drivers and objectives to its original purpose over the years. To this day, some common drivers of LFM projects have been: material recovery (recyclable and reusable materials), land reclamation, landfill capacity regain, pollution mitigation, landfill remediation, removal of deposits obstructing urban development, production of alternative fuels, aftercare and closure costs reduction, enabling the operation of regional MSW incinerators at full capacity, reuse of already available landfill infrastructure, simplification of the permitting process, among others (Hull, Krogmann, & Strom, 2005; Krook, Svensson, & Eklund, 2012).

Moreover, a holistic concept, ELFM, has been developed during this decade. This approach envisages the combined and integrated waste valorization of old and future waste deposits as both materials (Waste-to-Material, WtM) and energy (Waste-to-Energy, WtE); while respecting most stringent ecological and social criteria (Jones, Geysen, Rossy, & Bienge, 2010).

Since landfills were for decades the sole disposal solution for all types of waste with any segregation, they represent a heterogeneous source of materials (Kaartinen, Sormunen, & Rintala, 2013). Previous investigations made by Krook et al., 2012; Cossu, Motzo, & Laudadio, 1995, Kaartinen et al., 2013; Prechthai, Padmasri, & Visvanathan, 2008; Quaghebeur et al., 2013; Zhao, Song, Huang, Song, & Li, 2007 report that landfill-mined waste normally consists of 20-30 wt.% combustible materials, 50-60 wt.% fine-grained degraded matter, 10 wt.% inert materials and a small percentage of metals. For further references predating year 2011, a broad worldwide overview of over 60 LFM projects

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and the rough composition of landfilled waste from over 20 landfill dismantling and exploratory drilling projects can be consulted in the work made by Bockreis & Knapp, 2011.

A more detailed material composition (Van Vossen & Prent, 2011), obtained from information found in literature of 60 landfill mining projects, plus the outcomes of most recent investigations (after year 2011) are presented in Table 1.

In this information (Table 1) it can be noticed that the fine fractions (referred sometimes as "soil", "soil-like" or "soil-type" fractions, due to their appearance, organic matter and mineral contents and relatively homogeneous composition compared to the coarser fractions) are commonly to a great extent the largest fraction of the whole excavated amount in a LFM project. These fractions typically contain mainly degraded garden and food materials (Quaghebeur et al., 2013). This degradation process can be compared to the natural humification process during soil formation. Since the US EPA reported that around 75% of the LFM material corresponds to mineral landfill liners and degraded organic waste (Landfill Reclamation, 1997), a comparison

TABLE 1: Material composition of excavated waste from previous LFM investigations.

Parameter	Van Vossen and Prent, 2011 (various countries)	Jani et al., 2016 (Högbytorp, Sweden)	Kaartinen et al., 2013 (Kuopio, Finland)	Bhatnagar et al., 2017 (Kudjape, Estonia)	Wolfsberger et al., 2015 (Lower Austria, Austria)	Quaghebeur et al., 2013 (REMO, Belgium)
Type of waste disposed of	Various	MSW + C&D	MSW	MSW	MSW	MSW
Age of waste [a]	Various	5	5 - 10	10	13 - 20	14 - 29
Fraction(s) considered	All	10 - 40 mm	All	All	All	All
Average moisture content	-	-	-	-	29.0 - 55.0%	53.0 - 68.0%
Fines / Sorting residue / Soil-type material	54.8%	27.3%	50.0 - 54.0%	28.7%	47.0%	44.0 ± 12.0%
Stones	2.5%	28.1%	-	17.5%	-	-
Minerals / Inert	5.8%	-	-	-	6.0%	10.0 ± 6.0%
C&D	9.0%	-	-	-	-	-
Limestone	-	4.8%	-	-	-	-
Asphalt	-	3.2%	-	-	-	-
Glass / Ceramics	1.1%	5.6%	-	4.6%	1.0%	1.3 ± 0.8%
Plastics	4.7%	-	23.0%	22.4%	18.0%	17.0 ± 10.0%
Soft plastics	-	0.7%	-	-	-	-
Other plastic / Composites	-	6.8%	-	-	4.0%	-
Organic / Kitchen waste	5.3%	-	-	-	-	-
Paper & cardboard / PPC	5.3%	-	4.0 - 8.0%	5.1%	3.0%	7.5 ± 6.0%
Paper	-	4.5%	-	-	-	-
Wood	3.5%	15.2%	6.0 - 7.0%	4.7%	-	6.7 ± 5.0%
Textiles	1.6%	2.7%	7.0%	-	6.0%	6.8 ± 6.0%
Leather	1.6%	-	-	-	-	-
Rubber	-	0.2%	-	-	-	-
Wood, leather and rubber	-	-	-	-	9.0%	-
Total metals	2.0%	-	3.0 - 4.0%	3.1%	5.0%	2.8 ± 1.0%
Fe metals	-	0.5%	-	-	-	-
Non-Fe metals	-	0.5%	-	-	-	-
Other / Rest	2.6%	-	2.0%	13.4%	1.0%	-
Non-MSW	0.3%	-	-	-	-	-
Notes:	*Information organized according to age of waste Totals may not add exactly 100% due to figures´ rounding Figures have weight and wet basis MSW - Municipal solid waste C&D - Construction and demolition waste PPC - Paper, paperboard and cardboard*					

to natural soils which also contain fine-grained mineral and organic materials can be drawn. However, the different genesis of the fine fractions in landfills and of soils, and the lack of separation of the fine fractions from other materials in the landfill, do not allow addressing the fine fractions from landfills as soils.

Fine fractions (frequently defined as material with a particle size < 60 mm to < 10 mm) account for 40-80 wt.% of the mined material in previous studies (Hogland, 2002; Masi, Caniani, Grieco, Lioi, & Mancini, 2014; Kaartinen et al., 2013; Kurian, Esakku, Palanivelu, & Selvam, 2003; Rettenberger, 2009; Hull et al., 2005; Mönkäre, Palmroth, & Rintala, 2016; Quaghebeur et al., 2013; Maul & Pretz, 2016; Van Vossen & Prent, 2011; Wiemer, Bartsch, & Schmeisky, 2009; Wolfsberger et al., 2015). Therefore, regardless of the particle size used to define the fine fractions, its quantity will always be an important factor to be considered in LFM and ELFM projects.

The main purpose of the present review is to gather information regarding the fine fractions of previous LFM investigations, in order to identify their composition and properties, so that the possibility of material and energy recovery from these fractions can be assessed in forthcoming research, as well as to identify key aspects to be taken into account while designing the processing approach in future LFM and ELFM investigations.

2. MATERIALS AND METHODS

The present study comprises a review of several previous LFM investigations found in scientific literature. The main focus of this review paper is on the material characterization of the fine fractions. The scope envisages scientific papers published in international peer-reviewed journals, as well as a minor amount of other review papers and international conference proceedings, books, guidelines, standards and legislation.

3. REVIEW AND DISCUSSION

There have been plenty of LFM projects and investigations carried out up to now; nevertheless, not much attention has been paid to the fine fractions in terms of their potential for material recovery, alternative fuels production and possible alternative uses (e.g. as cover layer in operating landfills, as filling material for leveling purposes or the construction of embankments, as soil improver for growing nonedible crops, etc.). In most LFM projects recycling has been restricted to the coarse fractions, while the fine fractions have been re-directed to the landfill with poor or no treatment beforehand, mainly due to technical and economic challenges, despite their recovery potential (Bhatnagar et al., 2017; Münnich, Fricke, Wanka, & Zeiner, 2013).

According to previous investigations (Kaartinen et al., 2013; Mönkäre et al., 2016; Wolfsberger et al., 2015) the amount of fines to be obtained in a LFM project mostly depends on the excavation procedure, the age of the waste and the selected cut-off diameter to define a certain particle size as upper limit for the fine fractions. For example: (i) the implementation of borehole sampling via drilling activities can increase the amount of fine fractions in the samples, (ii) the amount of fine fractions has been found to raise with age in some investigations and (iii) the amount of material passing the screen tends to increase with the increase in size of the cut-off diameter of the screen. However, these factors might be correlated with one another and each of them can increase or decrease the amount of fine fractions by itself. Therefore, the specific setup employed in a particular LFM project is to be analyzed in a single-case base, in order to determine the overall effect of these factors on the total amount of fine fractions to be obtained.

Additionally, the characteristics of the fine fractions of landfill-mined material can be influenced by the chosen processing, e. g. sieve size affects utilization and disposal methods of the sieved materials, as it has been observed that the methane potential rises with the increase of particle size (Mönkäre et al., 2016). Moreover, the amount of fine fractions increases with time due to the decomposition processes (Jani et al., 2016). Long disposal time leads to degradation processes of the organic matter, which leads to a higher amount of fines (Maul & Pretz, 2016).

Because of the lack of economic value, the characterization properties of the fine fractions have not been thoroughly investigated (Mönkäre et al., 2016). Nonetheless, in order to evaluate the specific recycling potential of a landfill, adequate and proper quantitative and qualitative characterizations of the disposed waste are to be performed (Prechthai et al., 2008).

It is important to point out that much care needs to be taken when comparing information between different investigations directly, since there are many factors, such as; characterization conditions and procedures, laboratory analyses and followed standards, age of the waste material, defined particle size for the fines fraction, among others, that might play an important role during their execution and may differ significantly from investigation to investigation. The implementation of different approaches for the material characterization of waste remains to be one of the crucial challenges for the elaboration of comparable and accurate compiled studies.

3.1 Material composition

In this paper, material composition refers to the kind of material, e.g. “plastic” or “textiles”, whereas chemical composition refers to the elemental composition and mineralogical composition to the phase composition. On the basis of the result of previous studies on the material composition of the fine fractions, of excavated waste from landfill, some tendencies can be recognized. Table 2 shows a compilation of these studies and their reported results. Some clear trends that can be noticed, apart from the already stated clear dominance of the amount of fines over the total, are the considerable amounts, in some cases, of inert materials (mainly stones and glass), plastics, textiles, paper and metals present in these fractions. This information allows grouping the sub-fractions that constitute the fine fractions in to major constituents, which are degraded organic and mineral materials, and minor constituents, which are plastics, textiles, metals paper and

TABLE 2: Material composition of excavated fine fractions from previous LFM investigations.

Parameter	(Filborna, Sweden) in Kurian et al., 2003	Wolfsberger et al., 2015 (Lower Austria, Austria)	Hull et al., 2005 (BCRRC, USA)	Kurian et al., 2003 (Perungudi, India)	Kurian et al., 2003 (Kodungaiyur, India)	(Deonar, India) in Kurian et al., 2003
Type of waste disposed of	MSW	MSW	MSW + C&D + IW	MSW	MSW	MSW
Age of waste [a]	-	13 - 20	1 - 11	0 - 10	10	-
Particle size [mm]	< 40	< 40	< 25.4	< 20	< 20	< 8
Amount of fines from the whole	65.7%	68.0%	≥ 50%	58.9%	32.3%	34.6%
Sorting residue	-	65.6%	-	-	-	-
Stones	19.0%	-	-	18.5%	28.3%	31.5%
Minerals / Inert	-	6.6%	-	-	-	-
Glass	0.5%	1.4%	-	0.8%	0.4%	-
Wood	-	-	8.5 - 11.7%	-	-	-
Wood, leather, rubber	15.7%	5.9%	-	26.1%	1.0%	1.2%
Textiles / Rubber	4.5%	1.9%	5.4 - 6.8%	2.3%	0.6%	-
Plastics	18.1%	11.6%	10.5 - 19.5%	11.0%	1.9%	1.5%
Composites	-	1.0%	-	-	-	-
PPC	-	3.0%	-	-	-	-
Paper	-	-	6.8 - 14.4%	-	-	-
Cardboard	-	-	4.6 - 14.9%	-	-	-
Total metals	7.9%	1.9%	-	0.2%	0.1%	0.4%
Fe metals	-	-	5.5 - 12.6%	-	-	-
Others	-	1.1%	-	-	-	-
Problematic substances	-	0.1%	-	-	-	-
Notes:	*Information organized according to particle size* *Totals may not add exactly 100% due to figures´ rounding* *Figures have weight and wet basis* *MSW - Municipal solid waste* *C&D - Construction and demolition waste* *IW - Industrial waste* *PPC - Paper, paperboard and cardboard*					

cardboard, among others. Below, particular tendencies and aspects that have been found in former investigations are discussed.

The fine fractions have been found aesthetically unpleasant due to the presence of non-soil materials, such as plastic and paper flakes and broken glass (Hull et al., 2005). Zhao et al., 2007 reported that the fine fractions showed similarities to black soil and suggested, therefore, a use for green construction, organic fertilizer or as bioreactor media for biological treatment of leachate. Bhatnagar et al., 2017 identified the fraction < 40 mm as composed of 22 wt.% inert materials (glass and stones), 5.4 wt.% biodegradables (paper and wood), 5 wt.% combustibles (mix of plastic textile and rubber), 1 wt.% metals (Fe, Cu and Al) and a larger amount of non-identified material. An amount of about 20 wt.% DM of the fraction < 25.4 mm was reported by Hull et al., 2005 to be accounted for the sum of metal, plastic, glass, textile/rubber/leather and stone/brick/concrete materials.

According to the findings of Prechthai et al., 2008 and Masi et al., 2014, the fractions < 25 mm and < 10 mm, respectively, were mainly composed of organic matter and fine-grained mineral matter, pieces of wood, metals, glass and plastics. The fraction 0.425-6.3 mm was mainly constituted by degraded organic matter mixed with broken glass and ceramics, whereas the fraction < 0.425 mm was composed mainly of mineral particles (Jain, Kim, & Townsend, 2005). A soil-like mixture of minerals and organic matter was the most abundant material in the fraction < 20 mm identified by manual sorting reported by Kaartinen et al., 2013; this fraction was particularly dominant within the fraction < 4 mm.

The high content of fine-grained aggregates of mineral and organic particles in the aged MSW is likely the result of the daily covering soil (Chen, Guan, Liu, Zhou, & Zhu, 2010) and the humification of organic matter in fresh MSW. Organic waste normally degrades and cannot be identified after some years of being landfilled (Quaghebeur et al., 2013), since the material is gradually transformed into humus. Humus is the stable state reached by organic matter after being degraded down to the point where organic matter resists further degradation and constitutes one of the main components of soil, together with liquids, gases, minerals and living microorganisms (Stevenson, 1994).

Prechthai et al., 2008 found no significant variation in waste-type composition of waste among different sampling locations within one landfill. However, he reported variations in the composition of the fine fractions along the vertical profile, suggesting a variation depending on the degree of biodegradation of waste over time in the dumpsite. Changes regarding the content of individual fractions over time were also observed by Chen et al., 2010; Hull et al., 2005; Kaartinen et al., 2013; Quaghebeur et al., 2013; Sormunen, Laurila, & Rintala, 2013, where the mass fractions of paper and cardboard, textiles and wood were lower in older wastes. This was also the case for Jain et al., 2005, where the lower percentage of paper found in samples of an older part of the landfill than in the more recent one suggest the decomposition of paper over time. The quantity of paper-cardboard in excavated waste seems to be a useful parameter regarding the stabilization state of the material that can be easily determined on site (Francois, Feuillade, Skhiri, Lagier, & Matejka, 2006).

Further data on the material composition of the fine fractions shows that the amount of degradable components decreased over time, while the amount of degraded components increased up to 60 wt.% (Francois et al., 2006). Such changes can be also attributed to the composition differences of the landfilled waste due to waste management systems, legislation, changes in the consumption and production trends during the landfilling lapse (Quaghebeur et al., 2013; Spooren, Nielsen, Quaghebeur, & Tielemans, 2012).

The data on the amount of plastic, metal, glass and inert materials did not present a significant variation in time, matching with the expected behavior for slow- or non-biodegradable components (Francois et al., 2006). Plastics recovered from landfills show similar properties to those of plastics from MSW and other secondary plastics (Maul & Pretz, 2016).

Therefore, it can be concluded from previous studies that the amount of biodegradable materials, from the initial quantity, in landfill sites tends to decrease with time, while the amount of slow- and non-biodegradable materials tends to remain without high variations.

According to Wolfsberger et al., 2015 the amount of recyclables and materials for energy recovery in the fine fractions (< 40 mm) was significantly lower than in the coarse fractions, identified around 33 wt.% for the fine fraction. This value, despite being lower than for the coarse fractions, represents an interesting amount of material due to the fact that the fine fractions make up most of the excavated material. For instance, according to Bhatnagar et al., 2017, it would be possible to obtain 23% revenue, with respect to the total income from material recovery, via individual materials from the fraction < 40 mm.

Quaghebeur et al., 2013 reports that for certain waste fractions (i.e. metals, plastics, glass/ceramics, stones and textile) the amount found in the excavated material was comparable to the amount originally present in the waste when initially landfilled. Therefore, records with regard to the composition of fresh waste sent to a landfill over time can be a good source of information to estimate the composition of the disposed material at the site; with exception of biologically degradable materials, especially biowaste, followed by paper and paperboard, which degrade over time (Quaghebeur et al., 2013).

The previous data suggest that the fine fractions might result interesting as source of potentially recoverable materials as metals, plastics and soil-like material, as well as a source of inert materials like sand, glass and ceramics. However, when the quality of paper and cardboard, plastics, textiles and wood (calorific fractions in general) recovered from a landfill is too low or when concentrations of specific compounds, whose amounts are restricted in certain recycling routes are exceeded, waste-to-energy could be the most suitable valorization path (Quaghebeur et al., 2013). Nevertheless, there are also limit values for certain pollutants that apply to WtE and need to be taken into account. For example, an Austrian investigation accounts for a case in which several calorific fractions from LFM material did not meet the limit values applicable in Austria (Wolfsberger et al., 2015). For instance, washing of plastics from LFM reduced the contents of most heavy metals, but not of antimony (Sb) which was incorporated into the polymer (Liebetegger, 2015).

3.2 Particle size distribution

Table 3 gives an overview of the results on the particle size distribution of different studies: for most of the studies, the biggest amount of material belongs to the fine fractions, followed by the coarse fractions and the intermediate fractions. Logically, their amounts will depend mainly on the set particle size for the sieving process, but, however, most of the reviewed previous investigations have shown a consistent dominance of fractions < 40 mm upon the coarser fractions. Information of this kind is difficult to compare, since same particle sizes for the sieving of the excavated material are hardly used in different investigations.

A study on the physico-chemical characteristics of landfilled municipal solid waste of various ages (3, 8, 20 and 30 years old) at four different sites realized by Francois et al., 2006, shows that the particle size distribution of the waste (considering materials ≥ 100 mm as coarse fraction, materials < 100 mm but >= 20 mm as middle fraction and materials < 20 mm as fine fraction) changes from predominantly coarse fraction (approx. 50 wt.%) for 3 years old waste to mainly fine and middle fractions for 30 years old waste (approx. 46 wt.% and 40 wt.%, respectively). The data obtained from the time in between, 8 and 20 years old, show a clear gradual amount reduction of the coarse fraction, as well as a clear gradual amount increase of the fine fraction. The data for the middle fraction shows some fluctuation over time, as it would be logically expected.

Landfill mining tests carried out at a MSW landfill in Sweden by (Hogland, Marques, & Nimmermark, 2004) revealed that about 70-80 wt.% of the fraction < 18 mm (17-22 years old waste) in all excavated depths was within the size range 10-1 mm.

Figure 1 depicts the particle size distribution within the fine fractions of additional studies. Most of these studies present similar results to those of Hogland et al., 2004, where the majority of the fine fractions of excavated land-

TABLE 3: Particle size distribution of excavated waste from previous LFM investigations.

	Parameter	Prechthai et al., 2008 (Nonthaburi, Thailand)	Jani et al., 2016 (Högbytorp, Sweden)	Jain et al., 2005 (ACSWL, USA)	Mönkäre et al., 2016 (Kuopio, Finland)	Kaartinen et al., 2013 (Kuopio, Finland)	Hull et al., 2005 (BCRRC, USA)	Zhao et al., 2007 (Shanghai, China)	Hogland et al., 2004 (Maasalycke, Sweden)	Quaghebeur et al., 2013 (REMO, Belgium)	Mönkäre et al., 2016 (Lohja, Finland)	Masi et al., 2014 (Lavello, Italy)	Masi et al., 2014 (Lavello, Italy)
Type of waste disposed of		MSW	MSW + C&D	MSW	MSW	MSW	MSW + C&D + IW	MSW	MSW	MSW	MSW + C&D + soil	MSW	MSW
Age of waste [a]		3 - 5	5	3 - 8	1 - 10	5 - 10	1 - 11	8 - 10	17 - 22	14 - 29	24 - 40	30 - 60	30 - 60
Particle size	> 100 mm	-	-	-	-	31.0 - 34.0%	-	-	-	-	-	-	-
	> 50 mm	69.0%	-	-	-	-	-	-	48.2 - 59.2%	-	-	-	-
	> 40 mm	-	24.0%	-	-	-	-	25.5 - 70.6%	-	-	-	-	-
	> 6.3 mm	-	-	40.9%	-	-	-	-	-	-	-	-	-
	40 - 100 mm	-	-	-	-	16.0 - 17.0%	-	-	-	-	-	-	-
	25 - 50 mm	13.0%	-	-	-	-	-	-	-	-	-	-	-
	18 - 50 mm	-	-	-	-	-	-	-	21.8 - 31.4%	-	-	-	-
	20 - 40 mm	-	-	-	-	6.0%	-	-	-	-	-	-	-
	15 - 40 mm	-	-	-	-	-	-	14.9 - 32.6%	-	-	-	-	-
	10 - 40 mm	-	38.0%	-	-	-	-	-	-	-	-	-	-
	0.425 - 6.3 mm	-	-	14.5%	-	-	-	-	-	-	-	-	-
	< 25.4 mm	-	-	-	-	-	50.0 - 52.0%	-	-	-	-	-	-
	< 25 mm	18.0%	-	-	-	-	-	-	-	-	-	-	-
	< 20 mm	-	-	-	38.0 - 53.9%	43.0 - 47.0%	-	-	-	-	39.8 - 73.6%	-	-
	< 18 mm	-	-	-	-	-	-	-	14.8 - 24.7%	-	-	-	-
	< 15 mm	-	-	-	-	-	-	12.8 - 45.3%	-	-	-	-	-
	< 10 mm	-	38.0%	-	-	-	-	-	-	44.0 ± 12.0%	-	70.4%	-
	< 4 mm	-	-	-	-	-	-	-	-	-	-	-	63.6%
	< 0.425 mm	-	-	44.6%	-	-	-	-	-	-	-	-	-
Notes:		*Information organized according to age of waste* *Figures have weight basis* *MSW - Municipal solid waste* *C&D - Construction and demolition waste* *IW - Industrial waste*											

fill MSW / MSW + C&D / MSW + C&D + IW / MSW + C&D + soil with various waste ages were composed of a particle size over 1 mm.

On the other hand, according to Jani et al., 2016 the fraction < 10 mm represented 38 wt.% of the total excavated material (5 years old MSW + C&D material) and were composed mainly of soil-like material and minerals; from which 98 wt.% were smaller than 4 mm and 80 wt.% were smaller than 2 mm.

Previous LFM studies have similar results. For instance, Mönkäre et al., 2016 reported that about 78-81 wt.% of the fraction < 20 mm was smaller than 11.2 mm and about 51-52 wt.% of it was smaller than 5.6 mm in a landfill containing 1-10 years old waste (MSW), whilst a site with 24-40 years old waste (MSW + C&D + soil) presented ratios of 88-93 wt.% and 66-74 wt.% (except one sample having 40 wt.% under 5.6 mm), respectively, for the same fraction.

Miller, Earle, & Townsend, 1996 reported that most (around 99%) of the landfill cover soil passed through a sieve of 0.425 mm, while retaining a majority of the biodegradable material. The fine fraction < 0.425 mm was composed mainly of sand, which had the lowest organic matter content of all three fractions. Moreover, together the fractions < 0.425 mm and 0.425-6.3 mm constituted about 60

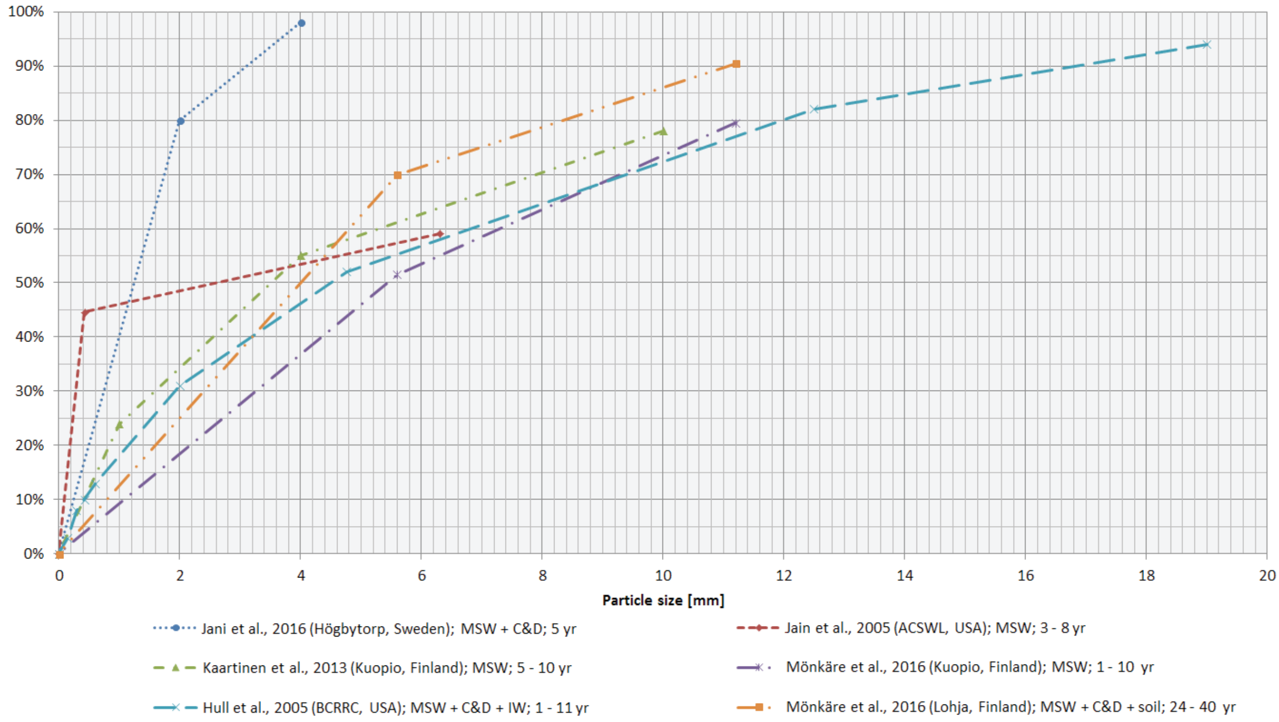

FIGURE 1: Particle size distribution within excavated fine fractions from previous LFM investigations.

wt.% of the excavated landfill MSW (3-8 years old) by Jain et al., 2005; where 44.6 wt.% corresponded to the fraction < 0.425 mm and 14.5 wt.% to the fraction 0.425-6.3 mm.

Excavated landfill MSW (10 years old waste), which consisted of around 54 wt.% material < 40 mm and about 46 wt.% material > 40 mm, exhibited a slight increase in the amount of the material < 40 mm with depth (Burlakovs, Kaczala et al., 2016; Bhatnagar et al., 2017). The results obtained by Kaartinen et al., 2013 indicated transport of the fraction < 20 mm (5-10 years old excavated MSW) towards the bottom layer of the landfill as well. The age of the disposed waste can affect the particle size distribution in a landfill (Hull et al., 2005); several fractions of the older waste (7-11 years old MSW + C&D + IW) presented greater amounts of material < 25.4 mm. Thus, it can be inferred that the amount of fine fractions might increase over time due to the reduction of the particle size of certain waste materials, mainly organic materials, driven by biodegradation and weathering effects. However, it is relevant to point out that a larger amount of fine particles can be found in deeper layers of the landfill due to vertical transport (i.e. downward migration due to gravitational force) rather than biodegradation and weathering effects, which could mislead to the consideration of higher values for the decrease in particle size due to degradation of waste over time.

Other interesting findings include that, for example, a visual inspection by Kaartinen et al., 2013 indicated that the fraction < 4 mm was predominantly composed of soil. Spooren et al., 2012 reported an average of 43 wt.% for the fraction < 10 mm from excavated landfill MSW (14-29 years old material). Hull et al., 2005 suggested that in order to remove all visual contaminants a 2 mm screen is to be employed, since non-soil materials such as plastic, paper flakes and broken glass generally did not pass through; in this manner the mass of the fraction < 25.4 mm could be reduced by about 70% as well.

According to Spooren et al., 2012, common industrial waste separation techniques are unable to sort materials with a particle size below a certain threshold, which often lies within the range of 2-10 mm.

From the gathered information above it can be extracted that: the amount of the fine fractions in landfilled MSW seems to increase over time, whereas their particle size seems to decrease; in a landfill a larger amount of fines could be expected with depth; most of the material composing the fine fractions from excavated landfill MSW is likely to have a particle size larger than 1 mm; most of non-soil materials such as plastics, paper, textiles, stones and broken glass and ceramics could be removed through sieving (probably around 2 mm); the under-sieve material could be expected to be mainly soil-like material (including inert materials) and landfill cover soil and fine inert materials could be recovered via further finer sieving (probably around 0.5 mm).

Therefore, it is relevant to emphasize that in LFM and ELFM future investigations the particle size will be a key parameter for the separation of the fine fractions into exploitable resources and the minimization of the material to, if the case, be sent back for re-landfilling. For this, the fine fractions may be classified into certain particle size ranges, selected according to the results of the material characterization and particle size distribution during the exploration phase of a LFM project, to determine the cut-off diameter size for the fine fractions and enable more efficient material recovery, for different purposes (e.g. recycling and alternative fuel), and recuperation of soil-like and inert materials in the corresponding processing techniques (e.g. density, magnetic and eddy-current separators, among others). This can result very useful to implement a material processing approach especially designed for the particular characteristics of each particle size range, as well as to concentrate some of the moisture and undesired substances (e.g. heavy metals) into a few of the finest particle size ranges.

3.3 Moisture and organic content

The moisture content of the excavated waste is an important characteristic that determines the environmental conditions in the landfill and plays an important role when considering the material processing (Hull et al., 2005). In a landfill it depends on many interrelated factors, such as waste composition (e.g. percentage of organic matter, plastics, inert, etc.), waste type (e.g. MSW, C&D, Industrial waste), waste properties, local climate and weather conditions, landfill operation procedures, gas and leachate collection systems, water generation and consumption due to microbiological activity, between others (Qian, Koerner, & Gray, 2002). Moisture is predominantly present in the fine fractions, as small pores hold water stronger than large pores (capillary action). This is why moisture is a key parameter regarding the treatment of the fine fractions.

Moreover, moisture is one of the most relevant factors influencing the biodegradation of organic matter, playing a vital role in all microorganism´s metabolism, and, hence, it is highly interrelated with the organic content in a landfill (Bäumler & Kögel-Knabner, 2008). The water content is also related to the organic content because organic matter can store a manifold of its own weight of water; this is also valid for certain types of clay minerals. Furthermore, the microbial activity and organic matter play a very important role in the absorption and mobilization of metals (Bozkurt, Moreno, & Neretnieks, 1999; Bradl, 2005).

The water content of the excavated waste can vary significantly and needs to be taken into account when assessing the valorization and treatment options for ELFM (Quaghebeur et al., 2013). It is to be noted that the sampling procedure and the approach with which the water content is determined might have relevant effects on the determined value and, hence, the real water content might differ from the calculated value. For example, the calculated water content can result in a lower value due to water losses during sampling and sieving activities.

Previous experiences include that moisture contained in excavated waste did not impede its processability, but it might have affected the processing efficiency (Kaartinen et al., 2013). Thus, some studies have recurred to the drying of the fines fraction for better results (Hull et al., 2005; Jain et al., 2005; Kaartinen et al., 2013; Kurian et al., 2003; Prechthai et al., 2008; Quaghebeur et al., 2013). Drying of the fine fractions could: (i) reduce the amount of surface defilements; increasing the quality of the recyclable materials and raising the efficiency of sorting processes, especially for the sensor-based sorting technologies, such as near infrared (NIR) and color recognition (VIS), (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.

An additional study by Jain et al., 2005, investigated differences regarding physical appearance, such as presenting darker color, smaller particle size and higher degree of degradation for landfill-mined material which has been previously exposed to leachate recirculation; while no significant difference was observed in the mean moisture content when compared with landfill-mined material without leachate recirculation.

Like the moisture also the organic matter is enriched in the fine fraction, as degradation processes of biowaste decrease its grain size over time in a landfill. Table 4 shows the results obtained on moisture and organic contents from various LFM studies. Into this respect it can be observed that the moisture content varies between 16 and 54 wt.% and the organic matter content between 9 and 21 wt.% (dry matter) for landfills with comparable ages (up to 10 years) and type (MSW) of disposed material, as well as similar particle sizes (< 20 mm); while for older excavated material (17 to 40 years old) the moisture and organic content seem to decrease slightly to ranges of 18-40 wt.% and 5-14 wt.% (dry matter), respectively.

The decrease of organic matter content with the increase of the age of the waste was also observed by Mönkäre et al., 2016 and Hull et al., 2005. This showed congruency with the results obtained by Francois et al., 2006 as well, indicating that younger material is less degraded than older material. Thirty year old material presented volatile solid contents (VS) characteristics for stabilized material (Kelly, 2002). Ayuso, Hernández, García, & Pascual, 1996 reported that the organic matter content of 30 year old material were close to the characteristics of soil. In this respect, a model (Tabasaran & Rettenberger, 1987) can be used to estimate the organic decay in landfill through the prognosis of landfill gas generation.

It has been observed that organic matter influences the capacity of waste to hold water, known as field capacity (Sormunen et al., 2013; Zornberg, Jernigan, Sanglerat, & Cooley, 1999). The higher the content of organic matter, the higher is the water content to be expected (Hull et al., 2005). The biodegradable organic matter content in the waste reported by Zhao et al., 2007 was significantly higher than in the cover soil used at the landfill.

The contents of total and volatile solids determined by Mönkäre et al., 2016 showed no trends regarding site or depth and her results indicate that organic matter can remain for a long time in a landfill, which is explained by the formation of stable humic substances during the biodegradation of organic matter. Volatile solids, despite not a measure of available organic matter, might be a simple and inexpensive way to assess the potential degradability of the excavated waste from a landfill (Hull et al., 2005).

The composition of the cover layer employed at a site seems to play a relevant role regarding the waste degradation rate; meaning that the degradation process could be significantly faster with the use of a high to medium permeability material (e.g. compost) than with a low permeability one (e.g. clay) (Francois et al., 2006). This can be explained by the fact that the use of a permeable material as cover

TABLE 4: Organic content, total solids and water content of excavated fine fractions from previous LFM investigations.

Parameter	(Filborna, Sweden) in Kurian et al., 2003	Bhatnagar et al., 2017 (Kudjape, Estonia)	Hull et al., 2005 (BCRRC, USA)	Kurian et al., 2003 (Perungudi, India)	Mönkäre et al., 2016 (Kuopio, Finland)	Kurian et al., 2003 (Kodungaiyur, India)	Mönkäre et al., 2016 (Lohja, Finland)	Gutiérrez-Gutiérrez et al., 2015 (4 sites in UK)	Hogland et al., 2004 (Maasalycke, Sweden)	Jani et al., 2016 (Högbytorp, Sweden)	(Deonar, India) in Kurian et al., 2003
Type of waste disposed of	MSW	MSW	MSW + C&D + IW	MSW	MSW	MSW	MSW + C&D + soil	MSW and MSW + C&I	MSW	MSW + C&D	MSW
Age of waste [a]	-	10	1 - 11	0 - 10	1 - 10	10	24 - 40	-	17 - 22	5	-
Particle size [mm]	< 40	< 40	< 25.4	< 20	< 20	< 20	< 20	< 19	< 18	< 10	< 8
Organic content	-	-	-	8.9 - 15.8%	8.8 ± 0.4 - 16.9 ± 0.7%	8.9 - 20.7%	4.9 ± 0.4 - 14.3 ± 0.8%	25.0 ± 10.0 - 41.0 ± 9.0%	-	16.6%	14.5%
Total solids	-	-	-	-	46.2 ± 1.7 - 63.7 ± 1.7%	-	59.6 ± 1.6 - 81.6 ± 1.0%	-	-	-	-
Water content	30.0 - 38.0%	≤ 40.0%	16.0 - 43.0%	21.4 - 52.0%	-	15.5 - 46.0%	-	58.0 ± 70. - 79.0 ± 9.0%	22.3 - 28.8%	23.5%	14.0%
Notes:	*Information organized according to particle size* *Figures have weight basis* *MSW - Municipal solid waste* *C&D - Construction and demolition waste* *IW - Industrial waste* *PPC - Paper, paperboard and cardboard*										

layer in a landfill could favor the aerobic biological degradation of certain organic components, which degrade at a faster rate in aerobic than in anaerobic conditions. This could mean that waste from landfill sites where low permeability cover materials were used might present higher organic content, which could eventually be found in the fine fractions. Nonetheless, the use of compost as cover material might raise the organic content of the fine fractions as well.

Studies in Europe on fresh organic MSW have revealed that the average total organic carbon remains relatively stable, around 43-44 wt.%, despite the heterogeneity of the organic matter content (Baky & Eriksson, 2003; Iglesias Jimenez & Perez Garcia, 1992), suggesting that the variation of this parameter in the fine fractions from LFM depends strongly on the treatment of the MSW before disposal and the conditions at the landfill site.

From this information it can be concluded that significant amounts of organic matter and moisture are likely to be present in LFM material and, eventually, to be found in the fine fractions. These interrelated parameters are of critical relevance, since processing routes and possible end-uses for these fractions will depend on their quantities. Depending on the amount of moisture it can be determined if a dry or wet further processing of the material is to be employed and the efficiency of the chosen method will depend considerably on the moisture content. The organic content can be used to determine if material or energy recovery routes should be pursued or, otherwise, if the material is suitable to be re-landfilled according to the legislation in force.

Fine fractions might be used in the future to predict the moisture content in samples containing all fractions, as results on moisture between samples of fines and samples containing all fractions have shown good correlation and representative samples of the fines are easier to take than those containing all fractions (Hull et al., 2005).

3.4 Physico-chemical properties

Table 5 displays data on some physico-chemical properties of the fine fractions obtained in the reviewed investigations. This information shows that most of the compared parameters presented roughly similar ranges within different sites, taking into account the differences among them (mainly in particle size and age of waste).

Characteristics such as calorific value, amount of organic carbon, total carbon, ash content, hydrogen and nitrogen contents are needed to assess the efficiency for WtE applications (Quaghebeur et al., 2013).

Most physical, chemical and microbial processes, such as dissolution of waste materials and metabolites and emissions of volatile substances, as well as the pressure conditions, in a landfill are affected by temperature (Hull et al., 2005). The temperature of landfilled waste has been seen to increase with a rate of approximately 1 °C per m of depth (Attal, Akunna, Camacho, Salmon, & Paris, 1992; Gurijala & Suflita, 1993; Hull et al., 2005; Zornberg et al., 1999). Temperature also plays a decisive role regarding microbiological activity and biochemical reactions inside the landfill, which are linked in parallel to the moisture, organic matter content and pH, among others.

The calorific value of a waste fraction is mainly driven by the amount of carbon (usually measured as total carbon or total organic carbon) and the ash and moisture contents.

Chemically, fine fractions of landfill mining consist mainly of SiO_2, CaO, Al_2O_3 and FeO (Spooren et al., 2012). However, the recovery of these compounds is limited by the presence of environmentally problematic elements, especially heavy metals. Nevertheless, the environmental impact of a material does not depend on its total content of hazardous substances, but on their mobility. Some parameters of significance in control of metal mobility such as pH, sulfide, sulfate and chloride contents in a landfill have been studied by Gould, Cross, & Pohland, 1990; the latter reported that a broad range of attenuating mechanisms limiting the mobility of toxic metals in the leachate from columns simulating landfill material with shredded MSW occurred, suggesting that MSW has a capacity for minimizing the mobility of heavy metals.

The values for pH and phosphorous of the fines fraction complied with the limits for compost standards, while the contents of N and K did not (Prechthai et al., 2008). The variation of pH towards the bottom of a dumpsite show the varying decomposition rate of organic waste (Das, Smith, Gattie, & Hale Boothe, 2002; Townsend, Miller, Lee, & Earle, 1996). It is known that the pH of waste varies over time in a landfill depending on the phase the waste is going through, i.e. aerobic phase, anaerobic phase (acidic and methanogenic phases) and humic phase; where the waste presents a pH close to 7 units at the beginning, decreases to about 4-6 units in the acidic phase, raises back to around 7-8 units during the methanogenic phase and remains slightly basic, approximately 8 units, for the humic phase (Bozkurt et al., 1999). The humic phase is reached when all the readily degradable organic matter has been degraded and remains in the waste in a very stable substance state (Bozkurt et al., 1999). This suggests that the pH of the waste in a landfill can be expected to decrease with depth in some cases; depending on the characteristics of the site, operational procedures and type of waste contained. Nonetheless, it is to be considered that due to the strong heterogeneity in a landfill, different parts of the landfill might develop at different rates (Bozkurt et al., 1999) and, thus, a decreasing tendency of the pH with depth might not be always the case. A pH variation within the range of 4-9 units was observed in the reviewed investigations displayed in Table 5. Low pH and high TOC values are indicators of incomplete biodegradation of the material (Kurian et al., 2003).

The nutrient contents were low for two observed landfills; especially total phosphorus was below detection limit of 10 mg/kg (dry matter) in most sampling points (Mönkäre et al., 2016). Results obtained by Hogland et al., 2004 for total phosphorus showed similar concentrations. Values for fresh biowaste from household waste in Denmark

TABLE 5: Physico-chemical properties of excavated fine fractions from previous LFM investigations.

Parameter	(Filborna, Sweden) in Kurian et al., 2003	Wolfsberger et al., 2015 (Lower Austria, Austria)	Hull et al., 2005 (BCRRC, USA)	Prechthai et al., 2008 (Nonthaburi, Thailand)	Kurian et al., 2003 (Perungudi, India)	Mönkäre et al., 2016 (Kuopio, Finland)	Kaartinen et al., 2013 (Kuopio, Finland)	Kurian et al., 2003 (Kodungaiyur, India)	Mönkäre et al., 2016 (Lohja, Finland)	Hogland et al., 2004 (Maasalycke, Sweden)	Jani et al., 2016 (Högbytorp, Sweden)	Kaczala et al., 2017 (Kudjape, Estonia)	Quaghebeur et al., 2013 (REMO, Belgium)	(Deonar, India) in Kurian et al., 2003
Type of waste disposed of	MSW	MSW	MSW + C&D + IW	MSW	MSW	MSW	MSW	MSW	MSW + C&D + soil	MSW	MSW + C&D	MSW	MSW	MSW
Age of waste [a]	-	13 - 20	1 - 11	3 - 5	0 - 10	1 - 10	5 - 10	10	24 - 40	17 - 22	5	5 - 6	14 - 29	-
Particle size [mm]	< 40	< 40	< 25.4	< 25	< 20	< 20	< 20	< 20	< 20	< 18	< 10	< 10	< 10	< 8
Ash content	78.9%	-	-	68.6%	84.2 - 91.1%	-	-	79.3 - 91.1%	-	87.3 - 90.2%	-	-	64.4- 85.0%	-
Bulk density [kg/m³]	400 - 500	-	370 - 1,206	-	745 - 1,147	-	-	853 - 1,254	-	-	690	-	-	-
Calorific value [MJ/kg]	-	4.4 - 9.0	-	-	-	-	-	-	-	0.4 - 0.9	1.7	-	2.2 - 4.8	-
pH	4.0 - 5.0	-	-	7.7 ± 0.3	7.6 - 8.6	6.8 - 7.6	8.1 ± 0.1 - 8.3 ± 0.1	6.9 - 8.1	7.2 - 7.9	7.0 - 7.3	7.7	7.1 - 8.3	-	7.2
Total organic carbon	13.0%	10.0 - 20.0%	-	-	5.2 - 7.9%	4.7 - 5.6%	4.7 ± 0.8 - 5.8 ± 1.6%	4.5 - 10.4%	-	-	5.6%	0.2 - 0.4%	7.6 - 12.4%	5.8%
Notes:	*Information organized according to particle size Percentage figures have weight basis MSW - Municipal solid waste C&D - Construction and demolition waste IW - Industrial waste*													

for nitrogen content reported by Riber, Petersen, & Christensen, 2009 were significantly higher than the ones from landfilled material reported by Hogland et al., 2004 and Mönkäre et al., 2016. This reflects the consumption and migration of the nutrient content in the fine fractions after disposal. Total nitrogen and phosphorus measurements by Zhao et al., 2007 in fine fractions indicated that nitrogen levels decrease over time, whereas phosphorus levels remain steady; the measured contents of these elements in the fine fractions were higher than in the cover layer.

The chemical oxygen demand and heavy metals, chlorides and fluorides contents in leachate of the fine fractions can be useful to identify the ability to reuse these fractions as a construction material outside the landfill, landfill cover material or for landfilling as inert material (Jani et al., 2016). The sulfur, chlorine, fluorine and bromine content of waste is needed to assess the emission levels during thermal conversion (Quaghebeur et al., 2013).

According to Kaartinen et al., 2013, the fine fractions exhibited generally non-hazardous properties in leaching tests performed to assess landfill acceptability; nevertheless, leaching of dissolved organic carbon from fine fractions of young disposed MSW may be challenging for landfill disposal in the EU. Therefore, pH-dependent leaching tests (e.g. European standard EN 14429) might result essential to identify the recycling possibilities of waste materials like the fine fractions.

Jani et al., 2016 reported that the calorific value, methane gas potential and total organic carbon decreased with the time waste has been disposed of, since lower values were obtained for the waste in older layers of the landfill, where the organic materials showed a larger decomposition. The determination of the methane potential could be used to identify the suitability of the fine fractions for energy recovery or the need for stabilization to prevent emissions (Mönkäre et al., 2016).

As shown in Table 5, mostly low heating values (between 0.4-4.8 MJ/kg) were reported for the fine fractions in previous LFM studies (Hogland et al., 2004; Jani et al., 2016; Quaghebeur et al., 2013). Nonetheless, Wolfsberger et al., 2015 obtained higher values (around 4.4-9 MJ/kg). As already stated, moisture, carbon and ash contents are highly interrelated with respect to the calorific value and, despite the lack of information regarding these parameters for some of the compared studies, it could be observed that the highest calorific value range (4.4-9 MJ/kg) corresponds to the highest total organic content range (10-20 wt.%), whereas the lowest calorific value range (0.4-0.9 MJ/kg) corresponds to the highest ash content range (87.3-90.2 wt.%).

There was not enough comparable information in the studies presented in Table 5 to identify a correlation between particle size and age of waste regarding the calorific value. Nevertheless, additional previous investigations reported that the calorific value and total organic carbon concentration decreased with increasing storage time of the waste in the landfill; what is most likely the result of decomposition of carbon-rich material into landfill gas over time (Quaghebeur et al., 2013). The analyses from Masi et al., 2014 show that the fraction < 4 mm has a percentage of the total organic carbon more than six times higher than a conventional agrarian soil.

The bulk density showed a variation from 370 to 1,254 kg/m^3 in Table 5; where the lowest density range (400-500 kg/m^3) corresponds to the coarsest particle size (< 40 mm) from the compared studies. The highest bulk density ranges (853-1,254 and 745-1,147 kg/m^3) corresponded to a particle size of < 20 mm, which was not the finest particle size from the compared studies but presented the highest ash content ranges (79.3-91.1 and 84.2-91.1 wt.%).

The cellulose content, cellulose-to-lignin or cellulose-to-VS ratios have been used in MSW decomposition studies as an indicator of degradation grade of the waste in landfills (Bookter & Ham, 1982; Ham, Norman, & Fritschel, 1993; Jones, Rees, & Grainger, 1983; Mehta et al., 2002; Wang, Byrd, & Barlaz, 1994). A cellulose-to-lignin ratio of < 0.2 (30 year old waste) indicates relatively well stabilized waste compared to less degraded waste with a ratio of 0.9-1.2 and fresh waste with a ratio of 4 (Bookter & Ham, 1982). These ratios could be used to determine the degree of degradation of the fine fractions and, thereby, evaluate their material or energy recovery potential.

Phytotoxicity was tested by Masi et al., 2014, which reported that the acute tests did not demonstrate particularly adverse effects on the growth of test species for two of three species. According to the results obtained by Prechthai et al., 2008 the phytotoxicity of waste to inhibit the germination of rice seed was relatively low and signified the completed degradation of organic matter in the fines fraction; suggesting the safe and suitable usage of the material as compost for non-edible crops. For this, materials like stone, glass, metal and plastics, which can be a problem in the soil, are to be removed (Masi et al., 2014; Prechthai et al., 2008). However, it has to be considered, that limit values with respect to the total and leachable contents of environmentally problematic substances can be far below the concentrations which yield a visible effect in ecotoxicity tests. Consequently, the lacking phytotoxicity of a material does not allow its recycling per se.

3.5 Metals content

Table 2 shows that the contents of total metals in LFM fine fractions were relatively low regarding metal recovery potential; except for a site in the USA, where the disposal of industrial waste (IW) and construction & demolition waste (C&D) together with MSW was registered, and a site in Sweden, where the upper limit of the fine fraction was set at a coarser particle size (< 40 mm). The latter presented a considerably higher amount of metals than a study in Austria that contained the same type of waste (only MSW) and used the same particle size as upper limit for the fines fraction (Wolfsberger et al., 2015). Spooren et al., 2012 found an amount of 3±2 wt.% of ferromagnetic material (such as ferromagnetic metals and metal oxides) in the fraction < 10 mm of excavated landfill MSW with respect to the same fraction.

Furthermore, amounts of around 99.9% metals and around 90% non-metals, of their whole amount within the fines, have been reported as still found in the fines at the beginning of the humic phase by Belevi & Baccini, 1989 and

Bozkurt et al., 1999, showing low degradation of metals in a landfill.

It is relevant to mention that the removal of metals could be negatively influenced when the reach of the metal sorting equipment is limited to larger particle sizes or the metals present poor quality, as they might be oxidized and in degraded conditions.

Nonetheless, there have been additional studies where the amount of metals has been found in significant concentrations, such as a study on the content of metallic elements in the fraction < 10 mm, at different depths, of the excavated waste from a MSW landfill (Burlakovs, Kaczala et al., 2016), which unveils that interesting concentrations of several metals with respect to material recovery, i.e. Fe (average concentrations above 10,000 mg/kg), Mg and Zn (average concentrations above 1,000 mg/kg), can be found in the fine fractions of landfilled waste, as well as concentrations above 100 mg/kg of metals like Mn, Ba, Cu, Pb and Sr. Also, results on the fraction < 40 mm show that the metal content (mainly Fe, Al and Cu) was about 0.6 wt.% of the same fraction (Burlakovs, Kriipsalu et al., 2016). Significant concentration ranges of Al (12,079-17,274 mg/kg) and Cu (1,027-2,595 mg/kg), in waste mined from landfills were reported by Gutiérrez-Gutiérrez, Coulon, Jiang, & Wagland, 2015. Bhatnagar et al., 2017 identified 1 wt.%, of the total amount of the fraction < 40 mm, of Fe, Cu and Al in the same fraction.

Additionally, the amount of magnetic metals recovered from a full-scale process by Kaartinen et al., 2013 was around 1 wt.% from the total processed waste; moreover, this amount was smaller than the amount of total metals separated by manual sorting, which was 3-4 wt.%. Al and Fe recovery from the fine fractions are of interest; as their concentrations in the fines could yield around 2-2.5 wt.% of Al and 1.5-2 wt.% of Fe of the total amount of the same fraction (Kaartinen et al., 2013).

Chemical analyses of fine fractions do not give information about the oxidation state of metals. However, under landfill conditions it is obvious that metals like Ca, Mg, K and Na, but also a significant proportion of Fe are present in oxidized form as minerals. Among these, Ca and Fe were the metals with higher concentration ranges, with 70,000-80,000 mg/kg and 30,000-50,000 mg/kg, respectively, in the fraction < 10 mm; followed by Mg, K, Na and Zn (concentrations between 500-20,000 mg/kg), Mn, Cu and Pb (concentrations around 150-400 mg/kg) and Cr, Ni, Co and Cd (concentrations below 150 mg/kg), according to Bhatnagar et al., 2017. A similar trend in terms of metal concentrations was found between all excavated pits (Bhatnagar et al., 2017). Interesting concentrations of zinc, copper, barium and chromium for metals recuperation were found by Jani et al., 2016 in the fine fractions.

It is relevant to note that the mineralogical bonding of the individual metals has to be considered when assessing the metals recovery potential of a particular site; as the total amount (mixed with minerals and other materials), the metallic amount and the amount found in compounds (e.g. oxides) of these elements play a crucial role in the determination of the recoverable amount and their speciation.

Moreover, fractions (< 10 mm) of excavated industrial waste can contain higher magnetic metals concentrations than mined MSW, seeing that a concentration of around 0.5-5.3 wt.% of ferromagnetic material was obtained from MSW and one of about 25-29 wt.% from IW by Spooren et al., 2012 and Quaghebeur et al., 2013. This can be corroborated with the results obtained by Hull et al., 2005 presented in Table 2; in which the metals content was higher in a site where MSW, C&D and IW were landfilled than in the sites where solely MSW was registered. The analysis of previous studies focused on mining industrial waste from landfill is not within the scope of the present review; nonetheless, the recuperation of metals from landfills for industrial waste might result interesting for future review.

According to the preliminary results obtained by Quaghebeur et al., 2013 the removal of the magnetic metals from the fraction < 10 mm could result in a reduction of more than 50 wt.% of the total amount of metals in the same fraction.

The previous information suggests that mechanical processing technologies still have optimization potential for higher yields.

Table 6 encompasses data gathered on specific metals, mainly heavy metals, found in the fine fractions in LFM studies carried out in the past. Heavy metals might accumulate in the fine fractions due to their high specific surface area for interaction (Jain et al., 2005; Wolfsberger et al., 2015). This suggests that a significant part of these heavy metals occur as dissolved species in the pore water or as oxidized precipitates at particle surface. Consequently, the recovery of these metals must include reduction to their elementary state, which would be associated with significant effort.

The results obtained by Masi et al., 2014 showed that the composition of very old dumpsites is relatively uniform and that the concentrations of heavy metals in the fraction < 4 mm were, on average, 30% lower than in the fraction 4-10 mm. This suggests that besides the part which is dissolved, adsorbed or that occurs as fine-grained precipitates, another portion of heavy metals might occur in its metallic state. Furthermore, the classification of the fine fractions into determined particle size ranges might be a way to identify and select more accurately the necessary mechanical processing for each particle size range; enabling a more efficient and appropriate processing according to the properties and material recovery potential of each particle size range.

The heavy metal concentration in the waste fractions to be revalorized as refuse derived fuel (RDF) is also to be taken into consideration (Rotter, Kost, Winkler, & Bilitewski, 2004). It is possible that high concentrations of hazardous substances and heavy metals are found in local pockets (Kurian et al., 2003), since elements such as As, Cd, Co, Cr, Cu, Ni, Pb, Hg and Zn can be found in household products (Slack, Gronow, & Voulvoulis, 2005) and, therefore, in landfill leachate (Reinhart, 1993) depending on the solubility of the respective phases.

For instance, the highest concentrations of Cr and Pb found by Prechthai et al., 2008 were in the fine fractions; which was in accordance with the findings reported by Hogland et al., 2004. High concentrations of Cd and Pb

TABLE 6: Metals content in excavated fine fractions from previous LFM investigations.

Parameter	Wolfsberger et al., 2015 (Lower Austria, Austria)	Hull et al., 2005 (BCRRC, USA)	Prechthai et al., 2008 (Nonthaburi, Thailand)	Kurian et al., 2003 (Perungudi, India)	Kaartinen et al., 2013 (Kuopio, Finland)	Kurian et al., 2003 (Kodungaiyur, India)	Hogland et al., 2004 (Maasalycke, Sweden)	Zhao et al., 2007 (Shanghai, China)	Jani et al., 2016 (Högbytorp, Sweden)	Kaczala et al., 2017 (Kudjape, Estonia)	Bhatnagar et al., 2017 (Kudjape, Estonia)	Quaghebeur et al., 2013 (REMO, Belgium)	Masi et al., 2014 (Lavello, Italy)	Jain et al., 2005 (ACSWL, USA)	Masi et al., 2014 (Lavello, Italy)	Jain et al., 2005 (ACSWL, USA)
Type of waste disposed of	MSW	MSW + C&D + IW	MSW	MSW	MSW	MSW	MSW	MSW	MSW + C&D	MSW	MSW	MSW	MSW	MSW	MSW	MSW
Age of waste [a]	13 - 20	1 - 11	3 - 5	0 - 10	5 - 10	10	17 - 22	8 - 10	5	5 - 6	10	14 - 29	30 - 60	3 - 8	30 - 60	3 - 8
Particle size [mm]	< 40	< 25.4	< 25	< 20	< 20	< 20	< 18	< 15	< 10	< 10	< 10	< 10	< 10	0.425 - 6.3	< 4	< 0.425
Ag [mg/kg]	-	-	-	-	-	-	-	-	-	-	-	-	-	< 1.5 - 24.6	-	< 1.5 - 23.8
Al [g/kg]	-	-	-	-	51.0 ± 10.0 - 57.0 ± 3.1	-	-	-	-	-	-	-	-	2.5 - 169.0	-	1.9 - 76.1
As [mg/kg]	16.0 - 23.0	9.1 ± 8.6	-	0.1 - 1.6	-	0.8 - 5.6	< 0.4	-	5.1 ± 1.7	-	-	27.1 ± 15.0	73.0	1.1 - 58.6	68.0	0.2 - 10.1
Ba [mg/kg]	-	-	-	-	1,100 ± 100 - 900 ± 300	-	-	-	468.0 ± 143.0	-	-	-	-	13.8 - 681.0	-	8.2 - 70.0
Ca [mg/kg]	-	-	-	-	85,000 ± 39,000 - 65,000 ± 12,000	-	-	-	-	-	20,000 - 60,000	-	-	-	-	-
Cd [mg/kg]	1.6 - 4.8	1.2 ± 1.2	4.2	0.8 - 1.8	≤ 100	0.9 - 3.1	0.9 - 1.2	1.1 - 10.7	2.1 ± 0.6	-	0 - 5	5.9 ± 3.8	54.0	< 0.3 - 40.0	55.0	< 0.3 - 13.8
Co [mg/kg]	6.6 - 17.0	-	-	-	-	-	-	-	23.3 ± 5.8	-	5 - 10	-	-	< 0.5 - 86.7	-	< 0.5 - 32.9
Cr [mg/kg]	130.0 - 170.0	26.0 ± 24.0	166.6	110.0 - 261.0	100 ± 100 - 200 ± 100	191.0 - 657.0	47.0 - 78.0	73.5 - 252.1	254.0 ± 54.0	-	10 - 100	495.7 ± 118.0	145.0	9.5 - 531.0	117.0	2.5 - 151.0
Cu [mg/kg]	-	-	2,245.0	75.0 - 217.0	800 ± 1,200 - 200 ± 100	127.0 - 968.0	34.0 - 36.0	-	1,460.0 ± 684.0	45.3 - 105.2	100 - 300	339.3 ± 55.3	1,067.0	5.8 - 5,530	538.0	0.7 - 170.0
Fe [mg/kg]	-	-	-	-	37,000 ± 1,700 - 41,000 ± 2,100	-	-	-	28,724 ± 8,108	-	20,000 - 60,000	27,000 ± 750	-	4,600 - 61,800	-	800 - 28,200
Hg [mg/kg]	0.4 - 0.6	0.4 ± 0.4	-	0.04 - 0.8	-	0.6 - 2.7	0.2 - 0.3	-	0.7 ± 0.2	-	-	0.7 ± 0.5	-	0.04 - 9.0	-	< 0.04 - 1.8
K [mg/kg]	-	-	0.2 ± 0.1	-	16,000 ± 2,900 - 17,000 ± 2,100	-	-	-	-	-	500 - 20,000	-	-	-	-	-
Mg [mg/kg]	-	-	-	-	12,000 - 12,000 ± 600	-	-	-	-	-	500 - 20,000	-	-	-	-	-
Mn [mg/kg]	-	-	947.0	-	700 ± 100 - 1,300 ± 400	-	-	-	-	-	100 - 500	-	3,385.0	50.4 - 14,700	1,241.0	6.2 - 993.0
Mo [mg/kg]	-	-	-	-	< 100 - 100 ± 100	-	-	-	18.8 ± 3.9	-	-	-	-	-	-	-
Na [mg/kg]	-	-	-	-	19,000 ± 2,100 - 21,000 ± 2,500	-	-	-	-	-	500 - 20,000	-	-	-	-	-
Ni [mg/kg]	45.0 - 60.0	-	47.8	21.0 - 50.0	100.0	31.0 - 247.0	14.0 - 15.0	-	111.4 ± 33.7	-	10 - 100	176.3 ± 60.7	138.0	6.4 - 743.0	89.0	0.8 - 340.0
Notes:	*Information organized according to particle size / MSW - Municipal solid waste C&D - Construction and demolition waste / IW - Industrial waste*															

were found in the fine fractions by Wolfsberger et al., 2015. Arsenic was determined to come mainly from the waste rather than from the soil cover (Jain et al., 2005).

In general, heavy metals commonly demonstrate high levels of sorption and precipitation; mercury has been predominantly found as resistant to leaching in landfills (Slack et al., 2005). Only trace levels of volatile heavy metals (As and Hg) have been detected in landfill gas whereas particulate matter contributes more to the emissions of heavy metals from landfills (Parker, Dottridge, & Kelly, 2002). Most of the contaminants released from fine fractions can be found in particulate matter (Kaczala et al., 2017); this suggests that they could be removed using physical methods. Mn, Cd and Zn showed low mobility potential under aerobic conditions in the study of Prechthai et al., 2008 as well. Poor solubility of heavy metal containing phases and slow leaching kinetics of heavy metals in water was reported by Kurian et al., 2003, since the heavy metal concentrations in water extract were lower than of leachate.

Results (Gutiérrez-Gutiérrez et al., 2015) indicate that the leachate is not mobilizing vertically critical metals (critical raw materials – CRMs, according to EU´s criteria) in the landfill, since no direct relationship between depth and concentration has been reported consistently for these metals. Leaching tests done by Kaczala et al., 2017 on landfill-mined material (particle size < 10 mm) for heavy metals such as Zn (0.5-0.9%), Cu (0.2-0.6%) and Pb (0.9-1.1%) have shown low average leaching ratios (leached amount/amount in solid waste matrix). Low leaching ratios for Cu (0.2%), Zn (6.1%) and Pb (0.7%) from under-sieve (< 20 mm) residues, prior to landfilling, were obtained by Cossu & Lai, 2012 as well. However, limit concentrates in the leachate are often very low (< 1 mg/kg dry matter) so that even the leaching of few per mil of the total heavy metal content leads to an excess of limit values.

Other studies on landfills have also reported very low leaching of heavy metals out of the landfill compared to the accumulated amount (Baccini, Henseler, Figi, & Belevi, 1987; Belevi & Baccini, 1989; Finnveden, 1996; He, Xiao, Shao, Yu, & Lee, 2006; Øygard, Måge, & Gjengedal, 2004); where most of the heavy metals were found retained within the waste matrix. Contaminants such as heavy metals will remain in the waste unless leached out (Jain et al., 2005).

Moreover, studies have reported low leaching rates of heavy metals over relatively long periods of time (Esakku, Palanivelu, & Joseph, 2003; Gould et al., 1990; Kjeldsen et al., 2002; Reinhart & Basel Al-Yousfi, 1996; Ross, Harries, Revans, Cross, & NATHANIEL, 2000).

These previous results and experiences suggest that these elements are most likely to be found in the fine fractions even in older landfills and that their relevance for the feasibility of landfill mining should not be overseen.

Nonetheless, the leaching rate of heavy metals in landfill waste can be influenced by several factors. For example, in an open dumpsite, where the oxygen diffusion rate can be high, the conditions for the leaching of heavy metals out of the waste matrix could be favored (Martensson, Aulin, Wahlberg, & Agren, 1999). The content of organic compounds in landfilled waste considerably influences the mobility of metals; it either tends to increase the sorption of metals and delay their release or promote their mobilization (Gutiérrez-Gutiérrez et al., 2015). So the particular conditions of each site must be carefully assessed as one of the first steps.

Younger landfills show lower concentrations of As, Cd, Cr, Cu, Hg, Ni, Pb and Zn in the fine fractions than old ones; this can most likely be attributed to an improved initial quality of the fresh MSW over time (Quaghebeur et al., 2013), e.g. due to separate collection of waste electric and electronic equipment (WEE).

Elements like copper, iron, zinc and rare earth metals are being depleted in their primary sources and there is an increasing demand on these elements due to the development of the standard of living (Jani, Marchand, & Hogland, 2014). As showed by previous studies (Quaghebeur et al., 2013), the fine fractions from landfill-mined waste might contain high concentrations of certain metals, offering attractiveness for feasible material recovery. Special attention should be paid to critical and rare earth metals, as their prices can reach high levels (Bhatnagar et al., 2017). However, the recovery of rare earth metals from landfilled waste could be used as an additional source of revenue sometimes and not as one of the main drivers for LFM, since these metals are used in very specific applications, which were not predominantly disposed of in MSW landfills and, hence, their primary recovery route for recycling might remain to be the separate collection of the products in which they were used.

Rare earth elements (REEs), platinum group metals (PGMs), Li, In, Co and Sb have been identified as high risk of supply shortage and increased impact on the economy (Hislop, 2011 in Gutiérrez-Gutiérrez et al., 2015).

Concentrations of critical metals, such as Co (11 mg/kg), Ga (2.2 mg/kg), Nb (2.5 mg/kg), Ta (1.2 mg/kg), and W (56 mg/kg); REEs, such as Gd (0.75 mg/kg), Nd (7.26 mg/kg), Pr (1.9 mg/kg) and Y (7.85 mg/kg); PGMs, such as Pt (59 µg/kg), Rh (0.092 µg/kg) and Ru (0.5 µg/kg); Ag (5.3 mg/kg) and Au (0.4 mg/kg) have been found present in MSW (Morf et al., 2013). Low recycling rates of REEs, Sb, In, Co and Li have been also reported in Graedel et al., 2011. However, the observed concentrations are far below the cut-off grades which are in the range of several wt% REE oxides, for example (Lehmann, 2014).

Furthermore, the proportion of REEs in the fine fraction reported by Burlakovs, Kriipsalu et al., 2016 between four sites did not present a significant variation; being Ce, La, Nd and Y the elements found in higher amounts with around 35.5 wt.%, 19 wt.%, 18 wt.% and 9 wt.%, respectively, with respect to the total REEs amount. Ce, Nd, Li, Sb and Co were the most abundant metals in the excavated waste reported by Gutiérrez-Gutiérrez et al., 2015 and did not present significant variations between the studied landfills either. Li had the greatest concentration range of all examined critical metals (11.17-27.66 mg/kg), followed by Co (8.72-14.14 mg/kg), Sb (6.40-15.15 mg/kg) and In (0.04-0.10 mg/kg) (Gutiérrez-Gutiérrez et al., 2015). The range concentrations of Ce and Nd were 13.85-25.20 mg/kg and 8.34-11.75 mg/kg, respectively (Gutiérrez-Gutiérrez et al., 2015). Nevertheless, the amounts of Cu, Ag and Au found were highly variable between different sites and the PGMs concentrations, except for Pd (0.41-0.77 mg/kg), were found low (Gutiérrez-Gutiérrez et al., 2015).

The recovery of metals (critical metals and other metals such as Al, Cu, Ag and Au) together with other materials of value (e.g. recyclables and RDF) may result in a feasible business model for LFM (Gutiérrez-Gutiérrez et al., 2015). According to Van Vossen & Prent, 2013, an amount of 2.5 vol.%. metals recovery could reduce the costs of landfill mining about 20%, which could be raised to 30-40% by future prices rises due to increasing raw materials scarcity. Thus, the potential of the fine fractions as secondary source of important metals is emphasized by the considerable amounts of them that can be found in the fines (Bhatnagar et al., 2017).

4. CONCLUSIONS

The results from the previous investigations reveal that around 40-80 wt.% of the total excavated material out of a landfill correspond to fine fractions. Each landfill site has its own potential regarding LFM or ELFM. Factors such as the age and type of the landfill, as well as its location and operation procedures might have a relevant impact on their content and valorization potential.

According to previous studies the amount of fine particles to be obtained in a LFM project mostly depends on the excavation and processing techniques, the age and type of the waste and the selected cut-off diameter to define a certain particle size as upper limit for the fines fraction.

Fine fractions can be considered as a relevant source of metals and calorific fractions, as well as a fraction suitable for inert and soil-like material recovery. This situation makes their recuperation from landfills interesting; since old landfills are not just a potential source for such elements, but this might also be a decisive factor to achieve economic feasibility in LFM and ELFM projects.

Moisture, organic content and other physico-chemical properties are interrelated parameters of critical relevance, since processing routes and possible end uses for the fine fractions will depend on their quantities to a certain extent.

Drying of the fine fractions could: i) reduce the amount of surface defilements; increasing the quality of the recyclable materials and raising the efficiency of sorting processes, especially for the sensor-based sorting technologies, such as near infrared (NIR) and color recognition (VIS), ii) enable a more efficient and precise particle size classification in the screening and sieving processes, iii) decrease the total amount of material and material flowrate to be processes and, perhaps, transported and iv) raise the calorific value.

The particle size will be a key parameter in LFM and ELFM future investigations for the separation of the fine fractions into exploitable resources and the minimization of the material to, if the case, be sent back to re-landfilling. To achieve this, the fine fractions may be classified into certain particle size ranges, selected according to the results of the material characterization and particle size distribution during the exploration phase of the project, to determine the cut-off diameter size for the fine fractions and enable more efficient material recovery, for different purposes (e.g. recycling and alternative fuel), and recuperation of soil-like and inert materials in the corresponding processing techniques (e.g. density, magnetic and eddy-current separators, among others).

Much care needs to be taken when comparing information between different investigations, since there are many factors, such as characterization conditions and procedures, laboratory analyses and followed standards, age of the waste material, defined particle size for the fines fraction, among others, that might play an important role during their execution and may differ significantly between investigations.

The implementation of different approaches for the material characterization of waste remains to be one of the crucial challenges for the elaboration of comparable and accurate compiled studies.

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WASTE MANAGEMENT IN REMOTE RURAL COMMUNITIES ACROSS THE CANADIAN NORTH: CHALLENGES AND OPPORTUNITIES

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ABSTRACT

We report results from a two-phase mixed methods research study to illustrate challenges and opportunities of waste management within the rural circumpolar and boreal regions of Canada. In the qualitative research phase, data were obtained from archives, semi-structured interviews with community partners, an information meeting with a community grassroots organization, and a participatory action meeting to develop a case study of the Labrador waste management system. Like many regions across the world's boreal belt, the study area consists of a population centre surrounded by diffuse, rural communities (many of which are inhabited by Indigenous and First Nations persons), multiple land uses, and complex governance considerations. The area faces harsh climatic conditions (e.g. frigidly cold temperatures and extended winters) that challenge biological processes and organic waste decomposition. These regions are often highly reliant upon natural resources and temporary labour forces to drive economic development, and they bear the environmental consequences of legacy wastes after project closures. In the qualitative research phase, we identify factors contributing to the accrual, management, and transport of inorganic waste across the study region to select a priority waste stream for an economic analysis in the quantitative study phase. In the quantitative phase we build an economic enterprise budget to assess costs associated with converting waste biomass from the construction of the Muskrat Falls hydroelectric dam into biochar with either a fast or slow mobile pyrolysis system. We include photos of the study region and we present an Excel-based spreadsheet tool as a supplemental file.

1. INTRODUCTION

1.1 Challenges and opportunities of waste management in circumpolar and arctic communities

This study utilizes a two-phase mixed methods research process to characterize waste management practices and socio-economic issues within circumpolar and boreal regions of Canada. We first report results from the qualitative phase of a mixed methods research project to create a case study of Happy Valley-Goose Bay, a coastal region situated in northeastern mainland Canada, within the eastern-most province of Newfoundland and Labrador (see Figure 1). In the quantitative research phase, we develop an economic enterprise budget for converting biomass waste (primarily black spruce) from construction of the Muskrat Falls hydroelectric dam into biochar. The enterprise budget and accompanying report are included as supplemental files.

We assert that parallels can be drawn between the study area and other regions of Canada's North, as well as the rest of the world with similar climate and population. These regions merit attention because they are often on the peripheries of geo-political spaces in developed nations (Vodden, Baldacchino, and Gibson, 2015). These communities disproportionately shoulder the social costs (including the subsequent waste streams) of projects that propel the economic interests of the nation as a whole. Resource extraction, hydroelectric power development, and military installations for example yield subsequent waste flows and environmental impacts (Hird, 2016). Implementation of these large natural resource projects not unexpectedly leads to large quantities of organic wastes, as well.

For example, the recent Muskrat Falls hydroelectric dam project located in the study region has generated

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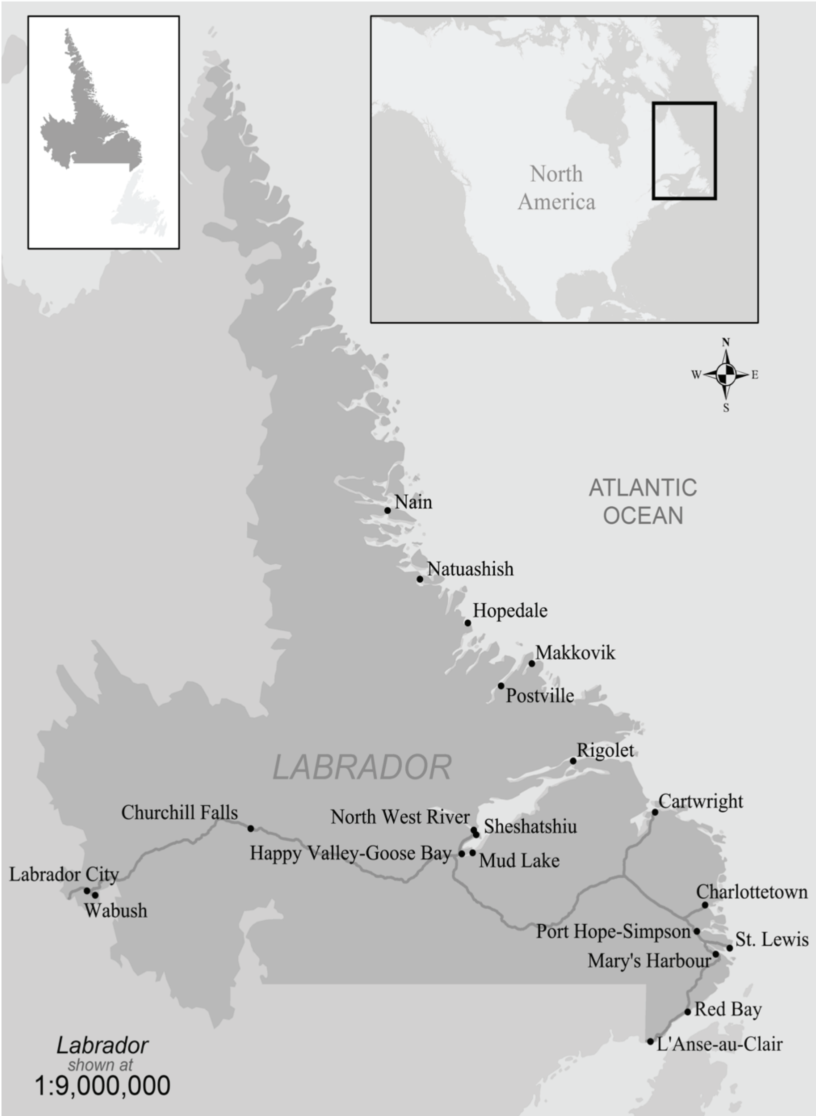

FIGURE 1: Map of Labrador. Cartography credit: Myron King and Morgon Mills.

more than 2 million m^3 of wood, equivalent to roughly 1.25% of Canada's annual timber harvest. Although the waste accumulates in Labrador, the economic benefits and energy security are often realized by other provinces such as nearby Québec, Ontario, and Nova Scotia, as well as by Canada as a whole, through the sale of hydroelectric power to the United States (Canadian Hydropower Association, 2018; Lorinc, 2016). There is also disparity between the geographically separated areas within the province. A key objective of the Muskrat Falls hydroelectric project is to improve the province's energy security and to lower electricity prices, which are among the highest in Canada. However, Labrador communities sustain the land impacts, including the costs of the mobile labour force that largely consists of Newfoundland residents who fly in and out of Labrador leaving behind household wastes.

Furthermore, the Labrador Indigenous communities, whose self-identities are formed with the land and natural attributes, disproportionately endure negative environmental and social consequences (Bharadwaj et al., 2006; Waldron, 2015). There are negative linkages between waste management practices and the quality and quantity of "country foods" that are hunted or harvested by Indigenous communities where subsistence hunting and foraging are integral to food security (Schiff and Bernard, 2018). Zagozewski et al. (2011) explicitly document that, "...solid waste disposal has been identified as a major environmental threat to First Nations Communities" (p. 9). According to the authors, one of the first well-documented cases of the presence of environmental contaminants on First Nations communities includes mercury poisoning in the Asubpeeschoseewagong Netum Anishnabek community in the 1960's as a result of the Dryden Chemical Company discharging chemical waste directly into the English-Wabigoon river system, a main water source for the Grassy Narrows First Nations Community located north of Kenora, Ontario. Similar environmental concerns have been expressed by Labrador Indigenous communities, as well.

Environmental justice, scientific concerns about methane emissions generated by filling the Muskrat Falls hydroelectric dam site, and the potential for methylmercury contamination of Labrador communities (Levin et al. 2007), and Indigenous communities in particular (Calder et al., 2016), has generated contentious activism that has been chronicled extensively in the media (Breen, 2017; CBC News, 2017). Understandably, this has made collaboration with the Nalcor, the agency overseeing dam construction, difficult on many levels, including coordination of waste management practices. In sum, clearly Labrador is encumbered with a disproportionate burden of waste in the effort to advance goals that benefit the province of Newfoundland and Labrador, and Canada as a nation.

1.2 Summary of literature and study region

As shown in Figure 1, Labrador is a large (269,134 km^2) coastal region in north-eastern mainland Canada. Its climate is predominantly subarctic in inhabited areas, with boreal forests broken by alpine and coastal barrens, giving way to tundra in the north. The largest population centre (10,227 people, or 38% of Labrador's population) is the Upper Lake Melville area around Happy Valley-Goose Bay in central Labrador (Statistics Canada, 2017), at the head of the Hamilton Inlet estuary, which drains the Churchill River and several other major watersheds. Within the Churchill River valley in particular, soils are derived mainly from glaciofluvial sands and silts, with pockets of organic soils in poorly drained areas and acidic, sandy soils elsewhere, including in Happy Valley-Goose Bay proper (Walker, 2012; Fonkwe, 2016), where soils have low organic content and little ability to immobilize nutrients (Abedin, 2015) or pollutants (Abedin, 2017a; Abedin, 2017b). However, despite its subarctic designation, the area has warm summers, no permafrost, and an average frost-free period of 104 days (St. Croix, 2002).

Demographically, the region is majority Indigenous persons, with 52% of Happy Valley-Goose Bay's population reporting an Indigenous identity (Statistics Canada, 2011). Indigenous majorities are in the three outlying communities of North West River (about 550 people) and Sheshatshiu Innu First Nation (a reserve of about 1300 people), both 35 km to the north, and Mud Lake, a hamlet of about 50 people a few kilometres east and across the Churchill River. Local municipal histories are comparatively short, with a settled population of 129 in the whole area as recently as 1901, predominantly representing mixed Inuit-European trappers, fishers, and subsistence hunter-gatherers, with some timber harvesters at Mud Lake and an additional, uncounted nomadic population of Innu (a First Nations people) living traditionally on the land.

In Labrador, like much of the Canadian North, remoteness, low population densities, limited local capacity, and lack of local control in public decision-making make waste management complex. Waste accumulation from extractive industries and landfilling practices imperils drinking water quality. In a study of the Canadian Arctic and the Canadian territory Nunavut in particular, Daley et al. (2015) note that most of North America relies on wastewater buried conveyance systems, but it's impractical for the Canadian North due to extremely low temperatures. In Nunavut, water storage is separated into two tanks, one for drinking and one for wastewater. Municipal trucks attempt to provide drinking water delivery and wastewater removal daily, which is frequently disrupted by adverse weather (Daley et al., 2015). One engaged community partner, Healthy Waters Labrador, developed a Comprehensive Environmental Management Plan (2012) promulgating six critically important waste issues; of primary concern is the lack of sewage and wastewater treatment facilities in watershed communities and the need for improved management of municipal solid waste.

As a province, Newfoundland and Labrador has one of the highest waste disposal levels per capita in the country. According to the Multi Materials Stewardship Board (MMSB), it is estimated that more than 400,000 tonnes of municipal solid waste (MSW) materials are generated each year in the province; organic waste comprises as much as 30% of all waste generated. Freezing temperature and snowfall make transport difficult much of the year, and the low temperatures slow decomposition so most organic and inorganic waste are landfilled (Zhang et al., 2013; Walker, 2012). For example, it is not uncommon for snow cover to remain for six consecutive months in the Happy Valley-Goose Bay population centre, and somewhat longer in other northern communities. Storage space becomes increasingly scarce and pedestrian/vehicle access to storage areas becomes increasingly difficult during winter months and towards the end of spring. The Gulf of St. Lawrence freezes and coastal transportation is unavailable for much of the year; the southern coastal ferry service runs about July-October, depending on ice conditions. Thus, it may be infeasible to store waste at on-site locations, or to transport materials elsewhere in a timely manner from the fall through spring months (October through April). There are also indirect climate impacts on settlement patterns, industry, and community systems. These factors exert pressure on households to dispose of waste illegally by land or sea, a behavior that has been noted in the past (Ritter, 2007), but that has also been demonstrated to take place at an increasing rate across the province after localized dump sites were closed (Harris Centre, 2017; Storey et al., 2017; Neil, 2017).

The waste management issues noted in the study region are similar to those noted across the Circumpolar North (generally defined as the ecosystem north of 60° latitude) and within the boreal forest biome 50°-60° latitude. These regions are sparsely populated, ethnically diverse with a large population of Indigenous communities, (approximately 40 different ethnic groups reside in the Circumpolar North, according to Cunsolo Willcox et al., 2015), and highly reliant upon mineral extraction and forestry to drive the economy. They also reflect a sustained history of disruption from colonialism and military installations.

Several waste management studies have been conducted in boreal climates within Europe, including Greenland (Eistead and Christensen, 2013), Siberia (Starostina et al., 2014), and other parts of northern Canada (Chouinard et al., 2014). Incineration and open dumping into unlined landfills are common practices in these rural regions. The ground is often too cold to allow organic waste to disintegrate, and toxic emissions dispel from leachate (Eisted, 2013; Samuelson, 2013). Studies have found ammonia nitrogen, polycyclic aromatic hydrocarbons, and total petroleum hydrocarbons, putting human and environmental health at risk (Samuelson, 2013). In addition to potential health impacts stemming from contaminated waste water and food sources, Czepiel et al. (2003) note that sanitary landfills are the leading anthropogenic source of methane emissions, which constitute roughly 25 times as much global warming potential as carbon dioxide.

We believe that parallels can be drawn between our study area and other regions of the world, like Tibet, that are experiencing waste management issues associated with infrastructure development projects and tourism. At an average altitude of over 4,000 metres, the Tibetan Plateau also has a sensitive, fragile ecosystem (Jiang et al., 2009). With over 41,000 glaciers and vastly expanding permafrost, Tibet is a major source of drinking water for over half of the world's population, but since the region has limited waste management facilities, waste burial or burning waste are the most common (Dong, Tan, & Gersberg, 2010). The Qinghai-Tibet railway, a state project, has led to an influx of tourism. A "floating population" is projected to increase municipal solid waste by approximately one-third, to 4942 tonnes/day in 2020 from 3597 tonnes/day in 2006 (Jiang et al., 2009). Studies by Ding and Wang (2018) note project that the proportion of municipal solid waste produced by tourists in Tibet increased from 2.99% to 20.06% over two years, and it is estimated that it will increase to 33.49% in 2025. Large cities have been equipped with sanitary landfills, however, surrounding Tibetan communities have dealt with the influx of solid waste by dumping into garbage piles, or rivers and streams (Jiang, 2009; Dong, Tan, & Gersberg, 2010). This rise of municipal waste by means of an industrial public project and a transient population, like the Qinghai-Tibet railway, draws parallels to the struggles Labrador finds with waste management due to industrial military projects, mobile populations, and resource extraction projects.

In summary, based upon a review of the literature conducted in similar climates, we assert that we are the first to engage in in a mixed methods research approach to study waste in Labrador and in climatically similar regions. We believe that a mixed methods research methodology facilitates local community empowerment in that it has yielded a focused economic budgeting study of a waste stream that is an environmental priority, and that may also present potential economic development opportunities. Results from our study could inform waste management and community development projects in similar regions like Tibet and elsewhere in the boreal and artic regions of the North.

2. METHODS

2.1 Mixed methods research approach

Waste management is one of many inter-related public systems for community well-being, and it is therefore best understood from a holistic perspective that takes a wide view of agency, including not only diverse municipal and high-level governmental structures, but also community values and practices. This approach is adapted from well-established literatures on community health and well-being (e.g. Srinivasan, O'Fallon, and Dearry, 2003; Parlee and Furgal, 2012), and ecohealth (Charron, 2011). It is especially relevant in the context of Northern and Indigenous communities where governance systems and cultural practices may not conform with external structural paradigms. Similar recognition in climate change studies has led to a broader appreciation of the importance of taking into account the inter-relatedness of community systems, especially as they intersect with issues of community resilience (Ruscio et al., 2015), connectivity to the land (Cunsolo Willox et al., 2013), and the impacts of extractive resource development (Parlee, 2015; Southcott, 2015).The present research project has therefore employed a community-based perspective in all phases, beginning with a foundational partnership with the local municipality of Happy Valley-Goose Bay, who originated the underlying research program. This study is situated as part of a larger multi-institutional, multi-disciplinary project on facilitating sustainable communities and sustainable resources in the Arctic. Specifically, the overall objective of the waste management project is to investigate how resource development may provide communities with economic and strategic opportunities to overcome existing challenges, while simultaneously acknowledging the social and economic costs associated with resource development-related waste production. Like others before us have shown, we assert that these are inextricably linked, in that costs associated with waste management may be recovered and transformed into opportunities. We posit that this premise holds true across the Canadian North and in other similar regions in the world with similar climates and social structures. Identification of the waste characteristics is a first step in actualizing these benefits, and it lays the groundwork for a cost-benefit analysis of priority waste streams.

Mixed methods research is a broad process for a line of scientific inquiry that integrates and synthesizes both qualitative and quantitative processes, either simultaneously or sequentially (Newman et al. 2003). The methodology we employed was consistent with an integrative two-stage qualitative-quantitative research typology described in Tashakkori and Teddlie (2010). There is a vast literature on qualitative research techniques. Creswell (2003) notes that 19 complete qualitative procedures have been outlined in the sociological literature alone that form a continuum of qualitative research strategies. This continuum ranges from unstructured ethnographic data collection techniques where the researcher is a passive observer who listens to the language of the natives (Spradley 1979), to highly structured interview or case study methods where the interviewer controls the delivery of the questions with almost rigid precision (Yin 2003).

One important outcome is the creation of a quantitative model through theoretical data triangulation (interpreting data from more than one theoretical or disciplinary perspective) and the separation of qualitative data. In his seminal publication, The Research Act in Sociology, Denzin (1970) advocates for the use of triangulation, which is defined as the integration of multiple methods to study a research problem. Lewis-Beck, Bryman and Futing Liao

(2004) also emphasise that using multiple research methods presents a good validity check for research methodology and findings. There is some debate over the nomenclature of "mixed methods," but most authors have reached consensus that mixed methods research involves a degree of synthesis.

The downside of mixed methods research is that the division between quantitative and qualitative research may become unclear as the approaches become increasingly integrated. However, Tashakkori and Teddlie, editors of pivotal mixed methods research texts (2010), summarize a prevailing thought in the field: combining both qualitative and quantitative approaches in social sciences seems to many researchers to be a natural or intuitive process (2010). To frame our contribution in the context of Denzin's work, some engineering and waste management studies involve a narrower version of methodological triangulation by using focus groups to develop a survey for quantitative data collection. In contrast, we integrate more than one qualitative research approach to collect and interpret data for a complex system. We utilize multiple qualitative research approaches due to the limited number of individuals dispersed across a large region who have knowledge about the region's waste management processes.

The specific objective of the qualitative research phase was to identify the sources and types of waste streams in Labrador and to contextualize these findings within the waste management literature. Based upon the waste streams identified in the qualitative research phase, we selected one priority waste stream to apply quantitative methods to develop an enterprise budget of the costs of converting waste biomass from the construction of the Muskrat Falls hydroelectric dam into biochar. Results from the quantitative phase an Excel spreadsheet budget tool and supporting project report are available as a supplemental supporting document.

2.2 Qualitative data collection

In the qualitative research phase, from March 2016 through June 2017 we obtained historical public records archived at the Town of Happy Valley-Goose Bay and within the Labrador Institute of Memorial University of Newfoundland to determine waste generation sources in Labrador, with the goal of selecting a priority waste material for additional quantitative analysis as well as policy recommendations for additional analysis. Data were collected and compiled from archival literature reviews and consultation with community partners. We also conducted semi-structured, participatory interviews and discussions with 21 community partners, where at least one investigator asked a similar series of open ended questions, and data were transcribed in real time and clustered by themes (Keske et al., 2011; Creswell, 2003). Partners that have been affiliated with the project, all of whom directly work with community or municipal waste management processes or northern environmental conservation, are listed in the Acknowledgements. Other partners, some of whom are involved with waste management through contracting or other research projects were individually approached at the May 2017 Labrador Research Forum. In addition, we conducted a qualitative, participatory action meeting in May 2017 at the Labrador Research Forum, where approximately 35 persons were invited to share their perspectives about challenges to northern waste management systems as part of an ideas generating session. This session was promoted as an opportunity to provide a context for dialogue about research addressing solid waste management issues, especially in rural and remote areas, by bringing together researchers, funders, and potential partners to brainstorm about already identified needs and research gaps, and to explore new possibilities specific to Labrador. Data from the Labrador Research Forum were recorded by two note takers, and interview data were otherwise recorded by a single note taker. These data were categorized by theme and frequency by which they were raised. Following the Labrador Research Forum, we conducted an information meeting with the Happy Valley-Goose Bay Recyclers, a community grassroots organization.

2.3 Quantitative data collection

After potential waste streams were identified in the qualitative research phase a priority waste stream, black spruce and fir biomass from the construction of the Muskrat Falls hydroelectric dam, was selected for additional economic study. Consultation with the community partners revealed interest in recent research results on the potential for biochar as a soil amendment in Labrador for agriculture and soil remediation (Abedin, 2015; Abedin, 2017a; Abdein, 2017b). Biochar is essentially charcoal made from organic matter, and the efficacy of biochar from various biomass inputs has received increasing attention in the scientific literature during the past decade. Quantitative data for the economic analysis were obtained from a literature review of biomass and biochar literatures, including biochar production in northern climates with an abundance of biomass, and slower degradation processes due to lower temperatures. Eighty different sources were used to construct the budget tool.

The enterprise budget consists of fixed and variable costs for building and operating a prospective biochar production project co-located at the Muskrat Falls hydro-electric mega-project under construction by Nalcor Energy. We provide the Excel-based tool as a supplemental document. Spreadsheet users may select from a range of parameters like hours of production per day, technical efficiency, and pre-processing equipment, depending upon desired production input and output variables. Hence, the enterprise budget could be adapted according to different locations in Labrador or rural, northern Canada with a similar ecosystem.

The range of production inputs, cost structures, and parameters were informed, in part, by the local community partners who participated in the qualitative data phase, and who had knowledge of the volume and locations of waste biomass piles, and waste patterns of the study region. The budget was developed to summarize the costs relevant to instituting a new biomass to biochar production facility in Labrador, with the goal of attracting either potential investors or government partners interested in pursuing commercial biochar production or co-locating a multi-purpose biomass waste management facility with the hydroelectric dam. Future investors or government agencies can impute their customized values into the spreadsheet to reflect proprietary cost information. Revenue streams were not included in the budget tool, although this would be a logical extension for the next phase of research.

3. RESULTS

3.1 Labrador waste flows and waste management

Following the qualitative data collection process, we developed an overview and timeline of waste management practices, summarized in Figure 2. The timeline captures the relationship and co-existence between military legacy waste, colonialism, and resource extraction that is similar to many other communities across the Canadian Arctic, Alaska (U.S.), Greenland, and Iceland. Directly following World War II, sixty-three temporary military settlements, including Happy Valley-Goose Bay and other Labrador communities, were erected to form the Distant Early Warning (DEW) radar line to defend against potential Soviet attacks in northern Canada. The construction and operation of a military base at Goose Bay led to rapid population growth and a shift away from Indigenous and pre-industrial waste management practices towards a series of disjointed waste management strategies. Until 1990 household wastes were discarded in nearby landfill locations, compacted and bulldozed. These practices and other regulatory shortfalls have left a legacy of contamination, particularly by petrochemicals, polycyclic aromatic hydrocarbons (PAHs), polychlorinated biphenyls (PCBs), pesticides, and heavy metals, which has prompted a range of site assessments, monitoring studies, and remediation projects since the early 1990s (summarized in Fonkwe, 2016). The most substantial, long-lasting environmental impacts of military development have been at Hopedale, Saglek, and Happy Valley-Goose Bay. Military operations have clearly influenced the waste management and socio-economic community changes within the study region, and in other northern communities across the world. Additional discussion about the chronology and evolution of the Happy Valley-Goose Bay landfill is presented in Keske et al. (2018). Hird (2016) also provides detailed discussion about legacy military wastes left behind from the DEW Line.

Wastes that are an extension of colonialism and military disturbances have created an endogenic relationship with the present-day community infrastructure. For example, community infrastructures and lifestyles were shaped by natural resource projects, like the Churchill Falls hydroelectric station and the Muskrat Falls hydroelectric dam that is currently under construction (Mills and Keske, 2018). However, these community infrastructures are also at odds with the natural waste decomposition processes of the arctic and boreal environments; additional resource development projects perpetuate the cycle of unsustainable waste management practices. As demonstrated in Figure 2, population increases correspond with military installations and with resource development. These population influxes shape the culture of the community and leave behind legacy wastes. To illustrate this, we now briefly discuss the present-day waste management practices within two communities Happy Valley-Goose Bay and Nain.

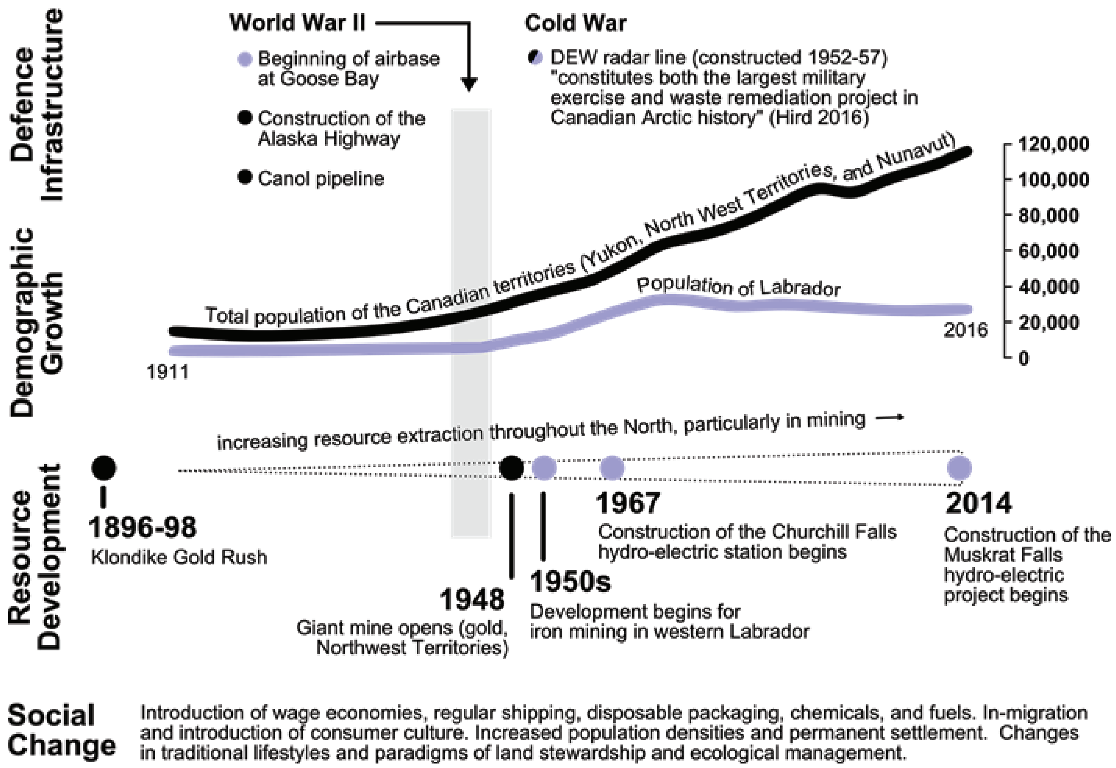

FIGURE 2: Defence infrastructure, demographic growth, resource development and social change as colonial impacts on waste systems in Labrador and across the North.

3.1.1 Happy Valley-Goose Bay

Figure 3 illustrates the different sources of waste generated in the Happy Valley-Goose Bay study region; some waste is reused or reclaimed by grass roots community organizations. Municipal waste is collected from dispersed locations and processed in a number of different ways, including incineration and composting, but waste is chiefly disposed at the municipal landfill. Photo 1 shows an aerial photograph of the Happy Valley-Goose Bay landfill.

As shown in Figure 3, waste is also collected by a provincially-contracted recycling depot for prescribed materials; by independent, commercial waste management contractors; by the health authority (hospital-generated medical wastes); through a municipal drop-off location for organic yard wastes; by commercial scrapyards; by a new municipally-run compost initiative; and by grassroots resident-led strategies, such as composting, burning (including as heating fuel), reuse, unauthorized dumping and littering, and a volunteer-led recycling initiative, the Happy Valley-Goose Bay Recyclers group, which pays for the trucking of household recycling to a plant in Newfoundland.

Not surprisingly, after years of sustained outside contact, food consumption patterns have also changed correspondingly. Several at-risk Indigenous communities have a higher incidence of consuming pre-packaged foods (Schiff and Bernard, 2018), a practice that is associated with lower income households. The municipality and waste collection service providers and community partners have noted a higher incidence of food packaging waste in lower income communities compared to the rest of the community. This increases the amount of bulk food packaging waste sent to the landfill. This places communities in a difficult position where they collect disproportionately more waste from some areas, but generally opt not to impose extra fees out of concern that residents will dispose of waste illegally

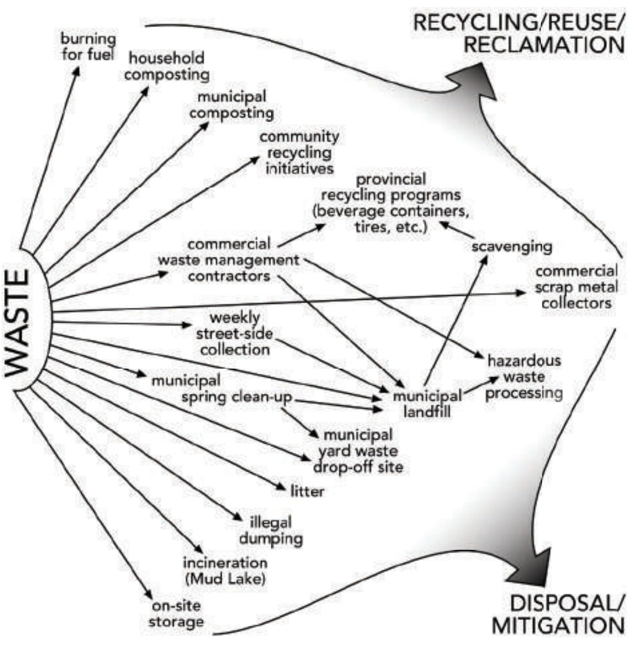

FIGURE 3: A flowchart of selected household waste management flows in Happy Valley-Goose Bay.

PHOTO 1: Aerial photograph of Happy Valley-Goose Bay municipal landfill. Photo credit, Jason Dicker. The Happy Valley-Goose Bay landfill is surrounded by wooded biomass. The Goose River flows nearby, which serves as a source to Mud Lake.

(Harris Centre, 2017). The Town of Happy Valley-Goose Bay notes that it is not uncommon for waste to be dumped at the landfill entrance outside of business hours when the gate is closed.

A number of factors also contribute to high levels of landfill harvesting (colloquially known as “scavenging”). Despite being illegal, landfill harvesting practices are common at the Happy Valley-Goose Bay municipal landfill, as well as in small communities across Labrador’s northeastern coast. In 2016, landfill harvesting resulted in the death of a minor child (CBC News, 2016). Reclamation of commercial wastes like lumber or construction supplies is also common, but this unfolds differently, because it is more predictable. For example, once a week a company may make materials available for public collection, which assists interested parties in a safe and planned transfer of reused materials.

Apart from the municipal landfill, solid waste in Happy Valley-Goose Bay is processed in various ways. Since 1996 waste diversion and recycling programs in Newfoundland and Labrador have been funded and managed through a British Crown agency, the Multi-Materials Stewardship Board (MMSB). The MMSB is funded by levies on beverage containers and tires, as well as the sale of recyclable materials collected under its programs (MMSB, 2017).

The most significant quantity of household waste is not collected weekly. It consists of items too voluminous to be bagged or boxed — such as demolition or renovation debris, yard waste, old appliances and furniture, and other bulky household items. Water-using appliances such as water heaters, washing machines, and dishwashers are particularly frequently discarded, given the medium-term corrosiveness of the municipally-supplied water (Fonkwe, 2016; Fonkwe and Schiff, 2016; CBC News, 2008). Our site visits of the municipal landfill confirm a large number of water heaters among a number of used appliances, though the rate at which these appliances are discarded compared to the rest of Canada has not been assessed, though a comparison may be of merit. An example of the high volume of appliance waste is shown in Photo 2.

As shown in Photos 3 and 4., as part of the annual "Spring Cleaning" waste sorting and collection event, furniture, appliances, construction debris, wood for burning. Spring cleaning is presented at the side of the curb, and community members are encouraged to collect and transport items that they wish to reuse.

Mobile labour forces also contribute to landfill and household wastes. Table 1 summarizes differences in the wastes generated by different communities (SNC Lavalin, 2016, p. 7), with a disproportionate amount of waste in the Happy Valley-Goose Bay area arising from the Muskrat Falls hydroelectric development project and the temporary labour force. In order to house workers for the development and operation of these large projects, some resource developments have established temporary camps (Muskrat Falls), permanent settlements (Churchill Falls), or created entirely new public municipalities (Labrador City). The autonomous governance and the scale of these large projects, coupled with an understandable need for privacy places a dilemma on municipal waste management. There is a need to accommodate the additional waste stemming from the resource related development documented in Table 1. If the Muskrat Falls project could collaborate with the Town of Happy Valley-Goose Bay, there would be synergies between recycling and scalable waste management practices, along with landfill disposal fees that were commensurate with landfill access. However, difficulties with doing so prompted the Muskrat Falls project to take on their own waste management practices. In spring 2017, Muskrat Falls begun incinerating its own waste in response to an increase to landfill tipping fees incineration (Barker, 2017).

Aside from the Muskrat Falls construction and mobile labour household waste, waste biomass exists from forest clearing associated with the Muskrat Falls reservoir and transmission line clearcutting. The total volume of merchantable timber produced from has been estimated at 2,172,300 m3 (Nalcor, 2009, p. 10), but to date this substantial material resource has been underused. Some wood has been made freely available for domestic use by residents of Happy Valley-Goose Bay, North West River, and Sheshatshiu (Nalcor, 2015), but as frequently reported in the media (e.g., Canadian Press, 2014), the cost of transport has been a major barrier for initiatives to commercialize the resource. Processing the material on site might present a cost-effective approach to managing the waste.

The homogeneity and availability of the waste biomass prompted a subset of community partners to prioritize biomass waste streams for further economic analysis. More-

PHOTO 2: The accumulation of appliance wastes in the Happy Valley-Goose Bay landfill.

PHOTOS 3-4: Photographs of Happy Valley-Goose Bay Spring Clean-up taken Tuesday, June 6, 2017. Photo credit, Morgon Mills. As shown in the photographs, there can be a large quantity and an assortment of household items. Many households sort the wastes to make collection and trading easier for others.

TABLE 1: Estimated waste production (t/year) in Labrador communities near Happy Valley-Goose Bay.

	Population	Dwellings	Estimated Waste (t/yr)
HVGB	7552	2843	6919
North West River	553	225	262
Sheshatshiu	1314	291	748
Mud Lake	539	23	912
TOTAL	**9473**	**3382**	**7955**

over, a series of studies indicated that converting biomass from the waste wood to biochar was technically feasible (Abedin, 2015), and that biochar application in the study area shows promise for mine tailing remediation (Abedin, 2017b) and reduction in toxic leachate and greenhouse gases from the municipal solid waste stream (Abedin, 2017a).

3.1.2 Labrador's remote, self-contained communities

Since waste management also presents challenges specific to rural northern communities that are accessible only by air and sea, it is worthwhile to extend our discussion to the community of Nain, Labrador. Nain also exemplifies other northern Indigenous communities that have experienced periods of disruptive, ongoing colonialism and the Voisey's Bay nickel deposit is a source of both economic development and waste.

With a population of 1,125 (Statistics Canada 2017), Nain is the largest of the five communities in the Labrador Inuit Land Claims Area, under the regional jurisdiction of the Nunatsiavut Government. It was also the site in 1995 of a case study in Inuit community waste management, conducted in partnership by the Inuit Tapirisat of Canada and the Labrador Inuit Association (Harris et al., 1995). As noted in the case study, waste management is a community responsibility, shared by local residents, and requiring leadership from local government (i.e. regionally today, the Nunatsiavut Government, and municipally the Nain Inuit Community Government or NICG).

The NICG operates a dump site and regular weekly residential and commercial garbage collection. The dump is sorted by waste stream (e.g. household wastes, automotive, appliances, oil drums, scrap metal). Since winter weather, permafrost, and the scarcity of topsoil make landfill burial impractical, combustible waste streams are processed by incineration. Recent photograph of the dump are presented in Photo 5 and Photo 6.

Even with the summer solstice imminent, snow remains along the narrow pathway to the open landfill. Water and waste accumulate into several sludge pools. Other sections of the landfill serve as a place where auto parts and appliances may be recovered. At the time these photos were taken, one of the researchers observed employees conducting an open burning at the landfill that had a noticeable effect upon air quality in the hamlet of Nain, where smoke tends to linger into town from over the hill. Although the skies are blue at the landfill, the air quality in town was smoky, filled with odor, with unclear visibility.

Photo 5 and Photo 6 illustrate the realities of landfill harvesting during summer in Nain. The red truck provides a frame of reference. In May 31, 2017 at the beginning of summer there is substantially more waste than August 23, 2017, when volumes of waste have been collected after three months of summer, and before snowfall begins.

Given the remoteness and ecological sensitivity of the area, as well as the reliance of the Inuit population on wildlife and forage for food security, environmental concerns associated with the dump site have special significance. One challenge is its location near the ocean without lining or leachate remediation measures, potentially leading to impacts on aquatic ecosystems and human health. As in many rural landfills, terrestrial wildlife are also potentially affected, with wildlife scavenging resulting in potential contamination of the food web and increased risk of adverse human-wildlife contact. This is exacerbated by the lack of organic waste diversion, since composting for agriculture is not a major strategy due to the cold climate, permafrost,

PHOTOS 5-6: Photo 5 shows waste in the Nain open landfill on May 31, 2017. Photo 6 presents waste in the Nain open landfill on August 23, 2017. Note the same red truck appears in both photos, but auto parts and other wastes have been scavenged.

and poor soils. Ammunition is a particular concern, especially given its prevalence in a community reliant on hunting. Its inclusion in household waste destined for incineration can potentially endanger workers, landfill users, and equipment. Harsh winters with heavy snowfall also lead to widespread litter, since snow and ice often block garbage bins and impede waste transportation, causing inefficient roadside pickup and scattering of household waste by stray dogs and crows.

3.2 Quantitative research results

As previously discussed, consultation with a subset of community partners led to the prioritization of waste biomass for the economic study. Several options were considered that could accomplish dual goals of waste collection and co-production of a desirable output or commercial product. For example, conversion of municipal household waste into biogas either at the household level or as a large community greenhouse (for food production) were considered. Other waste sources identified during the qualitative research phase were also considered, but not prioritized because the dispersed collection points made cost control challenging. The homogeneity of the woody biomass makes this waste management project tractable, in part because dedicated capital equipment can be purchased for the specific waste, and the biomass is consistently available. Moreover, as Abedin (2017b) demonstrates, biochar presents a potential to reduce toxic leachates emanating from extraction projects.

Due to the number of log piles (pits) left after forest clearance, the enterprise budget focused on the use of a mobile pyrolysis unit to reduce capital costs and to take advantage of the waste accumulations in different areas. Photos of the waste biomass are shown in Photos 7 and 8. Most piles have been present since being clear cut in during 2012-2013. The largest of these pits is located right next to the Muskrat Falls site. For the operation described in this report, biochar production would take place at the pit nearest Muskrat Falls, while Happy Valley-Goose Bay would be the business headquarters.

The enterprise budget provides options for the user to choose between using slow and fast pyrolysis units. Slow pyrolysis serves as the default because it usually produces more biochar than fast pyrolysis (Pratt & Moran, 2010; Kung, McCarl, & Cao, 2013; Ahmed, Zhou, Ngo, & Guo, 2016; Ronsse, Van Hecke, Dickinson, & Prins, 2013), and the proposed Labrador project is primarily focused on producing biochar as an output. Slow pyrolysis also has lower pre-treatment costs (Ahmed, Zhou, Ngo, & Guo, 2016; Wrobel-Tobiszewska, Boersma, Sargison, Adams, & Jarick, 2015), which is favourable for a small-scale operation, like the proposed mobile pyrolysis units.

Fixed and variable costs (such as permitting fees and labour wage rates, respectively) have been customized to be consistent with the study region, although the tool allows the user to modify values that may be more reflective of elsewhere. For example, the spreadsheet user can choose whether production will be an outdoor seasonal operation, an indoor operation during winter months with an outdoor operation during summer months, or a year-round outdoor operation. While the year-round outdoor operation might not be very realistic, it is still the most desirable operating condition and it is therefore set as a default.

The Enterprise Budget Excel file is broken into four Excel sheets, and specific instructions, along with references, are provided in an accompanying report. The first Excel sheet contains the fixed costs for biochar production, which do not change with the amount of biochar produced. This includes machinery costs, miscellaneous costs, overhead charges, and total fixed costs. Total fixed costs are equal to the sum of repair and maintenance, insurance, depreciation, interest, and tax.

The second sheet contains all variable costs for biochar production, fixed costs from the first sheet, and total fixed and variable costs. The variable costs include fuel, oil and lubricants, labour, and other miscellaneous costs. The value of the variable cost per tonne of biochar produced is calculated in the same way as the fixed costs are calculated. The annual variable cost is divided by the actual days worked per year multiplied by technical efficiency, with the sum divided by the tonnes of biochar produced in a day.

The third sheet contains all the parameters for the model that can be changed depending on the operation's expected efficiency, such as hours per day for machinery use, pre-processing, and bagging. The fourth sheet contains data tables and enterprise budget calculations. These ta-

PHOTOS 7-8: Wood piles cleared for Muskrat Falls transmission lines. Photo credit: Morgon Mills.

bles include data on business permits and fees, administrative costs, labour rates, days worked and paid, equipment depreciation, maintenance costs, fuel consumption, lubricant consumption, and miscellaneous other data.

The largest constraint to using a mobile pyrolysis unit is the limited amount of biochar that can be produced. Additional equipment is still needed for preprocessing and pyrolysis, even though the scale of the operation is very small, and many of these machines will not be used to their full capacity. Examples of the additional equipment include a horizontal grinder, rotary screener, and a biochar bagger. Total costs would most likely not rise at the same level as the operation's scale, meaning that units capable of producing more biochar could lower the per-tonne cost of production.

An obvious drawback of this project would be the difficulty in finding a market for the biochar, and potentially high shipping costs to a distribution centre. Since Happy Valley-Goose Bay is a remote community in northern Canada additional shipping costs would likely need to be added to the budget. The community is optimistic that some of the biochar may be retained locally for agricultural or remediation purposes. Abedin (2015) has already demonstrated that biochar application can significantly improve soil fertility and increase crop production in Happy Valley-Goose Bay soils, and that there is potential for mine site remediation (Abedin, 2017b). However, production and storage of the biomass provides an option value for using the biomass, which has already been clear cut and is otherwise slowly beginning to decay.

4. DISCUSSION AND CONCLUSIONS

4.1 Changing the narrative: Engaging local communities for collaboration and co-investment

Though it's been said before, engaging community partners is critically important to creating synergies that will take root in a community. In our case, the community partners from the Town of Happy Valley-Goose Bay facilitated the qualitative data collection process, and their feedback led us to pursue the development of an enterprise budget tool to evaluate costs of establishing and operating a biochar processing plant in the study region. Furthermore, the presence of one our authors in his hometown of Nain serves to ground truth many of the observations in each of the respective study areas, and it provides better insight into the different areas of Labrador in relation to one another.

4.1.1 Local education and community waste recovery initiatives

Grass roots organizations provide an important role in educating the community about waste reduction and recovery. There must be incentives for households and commercial operations to adhere to waste management practices in order to achieve societal benefits, though the impacts that each household exerts onto the community or on the environment may not be immediately recognizable. Like other public utilities including electricity generation (Fox-Penner, 2010), there are paradoxical tensions between ensuring that there is enough waste volume to facilitate municipal waste collection as a public service and providing incentives for households to change their behavior to achieve societal goals. This creates perverse incentives for unsustainable waste management practices. Specifically, households may not see the value of reducing their waste volumes, in part because they are being encouraged to routinely supply waste as part of the weekly household waste removal cycle. Yet, policy makers have expressed that they are uncomfortable with setting a limit on the number of bags allocated to each household, out of valid concern that it will increase the incidence illegal dumping. Thus, households receive positive reinforcement for sending unsorted waste streams for municipal collection without a clear sense of how their waste affects environmental quality and the region's waste management practices in general.

Community collaboration and grass roots efforts play an important role in disseminating information and shaping behavior. Waste management practices must be socially acceptable for widespread community adoption (Berhe et al., 2017). Since the beginning of the project, the grass roots Happy Valley-Goose Bay Recyclers group has continued to build momentum and to generate support for sustainable waste collection and landfill diversion of both household and light industrial wastes. As of May 2018, the group's Facebook page had 510 members, and they report 30 dedicated households whose donations and dropped-off recyclables have established the feasibility of their model provided that residents who use the service continue to remain willing to fund it. This group is hosted without cost in a large storage shed owned by a local small business, and relies upon volunteer labour to accept, sort, and palletize recyclable materials. Benefits identified by the group are above and beyond the usual advantages of waste diversion, and they include the education of children and the furtherance of a culture of environmentalism. Organizers have pursued funding opportunities with the municipality, provincial government, and local businesses, and an account has also been set up whereby residents locally dropping off beverage containers for the provincial MMSB recycling program can have the refunds donated to the Happy Valley-Goose Bay Recyclers group. They state that the chief limitation for the program is one of scale, as there is limited potential for expansion without significant infrastructure acquisitions, although the door is open for future collaboration.

4.1.2 Community co-investment and management in waste management projects

If the proposed biomass to biochar pyrolysis project is to truly take hold in the community, it's critical for Nalcor, the Indigenous communities, and the Town of Happy Valley-Goose Bay to engage collaboratively and to have joint ownership in the financial and operational aspects of the proposed centre. An obvious next step is that biochar production and any waste management project co-located with the Muskrat Falls hydroelectric dam provide communities with co-ownership and co-investment opportunities. These partnerships are critical in order to develop appropriate incentives, including a distribution of benefits and management over time and space (in other words, in a sustainable manner for all). Ideally, this would foster increased community participation in decision-making and governance of the waste management process. As efforts gain momentum and as communities grow, this may even extend to Indigenous owned waste management services. Citizen action boards must also be comprised of representation from Indigenous governance, in order to ensure decisions are made with the best interests of these communities in mind.

In fact, it is our hope that the enterprise budget will facilitate more conversation between the groups. Since we've identified biomass to biochar processing costs, there is more time to pursue constructive discourse, project financing, and locating buyers for the biochar. The budget tool is Excel-based, and there is flexibility for its use in other northern regions across the world where biomass is being considered for biochar production.

4.1.3 Frozen in time? Community response to military waste across the North

It is also important to recognize that military and legacy wastes from sustained periods of colonial contact have left enduring impacts on Indigenous and marginalized communities across the world. As warming of the Arctic continues, and legacy waste sites thaw, additional environmental impacts from previous disturbances will unfold. It will be important for those responsible for the waste to step forward and lead remediation efforts. We urge scientists and policy makers to focus more resources on raising global awareness of buried military waste across the North, and the importance of collaborating with Indigenous and settler communities on the socio-economic barriers that contribute to unsustainable waste management practices. Community engagement will be critical to effectuating safety and clean-up at all scales from local to international.

4.1.4 Turning trash into treasure with improved waste matching and sorting

More sophisticated waste matching and sorting practices are needed both within the landfill, and to divert waste away from the landfill. Either the Town of Happy Valley-Goose Bay or Indigenous Governments could create a social media trading post to match those who are in need of cardboard boxes of specific dimensions with individuals who are looking to discard these materials. The communities could allocate storage areas specifically for holding cardboard containers and building supplies, including auto parts or construction materials, so that these may be acquired in a relatively safe manner.

Sorting and organizing the landfill spaces to allow for specific waste streams (e.g. auto parts or furniture) would guide harvesters to more specific locations and times that would involve less contact with heavy equipment. Additional labour force training to facilitate these sorting practices could include community outreach programs that would facilitate improved matching and sorting processes. It may also be possible for the provincial or federal governments to provide financial incentives to divert some wastes (e.g. plastic bottles or appliances) away from communities but return revenues from bottle collection back to the communities of origination.

4.1.5 Policies that balance community health while recognizing the practicalities of waste management in the North

We believe that the precautionary principle, essentially taking preventive action in the face of uncertainty, is particularly important for informal waste recovery practices. Landfill and curbside harvesting are realities in the rural Canadian North, although it is difficult to ascertain the scale and scope of the waste streams recovered and the household prevalence of participation. Many of the activities are illegal and are likely underreported. However, acknowledging that these activities are common practice is a first step towards the goal of facilitating a safe environment for waste recovery and reuse, while planning a vision of other sustainable, safe waste management programs.

As previously discussed, many households already store wastes for an annual spring cleanup and engage in curbside waste harvesting. Since the public is already accustomed to these practices, there is opportunity to expand these programs during different times of the year. For example, a second fall clean-up event, combined with a dedicated storage unit for sorted waste materials, could provide households the option to access the materials and to repurpose these at other desirable times (e.g. preparation for the December holiday season, and February Carnival celebrations).

From a practical perspective, the province might consider creating legislation that would essentially indemnify the municipalities for facilitating waste trading and create more waste bartering/trading opportunities through the waste collection, sorting, transportation, and sorting processes. This may be a good place to start, though the implications of implementing such a policy would obviously require its own research study. As shown by Keske and Loomis (2008), there is some evidence to suggest that these policies work for natural resources on multi-purpose lands in rural communities.

4.2 Conclusions

In conclusion, waste management in Canada's circumpolar and Arctic North is complex. Moving forward, a multitude of socio-economic and climatic considerations must be considered in order to develop strategies, policies, regulations commensurate with 21st century lifestyles, and to accommodate uncertainty associated with global climate change. However, optimistically speaking, attention to the development of a sustainable waste management program provides opportunity to foster community cohesion and drive innovation, like converting waste biomass into a biochar operation.

We have been fortunate to partner with the municipality of Happy Valley-Goose Bay, who has been actively pursuing research and innovation in waste management and environmental stewardship for many years. Their vision prompted us to engage in a holistic approach to conducting a waste management study of Labrador, and to position waste management in the Arctic as a resource issue that affects all of Canada and the worldwide circumpolar North. We assert that this holistic approach is a strength of the paper. Presenting results from the qualitative phase is a unique contribution to the waste management literature for circumpolar and boreal regions of the world. Moreover, using the qualitative research results to prioritize an economic study should reinforce the importance of engaging communities in waste management decisions. We hope that others in northern communities across the globe will benefit from the first steps that this community has taken to evaluate the feasibility of a biomass to biochar project and to effectuate sustainable waste management.

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LANDFILL SITE SELECTION FOR MUNICIPAL SOLID WASTE BY USING AHP METHOD IN GIS ENVIRONMENT: WASTE MANAGEMENT DECISION-SUPPORT IN SICILY (ITALY)

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ABSTRACT
The goal of this work was to test a methodology, based on multi-criteria analysis and geographic information systems, aimed at identifying areas potentially suitable to host landfills for Municipal Solid Waste (MSW). Although the above-mentioned methodology was applied to three different areas (Western, South-western and Eastern) of Sicily, in this paper, we present the results of the western sector. The first step consisted of the division of the study area in excluded and potentially suitable sites, on the basis of the Italian current legislation. The suitable sites were subsequently re-evaluated based on additional criteria in order to choose the most suitable ones. This second step consisted of a multi-criteria analysis based on a scores and weights system. The Analytic Hierarchy Process (AHP) was applied to estimate the relative importance weights of the evaluation criteria. The suitability for landfill siting was finally evaluated with the aid of a simple additive weighting method. The resulting land suitability was reported on a scale of 0 to 10, respectively, from the least suitable to the most suitable sites. In order to reveal the most suitable sites, to provide a ranking and, consequently, a quick selection, a spatial clustering process was carried out. In relation to the data obtained, several suitable areas to host sites for MSW landfill in Western Sicily were identified. The application of multi-criteria analysis, together with the use of geographic information systems, provided a powerful tool for the identification of the most suitable site among those identified.

1. INTRODUCTION

Waste is an important cause of environmental load and a marker of dissipating resources. Proper waste disposal, without compromising natural reserves and environmental quality, has become an absolute necessity to avoid environmental and public health risks (Pires et al., 2010) and is one of the greatest tasks of our times. Landfill is the most popular method of waste disposal currently being used; however, it is not the only method. In fact, an integrated waste management system comprising recycling of materials and, to a lesser extent, incineration or combustion, should be the preferred method (Sakai et al., 1996; Seadon, 2006). Today, many areas and administration departments are re-evaluating the use of landfills because of the lack of available space and the presence of methane and other landfill gases, which lead to numerous contamination problems.

In Sicily, Italy, waste management involves a series of problems, such as difficulties in monitoring and verifying the efficiency of landfills, low recycling levels (less than 10% in 2009), reduced waste capacity of authorised landfills and delays in the realization of the pre-treatment plants to reduce waste volume.

Therefore, this paper aims to test a methodology based on the application of Analytic Hierarchy Process (AHP) combined with Geographic Information System (GIS) in order to obtain a map of areas suitable for landfill establishment. This methodology has been applied to an area within the Western sector of Sicily in order to identify suitable landfill sites for Municipal Solid Waste (MSW) and to ensure the effectiveness of the proposed method. The use of special tools, such as GIS software, has enabled us to quickly manage and efficiently process large amounts of input data within a specific geodatabase (geology, geomorphology, hydrology, meteorological and climatic aspects,

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constraints imposed by regulatory and legislation both national and regional, etc.). Firstly, this processing was based on a preliminary division of the Sicilian territory carried out within the frame of the SIGLOD Project (Sistema Intelligente di Supporto alla Gestione e alla Localizzazione delle Discariche di Rifiuti), an intelligent support system for the management and localisation of landfills and waste management plants, by researchers involved in the project. It will be an applied and integrated technological infrastructure that will allow public organisations to recover technological efficiency and competitiveness through the use of satellite data. This first subdivision identified unsuitable areas (in red colour), low suitable areas (in blue colour) and suitable areas (in white colour) for establishment of MSW landfills (Perricone et al., 2016), starting from constraints imposed by current national and regional legislation (Figure 1a), that follow the requirements of Directive 1999/31/EC on the landfill of waste. Subsequently, the suitable areas previously identified were analysed and reconsidered on the basis of more detailed factors in order to select the most suitable areas. Although this procedure has been applied to three different sectors of Sicily (Western, Eastern and Southern sectors), which showed a higher proportion of potentially suitable areas, this paper will show results pertaining the Western sector of Sicily (Figure 1b).

2. DATA AND METHODS

2.1 Study area

The examined area is located in the Western sector of Sicily between 12°51'00" E and 13°19'00" E and 37°59'00" N and 37°47'30" N. It includes twenty municipalities (Alcamo, Calatafimi, Castellamare del Golfo, Gibellina, Poggioreale, Salaparuta, Salemi, Santa Ninfa, Vita, Bisacquino, Camporeale, Contessa Entellina, Corleone, Monreale, Partinico, Piana degli Albanesi, Roccamena, San Cipirello, San Giuseppe Jato, Santa Cristina Gela) falling within the provinces of Trapani and Palermo (Figure 1b). It covers an area of 1,160 km^2, with an average elevation of 254 meters above sea level. The study area is predominantly hilly, which is typical of the Sicilian hinterland, and is mainly characterised by sandy and clay deposits with some limestone outcroppings. Agriculture is the predominant land use within the region. The closest main towns include Palermo, Trapani, Mazara del Vallo, Marsala and Castelvetrano; the area had a population of 205,812 in 2011 (ISTAT 2011).

2.2 Methodology

Recently, both in Italy and globally, the methodology adopted for the identification of areas potentially suitable to receive municipal solid waste (landfills) has been generally based on hierarchical analytical approaches associated with Geographical Information Systems (GIS) in order to examine various criteria, including geology and hydrogeology, land use, land slope, distance from roads, residential areas, protected areas and wind direction (Dörhöfer and Siebert, 1998; Kontos et al., 2005; Chang et al., 2008; Sumathi et al., 2008; Zamorano et al., 2008; Sharifi et al., 2009; Carone and Sansò, 2010; Moeinaddini et al., 2010; Nas et al., 2010; Sener et al., 2010; Gbanie et al., 2013). Each criterion is evaluated according to a system based on scores and weights and mapped using GIS techniques. The data collected are grouped into classes that express the greater or lesser suitability of an area.

For this study, we selected eight criteria for the evaluation of landfill suitability. A meso-scale method provided the identification of new factors or the use of a greater detail for those factors previously used in the phase of macro-scale localisation. The factors were grouped according to economic or environmental importance and each factor was assigned values from five to six classes with scores between 0 and 10.

Subsequently, a "weight" was then assigned to each factor by means of a hierarchical analysis that establishes a priority scale within the factors.

The suitability of an area was then assessed by the use of a simple system of weighted summation (SAW, Simple Additive Weighting). This system is widely used for the calculation of final values in issues using multiple criteria. The mathematical formulation of this system is described by the following equation (Yoon and Hwang, 1995):

$$Vi = \sum_{j=1}^{n} Wjvij \qquad (1)$$

where Vi is the suitability index for the area i, Wj is the relative importance of the weight given to the criterion j, vij is the priority value of the area i with respect to the criterion j, n is the total number of criteria.

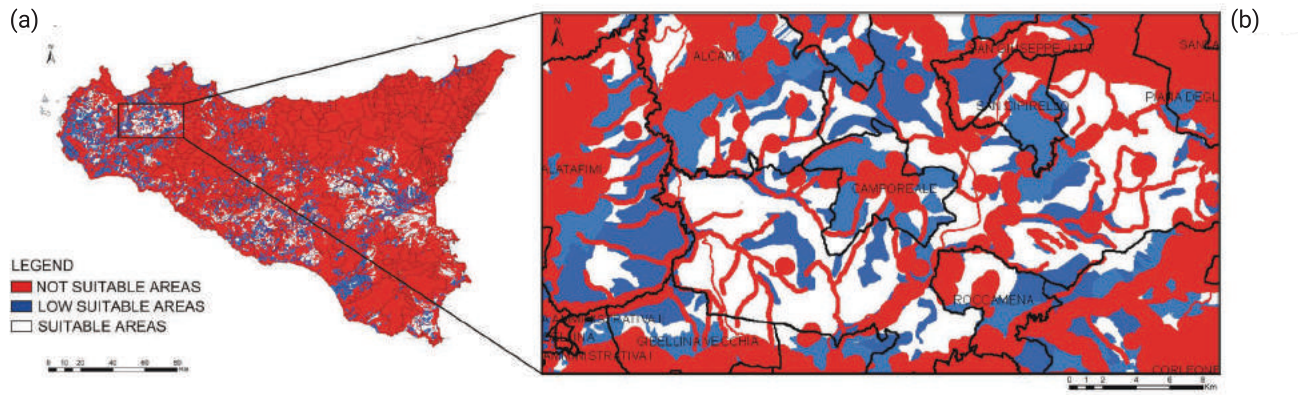

FIGURE 1: (a) Preliminary division of the Sicilian territory from constraints imposed by current national and regional legislation (Perricone et al., 2016); (b) the study area in the Western sector of Sicily.

The end result of this methodology was the evaluation of the territory on the basis of suitability indices. In this study, the used scale for such indices ranged from 0 (less suitable area) to 10 (most suitable area).

In this work, we applied a multi-criteria analysis for decision-making purposes (Multi Criteria Decision Analysis, MCDA). This method is directed at supporting the decision maker when he is working with numerous and disputing evaluations, allowing him to obtain an agreement solution.

There are several methods for multi-criteria analysis and MCDA is now used in various application fields, such as finance, planning, telecommunications and ecology.

Among the most commonly used method of MCDA, the so-called AHP (Analytical Hierarchy Process) method was applied in this work. The AHP method mainly allows to assign a priority to a number of decision alternatives and/or to relate criteria characterised by qualitative and quantitative assessments (often not directly comparable), combining multidimensional measures into a single scale of priorities (Saaty, 1980). AHP, in short, is a method to derive ratio scales from paired comparisons (Siddiqui et al., 1996; Mandylas et al., 1998; Balis et al., 1998; Kontos and Halvadakis, 2002; Kontos et al., 2003).

The comparison was performed using a nine-point scale that includes: [9, 8, 7, ..., 1/7, 1/8, 1/9], where 9 means extreme preference, 7 means very strong preference, 5 means strong preference, and so on down to 1, which means no preference (Table 1). This pair-wise comparison allowed for an independent evaluation of the contribution of each factor, thereby simplifying the decision making process. The pair-wise comparisons of various criteria were organised into a square matrix (Table 2).

The diagonal elements of the matrix were: 1. The principal eigenvalue and the corresponding normalised right eigenvector of the comparison matrix gave the relative importance of the criteria being compared. The elements of the normalised eigenvector were weighted with respect to the criteria or sub-criteria and rated with respect to the alternatives (Bhushan and Rai, 2004). The consistency of the matrix of order n was then evaluated. If this consistency index failed to reach a threshold level, then the answers to comparisons were re-examined. The consistency index, CI, was calculated as:

$$CI = \frac{\lambda_{max} - n}{n-1} \tag{2}$$

where CI is the consistency index, λ_{max} is the largest or principal eigenvalue of the matrix, and n is the order of the matrix. This CI can be compared to that of a random matrix, the Random Consistency Index (RI), such that the ratio, CI/RI, is the consistency ratio, CR. As a general rule, CR ≤ 0.1 should be maintained for the matrix to be consistent. For this study, RI is equal to 1.41 and calculated λ_{max} is equal to 8.599, producing a CI of 0.085. The consistency ratio CR was 0.060; thus indicating that a consistent matrix was formed. However, there may be a different judgment regarding the relative importance of the criteria when these

TABLE 1: The comparison scale in AHP (Saaty 1980).

Intensity of importance	
1	Equal importance
3	Weak importance of one over another
5	Essential or strong importance
7	Demonstrated importance
9	Absolute importance
2, 4, 6, 8	Intermediate values between the two adjacent judgments
Reciprocals of above nonzero	If activity *i* has one of the above nonzero numbers assigned to it when compared with activity *j*, then *j* has the reciprocal value when compared with *i*

TABLE 2: Square matrix of the pair-wise comparisons of various criteria.

Criteria	Rock Permeability	Average annual rainfall	Seismic activity risk	Distance from settlements	Land slope	Land use	Distance from roads	Distances from surface waters	Factor weights (%)
Rock permeability	**1.00**	5.00	5.00	7.00	1.00	3.00	9.00	3.00	0.299
Average annual rainfall	1/5	**1.00**	1.00	3.00	1/5	1/3	7.00	1/3	0.058
Peak ground acceleration	1/5	1.00	**1.00**	3.00	1/5	1/3	7.00	1/3	0.058
Distance from settlements	1/7	1/3	1/3	**1.00**	1/7	1/5	3.00	1/5	0.035
Land slope	1.00	5.00	5.00	7.00	**1.00**	3.00	9.00	3.00	0.299
Land use	1/3	3.00	3.00	5.00	1/3	**1.00**	7.00	1.00	0.113
Distance from roads	1/9	1/7	1/7	1/3	1/9	1/7	**1.00**	1/7	0.025
Distance from surface waters	1/3	3.00	3.00	5.00	1/3	1.00	7.00	**1.00**	0.113
Total	3.32	18.48	18.48	31.33	3.32	9.01	50.00	9.01	**1.00**

are compared in pairs. The decision process in multi-criterion problems is a subjective process, strongly linked to the decision maker. In complex problems, such as the identification of landfill sites, it is logical to expect people involved can have different opinions.

2.3 Criteria description and application

The assessment criteria used in this work were divided into two main categories: environmental criteria and economic criteria. To each factor, we assigned 5 to 6 classes of values, with a score between 0 and 10 (Table 3). Higher scores are representative of more favourable conditions of the location.

The environmental criteria included the five factors rock permeability, distance from settlements, average annual rainfall, peak ground acceleration and distance from surface water, as shown in Table 3.

The economic criteria included factors that affect the construction and the operations management of a landfill. The parameters here considered were land slopes, distance from roads and land use.

2.3.1 Environmental criteria

Among the environmental issues, great importance is attributed to the proximity of a landfill site from inhabited population centres. In fact, a landfill site located near an urban settlement poses many environmental problems, not least those related to the presence of unpleasant odours. Therefore, the direct distance of the examined sites from urban areas was taken into account, assigning the highest score with the increasing of this distance. In particular, in this work, we proposed bands at a distance gradually increasing and with an interval of 1,000 m (Figure 2). For distances less than the legal limits (500 m), we assigned a score of 0.The second parameter considered here is the rock permeability. National and regional law identifies the presence of exclusionary, penalising and preferential factors according to the coefficient of permeability and to primary and secondary porosity. During the application of the meso-scale method, the permeability factor has been re-considered and applied in order to add a greater level of detail. Rock classification on the basis of the permeability coefficient K is often arbitrary and subjective, mainly because of the wide range of K values of rocks and unconsolidated materials. However, a generalisation can be made in order to classify the study area into zones of permeability using, commonly accepted classification schemes from the literature (Colombo and Colleselli, 1994) and integrated with the data of permeability obtained from PAI (Piano di Assetto Idrogeologico) of Sicily for the types of terrains falling in the examined zones. In the present work, five classes were identified (Table 3), assigning a weight from 0 to 10 to the different degrees and permeability values identified (Figure 3).

Terrains with high permeability are considered unsuitable for establishing a landfill and were consequently attributed the lower value (0). In contrast, optimal sites are those with extremely low permeability or impermeable sites. A score of ten was attributed to the latter sites.

The third criterion, i.e. the average annual rainfall, was taken into account in order to assess the impact of rainfall

TABLE 3: Square matrix of the pair-wise comparisons of various criteria.

Main criteria	Classes	Scores
Environmental criteria		
Rock permeability	High ($K>10^{-3}$)	0
	Medium (10-3<$K<10^{-5}$)	3
	Medium-low (10-5<$K<10^{-7}$)	5
	Low (10-7<$K<10^{-9}$)	7
	Impermeable ($K<10^{-9}$)	10
Distance from settlements (meters)	d <500 m	0
	500 m < d < 1500 m	1
	1500 m < d < 2500 m	3
	2500 m < d < 3500 m	5
	3500 m < d < 4500 m	7
	d > 4500 m	10
Average annual rainfall (millimetres)	P > 800 mm	1
	700 mm < P < 800 mm	3
	600 mm < P < 700 mm	5
	450 mm < P < 600 mm	7
	P < 450 mm	10
Peak ground acceleration (g)	a > 0,150 g	1
	0,125 g < a < 0,150 g	3
	0,100 g < a < 0,125 g	5
	0,075 g < a < 0,100 g	7
	a < 0,075 g	10
Distance from surface waters (meters)	150 m < d < 300 m	1
	300 m < d < 450 m	3
	450 m < d < 600 m	5
	600 m < d < 750 m	7
	d > 750 m	10
Economic criteria		
Land slope (degree)	s > 45°	1
	26,5°< s < 45°	3
	18,5° < s < 26,5°	5
	9,5° < s < 18,5°	7
	s < 9,5°	10
Distance from roads (meters)	d > 2000 m	1
	1500 m < d < 2000 m	3
	1000 m < d < 1500 m	5
	500 m < d < 1000 m	7
	d < 500m	10
Land use	Urbanized and industrial zones	0
	Permanent crops	1
	Heterogeneous agricultural areas	3
	Arable land	5
	Pastures	7
	Quarry areas	10

in the leachate production in an MSW landfill although at present this parameter is not regulated by specific legislation which imposes specific limits.

The meteorological data were provided by SIAS (Sistema Informativo Agrometeorologico Siciliano), which is equipped with a network of stations for remote measurement, consisting of 96 automatic stations that store data detected with electric-electronics instrumentation. Monthly average rainfall data (in mm) refer to the 2003-2013 decade. Stations with higher values of cumulative rainfall were assigned the lowest score (Table 3) because they were regarded as areas of increased leachate production despite the predisposition of suitable drainage systems (Figure 4).

The peak ground acceleration criterion was taken into account using the method developed by Perricone et al. (2016) within the penalising factors, where a buffer of 300 m around the active faults was applied (see ITHACA catalogue, i.e ITaly HAzard from CApable faults,), excluding areas with peak ground acceleration higher or equal to 0.225 g of the map of the seismic hazard for Italy, realised by the Istituto Nazionale di Geofisica e Vulcanologia (INGV 2004). For the meso-scale method, it was re-applied to the areas potentially suitable already identified, considering the acceleration values less than 0.225 g. To detect the seismic zones, we used the nationwide reference seismic hazard map mentioned above. Based on the values of the expected maximum accelerations mostly identified in Sicily, the range of values of peak ground acceleration was re-classified, dividing it into five classes (Table 3). The lowest score was assigned to acceleration values between 0.150 and 0.225, while the highest score - indicative of more suitable areas - was assigned to acceleration values lower than 0.075 g (Figure 5).

The last environmental criterion, the distance from surface waters, according to the so-called "Legge Galasso" (L.n. 431 of 08/08/1985), includes all the areas included within a buffer zone of 150 meters from the shores of rivers, streams and waterways that are subject to landscape constraints (with the prohibition of building). This criterion, already taken into account during the macro-scale method in order to exclude all those areas comprised within this range of respect, was re-analysed at this stage in order to evaluate the influence of the greater or lesser distance of a potential landfill site to a water course concerning the risk of river pollution.

To assess this criterion, five different buffer zones were created, with a spacing of 150 m between a band and another, around all the river courses. These bands were converted into a grid-map with a 20x20 m resolution. To each class thus identified, we assigned a score according to the distance of the river. The maximum scores were obviously assigned to areas more distant from river shores (Figure 6).

2.3.2 Economic criteria

The land use criterion is not based on regulatory issues and can vary according to the study area. It is designed to protect "sensitive" areas, which are economically developing and can be negatively influenced by a landfill in their vicinity. In this paper, for the evaluation of the land use criterion, we referred to the map realised within the "CORINE Land Cover" project (updated 2012) that divides the country according to the characteristics of coverage and land use, with particular emphasis on environmental protection.

Analysing the various covering features, we decided to attribute the lowest score (0) to those levels previously considered within the exclusionary and penalised factors during the macro-scale method (e.g. forested areas, urban areas, water bodies, etc.). The remaining levels were organised into five classes by giving them the corresponding score depending on the greater or lesser suitability for hosting landfill sites (Figure 7). In particular, the lowest score was given to agricultural areas characterised by permanent crops (olive groves, fruit trees, etc.), while the highest score was assigned to mining areas, including landfills and abandoned mining areas (Table 3).

The criterion of distance from roads or proximity to roads should be taken into account for the selection of landfill sites because it represents the greater or lesser accessibility to the site itself. In fact, landfills should be located on sites that can be reached by alternative roads even in the presence of adverse weather conditions. On the other hand, such sites should not be positioned too far from existing road networks in order to avoid excessive costs for the construction of connecting roads. In addition, transportation of waste should not interfere with normal vehicular traffic. To evaluate this parameter, we created five buffer zones around all main roads (highways, main roads, secondary and urban roads), with interdistance of 500 m between one band and another. These bands were converted into a grid-map with a resolution of 20x20 m (Figure 8). To each class thus identified, a score was assigned according to the distance of the road from the landfill site, so that the maximum scores were assigned to roads closest to the potentially suitable site (Table 3).

The last economic criterion, the land slope or land morphology, constitutes a basic criterion for the realisation of landfill sites. In the proposed methodology, land morphology is evaluated on the basis of the slope angle, given in degrees (Table 3). Sites with excessive gradients are usually not suitable for the construction of landfills. The division into classes is based on the assumption that flat areas are most suitable for the construction of a landfill (Figure 9).

3. RESULTS AND DISCUSSION

According to the European waste management policy, landfilling is the least preferable option and should be limited to the necessary minimum. Where waste needs to be landfilled, it must be sent to landfills that comply with the requirements of Directive 1999/31/EC on the landfill of waste. The objective of the Directive is to prevent or reduce as far as possible negative effects on the environment, in particular on surface water, groundwater, soil, air, and on human health from the landfilling of waste by introducing stringent technical requirements for waste and landfills.

Taking into account the EU landfill directive, our Analytic Hierarchy Process (AHP) combined with Geographic Information System (GIS) allows discriminating the most suitable sites in a larger area already individuated as suitable for municipal solid waste landfill on the base of the parameters defined also by the regional requirements (Figure 1).

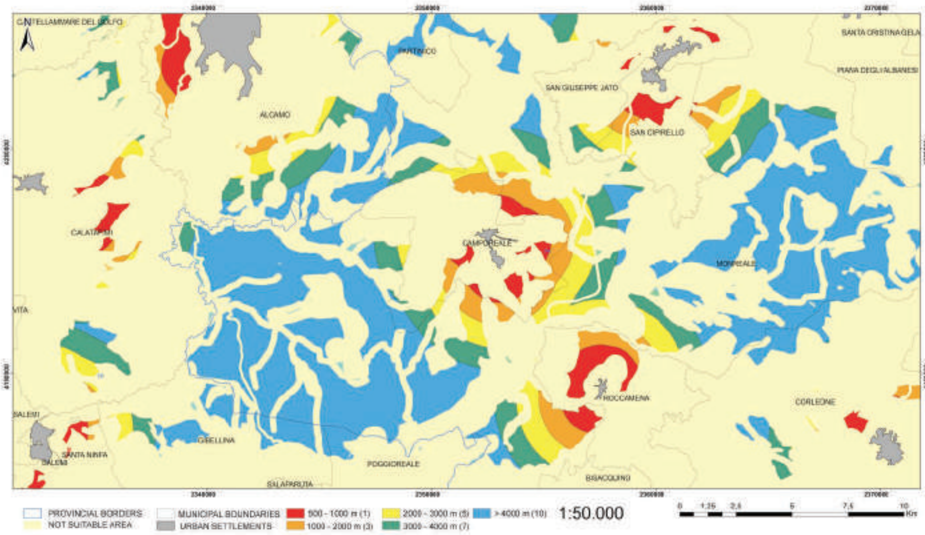

FIGURE 2: Distance from settlements criterion map for the Western sector of Sicily.

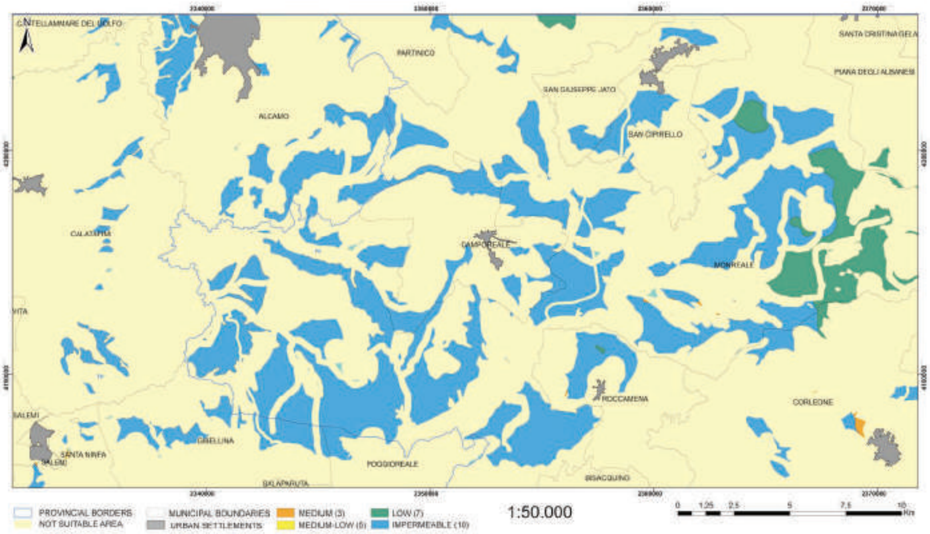

FIGURE 3: Rock permeability criterion map for the Western sector of Sicily.

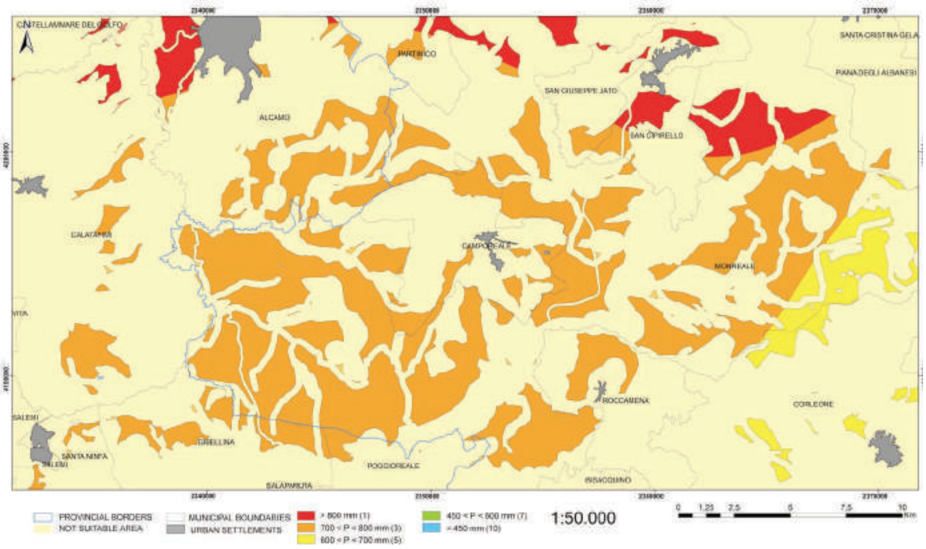

FIGURE 4: Average annual rainfall criterion map for the Western sector of Sicily.

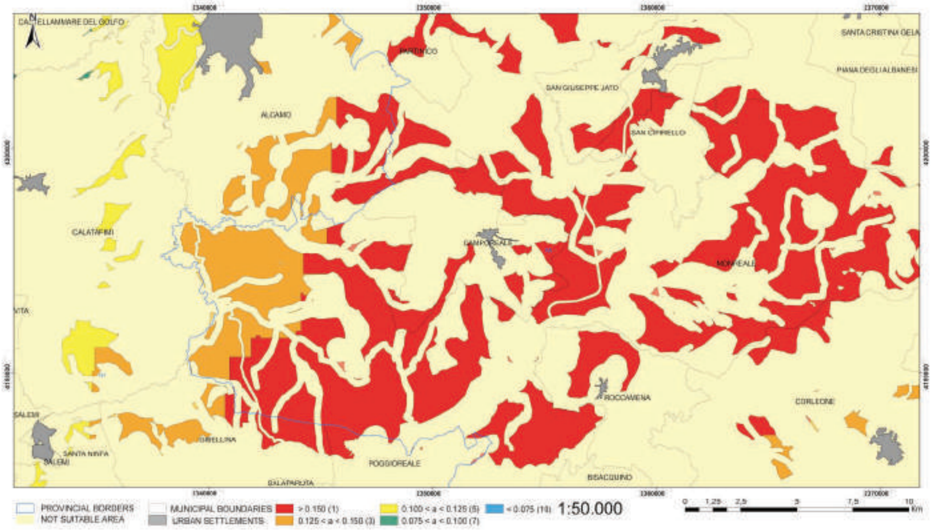

FIGURE 5: Peak acceleration ground criterion map for the Western sector of Sicily.

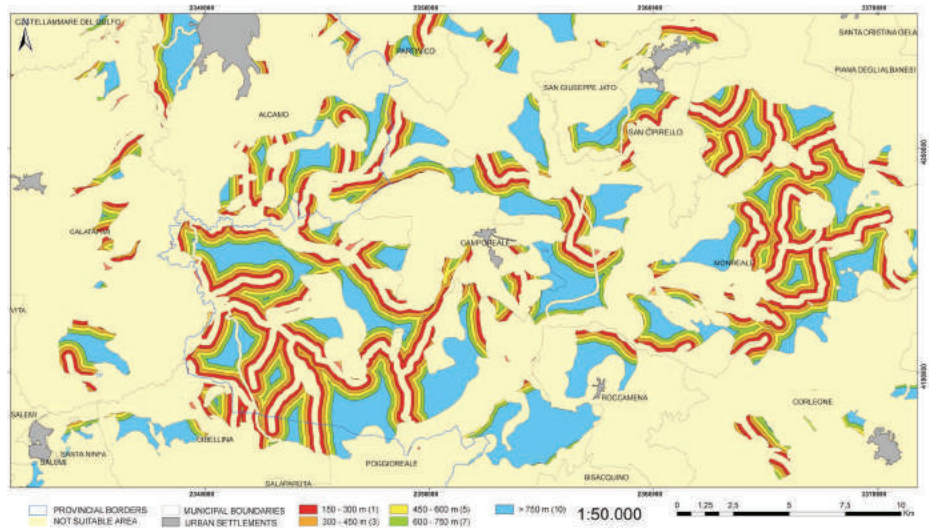

FIGURE 6: Distance from surface waters criterion map for the Western sector of Sicily.

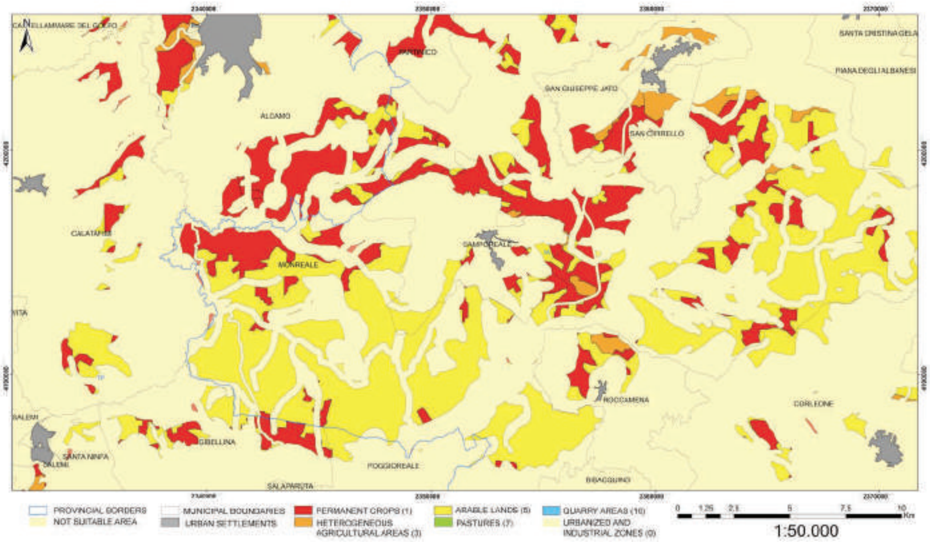

FIGURE 7: Land use criterion map for the Western sector of Sicily.

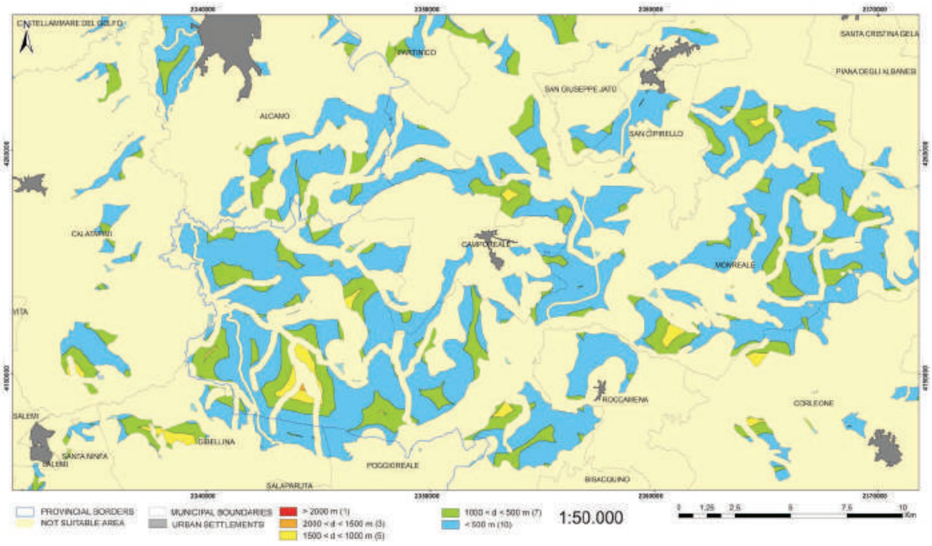

FIGURE 8: Distance from roads criterion map for the Western sector of Sicily.

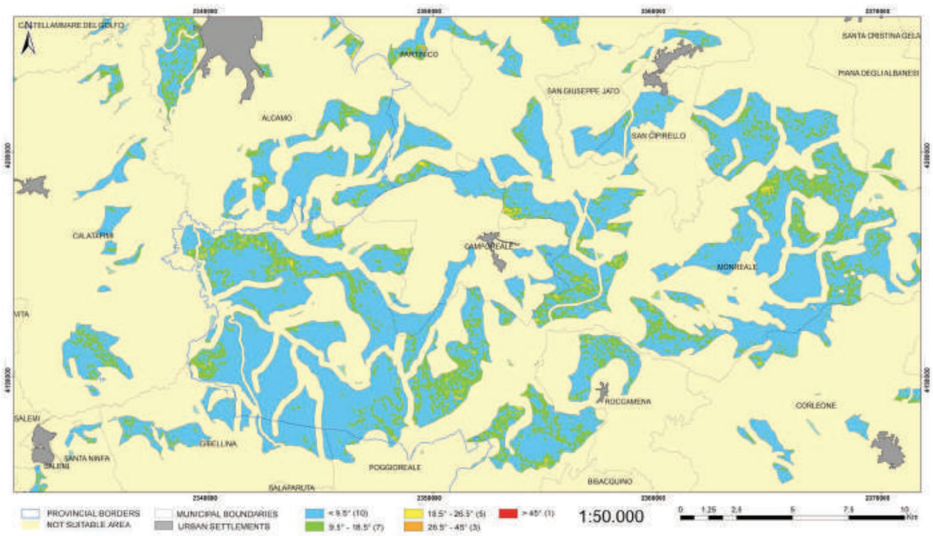

FIGURE 9: Land slope criterion map for the Western sector of Sicily.

Each criterion map was prepared using ArcGIS Spatial Analyst, and prepared maps were converted into ESRI Grid Format using weight values obtained from AHP. On the basis of the weight values assigned to each criterion, it is clear that one of the most important parameters is the permeability of the rocks (0.229 see Table 2). This parameter has been already introduced for the selection of the larger area, however for this purpose a lithological map of Sicily in a scale 1:250.000 has been used. In this second and more detailed step the analysis of permeability was based on the lithological characters of formations and synthems reported in the 1:50.000 National geological maps recently published for the studied area (Ispra Carg project, Sheet n. 607 "Corleone" and n. 619 "Santa Margherita di Belice"). The geological substrate of the area consists of Tertiary terrigenous sediments such as open shelf marls, molasse-type clays, sandstones and conglomerates, Messinian evaporites and Plio-Quaternary sandy clays, calcarenites and alluvial sediments. The nature and type of outcropping rocks (i.e. where highly permeable rocks crops out) has excluded large territories.

The second parameter considered equally important (0.229) was the land slope or morphology. Its importance derives both from the excessive costs of planning and construction in very steep areas and also from future problems of slope stability of the landfill itself. With regard to the remaining criteria both those associated with the presence of naturalistic-landscape constraints (for example land use) and those controlled by the Italian and EU directives (i.e. distance from surface waters), it was necessary to identify particularly large buffer areas.

Finally, the parameter of peak ground acceleration is also of particular importance because this area (more in general, a large belt running N-S across western Sicily) experienced in the 1968 a seismic sequence up to M 6, which is known as the earthquake of the River Belice Valley. Several municipalities were almost totally damaged and about 300 peoples died. No clear focal mechanisms were available for this seismic sequence, so the seismicity was attributed either to the compressional reactivation of a thrust sheet or to a NW-SE shear band running from the Castellammare gulf to Sciacca (Di Stefano et al., 2015). The epicentral zones of the 1968 seismic sequence was centred southward of the studied area, thus the parameter of peak ground acceleration has to be seriously considered for the assessment of the most suitable sites.

The final suitability map was derived with the aid of the map calculator function of ArcGIS and overlay analyses of ArcGIS spatial analyst (Figure 10).

After the constrained areas were extracted by masking, the land suitability of the study area was calculated by the SAW procedure (see equation 1). By considering the determined weights, the range of the landfill suitability index was between 2.99 to 9.46. Using an equal interval classification method, landfill suitability values of the studied area were classified into five groups: very high suitability (9.46-8.30), high suitability (8.29-7.54), moderate suitability (7.53-6.75), low suitability (6.74-5.81) and very low suitability (5.80-2.99). The results indicate that 3% of the study area has very low suitability, 26% has low suitability, 37% have moderate suitability, 26% has high suitability and 8% of the study area has high suitability for a landfill site.

Figure 11 schematically shows the used procedure by the AHP and GIS techniques in order to obtain a landfill suitability map.

In particular, four candidate sites (S1, S2, S3 and S4) were suggested for landfill use because these areas resulted as the most suitable and reach the largest ex-

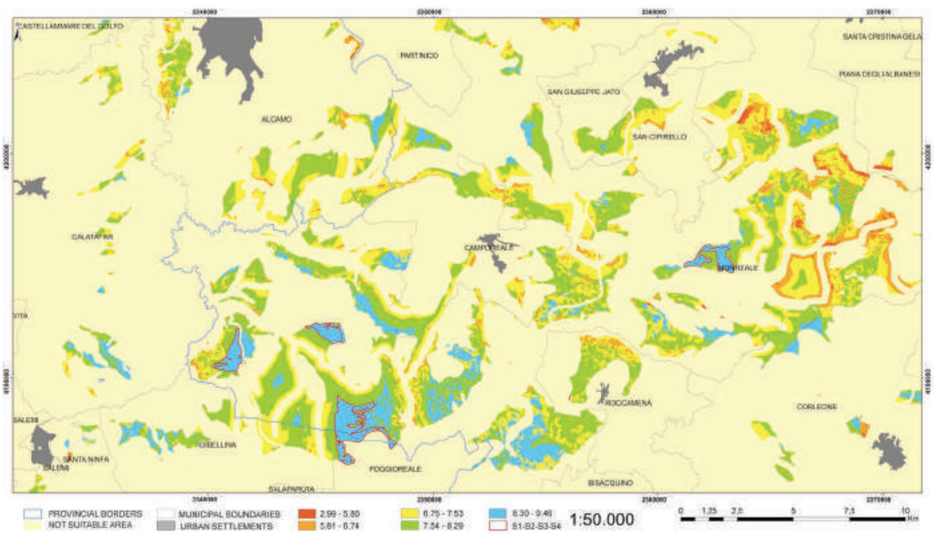

FIGURE 10: Map of the suitable area for MSW landfill locations.

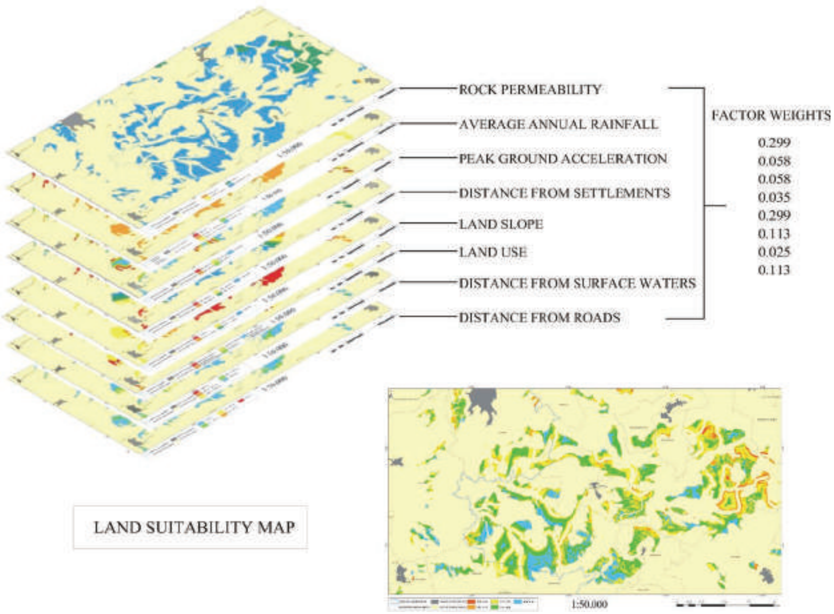

FIGURE 11: The flowchart of the methodology showing the overlay of different constrained maps and the finalland suitability map.

tensions. After the recognition of these four areas, the comparison between them on the basis of new criteria (both economic and environmental) was carried out. For example, among the new environmental criteria, the visibility of the landfill site from urban centres or the wind direction has been considered. Results of this procedure (named the micro-scale method) will be the subject of a later work.

4. CONCLUSIONS

The management of municipal solid waste in Italy is currently a very topical issue. In several regions of Southern Italy, the waste emergency has become a chronic problem, especially in Sicily, where one of the main problems is the landfilling of MSW.

The presence of a landfill in a specific area involves a series of inevitable consequences that have an impact both at socio-economic and environmental levels. In these areas, depreciation of the values of properties surrounding the landfill prevails, combined with deterioration of life and air quality and increased water and soil pollution in the surrounding environment.

It is therefore extremely important to define a scientific methodology for the identification of suitable areas for the location of MSW landfills; this enables the selection of sites that constitute the minimum environmental impact and allows the local population to make sensible choices.

In this paper, we addressed the problem of identifying the areas potentially suitable for the construction of a MSW landfill for a limited area of western Sicily, using a method based on a Geographic Information System (GIS) which is already applied in Italy and abroad.

However, the application of a multi-scale localisation procedure introduced in this paper, characterised by an increasing level of analysis from the regional up to the municipal level, has allowed to detail the examined area thus providing a tool as accurate as possible in the selection of the most suitable sites among those identified.

Despite the fact that environmental and cartographic data were not updated in some cases, several areas potentially suitable to host sites for MSW landfill in western Sicily were identified.

The methodology allows the introduction of several updates, such as the choice of different parameters or the application of different multi-criteria analysis methods. The potential offered by the computer-aided procedure and GIS, allows comparisons between different suitable areas and, above all, between areas characterized by the same score. The variation, for example, of the weight values assigned to each criterion or the introduction of new criteria specifically linked to the local situation, will further facilitate the choice of the most suitable area.

The next step will be the assessment of the sites identified through the computerised procedure described in this paper, thanks to the context analyses, starting from the field survey, sampling and laboratory analysis.

In conclusion, the study carried out in western Sicily can be an instrument of knowledge useful for territorial planning which enables the identification of areas suitable to host a MSW landfill that, on one hand, protects the environment and human health and, on the other hand, respects the constraints and restrictions imposed by law.

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COMPARATIVE ASSESSMENT OF INCINERABILITY OF MUNICIPAL SOLID WASTE OVER DIFFERENT ECONOMIES

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ABSTRACT

Municipal solid waste (MSW) generation rates have been steadily increasing over the years globally. Simultaneously, the composition of MSW has also been varying. Combined with the scarcity of land for landfilling, waste incineration is now gradually becoming an indispensable part of MSW management, even in developing countries. However, being an economy intensive process, assessment of the feasibility of incineration of MSW becomes vital prior to employing thermal Waste to Energy (WtE) techniques. A composite indicator to easily quantify the incinerability of MSW called incinerability index or *i*- Index has been developed in this regard. A comparative assessment of incinerability of MSW generated from different economies was drawn using this indicator. It was observed that incinerability of MSW decreased with a decrease in income. However, a marginal increase in incinerability was observed by 2025 in countries belonging to low-income groups.

1. INTRODUCTION

MSW management strategies adopted have been evolving over the years, especially in developing countries like India, owing to increasing generation rates, varying composition and growing environmental awareness. Economies that relied largely on composting and landfilling for management of generated MSW is now gradually inclining towards technologies that can handle the quantum of MSW generated at faster rates and occupy lesser space. Consequently, thermal Waste to Energy (WtE) treatment techniques like incineration are gaining prominence as an inalienable element of integrated solid waste management.

Figure 1 formulated using World Bank database demonstrates the disparities in MSW management in countries belonging to different economic groups (Hoornweg and Bhada-Tata, 2012). While open dumping is prevalent in low-income countries, more than 60% of the generated MSW in high-income countries is diverted from the landfills. With global waste generation stipulated to nearly double to 2.2 billion tonnes per year, it has become quintessential to renounce unsustainable management practices like landfilling with high greenhouse gas (GHG) potential. This has further boosted the growth of the thermal WtE sector, which can also serve as a source of renewable energy. Figure 2(a) developed using USEPA (2014) illustrates a gradual reduction in the quantum of MSW being landfilled, while the quantity of MSW incinerated is nearly constant after a steep increase in the initial stage. Thermal WtE techniques dominate the global WtE market, constituting about 88.2% of net market revenue (World Energy Council, 2016).

United States (US) and Europe alone have 86 and 455 waste to energy plants, respectively (ISWA, 2012). A significantly steady growth in this sector, however, was observed in China, with almost two-fold growth in WtE capacity in the period from 2010-13, as evident from Figure 2(b) (World Energy Council, 2016). The front-runner in exploiting WtE technology for MSW management is, however, Japan, with incineration of nearly 80% of the generated MSW (Lombardi et al., 2015). The WtE sector is envisaged to have escalated growth rates in Asia-Pacific region also due to the waste management initiatives in China and India. Attempts to incorporate thermal treatment into integrated waste management had started in the wake of the century in both the countries. However, higher organic fraction and the subsequent high moisture content (MC) and low calorific value tend to reduce the feasibility of WtE treatment. India had its first waste incineration plant set up in 1989, at Timarpur, Delhi. Despite a state of the art design, the plant was shut down after a few weeks of operation due to erroneous assumptions in the thermal characteristics of the MSW feed (Talyan et al., 2004). However, with improving standards of living, waste composition and characteristics have undergone appreciable changes. Singh et al. (2011) also reaffirmed the scope for thermal WtE technologies for MSW management in India.

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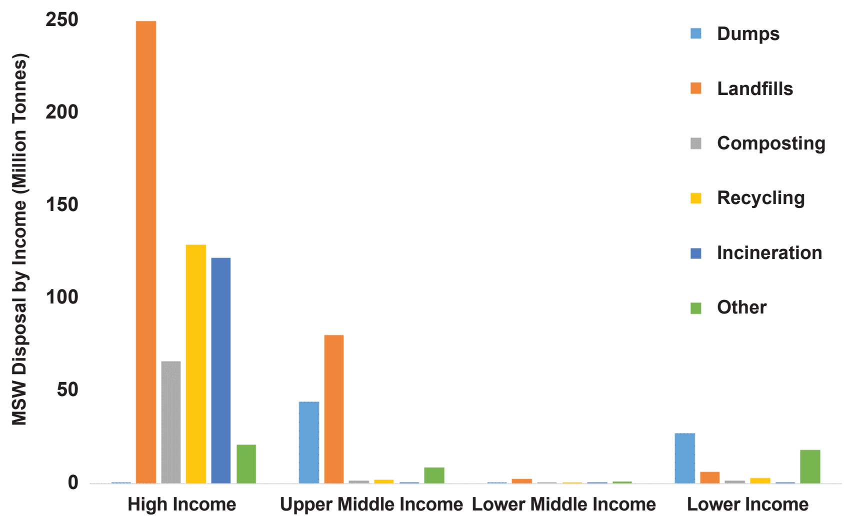

FIGURE 1: MSW disposal techniques adopted in different economies (Data source: Hoornweg and Bhada-tata, 2012).

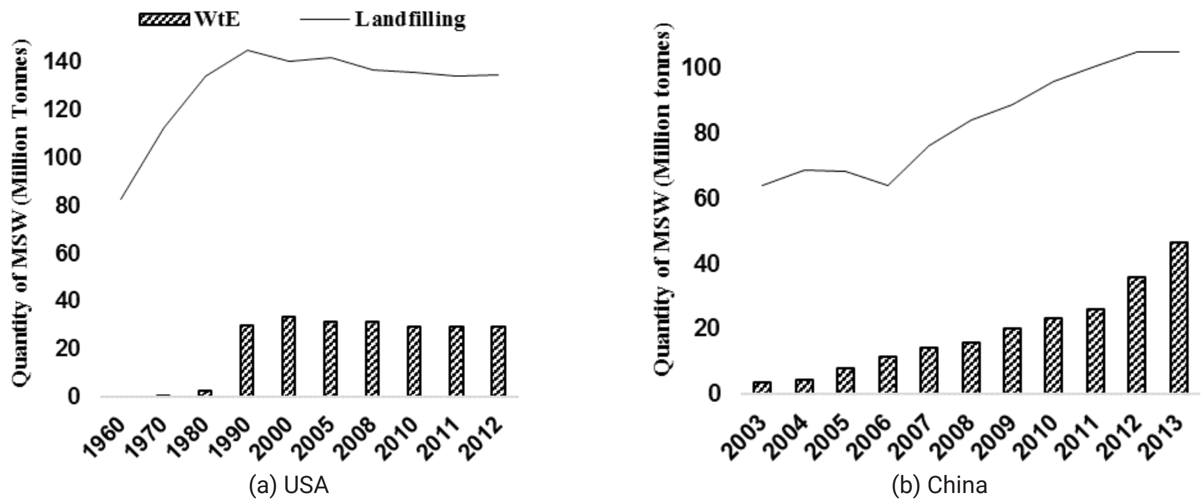

FIGURE 2: Quantity of MSW incinerated for energy recovery and disposed of in landfills over the years in the USA and China (Data source: USEPA, 2014; Robinson et al., 2009).

Being a relatively economically exhaustive technology during installation and operation, ensuring the feasibility of incineration of MSW becomes very crucial prior to its execution (Rand et al., 1999). The primary objective of the technology is the disposal of the MSW with minimal load on the environment. Nevertheless, energy recovery is considered an added advantage for the same. Moreover, low calorific values of the generated MSW may necessitate augmenting the feed with an auxiliary fuel which can make the process economically unfeasible.

The feasibility of incineration of MSW, or 'incinerability' of MSW may be defined as the amenability of MSW, to be burned completely to sterile ash, with minimal environmental impact, optimum energy recovery and economic sustainability. The feasibility of the incineration relies on the various properties of the feed to the furnace. While pre-treatment operations can considerably improve its thermal properties, it can prove to be economically exhaustive at the same time. A quick assessment technique for the incinerability of MSW can significantly aid in the decision-making process; incorporation of WtE techniques into integrated waste management as well as the need for pre-treatment operations can be determined by employing such a tool. A composite indicator for incinerability (CII) called incinerability index or *i*- Index of MSW has been developed for the quantification of incinerability of MSW; it encompasses the 3-E criteria which are fundamental for the viability of WtE techniques. *i*- Index is a novel estimation technique which can ascertain the incinerability of MSW prior to technology selection and framing of integrated waste management strategies.

The paper describes the application of this CII on MSW generated from countries belonging to different income groups. A comparative assessment of incinerability of MSW shall thus be drawn over different economies.

2. METHODOLOGY FOR THE FORMULATION OF CII: A BRIEF ACCOUNT

The formulation of the CII entails four main stages, viz.

- Selection of parameters that has the potential to affect incinerability of MSW;
- Determination of parameter weightages with reference to incinerability of MSW;
- Development of normalisation curves for rendering the parameters comparable;
- Aggregation of the normalised parameters.

Despite objectivity being a highly desirable factor for assessment of incinerability and subsequent decision-making process, lack of specific data makes the task subjective to some extent. To assuage the associated challenges, opinions of multiple panels of experts have been incorporated in the formulation of the quantification tool (Kumar and Alappat (2003); Saxena and Bhardwaj (2003); Kurian et al. 2005). A list of 13 parameters (Table 1) identified from exhaustive literature surveys was sent to a panel of 138 experts, comprising academicians, consultants, and regulatory authorities. Based on the scoring received on a scale of 1-5 and thorough scrutiny, a revised list of 8 parameters was finalised. The relative weightages of the chosen parameters were established using pairwise comparisons by analytic hierarchy process (AHP) instituted by Saaty (1980). Another expert panel consisting of 201 panellists comprising mostly of academicians and a section of industrial counterparts was approached for the same. 79 individual acceptable responses were acquired at this stage. Further, to transform the parameters to a uniform scale to facilitate their aggregation, graphical normalisation technique was employed. An expert panel comprising of 90 members were approached for developing rating curves for each parameter. An averaged curve was subsequently developed for each parameter based on the feedback from the panellists.

Figure 3 displays the averaged rating curves developed hence. The cumulative impact of these parameters was then estimated by choice of aggregation technique, the values of which can shed light on the incinerability of MSW. Weighted aggregation function was used for deriving CII for MSW.

$$\mathrm{CII} = \frac{\sum_1 w_i P_i}{\sum_1^n w_i} \qquad (1)$$

where, w_i - weightage of each parameter on a scale of 0-1, P_i - normalised score from rating curves on a scale of 1-100; n - number of parameters and CII - composite indicator called *i*- Index on a scale of 0-100.

Being an increasing scale indicator, a higher value suggests higher incinerability or better amenability to incineration. A step-by-step account of the formulation of *i*- Index has been presented by Sebastian et al. (2018). A comparison of the incinerability of MSW generated from countries belonging to different income groups, namely, high-income countries (HIC), upper-middle-income countries (UMIC), lower-middle-income countries (LMIC) and low-income countries (LIC), shall be drawn using CII.

2.1 Characteristics of the MSW generated from study areas

The composition of MSW generated in the afore-mentioned regions, reported by Hoornweg and Bhada-tata (2012), was used for the study. The same has also been illustrated in Figure 4. While organic fraction constituted more than 50% of the generated MSW in countries belonging to lower income groups, combustible components like paper and plastic formed a major fraction of MSW generated in HIC. Hence, the calorific value of the MSW in the former was considerably lower in comparison to the latter. The composition of the MSW influences the thermal characteristics and consequently the feasibility of incineration for energy recovery. This could also affect the auxiliary fuel requirement when the MSW is subjected to incineration. Furthermore, the pollution potential of MSW incineration also needs to be considered while estimating the incinerability of MSW. The various parameters for computation of CII were theoretically computed by the approach proposed by Kaiser (1966).

In order to estimate the incinerability of the MSW generated from study areas, the theoretically estimated input parameters were normalised using Figure 3. With the relative weightages and the normalised values known, the CII was computed subsequently by equation (1).

3. RESULTS AND DISCUSSION

The results of the assessment of incinerability of raw MSW over different economies have been tabulated in Table 2. Table 3 displays the values for the same projected to 2025.

The CII of raw MSW from HIC was found to be nearly 30% higher than that of LIC. Higher organic fraction and the consequent moisture content may have caused considerably low incinerability for MSW generated in LIC. Moreover, the SO_2 release potential was found to be appreciably higher for the MSW generated in LIC, in comparison to

TABLE 1: List of parameters sent to the panel of experts in the preliminary survey.

% Moisture content	% Inert content	Greenhouse gases (GHG) released/kg MSW feed
Heat content/kg of MSW feed	Total primary pollutants released/kg of MSW feed	Quantity of auxiliary fuel required per kg of MSW feed to maintain a particular temperature, say 10000 C
% Volatile content	Stoichiometric air required for incinerating/kg of MSW feed	Bulk Density of MSW feed
Ultimate analysis of MSW feed	Specific heat	Size of the MSW feed used for energy recovery
Time required for complete combustion of 1 kg MSW feed		

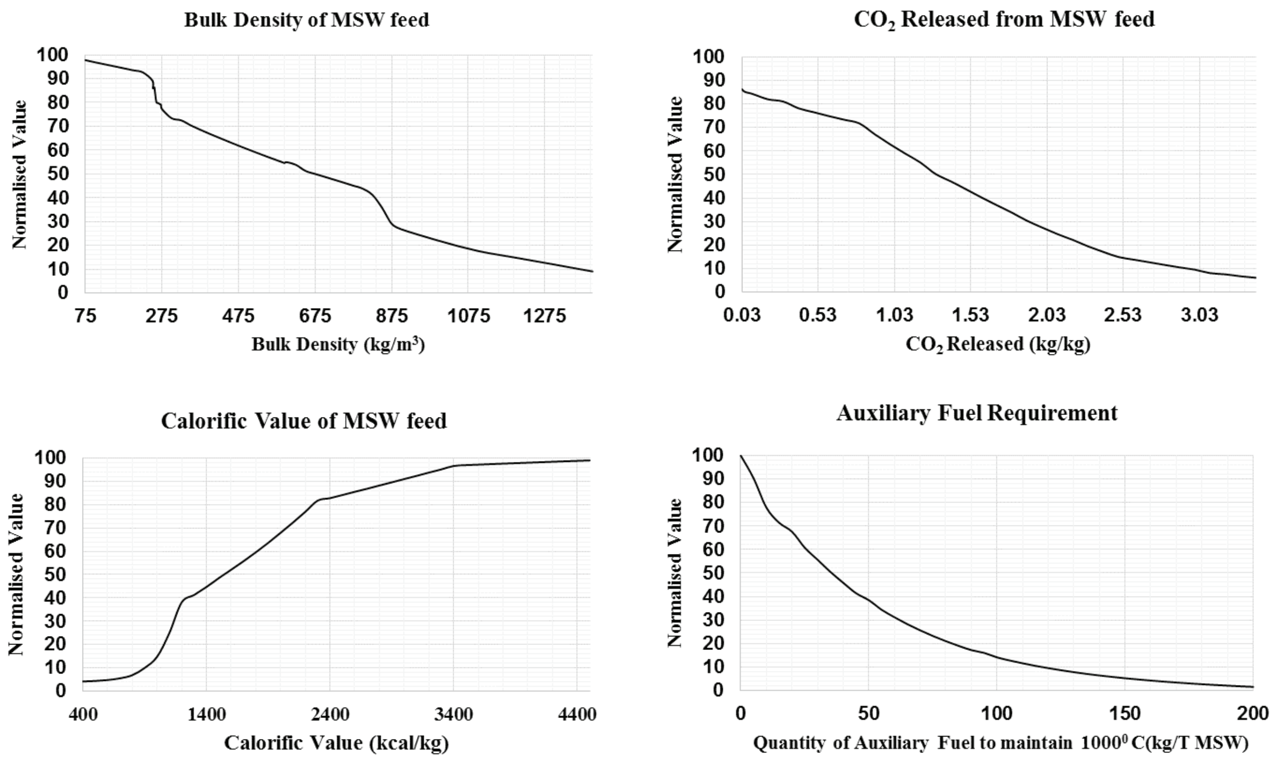

FIGURE 3 [i]: Normalisation curves for (a) Bulk density (b) CO_2 released (c) Calorific value (d) Auxiliary fuel requirement.

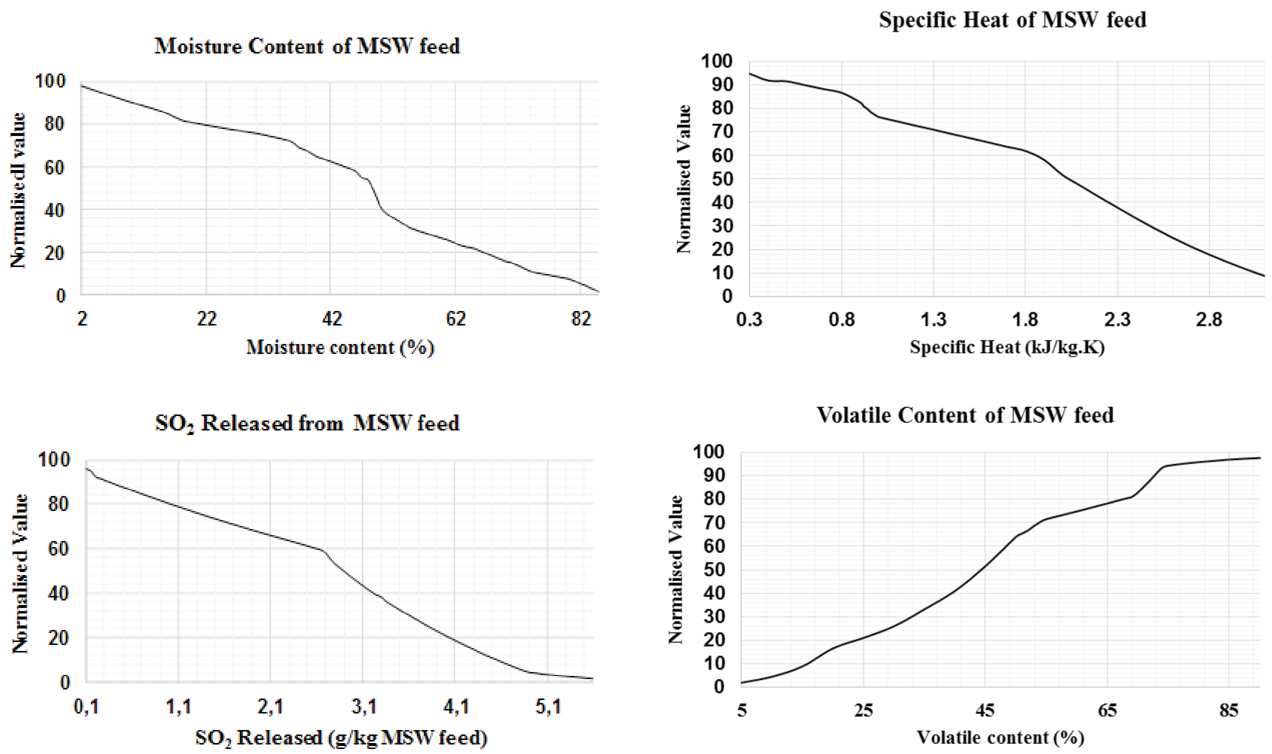

FIGURE 3 [ii]: Normalisation curves for (e) Moisture content (f) Specific heat (g) SO_2 released (h) Volatile content.

HIC. Within MIC, the biodegradable fraction is 8.47% higher in LMIC, in contrast to UMIC. Since this can increase the sulphur content in the MSW, the SO_2 release potential was higher in LMIC as well, relative to MSW generated in LMIC. Subsequently, the CII of raw MSW in UMIC was found to be 7.5% higher than LMIC. The composition of MSW generated in a particular region is affected by economic growth, standards of living, and cultural and climatic variations. This was further reflected in the incinerability of the generated MSW.

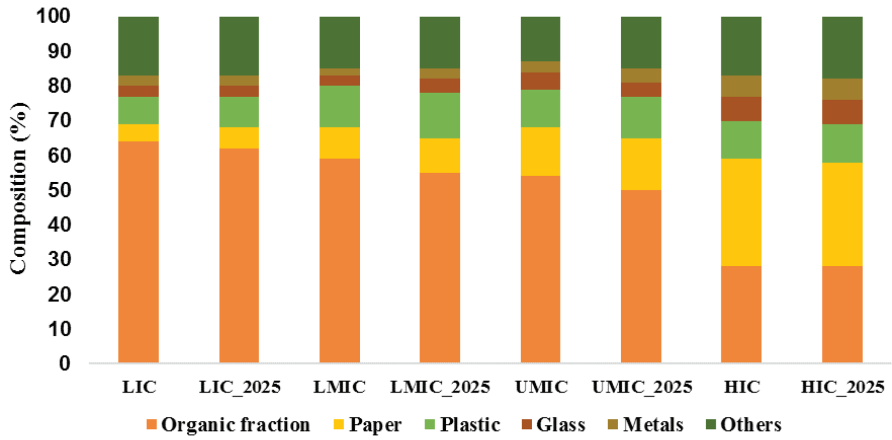

FIGURE 4: Percentage composition of MSW in different economies presently and projected to 2025. All the values are expressed in wet weight basis (Data source: Hoornweg and Bhada-tata, 2012).

An attempt to estimate the incinerability of MSW over different economies by 2025 was also carried out. While the incinerability of MSW improved marginally in LIC and MIC, a negligible drop in incinerability was observed in the MSW generated from HIC. Minor reduction in the paper waste generation in HIC may have triggered this trend in the latter. Ranking of the MSW generated in countries belonging to different income groups for the current year and projected values for 2025 is illustrated in Figure 5. The variation in the values of the indicator also suggests that a higher value of CII, towards 100 indicates an MSW of higher incinerability. Such MSW samples when fed to the furnace can be incinerated with energy recovery and minimal environmental impact, without being economically exhaustive. On the other hand, a lower value of CII hints at the unfeasibility of incineration. The indicator was also computed for refuse-derived fuel (RDF) which has a high heat content of about 4774 kcal/kg. While the MSW generated in HIC had the CII relatively closer to RDF, implying very high incinerability, the MSW generated in countries belonging to lower economic groups had significantly lower incinerability rating. Further, the flue gas released from incineration of RDF has been reported to have relatively lower pollution potential, which is evident from the incinerability rating. The flue gas generated from incineration of MSW generated in HIC was also observed to have lower pollution potential, in comparison to MSW generated in other economies. The parameter scores for SO_2 released substantiates this. The CII thus conveys the incinerability of MSW, while incorporating the 3-E concept, based on which decision-making can be accomplished.

Apart from ranking the generated MSW, the CII can also be used as a decision-making tool for the implementation of WtE techniques for MSW management. Based on the quantified incinerability, need for the pre-treatment can be identified. For instance, although the CII value for MSW from LIC is comparatively low, drying can considerably improve its thermal properties and hence the amenability to incineration. The choice of the pre-treatment techniques may be made using individual incinerability rating of the parameters under consideration. For instance, the individual incinerability score for bulk density of MSW generated in LIC was 55, as opposed to an appreciably high score of 96.3 for MSW generated in HIC. Pre-treatment operations that entail the removal of inert and other bulky constituents of MSW shall help improve the overall incinerability of MSW. Moreover, seasonal and annual fluctuations of the MSW generated in a particular locality may be computed to ascertain the incinerability of MSW, prior to adopting incineration for waste management. This data can help foresee the incinerability of MSW into future years, which can avert the closure of the plant due to poor thermal characteristics of the MSW feed.

4. CONCLUSIONS

With WtE techniques gradually becoming a significant element of integrated waste management, studies on the feasibility of incineration have become inevitable. For easy quantification of amenability to incineration of MSW, a composite indicator for incinerability called *i*-Index has been developed. Using the incinerability thus quantified, a ranking of MSW on the basis of its feasibility for incineration can be derived. MSW generated from different economies were used to demonstrate the application of this composite indicator. While MSW generated from HIC had high *i*- Index values of 73 that generated from LIC displayed a low *i*- Index value of 51.5. Consequently, MSW generated from LIC was deduced to be less amenable to incineration. Further, the incinerability of MSW was estimated to improve by at least 1-1.5% by 2025 in LIC and MIC, due to improving standards of living in those countries, which reflected in the composition of the generated MSW. However, a negligible decrease is anticipated in the incinerability of MSW generated from HIC by 2025. This was due to the marginal decrease in the paper fraction in the generated MSW. In addition to establishing the feasibility of incineration, this composite indicator may also be used for estimating the pre-treatment operations required to improve the incinerability of MSW. Being an economy-intensive technology, feasibility studies are crucial prior to its implementation. A tool like *i*- Index which can help promptly analyse the incinerability

TABLE 2: Incinerability assessment of MSW from different economies for 2012.

Parameter (of MSW feed)	Weightage (wi)	$Value_{HIC}$	N_{HIC} (p)	Unique $rating_{HIC}$	$Value_{UMIC}$	N_{UMIC} (p)	Unique $rating_{UMIC}$	$Value_{LMIC}$	N_{LMIC} (p)	Unique $rating_{LMIC}$	$Value_{LIC}$	N_{LIC} (p)	Unique $rating_{LIC}$
Bulk Density (kg/m^3)	0.092	125	96.3	9.897	250	90	9.250	500	60.6	6.228	600	55.5	5.704
CO_2 Released (kg/kg MSW feed)	0.148	1.092	88.5	12.547	1.019	90	12.760	1.024	90.4	12.817	0.941	91.4	12.958
Heat content (kcal/kg)	0.152	2467.5	93.5	15.026	2189.7	76	12.214	2166.06	75.5	12.134	1791.2	59.8	9.610
Moisture content (%)	0.09	18.81	81.8	7.893	30.67	75.9	7.324	33.02	75	7.237	35.41	73.4	7.083
Auxiliary Fuel required to maintain 10000 C (kg/T MSW feed)	0.134	0	99.9	13.4	31.5	53.1	7.13	40.86	42.8	5.74	71.75	22	2.95
SO_2 Released (g/kg MSW feed)	0.161	2.854	52	7.477	3.072	45.6	6.557	3.152	42.7	6.140	3.269	39.6	5.694
Specific Heat	0.097	1.961	53.2	5.446	2.545	28.3	2.897	2.659	22.2	2.272	2.747	19.1	1.955
Volatile content (%)	0.124	55.10	72	9.452	50.62	64.6	8.481	49.88	63.9	8.389	45.85	53.8	7.063
CII or *i*- Index				**73.31**			**61.56**			**59.24**			**51.55**

Value $_{HIC,\ UMIC,\ LMIC,\ LIC}$ are the parameter values of each MSW sample; $N_{HIC,\ UMIC,\ LMIC,\ LIC}$ denotes normalised parameter values.
Unique rating $_{HIC,\ UMIC,\ LMIC,\ LIC}$ denotes unique incinerability rating of individual parameters.

TABLE 3: Incinerability assessment of MSW from different economies projected to 2025.

Parameter (of MSW feed)	Weightage (wi)	$Value_{HIC\_25}$	N_{HIC_25} (p)	Unique $rating_{HIC\_25}$	$Value_{UMIC\_25}$	N_{UMIC_25} (p)	Unique $rating_{UMIC\_25}$	$Value_{LMIC\_25}$	N_{LMIC_25} (p)	Unique r $ating_{LMIC\_25}$	$Value_{LIC\_25}$	N_{LIC_25} (p)	Unique $rating_{LIC\_25}$
Bulk Density (kg/m^3)	0.092	319.8	72.3	6.69	277.6	76.5	7.08	273.1	78.3	7.25	286.4	74.6	6.91
CO_2 Released (kg/kg MSW feed)	0.148	0.8	71.9	10.67	0.8	72.2	10.72	0.7	72.3	10.74	0.7	73.3	10.89
Heat content (kcal/kg)	0.152	2436.9	82.5	12.57	2259.4	79.2	12.07	2225.4	78.3	11.93	1877.4	63.2	9.63
Moisture content (%)	0.09	18.8	81.2	7.28	28.9	76.2	6.83	31.1	74.7	6.69	34.5	72.8	6.52
Auxiliary Fuel required to maintain 10000 C (kg/T MSW feed)	0.134	0	99.9	13.42	26	60.2	8.08	31.6	53.1	7.13	64.3	28.6	3.84
SO_2 Released (g/kg MSW feed)	0.161	2.9	51.5	8.32	3	46.1	7.45	3	46.1	7.45	3.2	40.4	6.52
Specific Heat	0.097	2	54.6	5.27	2.5	31.6	3.05	2.6	26.4	2.55	2.7	21.2	2.05
Volatile content (%)	0.124	54.5	70.8	8.82	51.2	65.7	8.18	49.9	64	7.97	46.7	54.4	6.78
CII or *i*- Index				**73.04**			**63.46**			**61.71**			**53.13**

$Value_{HIC\_25,}$ $U_{MIC\_25,\ LMIC\_25,\ LIC\_25}$ are the parameter values of each MSW sample projected to 2025; $N_{HIC\_25,\ UMIC\_25,\ LMIC\_25,\ LIC\_25}$ denotes normalised parameter values projected to year 2025.
Unique rating $_{HIC\_25,\ UMIC\_25,\ LMIC\_25,\ LIC\_25}$ denotes unique incinerability rating of individual parameters projected to the year 2025.

of MSW shall be instrumental in framing MSW management strategies.

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COMPARISON OF WASTE MANAGEMENT SYSTEMS IN WESTERN AND TRANSITION ECONOMIES WITHIN THE WATRA PROJECT

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ABSTRACT

The transition from a planned to a market economy has been a great challenge for all post-socialist states of the former Eastern Bloc. Public services which were to a major extent previously subsidized by the state needed to be adapted to new economic realities. This paper will present some of the results of a country overview report. This report represents the first working package of the WaTra project, which aims to help understand and develop the waste management systems of selected transition economies. 13 European and post-Soviet countries were chosen with different starting conditions and economic states to investigate which are the crucial factors in enabling the development of modern waste management. The countries considered were compared with each other using indicators of waste management. Waste collection, waste treatment, landfilling, recycling, composting and incinerated waste per capita are the main aspects taken into account and set in relation to economic indicators. Western EU countries generate higher amounts of waste per capita and have higher recycling rates. Landfilling is the major waste treatment method in post-Soviet and Eastern European countries, even among the EU states. The change from practices of only landfilling to modern waste management systems with high recycling and recovery is more recognizable. The more developed a country's waste management system is, the more waste is generated according to the strengthening of its economy.

1. INTRODUCTION

1.1 WaTra Project

Transition from a planned to market economy has been a great challenge for all post-socialist states of the former Eastern Bloc. Public services which were previously to a major extent subsidized by the state needed to be adapted to the new economic realities. In this respect, the municipal waste management sector (WM) is usually the most problematic due to its chronic state of underfinancing, noticeable influence on the urban image, as well as significant negative impact on the environment. The goal of the currently running WaTra project (Waste in Transition Economies) is to support the sustainable reformation process of the waste management sector in Belarus and the Ukraine through the enhancement of international cooperation and capacity building at partner universities and other stakeholders in the field of waste management (see projects homepage: http://watra.boku.ac.at/).

1.2 Overview Report

As a first step in the WaTra project, the waste management systems of 13 post-socialist and post-Soviet countries as well as non-socialist EU countries were described and analyzed. Countries for analysis were selected based on population size, varying waste management performance, preferred treatment technologies and governance system (decentralized/centralized, democratic/autocratic), availability of information (for post-Soviet states). The results of this data collection are merged within Task 1.2: "Comparison of WM Systems in Western and Transition Economies (Overview report)". It consists of 415 pages and gathers all information about the countries' profiles and related waste management information, including comments on data availability and scientific validation.

The main results are presented in this paper (Wohmann et al. 2016). In the report all used references from the respective country overviews could be find, above this data from the European statistical office and from d-waste atlas (www.atlas.d-waste.com) has been used. Therefore in the paper it is referenced to this report and the primary sources are not listed in the references (Table 1).

The countries considered were compared with each other using waste management indicators. Waste collec-

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TABLE 1: List of countries in which waste management systems have been described in the WaTra project.

	Country	Political Development	Selection Reason for the Analysis
1	Poland	Post-socialist (EU)	Well-developed medium-performing large post-socialist country
2	GDR / Germany	Post-socialist (EU) / Non-socialist (EU)	Best performing large post-socialist country (former GDR) Best performing large non-socialist country with highest recycling rates (western Germany - former FRG)
3	Estonia	Post-Soviet (EU)	Best performing small post-Soviet country
4	Austria	Non-socialist (EU)	Best-performing mid-sized non-socialist country with highest composting rates
5	Denmark	Non-socialist (EU)	Best performing small non-socialist country with highest WTE rates
6	Italy	Non-socialist (EU)	Typical medium performing large non-socialist country with decentralized governance
7	UK	Non-socialist (EU)	Medium performing large non-socialist country with centralized governance
8	Belarus	Post-Soviet	Project main country; mid-sized post-Soviet country with centralized/autocratic governance
9	Ukraine	Post-Soviet	Project main country; large post-Soviet country with democratic governance (EU accession candidate)
10	Russia	Post-Soviet	WM system determining country before transition; large post-Soviet country with centralized/autocratic governance
11	Kazakhstan	Post-Soviet	Mid-sized post-Soviet country with centralized/autocratic governance
12	Moldova	Post-Soviet	Typical small post-Soviet country with democratic governance (EU accession candidate)
13	Georgia	Post-Soviet	Typical small post-Soviet country with democratic governance (EU accession candidate)

tion, waste treatment, landfilling, recycling, composting and incinerated waste per capita are the main criteria that were taken into account and set in relation to economic indicators. In that case, the years 1995 and 2014 were considered. Over the period from the 1990s until to today, the post-socialist EU states and the Soviet states had the same requirements in the beginning. However, the waste management systems differed strongly. The few available data for the period before the 1990s are also taken into account.

It will be shown how and for what reasons the post-socialist EU states and "old" EU countries developed much better than the post-Soviet states after the collapse of the Socialist bloc from a waste management point of view. Performance of the old socialist centralized waste management system and the challenges of the transition period in the post-socialist / post-Soviet states are addressed. The development paths for waste management in the countries considered during the last 25 years were compared and main influencing factors (economic, governance, etc.) determined.

1.3 Data collation

The gathering of all comparison data was done by the scientists and students involved during the project. It quickly became obvious that there is a relationship between the developmental state of the country and the data availability. The same conclusions can be drawn by looking at the waste management stage of the country and its related data. In countries with low waste management performance, it was difficult to get official statistical data. This was caused by poor administrative infrastructure and awareness of this topic. In these cases estimates often had to be made by the few experts and scientists working in this field in the lacking countries. For the comparison countries in Western Europe, official statistical numbers could also be used with a higher temporal resolution stretching back to the political turning point in 1990. For the time before that, the data situation was poor for every country considered. Therefore, it had been tried to gather data from the years after 1990, and similar values in the years before the breakdown of socialism are assumed.

2. COMPARISON

The waste management situation and the development of the waste industry in post-socialist states, the "old" EU member states and the post-Soviet states over the past 30 years were explained in detail in the related report as outcomes of the WaTra project. Waste sectors have developed differently in the countries considered – not only regarding treatment processes and operations, but also the temporary state of development. Waste management development of a country undergoes different development phases. Klampl et al. (2006) classifies 5 stages, see Table 2.

The countries considered represent the whole range of these developmental stages. The western European countries can be seen to have almost reached phase 5 with some potential for higher quotas of secondary raw material streams. Most new European member states or post-Soviet states are situated in phases 1 to 2, and even phase 0 can be found in rural areas.

In the report generated, waste, treated waste (which in this report means collected waste), landfilled waste, recycling, composting, incineration, the GDP and the unemployment rates of those countries will be compared.

For post-Soviet countries that have no strict information policy, data availability is poor. In order to become an EU member, comprehensive waste data management

TABLE 2: Phases of waste management development - adapted from Kampl et al. (2006).

Phase 0	Neglect
Phase 1	Collection and uncontrolled disposal
Phase 2	Controlled disposal
Phase 3	Collection logistics
Phase 4	Recovery solutions
Phase 5	Industrial cycle of (secondary raw) materials

is required. Information from EU member states is easily available, although the degree of aggregation has to be validated. Yet the information from post-Soviet states is fragmentary.

The countries Poland, Germany and Estonia will be regarded as post-socialist EU member states. Austria, Denmark and Italy are considered "old" EU member states. Post-Soviet states include the following countries: Belarus, Kazakhstan, Russia, Ukraine, Georgia and Moldova.

Municipal waste is considered to be waste collected through waste removal systems in private households or public institutions. It is used synonymously with the term "collection of waste." In the event that there is no waste removal system, the amounts are estimated by the participating research group. The "total waste treatment" depicts the treatment of the overall collected waste, and therefore unrecorded waste is excluded. The informal sector is estimated to differ strongly. Treatment methods of waste for this study are incineration, composting, recycling and dumping/landfilling.

Table 3 shows the percentages of incineration, composting, recycling and landfilling of the collected waste. The data are given in kg per capita per year.

As seen in Table 4, there was an evaluation established by BiPRO (2012) to assess the waste management developmental stage of a country. For the report, the 6 countries Belarus, Russia, Ukraine, Kazakhstan, Moldova and Georgia were calculated by the project team. It was found that as expected, the 6 countries are on a low waste managemental level compared to the EU member states for which the assessment was conducted in BiPRO (2012).

2.1 Generated Waste

The following figure shows the produced waste per capita in 1995 and 2014. In 1995, more waste was accumulated in post-socialist countries than in 2014. Until 2014 a small decrease can be seen. For example, in Germany, 623 kg of waste per capita was generated in 1995, and in 2014 it was 618 kg per capita (Figure 1).

The "old" EU member states produced less waste in 1995 than in 2014. The amounts increased especially for Denmark. Whereas there was only 521 kg of waste per capita in 1995, Danish population produced 758 kg of waste in 2014. The causes for Germany are known through the efforts of the environmental policy to decouple economic growth from the waste generated over the last 20 years.

Also, the post-Soviet states produced less waste in 1995 than in 2014. The biggest change was found for Belarus – the population produced 144 kg of waste per capita in 1995, and in 2014 it was 421 kg of waste per capita. This can be interpreted with the obvious positive correlation between economic development and waste amounts, unless countries are aiming to decouple explicitly in the policies, as is the case in Germany.

The increase for the transition countries can be explained by the industrial development and improved living conditions of the respective populations. Due to growing production, the ensuing supply of goods and improved liquidity, the consumption behavior of people has changed.

One has to bear in mind that the figure depicts merely the amount of collected waste. Any illegally collected or dumped waste cannot be taken into consideration and constitute unreported amounts.

2.2 Landfilling

When providing generation amounts of waste at the household level, data are often used from only a few small

TABLE 3: Waste amounts and rates for different treatment methods for 1995 and 2014 (UK, Belarus and Kazakhstan are not considered in this table).

	Waste generated [per capita in kg]		Landfill [% of generated waste]		Recycling [% of generated waste]		Composting [% of generated waste]		Incineration [% of generated waste]	
	1995	2014	1995	2014	1995	2014	1995	2014	1995	2014
Poland	280	272	99	52	0	21	1,8	11	0	15
Germany	623	618	42	1,5	26	46	13	17	17	34
Estonia	371	357	99	6	0	26	0,5	4,8	0	47
Italy	454	488	92	32	3,5	25	1,3	16	5	19
Austria	437	566	46	4	46	26	27	31	12	36
Denmark	521	758	18	1,3	14	26	10	17	56	54
Russia	101	330	90	90	10	10	n.d.	n.d	n.d.	n.d
Georgia	n.d.	318	100	100	0	0	0	0	0	0
Moldova	n.d.	650	100	100	0	0	0	0	0	0
Ukraine	195	268	99	99	0	0	0	0	0	0

n.d. - *No data*
0 - *considered not existing by scarse data*

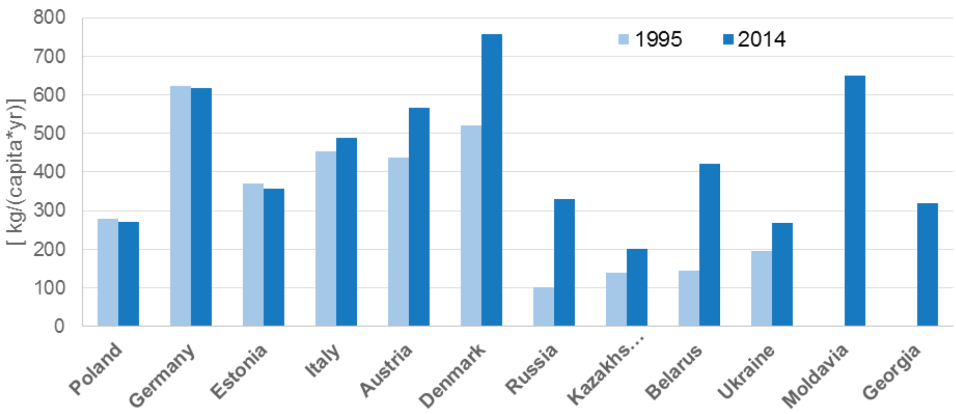

FIGURE 1: Waste generated per capita in 1995 and 2014 (for Ukraine, Moldova, Georgia, numbers for 2007 were used, no data were available for the period before 2000, and Russian data from 2004 and 2006), Wohmann et al. (2016).

surveys done voluntarily by related experts or by NGOs, especially in the countries considered with low waste management development. Sometimes generation rates are estimated by the amounts delivered to waste management facilities. In countries with a high share of landfilling, these data are similar to the waste generation. Therefore, it is useful to look at landfill amounts per capita. These numbers show only the amounts which are brought to official landfill sites, and in the better cases, these are already sanitary landfills with base sealing and suitable covering and collection systems for leachate and landfill gas.

Reducing landfilled waste amounts is always to be seen as one of the most important actions to improve a country's waste management situation. Landfilling leads to land consumption, landfill gas emissions and the depletion of resources that could instead be looped back into the economy through recycling and recovery (Maletz 2018).

Figure 2 shows the landfilling rate of waste for 1995 and 2014. This represents the amount of waste that was officially recorded or estimated by local experts at the public landfill sites as the rate of overall waste generation. Due to a lack of comprehensive statistical registering in some countries, partial estimates were used. It needs to be mentioned that post-socialist EU member states Estonia and Poland dumped 99 percent of their waste in 1995 while Germany dumped 42 percent of their waste in the same year. Until 2014, Poland reduced this amount to 50 percent. The reductions were higher in Estonia and Germany – Estonia dumped only 6 percent while Germany dumped 1.5 percent of the overall produced waste starting from a lower

TABLE 4: BiPRO Assessment for countries considered - BiPRO (2012) - and own calculations within the WaTra project, sorted from the highest to lowest result.

	Decoupling	Existing waste prevention program	Amount of municipal waste recycled (Score doubled for overall scoring)	Amount of municipal waste recovered (energy recovery) (Score doubled for overall scoring)	Amount of municipal waste disposed (Score doubled for overall scoring)	Development of municipal waste recycling	Existence of ban/restrictions for the disposal of municipal waste into landfills	Total typical charge for the disposal of municipal waste in a landfill	Existence of pay-as-you-throw (PAYT) systems for municipal waste	Collection coverage for municipal waste	Available treatment capacity for municipal waste	Forecast of municipal waste generation and treatment capacity in the WMP	Existence and quality of projection of municipal waste generation and treatment	Compliance of existing landfills for non-hazardous waste	Fulfilment of the targets related to biodegradable municipal waste going to landfills	Rate of biodegradable municipal waste going to landfills	Number of infringement procedures – WFD and Landfill Directives	Number of court cases – WFD and Landfill Directives	Overall score (2 last columns not considered for summation)
Austria	0	2	2	2	2	2	2	1	2	2	2	2	2	2	2	2	2	2	35
Denmark	0	0	2	2	2	2	2	2	1	2	2	2	2	2	2	2	2	2	33
GDR / Germany	1	0	2	1	2	2	2	2	2	2	2	2	1	2	2	2	2	2	32
United Kingdom	1	2	2	1	2	2	0	1	1	2	2	2	1	1	2	1	2	2	28
Poland	1	2	1	0	1	2	1	1	1	0	2	0	0	1	0	0	1	2	15
Estonia	2	0	1	0	0	0	1	1	1	0	2	0	1	2	2	1	1	1	15
Italy	0	0	1	1	1	0	1	2	1	2	0	0	0	0	2	1	0	0	15
Belarus*	1	2	1	1	0	2	1	0	0	0	1	0	1	1	0	0	n.c.	n.c.	13
Russia*	0	2	0	1	0	1	1	0	0	0	0	1	1	0	0	0	n.c.	n.c.	8
Ukraine*	0	2	0	1	0	1	0	0	0	0	0	0	0	0	0	0	n.c.	n.c.	5
Kazakhstan*	1	0	0	0	0	1	1	0	0	0	0	1	1	0	0	0	n.c.	n.c.	5
Moldova*	0	2	0	0	0	0	0	0	0	0	0	1	1	0	0	0	n.c.	n.c.	4
Georgia*	n.c.	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	n.c.	n.c.	2

* calculated in WaTra project
n.c. - not calculated

2	high performing according to this criterion / factor
1	medium performing
0	low performing or not existing

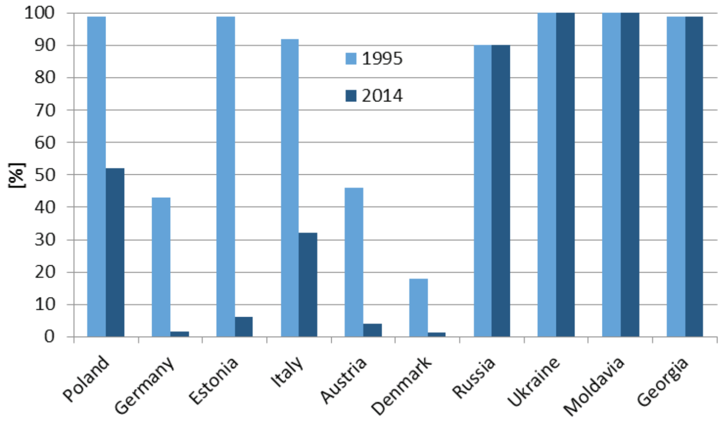

FIGURE 2: Landfilling rate in 1995 and 2014, Wohmann et al. (2016)

dumping level already. The example of Estonia can be seen as a best practice through massive efforts from the Estonian administration. A supporting effect is the small size of the country and the membership in the EU, where funding for an improving waste management situation can be requested, which was extensively used in several projects (Fischer 2013). Furthermore, this is caused by a strong enforcement of environmental law. Economic development also helps make investments in environmental technology possible.

There are still big differences between the "old" EU member states in a similar developmental stage. In 1995 Italy landfilled 92 percent of its waste, Austria disposed 46 percent and Denmark 18 percent of its waste in landfills. Until 2014 there were the following reductions: Italy reduced to 32 percent, Austria decreased to 4 percent and Denmark reduced the amounts to 1.3 percent.

For the post-Soviet states, data was available only for Georgia, Moldova, Russia and the Ukraine. These countries dump a similar amount of waste. Russia dumps circa 90 percent of its waste (VDMA 2015). If the waste accumulation is 330 kg per capita, this amounts to 297 kg of dumped waste per capita. Georgia and Moldova do not treat their waste in measurable amounts, and thus all waste is dumped. The Ukrainian numbers tell us that currently around 230 kg of waste is landfilled per capita and per year. The absolute amount of landfilled waste per capita can be adduced for estimating the environmental impacts through landfilling, such as the emission of climate gases like landfill gas and CO_2.

2.3 Recycling

Figure 3 indicates the amount of recycling per capita in 1995 and 2014. After collection, most of the waste produced goes through the procedure of materials recycling. For the post-Soviet countries, data could not be provided for all countries. Statements can only be made about the Ukraine, Russia, Moldova and Georgia.

During the 1990s, there was no materials recycling in Poland or Estonia. Compared to that, in 1995, Germany recycled 26 percent and in 2014, 46 percent of its overall waste produced. Around 50 percent of all waste is recycled in Germany and flows as secondary raw materials into the production cycle.

From the "old" EU member states Austria is most interesting, as the recycling rate decreased from 46 percent in 1995 down to 25 percent in 2014. This can be explained by the fact that Austria uses more of its municipal solid waste for waste-to-energy processes. The more common development towards higher rates can show Denmark and Italy, whereas Italy's recycling increased from 3.5 percent in 1995 to 26 percent in 2014, and Denmark increased from

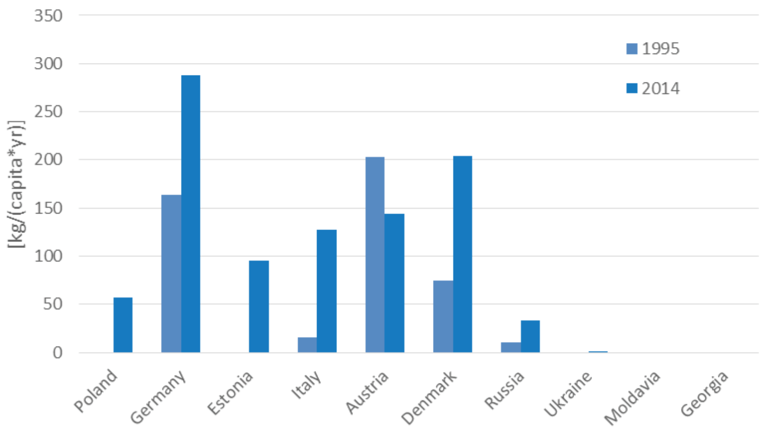

FIGURE 3: Materials recycling per capita in 1995 and 2014 (Belarus and Kazakhstan not considered), Wohmann et al. (2016).

14 percent to 26 percent.

Russia recycled only approximately 10 percent of its overall waste generated (VDMA 2015). Georgia and Moldova did not manage any kind of recycling in past years, though there is some small development apparent in its current situation. No statements can be made about Kazakhstan and Belarus, as there is a lack of data for both countries. The Ukraine recycled a very small part of it its collected waste, and the recycling rate given was approximately 0.09 kg per capita and per year.

2.4 Composting

Figure 4 indicates the amount of composted waste per capita in 1995 and 2014. A comparison ensues here merely between the "old" EU member states and the post-socialist EU states. In post-Soviet countries like Georgia, the Ukraine and Moldova, there were officially no composting amounts registered. For Kazakhstan, Russia and Belarus there were no numbers available.

The figure shows that the "old" EU member states compost much more than the post-socialist states.

As for the post-socialist EU member states, Germany composted most of its waste both in 1995 and 2014. In 1995, 13 percent, and in 2014 17 percent of the overall waste was composted in Germany. Poland and Estonia showed a significant increase – especially Poland went from 1.8 percent in 1995 to 11 percent in 2014. Estonia composted only 0.5 percent (1995) and 2.8 percent (2014) of its overall waste. In 1995, neither the state-of-the-art nor the capacities were sufficient to build composting plants or provide the required capacities. A higher share of waste was treated with other methods. The population's own efforts to compost in their own gardens were not included in the analysis, although they could raise the proportional amount of composting significantly, as green waste and garden waste amount to a considerable amount of waste. Especially for Belarus and the Ukraine as joint project partner countries, a very high share of home composting could be identified, and increasingly investigated in the rural areas.

Compared to the other "old" EU member states, Austria has the highest share of composting of its overall waste produced. The share increased from 27 percent in 1995 to 31 percent in 2014. In Denmark, composting rose during the same period from 10 percent to 17 percent, and in Italy from 1.3 percent to 16 percent.

2.5 Incineration

Figure 5 presents the incinerated waste amounts per capita in 1995 and 2014. There was no data generated in the post-Soviet countries Russia, Kazakhstan and Belarus. Statements can only be made about the Ukraine, Moldova and Georgia. For the post-Soviet EU member states, there

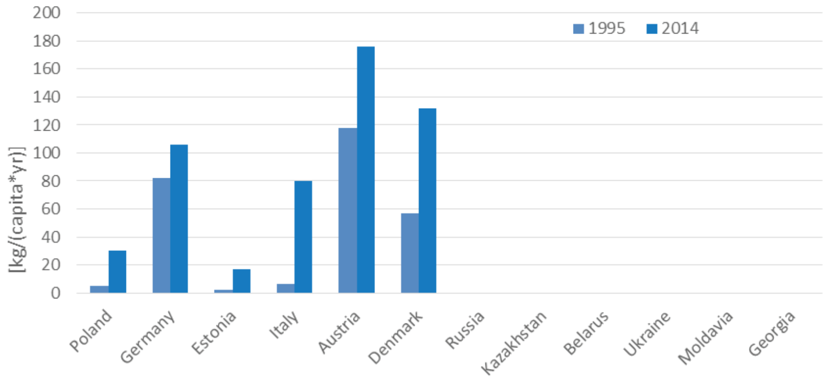

FIGURE 4: Composting per capita in 1995 and 2014, Wohmann et al. (2016)

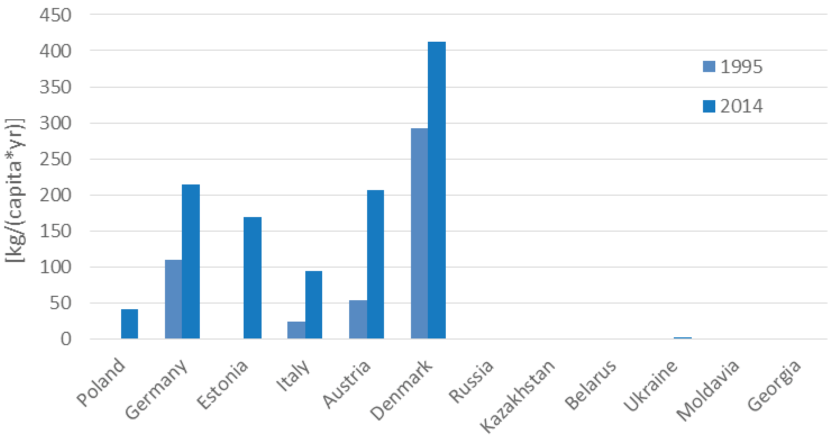

FIGURE 5: Total incineration per capita in 1995 and 2014 (no data for Belarus and Kazakhstan; in Georgia, Moldovia, Russia and the Ukraine, the amounts of incinerated waste were considered negligible), Wohmann et al. (2016).

was merely a comparable figure for Germany in 1995, as no incineration plants existed in Poland or Estonia. In 1995, Germany treated only 17 percent and in 2014, 34 percent of the overall collected waste in incineration plants. Energy for heating and electricity was generated through this thermal process. In 2014, Estonia captured 47 percent of its waste per capita as an energy resource through incineration. Due to the possible energy gain, incineration has a high importance in Estonia. In 2014, Poland treated 15 percent of the overall waste produced per capita in incineration plants. There are only a few Polish incineration plants, and thus the waste has to be treated in other ways.

Among the "old" EU states, Denmark has the highest amount of incineration. Small and economically strong countries like Denmark, besides Switzerland and the Netherlands, concentrate on waste to energy for their residual waste stream due to less geographical space and high energy and heat demand. In 1995, Denmark treated 56 percent and in 2014 54 percent of the overall waste produced per capita in incineration plants, which is 100% of the residual waste. Both in Italy and Austria, the amount of incinerated waste has risen significantly from 1995 until 2014. In 1995, Italy treated 5 percent and in 2014, 19 percent of its total waste by incineration. For Austria it was found that in 1995, 12 percent, and in 2014, 36 percent of the overall waste produced was incinerated. As for the post-Soviet states, only the Ukraine treated its waste, with very small share in incineration plants with a low technological standard. The figures here amount to 0.06 kg per capita. In Moldova and Georgia, there was no incineration of waste until now. The small incinerators which can be found partly in hospitals in these countries are not being considered in this comparison.

2.6 GDP and the unemployment rate

Developing a country's waste sector, social aspects and economic differences have to be taken into account. Among others, the development of a waste management system affects the gross domestic product (GDP) and the unemployment rate as the basic economic parameters. The following figures depict the differences concerning GDP and unemployment in post-socialist EU states, the "old" EU member states and the post-Soviet states. The social and financial differences merge in the willingness of citizens to introduce new waste systems, especially waste collection schemes.

The GDP summarizes the value of all goods and services of a particular amount of time generated in a person's country. It is of importance to check whether the economic effort is achieved by a national or foreign citizen.

Apart from that, the GDP is often compared to the prosperity of a country. Yet it remains problematic that GDP as an instrument to measure prosperity does not indicate whether the government's funds are invested wisely. Environmental exploitation and the waste of natural resources may have a positive effect on the economy and raise the GDP. Statistically, this would be an increase in the GDP. Apart from that, illegal employment, barter, shadow economy and subsistence economy cannot be ignored, as they form the livelihood for many poorer citizens. Yet these "industries" are not taken into the GDP's figures.

Furthermore, the GDP serves as an indicator for economic growth. It is indicated by the rise of the GDP. An increase in economic power is based on an increase in productivity, which is influenced by:

- physical capacity (machines)
- human capacity (employees)
- natural resources
- technical knowledge

Figure 6 depicts the GDP per capita in euros. It shows the increase of the GDP per capita from 1995 to 2014. In addition, the GDP in the "old" EU member states and the post-socialist countries was significantly higher than in the post-Soviet states.

In the post-socialist EU states, the 1995 GDP was clearly below the GDP average in 2014. Germany's GDP amounted to 23,000 euros per capita in 1995, and 37,100 euros per capita in 2014.

The GDP of the "old" EU member states is clearly above the overall average of the other countries compared. In 1995, Denmark's GDP amounted to 26,600 euros per capita, and in 2014 it was 46,800 euros per capita.

The GDP of the post-Soviet states is distinctly below

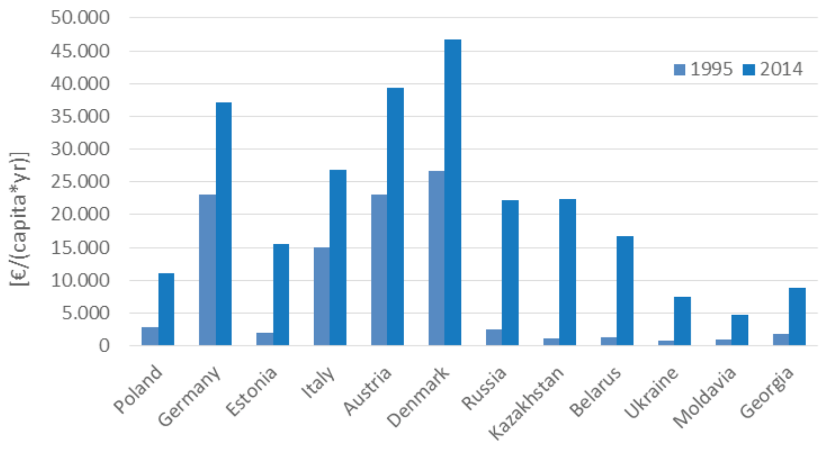

FIGURE 6: GDP per capita in 1995 and 2015, Wohmann et al. (2016).

the average of the EU member states. The Ukraine had the lowest GDP in 1995 – only 875 euros per capita. In 2014 the Ukrainian GDP rose to 7,500 euros per capita. The lowest GDP was in Moldova – 4,700 euros per capita.

From the figure above, it emerges that countries with a longer EU membership have a clearly higher GDP. Examples are Denmark, Austria, Italy and Germany. Poland and Estonia joined the EU later.

Unemployment is the lack of employment opportunities for parts of the population that are both able to work and seeking work. In many countries around the world, unemployment is one of the biggest macroeconomic challenges, as it causes high social costs.

Figure 7 illustrates the unemployment rate in 1995 and 2014. The unemployment rate was higher in 1995 than in 2014 in post-socialist countries. 8.2 percent of the German population able to work was unemployed in 1995. Until 2014, the unemployment rate sank to 5 percent.

In the "old" EU member states, a similar tendency prevails. Italy has the highest unemployment rate: it was 11.2 percent in 1995 and 12.7 percent in 2014. The Danish unemployment rate is, however, identical for 1995 and 2014 – 6.6 percent.

For the post-Soviet countries, the unemployment rates of 2006 and 2014 were compared, as no earlier data is known. There was a higher unemployment rate in 2006 than in 2014. Georgia had the highest unemployment rate: in 2006 it was 13.6 percent and in 2014 it was 13.4 percent. The biggest decrease in unemployment happened in Moldova – from 7.4 percent in 2006 to 3.9 percent in 2014, with its strong industry sector as one possible explanation.

The figures for unemployment are closely related to the financial concerns of the population. The more unemployed people, the more people struggle with financial problems and existential fears. This is a factor that influences the population's willingness to implement and accept a new industrial waste system.

The relationship between unemployment and GDP or economic growth is explained through Okun's Law. Arthur Okun first described the correlation between the two aspects based on his empirical observations. His law states that an increase in the unemployment rate by 1 percent costs 2.5 percent of economic growth. However, a reverse scenario can also be observed: It takes 2.5 percent of economic growth in order to decrease unemployment by 1 percent. One has to bear in mind that the exact percentage varies depending on the type of national economy and has to be adjusted anew.

To achieve a decrease in unemployment by boosting economic growth, it needs a so-called "employment surge." This surge characterizes a growth rate that is required as a minimum in order to secure current employment. Among other factors, the extent of the employment surge is defined by technological progress – because the higher the productivity, the less human capital is necessary to achieve the same GDP.

Figure 8 depicts the dependency of the GDP per capita and the unemployment rate in 1995 and 2014.

The figure illustrates the employment surge between 1995 and 2014. In 1995, Estonia, Germany, Austria and Denmark had low unemployment rates and sufficiently high economic growth. This means the employment surge was successful. The economic growth was sufficient to curb or decrease unemployment. In 2014 this applied to Germany, Austria, Denmark, Russia, Kazakhstan and Belarus. In 1995 the following countries had no successful employment surge: Georgia, Moldova, Ukraine, Belarus, Kazakhstan, Russia and Italy. As a consequence, unemployment rates went up. In 2014 this could be observed in Poland, Estonia, Italy, Ukraine, Moldova and Georgia.

Germany, Austria and Denmark achieved an employment surge in 1995 and 2014. On the one hand, these countries exhibit a consistent social and financial standard and on the other hand, a consistently positive development in the waste industry. It can be speculated whether and how these factors are related to one another. However, based on that assumption, broader support from the population for waste industry issues is visible. Apart from that, the economic and political interests pursue a constant improvement of the waste industry and the related improvements for the environment. Neither Georgia, nor Moldova, the Ukraine or Italy had a successful employment surge in 1995 and 2014. This is another hint that poor social and financial standards are related to a lack of willingness and opportunities for citizens, politics and the economy to contribute to change and improvement of the waste industry and the environment.

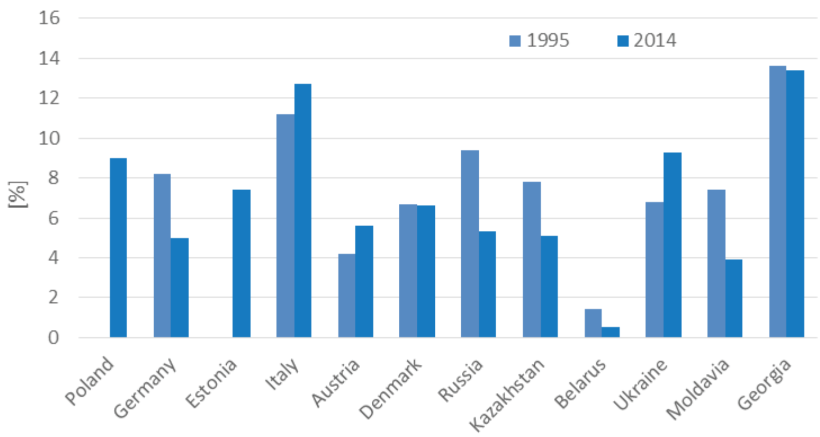

FIGURE 7: Unemployment rate in percent in 1995 and 2014, Wohmann et al. (2016)

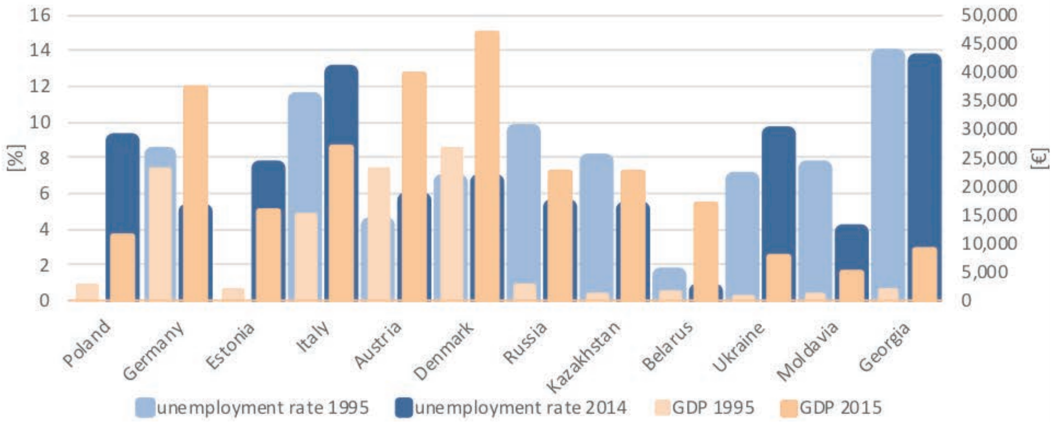

FIGURE 8: Relation of GDP per capita and unemployment rate.

3. CONCLUSIONS

Looking at the past socialist waste management system, there was a very efficient recycling system in the socialist economy. Shortage leads to efficient usage of material streams. Lots of uncontrolled dumping sites existed in the socalist countries. Recycling was not done for ecological reasons, but for economic. The partly ecoefficient waste management systems in socialist countries broke down after the collapse of the "Iron Curtain." For the future, it is necessary to implement a comprehensive data system. This is an important factor for measuring the performance and deriving improvements. Integration of the informal sector activities in the organized WM system can be a strong support for reducing the amounts of valuable waste streams going to landfills, especially where there is a lack of investment power in the waste management sector. Improving the waste sector has proven to be an ecological and economic opportunity for a developing country. Waste systems in the EU could be transferred to post-Soviet states (best practice examples for post-Soviet EU states. Countries with an EU orientation (GDR, former socialist EU states) had more opportunities to modernize their WM systems based on the model of existing western market oriented WM systems.

The project aimed to compare socialistic countries in Europe and the former Soviet Republic with some other European countries, because they have similar legislational conditions (EU-legislation) for their waste management development. Even the post soviet non EU states follow these regulations proved to be useful both for environmental protection and economic growth.

Further research could include other socialist countries not only European and former Soviet states. Waste compositions was not compared due to weak data availability, but should included in the further work in this field.

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USE OF ACRYLONITRILE-BUTADIENE-STYRENE FROM WASTE ELECTRIC AND ELECTRONIC EQUIPMENT WITHOUT AN ACCURATE PREVIOUS SEPARATION

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ABSTRACT

The aim of this work it to develop recycling strategies for Acrylonitrile-Butadiene-Styrene (ABS), coming from plastic WEEE stream, avoiding sorting by type. Self-compatibilization of ABS/HIPS blends, as well as the addition of Styrene-Butadiene-Styrene (SBS) as a compatibilizer to ABS/HIPS blends in order to improve mechanical properties, were studied. In this way, thermal behavior, mechanical performance and morphology of ABS/HIPS physical blends with two different proportions, 80/20 and 50/50, was analyzed comparatively with single ABS to assess self-compatibilization effectiveness. Obtained results indicate that ABS/HIPS self-compatibilization is effective. ABS can tolerate up to 50 wt% of HIPS conserving its properties with a slight improvement in ductility and strength. This allows a wider error in plastic sorting by type within plastic WEEE stream and consequently costs can be reduced. Same blends proportions with the addition of 2 wt% of SBS was also studied in comparison to their physical blends and single ABS. Mechanical properties of SBS-compatibilized blends were notably improved with respect to physical blends and consequently to ABS. Results are very promising for plastic WEEE recycling leading to a sustainable strategy that can promote the reuse of recycled ABS blended with other plastic WEEE instead of single ABS.

1. INTRODUCTION

Constant advances in high-tech products lead to an increment in electrical and electronic devices consumption, meaning that most of the replaced equipment are turned into scrap (Namias, 2013). The stream of waste from electrical and electronic equipment (WEEE) contains several valuable and recyclable materials like gold, silver, platinum, palladium, plastics and silica (Baldé et al., 2014; Buekens and Yang, 2014). While plastics are neither the main nor the most abundant, they occupy a lot of space in landfills because of their high volume due to low density and parts shape (Goodship and Stevels, 2012; Cui et al., 2003). Plastic from WEEE represents approximately 18 wt% of this waste stream consisting mainly of thermoplastics that can be recycled by reprocessing (Brennan et al., 2002; Bisio and Xanthos, 1995). For their recycling, they are generally chopped, washed and sorted by type involving relatively high costs (Baxter et al., 2014). Plastic resins from WEEE are similar which makes it very difficult to separate them by type using automatic sorting methods. There are several specific automatic techniques but their precision highly depends on plastic stream composition (WRAP, 2009). Near infrared spectroscopy (NIR) is the best known automatic method for plastic separation by type. However, its use for plastic WEEE classification is not optimum because these materials usually are dark colored making it difficult to classify by NIR. Also, the major amount of plastic WEEE contains styrenic resins with very similar molecular structures (like ABS and HIPS) and consequently, they are not well differentiated by NIR (Arends et al., 2015; Maris et al., 2015; WRAP, 2009). Also, the last problem strongly affects sorting by other techniques like density separation and impact milling (WRAP, 2009; Tall, 2000). Due to these reasons, and because devices used for automatic sorting are expensive and not easy to handle, lead to mainly manual sorting in the plastic recycling industry (Beigbeider et al., 2013). This is not optimum as the precision of this sorting method depends on human error and also, it can be hazardous and unhealthy for workers (Ceballos et al., 2014).

A sustainable option to manage plastics from WEEE would be to not separate WEEE plastics resins by type and recycle them together. This alternative could avoid sorting by type within plastic WEEE stream and consequently, re-

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duce recycling associated costs and promote it. It is well known, however, that direct melt blending of two or more thermoplastic resins causes phase segregation, low interfacial adhesion and consequently deteriorated mechanical properties (Utracki, 1991). Because of this, a specific compatibilization process is necessary to increases phase adhesion, reduce the interfacial tension, stabilizing morphology by inhibiting droplet coalescence and thus improving mechanical properties (Davis et al., 2000; Elmendorp et al., 1991; Wu, 1982).

An efficient blend compatibilization is the key to add value to mixed recycled plastics, but that is the challenge. Predominantly, plastics from e-scrap are complex composites with copolymers as a matrix and mineral particles (like calcium carbonate, carbon black, silica, etc.) as fillers (Hirayama et al., 2018; Vazquez and Barbosa, 2016; Buekens and Yang, 2014). These kind of materials also contain brominated substances as flame retardant additives making it even more difficult to recycle (Vazquez and Barbosa, 2016; Arnold et al., 2009). Within plastic WEEE stream, ABS (acrylonitrile-butadiene-styrene) and HIPS (high impact polystyrene) are two of the major components. ABS, is a block copolymer with good mechanical performance but its price is not as low as other plastics from e-scrap, like HIPS (Maris et al., 2015; Martinho et al., 2012). Both copolymers are styrenic resins which are very difficult to separate within plastic WEEE stream because of component similarities. In literature, there are several works of ABS/HIPS blends compatibilization studies on both, virgin and plastic from WEEE resins. Peyro Rasero et al. (2015) used SEBS (Styrene-Ethylene-Butylene-Styrene) as a compatibilizer in order to improve virgin ABS/HIPS blends ductility, as this material has similar polymeric segments. Results evidenced an increment in elongation at break with tensile strength decrement. Also, Arnold et al. (2010) claims that in ABS/HIPS blends obtained from virgin and plastic WEEE resins, final properties are deteriorated with respect to the initial materials and consequently, has poor added value. On the other hand, results obtained by Tarantilli et al. (2010) performing analysis on virgin ABS/HIPS blends shows an opposite behavior in which final properties were slightly improved.

Taking into account that plastic WEEE resins are composites as described above, conclusions obtained by working with virgin resins blends cannot be directly extrapolated to plastic blends from e-scrap. In this way, several authors obtained promising results working with ABS/HIPS blends from WEEE. Brennan et al., (2002) concluded that small amounts of ABS in HIPS and vice versa, improve tensile final mechanical properties respect on the corresponding properties of the major component in the blend. Moreover, de Souza et al. (2016) work evidenced that strength and stiffness of ABS/HIPS blends decrease with HIPS amount while ductility is improved. Also, their study indicates that smaller particle size of materials for injection molding, results in better mechanical performance. They conclude that ABS/HIPS recycled blends could lead to a decrease of processing costs.

Taking into account this background it is important to note that plastic WEEE used in some studies came from specific Electric and Electronic Equipment (EEE) sources (like informatic appliances) instead of a general plastic WEEE stream, that includes several EEE sources. General plastic stream from e-scrap is more conservative. As a consequence of using different kind of plastic sources, results are not always the same in studies of ABS/HIPS blends from plastic WEEE. All the studies have in common an important conclusion, however, which is to continue working these in order to find a sustainable solution to the recycling of these WEEE styrenic resins.

The aim of this work is to study direct melt blending of ABS with HIPS, both from WEEE, in order to develop recycling strategies for ABS avoiding plastic sorting by type within plastic WEEE stream using these blends as a direct replacement in same applications of single recycled ABS. In this sense, self and addition compatibilization of ABS/HIPS blends are studied. Self-compatibilization analysis allows to assess how much HIPS can be tolerated by ABS while conserving its properties (at least). Consequently, a wider error in plastic sorting by type within plastic WEEE stream can be accepted and then, the associated costs of plastic classification process can be reduced. Furthermore, the addition of 2 wt% Styrene-Butadiene-Styrene (SBS) as a compatibilizer of ABS/HIPS blends is also analyzed in order to improve phase adhesion and thus mechanical performance.

Two different proportion, 80/20 and 50/50, of ABS/HIPS are considered as base physical blends for both types of compatibilization studies. Compatibilization effectiveness is assessed through a comparative evaluation of thermal behavior, phase morphology and mechanical performance of all blends prepared and single ABS.

2. EXPERIMENTAL

2.1 Materials

ABS and HIPS from e-scrap were used as initial materials. They were kindly provided by Ecotécnica del Pilar S.R.L from Argentina. Each plastic sample used in this work were obtained by mixing 10 powder portions of 500 g from different places of a 25 Kg commercial bag in order to have a representative sample of each initial plastic e-scrap. A block copolymer of styrene-butadiene, SBS KIBITON® Q-Resin PB-5903 from CHI MEI Corporation with a MFI of 10 ml/10 min (ISO 1133 - 200°C×5KG) was used as compatibilizer.

2.2 Characterization

2.2.1 Blending

ABS/HIPS physical blends with 80/20 and 50/50 weight proportion were prepared in a batch mixer (Brabender Plastograph W50) under nitrogen atmosphere at 180°C and 30 rpm for 10 minutes. Physical blends with the addition of 2 wt% of SBS (compatibilized blends) were prepared under the same conditions . Batch mixing was used to simulate the expected processing in a twin-screw extruder as Brennan et al. (2002) suggested. In order to obtain a more representative and homogeneous blend, the following routine was performed for initial material and blends processing: a) each blend was replicated three times, b) resulting blends were milled and mixed and, c) with these chopped materials flexural specimens were obtained by compression molding. Table 1 summarizes the names and concentration of all blends prepared.

2.2.2 Glass transition

Glass transition temperatures (T_g) of initial materials and blends were determined by Modulated Differential Scanning Calorimetry (MDSC®) in a Discovery DSC from TA Instruments. An initial scanning was performed to erase thermal history followed by a cooling step and a second heating. Heat/cool/heat test were performed with a rate of 1°C/min from 60°C to 180°C. Second heating results were used to determine T_g values and perform the comparative analysis.

2.2.3 Mechanical properties

Flexural tests were performed at room temperature in the Universal Testing Machine Instron 3369. Test conditions and specimen dimensions were determined according to ASTM D790-03 standard for plastic. Flexural conditions were: rate of crosshead motion of 1.28 mm/min, support span of 48 mm and midspan deflection of 6.4 mm. Specimens of 100x10x3 mm were cut from plates prepared by compression molding at 180°C. Eight specimens for each sample were measured. Elastic modulus, ultimate strength, ductility and toughness were assessed from stress-strain curves.

2.2.4 Blend morphology

Blends morphology analysis was performed by Scanning Electron Microscopy (SEM) in a LEO EVO 40 XVP electron microscope, operated at 10 kV. Samples were cryofractured under liquid nitrogen, mounted on bronze stubs and then, coated with a gold layer (~ 30 Å) using an argon plasma metallizer (sputter coater PELCO 91000).

3. RESULTS AND DISCUSSION

In previous works it was demonstrated through High Resolution Modulated Thermogravimetric Analysis (Hi-Res™ MTGA™) that the ABS used in the present study contains 8.8 wt% of mineral fillers and an acrylonitrile/butadiene/styrene (AN/Bu/St) proportion of 32.5/30.6/28.1. Meanwhile, HIPS has 4.4 wt% of fillers with an AN/Bu/St proportion of 6.0/28.8/60.6, evidence that there is not neat separation of plastics WEEE by type, as it was expected. This fact also indicates that, plastics WEEE are composites with complex polymeric matrix. Additionally, it was determined that the total amount of bromine is under the maximum admissible content according European Directive about hazardous substances in WEEE (Vazquez and Barbosa, 2016 and 2017; European Union, 2011). This previous characterization is very important because mechanical behavior of ABS/HIPS blends will be determined by components proportion and their interaction with fillers.

Changes in glass transition temperature (T_g) give a first approximation of compatibilization effectiveness in polymer blends. In compatibilized blends, because of phase interaction enhancement, it is expected that T_g values of each component tends to converge (Utracki, 1991). T_g of ABS and HIPS from WEEE and all blends prepared are listed in Table 2 while the corresponding thermograms are presented in Figure 1.

It can be observed that A50/H50 blend has a T_g of 94.6°C, which value is between ABS and HIPS T_g's. This fact could indicate phase interaction improvement and consequently an effective self-compatibilization. On the other hand, A80/H20 blend has two T_g, one at 94.9°C and the other at 101.2°C. The first one is between glass transition temperatures of initial materials, probably indicating that styrene phases present in both matrix has interacted. However, the second T_g could evidence phase interaction decrement. It is probable that, during blending, initial AN domains suffer a coalescence and then, AN glass transition is manifesting by itself (Zhang et al., 2011). Despite this, it is not possible to ensure a non-effective compatibilization. For this reason, in order to corroborate claims made from thermal behavior, mechanical properties study along with morphology analysis is performed.

Mechanical behavior allows to better comprehend phase interaction and conclude respect to compatibilization efficiency. Changes in properties measured at high strain, like ultimate strength (σ_u) and ductility (ε_b) gives a measurement of compatibilization effectiveness. Toughness, the necessary energy for a material to break, is another property sensitive to compatibilization. Meanwhile, the Elastic Modulus (E), which gives an idea of material stiffness, is a zero-strain property and only depends on the internal structure of components and their relative proportions. For this reason, this property does not give any specific information about phase interaction (Utracki,

TABLE 1: Names and concentration of all blends prepared.

Name	ABS/HIPS (wt%/wt%)	SBS (wt%)
Physical Blends		
A80/H20	80/20	0
A50/H50	50/50	0
Compatibilized Blends		
A80/H20-2SBS	78.4/19.6	2
A50/H50-2SBS	49/49	2

TABLE 2: Glass transition temperatures of all blends prepared determined by MDSC (error < 5%).

Sample	Tg (°C)
HIPS	93.7
ABS	95.2
SBS	101.1
A80/H20	94.2/101.2
A50/H50	94.6
A80/H20-2SBS	*
A50/H50-2SBS	95.0

* *Transitions not clear*

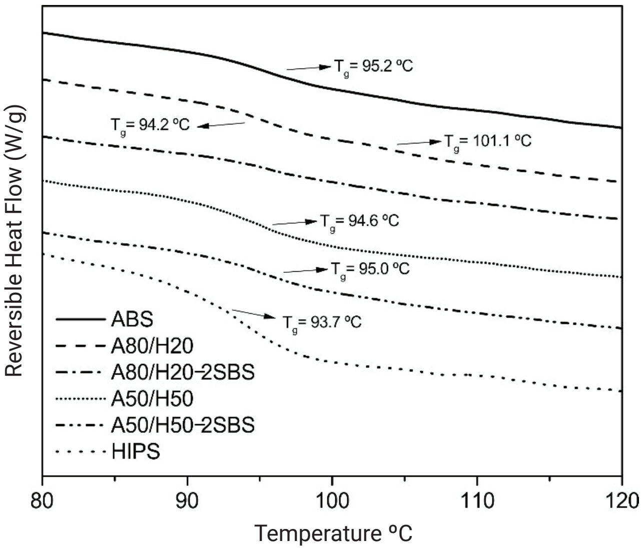

FIGURE 1: Thermograms (Exo-up) of ABS, HIPS and all blends prepared.

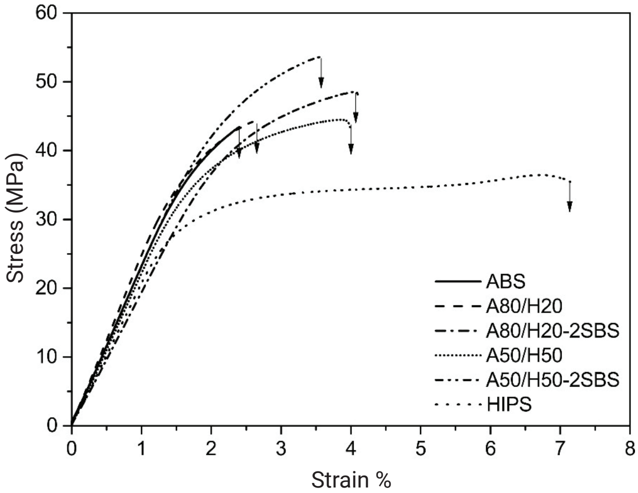

FIGURE 2: Stress-strain curves from flexural tests of ABS, HIPS and all blends prepared.

1991). Therefore, a thermal and mechanical behavior analysis combined with a morphology study allows to evaluate compatibilization effectiveness.

Figure 2 shows flexural stress-strain curves of ABS, HIPS and all blends prepared while mechanical properties (E, σ_u, ε_b and toughness) are detailed in Table 3. It can be observed that A80/H20 curve is very similar to ABS one. Also, an increase in stiffness and in less proportion in ductility and strength can be observed. This is evidence of the AN phase redistribution observation made from thermal analysis.

On the other hand, despite the coalescence of AN domains, it is possible to detect the strength and ductility increase in A80/H20 blend with respect to ABS which indicates phase adhesion improvement and therefore self-compatibilization effectiveness. This is consistent with morphological aspects of ABS and A80/H20 blend. Figure 3 shows cryofracture surface SEM micrograph of ABS while the corresponding A80/H20 blend SEM image is presented in Figure 4a. Both surfaces present sharpened edges, typical of a brittle fracture, but they are more pronounced in ABS. Also, it is possible to note that A80/H20 has bigger AN domains than ABS. This evaluation completely agrees with the coalescence of initial AN domains in ABS during processing resulting in strength improvement. Moreover, there are other small domains that correspond to rubbery ones (Bu phase) which form from rubber recoil after fracture therefore increasing ductility. In this sense, the obtained final properties of A80/H20 blends indicate that they could be used in housing manufacturing industry for Electrical and Electronic Equipment, instead of single ABS.

Regarding mechanical performance of A50/H50 blend (Figure 2 and Table 3), it is clear that fillers still govern its behavior at low deformations since stiffness did not change. The rubbery phase exists at higher deformations, however, resulting in lower stress in the A50/H50 blend than in ABS for same deformation values. It is possible that during blending rubbery phase had been relocated at the filler/matrix interphase reducing filler effect and increasing ductility (Figure 3). Changes observed in properties, particularly the notable improvement in ductility, evidenced by phase adhesion enhancement shown by glass transition analysis illustrate effective compatibilization. This hypothesis can be corroborated from the comparison between cryofracture surface SEM micrographs of A50/H50 (Figure

TABLE 3: Flexural mechanical properties (E, σ_u, ε_b and toughness) of ABS, HIPS, SBS and all blends prepared according ASTM D790-03.

Sample	E (MPa)	σ_u (MPa)	ε_b (%)	Toughness (J/m³) *
ABS [a,b]	2339 ± 29	43.4 ± 1.7	2.44 ± 0.16	0.63 ± 0.07
HIPS [a,b]	2068 ± 208	35.6 ± 1.6	6.96 ± 0.26	2.25 ± 0.16
SBS[b]	670±122	20.1±3.2	Not break	1.71 ± 0.22
A80/H20 [a,b]	2385 ± 113	43.7 ± 2.7	2.69 ± 0.14	0.67 ± 0.13
A50/H50 [b]	2148 ± 56	43.6 ± 2.3	4.03 ± 0.68	1.40 ± 0.18
A80/H20-2SBS [a]	1888 ± 57	47.6 ± 2.0	4.27 ± 0.61	1.36 ± 0.31
A50/H50-2SBS	2312 ± 52	53.1 ± 0.6	3.45 ± 0.25	2.26 ± 0.12

[a] *Vazquez and Barbosa, 2016.*
[b] *Vazquez and Barbosa, 2017.*
* *Toughness is the energy per volume necessary for sample break under flexural test. It is calculated as the area under stress-strain curves (up to break) obtained from this test.*

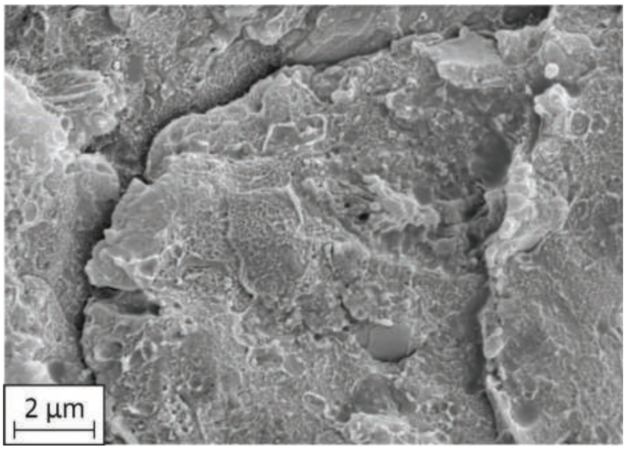

FIGURE 3: Cryofracture surface SEM micrograph of ABS (20000x).

4b) and ABS (Figure 3). Brittle fracture is less notable in A50/H50 blend in comparison with ABS. Also, it is possible to note that A50/H50 blend surface presents domains with fillers inside them agreeing with data from mechanical analysis. In this way, considering mechanical performance improvement with respect to the ABS, this blend can also be used as a replacement of ABS in EEE housing manufacturing industry. In addition, the higher ductility of A50/H50 blend allows use in a wider variety of applications.

In order to improve final properties and thus compatibilization effectiveness, the addition of 2 wt% of SBS as a compatibilizer to ABS/HIPS physical blends was analyzed. SBS is a copolymer with a typical rubbery behavior with low strength and stiffness. This copolymer was selected for having similar structure to ABS and HIPS (its molecules contain St and Bu). It is expected that St and Bu blocks interacts with those from ABS and HIPS improving their compatibility. Compatibilization effectiveness will depend on the relative ABS/HIPS proportion, among others factors like the presence of fillers.

Glass transition temperatures of compatibilized blends with SBS are listed in Table 2 while the corresponding thermograms are shown in Figure 1. In A80/H20-2SBS blend it was very difficult to assess a T_g because transitions were not well defined (Figure 1). Unfortunately, this does not give any evidence about phase interaction and therefore, it is impossible to ensure compatibilization effectiveness. On the other hand, A50/H50-2SBS blend shows a single T_g of 95.0°C, slightly higher than the A50/H50 one. The T_g of this blend presents an increment respect to A50/H50 physical blend one, as expected because of the addition of SBS which has a T_g of 101.1°C. This behavior indicates an improvement in phase interaction and a possible effective compatibilization. In order to assess SBS compatibilization effectiveness, mechanical properties and morphology of compatibilized blends were comparatively analyzed with ABS and physical blends.

Flexural mechanical behavior of compatibilized blends compared with physical ones and ABS are presented in Figure 2 while the assessed mechanical properties are listed in Table 3. A80/H20-2SBS stress-strain curve illustrates an increment in ductility and strength with respect to ABS and even to the physical blend. The increase in both properties indicates an improvement in phase adhesion which shows evidence of an effective compatibilization. On the other hand, A50/H50-2SBS present a notable increase in σu and εb respect to ABS, however, εb decrease respect to the physical blend although within the margin of error. This could be explained assuming that the addition of SBS produces the coalescence of rubbery domains and fillers unmasking. This last fact agrees with the higher stress in the whole range of deformation of A50/H50-2SBS blend in comparison with ABS and the corresponding physical blend. From mechanical behavior analysis it is possible to say that both SBS-compatibilized blends could be used as a direct replacement of separated recycled ABS and also of their corresponding physical blend, with improved resistance.

In order to corroborate all assumptions made from mechanical properties analysis, a morphological study was performed. Cryofractures surface SEM micrographs included in Figure 4 allows an analysis of compatibilized blends morphology. Regarding A80/H20-2SBS blend (Figure 4c), fragile fracture edges are evident just like in the A80/H20 blend (Figure 4a). Also, there are well distributed and dispersed domains with smaller sizes demonstrating an improvement in phase adhesion. This is consistent with the increment in ductility and strength (Table 3). On the other hand, the A50/H50-2SBS blend cryofracture surface (Figure 4d) shows more noticeable brittle fracture than the A50/H50 physical blend as illustrated by the sharpened edges. This morphology is consistent with a stiffer and more resistant material, agreeing with data from mechanical properties analysis. In addition, fillers are more evident and some of them are located inside rubbery domains. Also, domains and fillers are well distributed along fracture surfaces. These facts show rubbery phase redistribution by the addition of SBS, as shown from mechanical performance analysis.

4. CONCLUSIONS

Self-compatibilization as well as, addition compatibilization with 2 wt% SBS of ABS/HIPS physical blends from plastic WEEE stream were explored in order to design a sustainable strategy to recycle this kind of e-scrap with higher profitability. Self-compatibilization results indicate that ABS can tolerate up to 50 wt% of HIPS with an improvement in ductility and strength. Furthermore, the addition of 2 wt% of SBS to A50/H50 blend shows an increase of 22% and 40% in mechanical strength and ductility with respect to ABS. Also, the strength of A50/H50 physical blend was improved 22% with SBS addition, while ductility was slightly reduced. Furthermore, the addition of 2 wt% SBS to A80/H20 blend increase 9% strength and 65% ductility with respect to both ABS and physical blend.

These results are very promising for mixed plastic WEEE recycling. It was demonstrated that recycled ABS could be replaced in the housing manufacturing industry of Electrical and Electronic Equipment by materials resulting from ABS mixed with up to 50 wt% of HIPS. These materi-

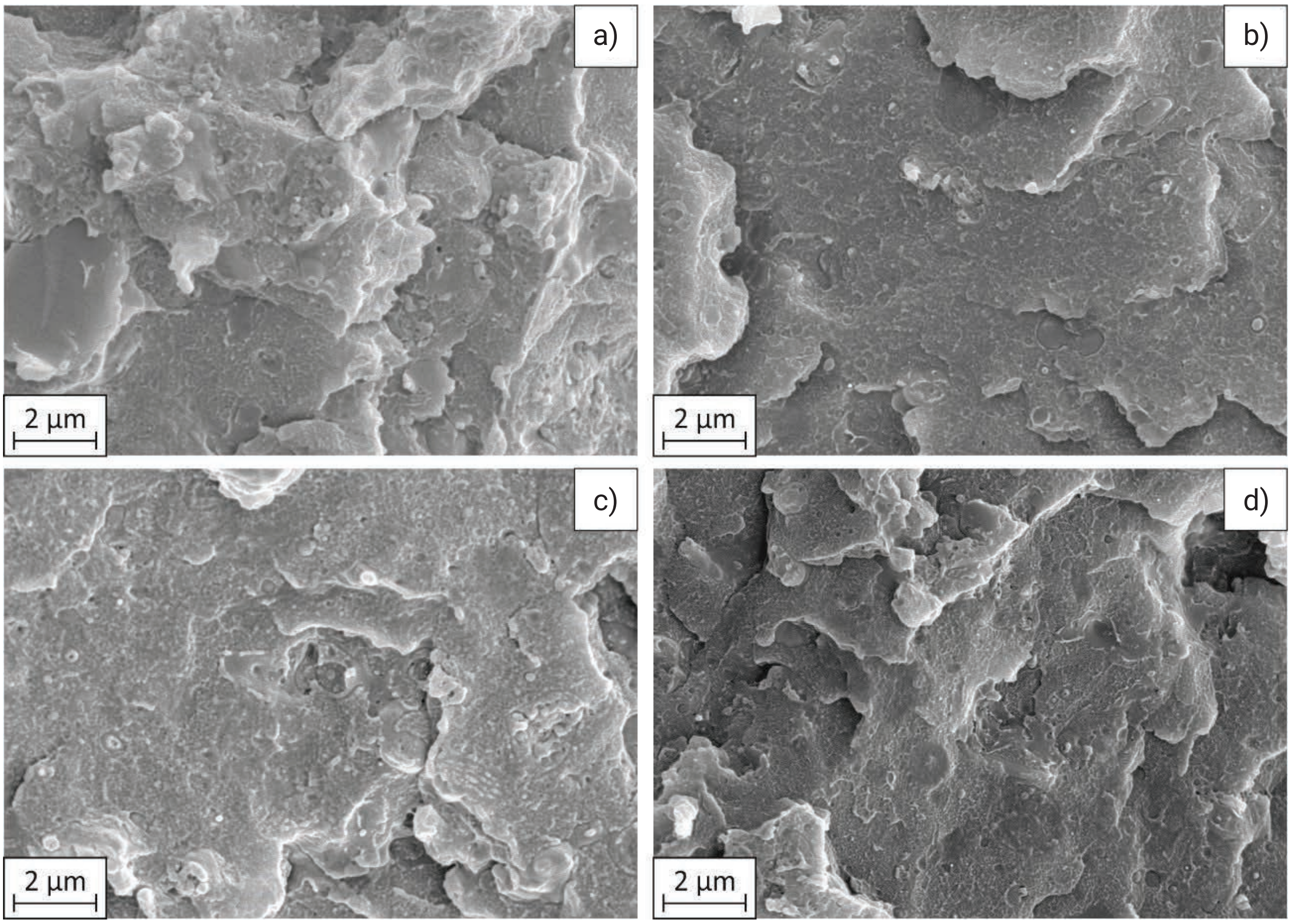

FIGURE 4: Cryofracture surface SEM micrographs (20000x) of a) A80/H20, b) A50/H50, c) A80/H20-2SBS and d) A50/H50-2SBS.

als can be even more improved with the addition of a small amount of a low cost compatibilizer. Clearly, the proposed strategies are effective because significant products are obtained through an easy and low-cost process: direct melt blending.

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SUSTAINABLE MANAGEMENT OF DEBRIS FROM THE L'AQUILA EARTHQUAKE: ENVIRONMENTAL STRATEGIES AND IMPACT ASSESSMENT

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ABSTRACT

This text deals with the management of construction rubble, in particular the debris generated by earthquakes and the relative environmental impacts of different choices. The contribution begins with a careful consideration of the cultural, legal and practical context, before presenting the results of an experience developed by the author following the L'Aquila earthquake of April 2009, involving a homogeneous area home to different municipalities. Differentiated handling and treatment options (centralised rather than localised) were evaluated in terms of their environmental impact, using an LCA methodology, and their technical feasibility. The study focused on identifying the potentialities and limits of various management strategies for the collection and reuse of debris caused by building collapses and demolition works. The results of the study make it possible to affirm that the criterion of the shortest distance appears to represent the better choice for the collection and treatment the demolition debris when an efficient network of inert material collection, treatment and recycling companies is in place. When this condition is met, this strategy offers various advantages by reducing distances and climate-altering emissions in addition to fostering new employment opportunities for the local community and businesses, in particular linked to the notion of the circular economy.

1. INTRODUCTION

Debris management is one of the key points of the highly complex system of governance supporting the reconstruction of historic centres damaged by earthquakes. This is demonstrated by on-going debate over the most appropriate ways to remove, transport, sort, store and recycle recovered materials.

From this point of view, criteria of environmental sustainability, including the reduction of new raw materials, waste, energy consumption and emissions related to the transportation of materials to and from landfills, and the maximum reuse of Construction and Demolition Waste (C&DW), represent best practices on which to base local regulatory instruments for the management of waste removal activities (Braungart M., McDonough W., 2002).

More generally, as concerns the sustainable management of C&DW, national and European rules and regulations provide some good starting points: Thematic Strategies for the Sustainable Use of Natural Resources (European Commission, COM (2005) 670), Public Procurement for a Better Environment (European Commission, COM (2008) 400), European Directive on Waste (European Commission, Directive 2008/98/CE), Italian rules on 30% use of recycled materials and products in public procurement (Italian Ministry of the Environment, 2003) and the recovery of 70% of C&DW by 2020 (Italian Government, 2010).

In addition, Italy's recent Public Procurement Code (Italian Government, 2016) issued mandatory "Minimal environmental criteria for the design and management of public administration buildings and sites" (Italian Ministry of the Environment, 2015).

Several authors have addressed the issue of sustainable C&DW management, examining aspects related to various ways of reusing waste as Secondary Raw Materials (SRM), the performance that can be obtained from building elements made from recycled aggregates (Ossa A. et al., 2016; Fan H.C. et al., 2016; Señas L. et al., 2016; Puthussery J.V. et al., 2017), as well as the environmental impacts of various types of collection, transportation and end of life scenarios (landfill, recycling, reuse), based on Life Cycle Assessments (LCA) (Butera S. et al., 2015; Silva A. et al., 2017).

More specifically, other authors have studied ways of managing earthquake debris, focusing on the usefulness of an effective operational plan (Lauritzen E.K., 1998; Brown

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C. et al., 2011; Sasao T., 2016; Askarizadeh, L. et al., 2017).

2. MATERIALS AND METHODS

Based on this knowledge framework, the study tried to understand the best strategy for the sustainable management of C&DW. The main objective of the study was to focus attention on the impacts on the environment as well as on local infrastructures and employment. Moreover, a secondary objective looked at developing a regulation that could be included in Reconstruction Plans (PdR) to guarantee the environmental sustainability of earthquake debris management activities.

With this objective in mind, the study applied the above-mentioned approach to a specific case study: the rubble generated by the earthquake that struck the city of L'Aquila in April 2009. The study focused on a homogeneous area, known as AREA 5, which includes the towns of Brittoli, Bussi sul Tirino, Civitella Casanova, Cugnoli, Montebello di Bertona, Ofena and Popoli. Beginning with a physical and infrastructural analysis of this territory, the study extend its attention to the rubble produced by this earthquake, compared to regional and national data. The analysis was also extended to the network of public and private companies authorised to treat C&DW materials, to understand their location and capacity to manage the debris produced by this earthquake. Finally, the study deepened the understanding of the debris management programme and legislative framework issued after the earthquake, to examine the feasibility of subsequent management hypotheses.

All of this information was used to define a set of possible rubble management schemes that were then compared. In particular, the centralised model indicated by the Reconstruction Commissioner, and a more widespread model based on the transfer of C&DW to the nearest authorised collection companies present in the area, selected based on the criterion of the shortest distance from the production site. This criterion was adopted as it allows for a reduction in the distances travelled by individual vehicles, travel times, traffic, loads and anticipated wear on road infrastructures.

The two models were compared in both physical and environmental terms. In the first case this involved assessing the distances travelled, and in the second assessing Greenhouse Gas (GHG) emissions created by transportation.

To simulate environmentally sound behaviour, it was considered important to make reference to the internationally codified Life Cycle Assessment (LCA) methodology (ISO 14040, 2006). In particular, considering the aim of the study, it was considered useful to adopt an expedited approach (screening LCA), more suitable to the development of a comparative assessment and the identification of improvement actions at a political stage. For this purpose, the study made use of LCA software (SIMAPRO, PRE' Consultant) and inventory data from literature or data banks associated with the software (ECOINVENT, PRE' Consultant). Moreover, it was considered appropriate to develop the assessments by relying on the IPCC 2001 (Climate Change) method, based on the characterisation of different GHG emissions according to their Global Warming Potential (GWP) values, as published by the Intergovernmental Panel on Climate Change (IPCC, 2001; Hischier R., Weidema B., 2010). At the same time, it was considered appropriate to refer to a time horizon of 100 years, in order to understand the middle-term effects of atmospheric lifetimes of different gases.

As known, GWPs are an index for estimating relative global warming contributions due to atmospheric emission of a particular GHG compared to the emission of carbon dioxide, used as a reference value with a GWP of 1. The measuring value is the carbon dioxide equivalent (CO_2eq) expressed in kg (Albritton D. L., Meira-Filho L. G., 2001).

3. RESULTS AND DISCUSSION

As mentioned, the study focused on the district known as the "L'Aquila Earthquake Crater", comprised of towns in varying states of destruction following the April 2009 earthquake.

To correctly appreciate the extent of the "rubble" produced by this event, it is worth noting that the Italian production of C&DW in that year was approximately 30,000,000 m^3 (Fischer C., Werge M., 2009) and that of the Abruzzo region in the range of 633,000 m^3 (Laraia R., 2011). The Institute for Building Technologies-Italian National Research Council (ITC-CNR) estimates the amount of rubble from the collapses and demolitions related to the L'Aquila earthquake at approximately 2,000,000-2,650,000 m^3, with some 1,125,640-1,305,617 m^3 in the town of L'Aquila alone (49% of the total). More specifically, AREA 5 had no particular problems related to the presence of rubble on public roadways to be removed, with the exception of the historic centre of Ofena, where the collapse of part of a housing block obstructed a street in this area. Nevertheless, the management of C&DW also concerns the repairing and/or reconstruction of buildings and spaces, both public (road works, infrastructures, pilot projects) and private, whose scale can only be evaluated following the presentation of Recovery and Consolidation projects.

Given such a huge volume to be dealt with in such a short time, and the necessity to ensure accessibility to and from construction sites in the historic centres, the companies operating in the province of L'Aquila authorised to treat C&DW materials provided an initial estimate of annual capacity of approximately 406,000 m^3/year. So, it appears evident that more temporary landfill, selection and treatments sites would be required, together with sites to be prepared for the storage of inert materials intended for reuse/recycling and possible disposal in non-dangerous and non-recyclable waste landfills after selection.

3.1 The Institutional Rubble Management Programme

To regulate flows of materials and debris, three different management sites (Figure 1) were identified by the reference legislative framework issued after the earthquake (Commissioner Delegated to Reconstruction, Decrees 18/2010, 49/2011, 51/2011; Italian Prime Minister, 2011):

- The former "EX TEGES" site in PAGANICA, for temporary rubble storage and selection, as well as the treat-

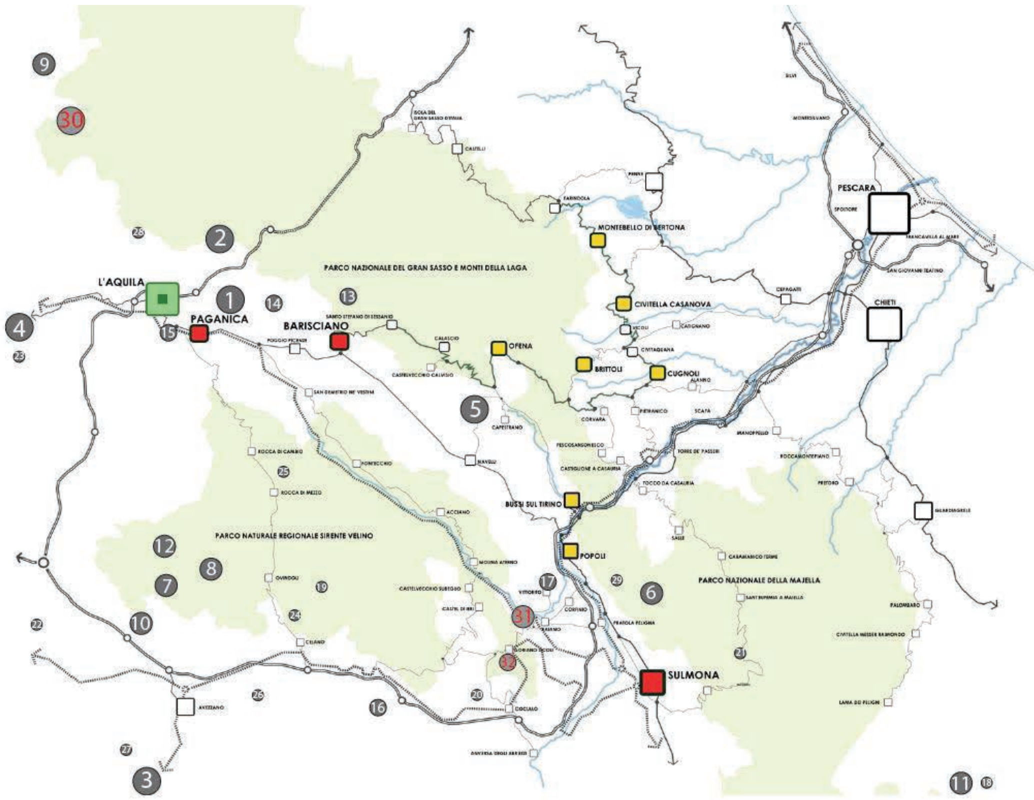

FIGURE 1: Temporary storage, treatment and disposal sites identified by Presidential Ordinance OPCM 3923/2011 et seq. (in red) and inert material collection and storage companies in the Province of L'Aquila, arranged by size (in grey).

ment, collection and storage of inert materials from collapses and demolitions for reuse;

- The TECHNOLOGICAL HUB in BARISCIANO, for temporary storage, treatment and disposal of waste from collapses, as well as from the demolition of damaged buildings;
- The "COGESA" LANDFILL in SULMONA, for the disposal of non-hazardous waste from the selection and treatment of rubble not suitable for recovery or reuse.

Furthermore, due to the high density of these historical centres, guaranteeing the ease of movement and temporary storage of rubble required individual municipalities to adopt specific Rubble Management Plans. These Plans identify one or more dedicated public areas, depending on the number of planned demolitions, easily accessible to collection and transportation vehicles. In these areas debris is arranged by homogeneous code categories, according to the Italian and European Waste Catalogue (EWC) (European Commission, 2000; Italian Government, 2006) and placed in metallic or fabric bins (i.e. big bags). To facilitate these activities and limit the public spaces required for temporary storage, private subjects must proceed with the selective demolition, separation and storage of C&DW on building sites.

Finally, the transportation of both public and private C&DW to treatment sites is assigned to the fire brigade, military or ASM SpA (L'Aquila's Multiservice Company), or to registered national environmental management companies. In particular, the public supply chain concerns rubble from collapsed buildings, demolitions of dangerous buildings or building works conducted by public authorities and are considered urban solid waste (Italian Government, 2009). Instead, private supply chain concerns rubble generated by private repair or reconstruction works, considered special waste to be managed under the normal collection and disposal programme.

3.2 The Impact Assessment of Organisational Choices

Without prejudice to the need to verify the environmental compatibility and legal compliance of temporary storage and treatment sites for rubble and C&D inert materials (activities outside the scope of this simulation), it is however possible to evaluate the physical and environmental impacts of organisational choices (Basti A., 2010), such as the selection of C&DW disposal sites.

The first hypothesis investigated is the "centralised model" indicated by the Reconstruction Commission and identified as SCENARIO 1. It contemplates the collection of C&DW at the Technological Hub in Barisciano dedicated to the temporary storage, treatment and disposal of waste from collapses and/or demolitions of damaged buildings. This solution presents some advantages owing to its location in the same location as the landfill for non-recoverable waste. In fact, this option means that this waste would not have to be transported to the "COGESA" LANDFILL in Sulmona for the disposal of the non-hazardous and/or non-recoverable or non-recyclable residues (Figure 2).

The second hypothesis investigated is the "widespread model" identified as SCENARIO 2, which contemplates the

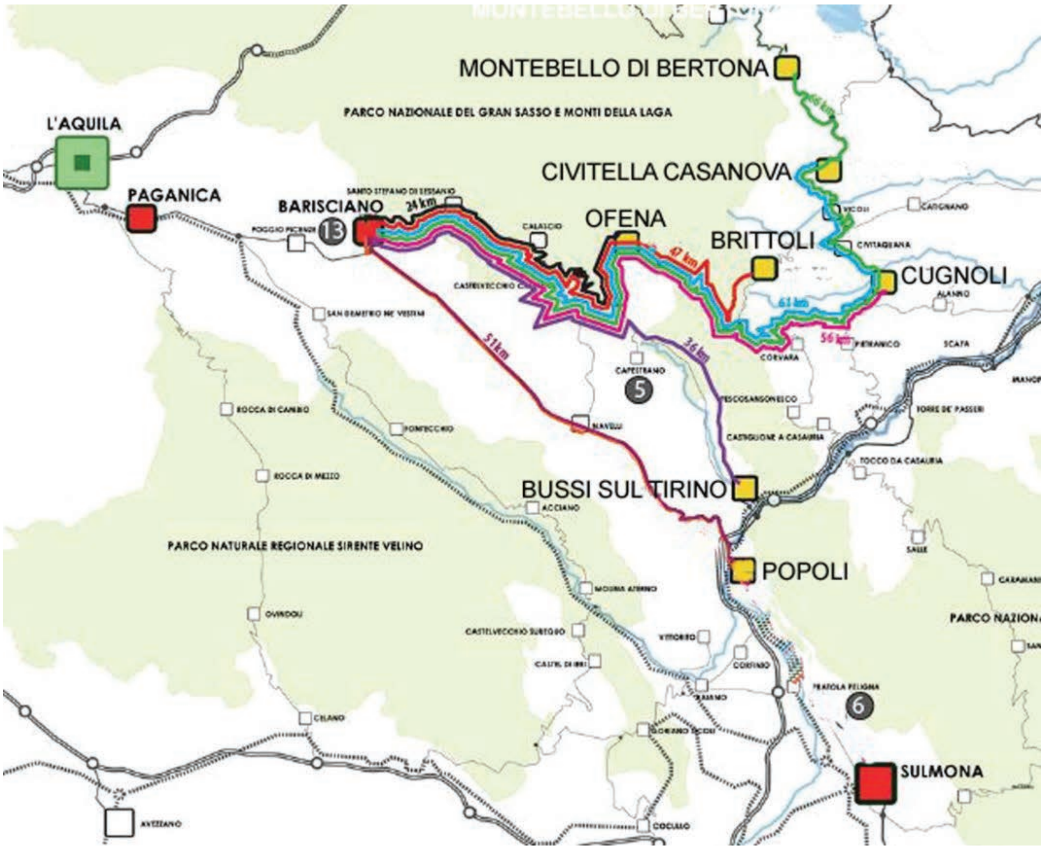

FIGURE 2: Identification of transportation flows for C&DW produced by reconstruction works in the towns of Homogeneous Area 5 (in yellow). SCENARIO 1: storage at the Technological Hub in Barisciano (AQ).

collection of C&DW by the nearest authorised collection companies present in the area, selected using the criterion of the shortest distance from the production site. Applying this criterion to the case study reveals that the towns at an altitude of 500 m above sea level (Brittoli, Civitella Casanova, Cugnoli, Montebello di Bertona and Ofena) and Bussi sul Tirino would benefit if they turned to Company 5 in Capestrano, while Popoli would benefit by turning to Company 6 in Pratola Peligna (Figure 3).

Comparing the two hypotheses by distance travelled, it is possible to note that in SCENARIO 1 the distance is approximately 284 km, while in SCENARIO 2 it is close to 151 km. However, to this last distance it is necessary to add the additional transport of non-recoverable waste from Capestrano to Barisciano (24 km) and from Pratola Peligna to Sulmona (12 km), for a total of 36 additional kilometres. For this reason the total distance of SCENARIO 2 is roughly 187 km. In light of this, it can be stated that in the second hypothesis the distance travelled by each vehicle could be reduced by about 133 km, or roughly 34%. This also means a consequent reduction in travel times, as well as loads and wear on the road infrastructures, already particularly tortuous and undersized.

When the two hypotheses are compared from an environmental point of view, with reference to the same vehicle (16t Lorry) and the same amount of waste transported (16t), it is possible to note that the GWP emissions generated in SCENARIO 1 are approximately 765 kg CO_2eq (kilograms of carbon dioxide equivalent), while in SCENARIO 2 this value is 504 kg CO_2eq. Consequently, it can be said that adopting the second hypothesis would reduce GWP emissions by an equal value of 34% (Figure 4).

Furthermore, regarding the vehicle selected for the study, it is useful to clarify that a specific comparison was made between some of the most common ones. Three vehicles were assessed: a 3.5 tonne van, a 16 tonne Lorry and a 21 tonne lorry traditionally used for waste collection. For each of these, the emitted kg of CO_2eq were considered, with reference to the unit value of 1t/1 km. The results demonstrate that the medium-large capacity vehicle (16t Lorry) is that with the lowest environmental impact, limiting transport-related emissions of GWP by roughly 75% (0.30 instead of 1.3 kg CO_2eq/t/km) compared to traditional vehicles (Figure 5). The results also confirm that the Reconstruction Commissioner's choice to use 16 tonne vehicles is consistent with the objective of limiting climate-altering emissions.

In conclusion, it must be noted that in order to arrive at the most economically, environmentally and socially sound choice, the above considerations must be placed in close relationship with other factors, such as the capacity of treatment structures, unit treatment costs and impacts on employment in specific territories. Factors that fall outside the scope of the present study.

3.3 Guidelines and Operating Suggestions

With reference to the purpose of the present study, i.e.

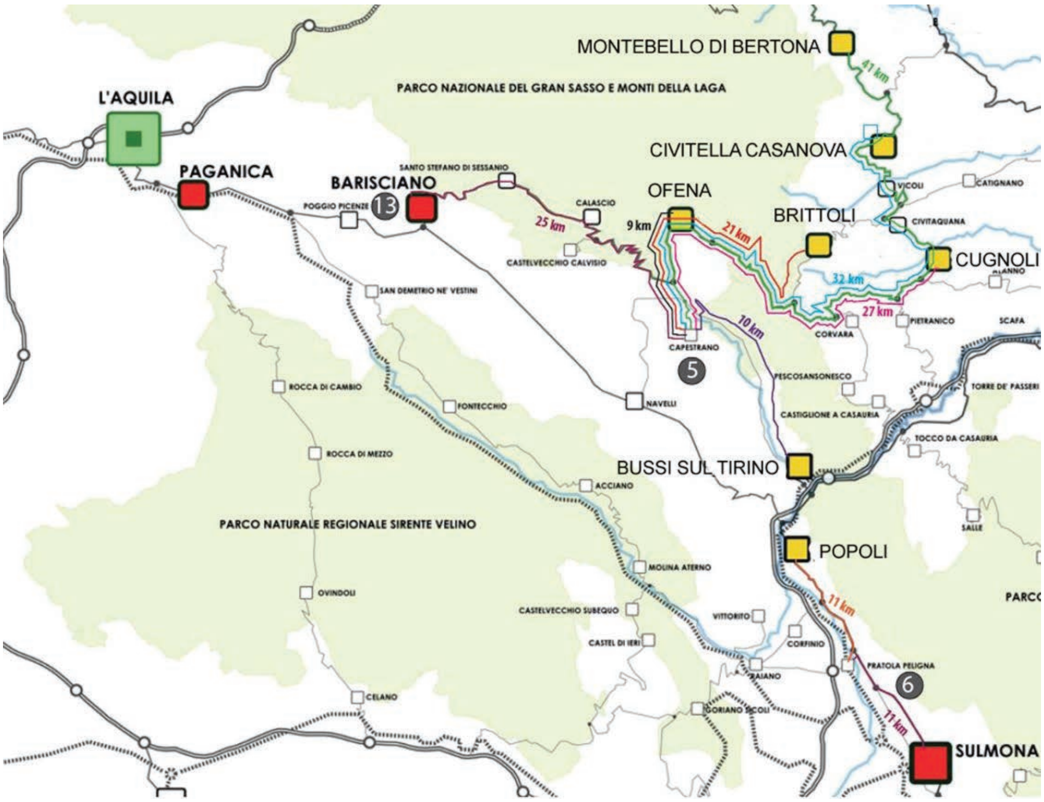

FIGURE 3: Identification of transportation flows for C&DW produced by reconstruction works in the towns of Homogeneous Area 5 (in yellow). SCENARIO 2: storage by collection companies, based on the shortest distance criterion.

the sustainable management of rubble generated from the April 2009 L'Aquila earthquake, it could be helpful point out that a Presidential Ordinance (OPCM, 2011) also establishes minimum objectives for the reuse and/or recycling of debris. Firstly, historic building elements (worked pantiles, bricks, ceramics, stones, wood and metal) are to be directly reused, which requires on-site storage. Additionally, the same document also calls for the use of recycled aggregates, preferably derived from rubble treatment, for all public works concerning the construction of buildings, environmental restoration, backfill, dikes and embankments. This also involves the procurement of recycled aggregates to be used for infrastructures, underground services, roadways and public squares, as well as drainage, underpinning, screeds and structural and non-structural concrete elements in public and private buildings, compatible with legal requirements (Ferrari G., Morotti A., 2008).

Many of these indications, however, are lacking a quantitative reference and method for evaluating their effective implementation. For this reason, the decision was taken to develop a set of exhaustive guidelines to be added to the Implementing Technical Standards (NTA) accompanying Reconstruction Plans (PdR). These guidelines integrate the provisions of the aforementioned Presidential Ordinances with such environmental sustainability criteria as:

- Reducing emissions caused by the transportation of materials to/from sites;
- Reducing waste sent to landfills;
- Recycling and reusing C&DW;
- Reducing the extraction of new raw materials.

One particular aspect of the procedure mentioned above involved the implementation of a C&DW management system comprised of the following phases:

- Direct reuse in the same buildings or building blocks of non-polluted soil and rocks, duly identified and certified under the procedures outlined in Ministerial Decree n. 161/2012 (Italian Ministry of the Environment, 2012). Recommended percentage: 75%;
- Direct reuse in the same buildings or building blocks of recovered elements with historic and architectural value, such as recovered pantiles, bricks, ceramics, worked stone, worked wood. Recommended percentage: 90%;
- Reuse of duly certified recycled inert aggregates (Italian Ministry of the Environment, 2005) in public works, in the design and construction of public infrastructures and environments for the creation of underground services, renovating public spaces, environmental restoration, backfill, dikes and embankments, to be procured in accordance with Abruzzo regional public works authorities. Recommended percentage 70%;
- Reuse of certified recycled inert aggregates in private construction projects for drainage, backfill, underpinning and non-structural concrete works, to be procured from temporary storage sites or collection facilities in Abruzzo, in accordance with the criterion of the shortest distance from the site, and, if distances are equal, the shortest travel time, encouraging the grouping of

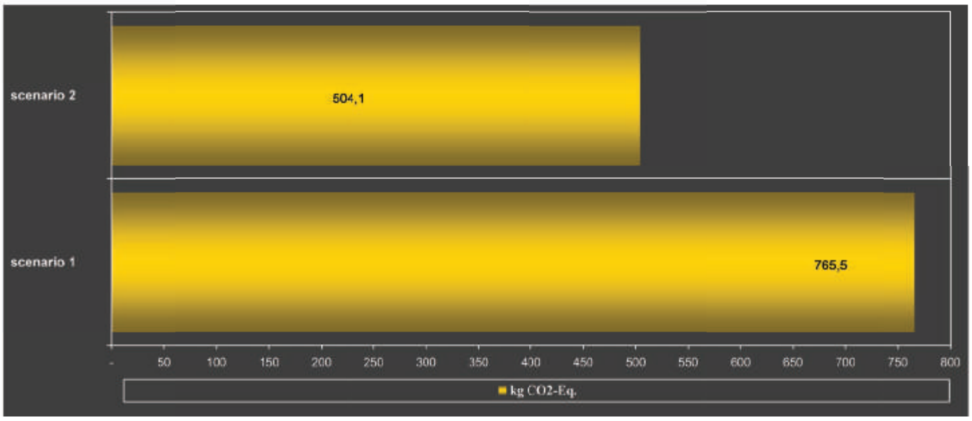

FIGURE 4: Simulation of kg CO²eq emissions potentially produced by the two scenarios, calculated based on vehicles carrying a maximum load of 16t, with a full outbound load and an empty return (SIMAPRO calculation code, ECOINVENT database, IPCC method 2001 GWP 100a).

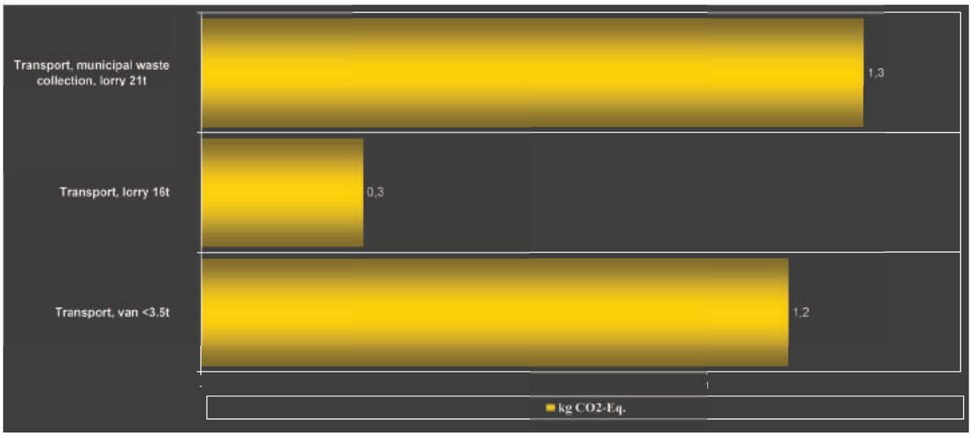

FIGURE 5: Simulation of kg CO_2eq emissions potentially produced by various vehicles, with reference to the unit value of 1 t/1 km (SIMAPRO calculation code, ECOINVENT database, IPCC 2001 method GWP 100a).

orders from several sites. Recommended percentage 50%;

- Accounting, as regards technical and economic assessments of the design and accounting of building repairs and/or construction aggregates for individual interventions, the quantity of C&DW selected and assigned to the collection and disposal system by homogeneous category (according to the EWC code), the quality and quantity of elements of historical and architectural interest that are recovered and reused, the recycling materials and inert aggregates used, all with respect to the total of the same type, indicating the original producing Site, Facility or Company.

However, several factors affect the effective implementation of these objectives, including:

- The traceability of materials of historic or architectural interest (lapidarium) for reuse in reconstruction sites. When these elements are so numerous as to prevent their easy storage in individual sites because they block or hinder normal construction activities, they would need to be categorised to link them with their original building; when they are not reused, they should be made available to other building sites in the same town to reduce the amount of time materials are kept in storage areas and to limit the procurement of new materials;
- The availability of recyclable inert materials for the organisation of reconstruction programmes involving the rebuilding of a significant portion of the buildings at the same time, in order to optimise materials procurement; in similar cases, quick access to the following types of information would be very useful:
  - The location of the treatment and recycling centre nearest the site;
  - The availability of recycled materials at the structure or at alternative structures. This type of organisation and information could help reduce flows of transportation traffic in areas with especially winding and narrow roads, making it easier to locate the nearest supplier. Another evolution of this organisation model could also facilitate the application of shared transportation systems, in which a single vehicle could serve several sites in a single trip. As well, as analysed above, the environmental advantages of using recycled aggregates also depends on the distance

between treatment plants and sites;
- The availability of specifications and unit prices. The qualification and certification of recycled aggregates (Italian Standardisation Body, 2008), as well as standardised pricing and the definition of adequate remuneration, would make it possible to compare and choose suppliers and encourage the use of recycled materials over new ones.

All issues will be specifically investigated in successive research activities.

4. CONCLUSIONS

Returning to a broader vision, it is important to highlight some aspects of the current supply system of quarry aggregates that, if re-examined in light of the considerations expressed above, could provide certain important food for thought.

A recent report by the Legambiente association (AA. VV., 2010) states that some 8,500,000 m^3 of gravel and sand are extracted in Abruzzo each year. This puts the region at the top of the list in Italy in terms of aggregates produced per capita. So, aggregates recovered from rubble could be used to produce recycled materials, reducing extraction for both reconstruction and other construction and infrastructure activities in the region. This could also have a significant effect on reducing the amount of rubble consigned to landfills, postponing the landfill depletion problem and reducing disposal costs.

Accordingly, reusing rubble could represent a small piece of the region's economy, when the reconstruction and socio-economic renewal of the areas covered by Reconstruction Plans propose an innovative system based on the knowledge economy linked to recycling.

From this point of view, it is possible to identify certain specific actions:

- The creation of a Rubble Management Observatory to collect and manage information related to the collection, treatment, selection and redistribution of recyclable materials for reconstruction, and more generally, the reconversion of the agglomerate procurement system at the regional level;
- The creation of a Research Centre for pre-competition design, experimentation and development of construction products and systems based on recycling Construction & Demolition Waste, available to public bodies and industries at the regional and national levels;
- The creation of a Business Incubator to train the people needed to manage the entrepreneurial and economic system related to the treatment and recycling of inert waste.

The opportunity to revamp the reconstruction and layout of urban outdoor areas and their technological underground works included in Reconstruction Plans could provide an opportunity to develop ad hoc design solutions and experimental applications. These pilots could fill the current knowledge gap between the experimental verification (in the laboratory) of residual uses of these recyclable materials and the effective services rendered by these materials when they are used in lieu of new materials for the development of long-term and reliable technological and building solutions.

As part of this vision, the planned creation and establishment of a Rubble Management Observatory to collect and monitor information related to the collection, treatment and distribution of recyclable materials for reconstruction, and more generally for the reconversion of the regional aggregate procurement system, could evolve to include the Provincial Waste Observatory planned in the Provincial Waste Management Plan for the monitoring, research and dissemination of good practices, support and development of actions disseminating Green Public Procurement processes, and study and research in the areas of recycling, eco-design, environmental audits and product lifecycles (Ma U., 2011).

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DETERMINING THE CLIMATE RELEVANCE OF REFUSE-DERIVED FUELS - VALIDITY OF LITERATURE-DERIVED VALUES IN COMPARISON TO ANALYSIS-DERIVED VALUES

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ABSTRACT

The adapted Balance Method (aBM) represents a cost efficient method for determining the fossil share in solid refuse-derived fuels (RDF). The method requires data on the elemental composition of the RDF on water-and-ash-free basis (TOX_{RDF}) and on the elemental composition of biogenic and fossil organic matter on water-and-ash-free basis present in the RDF (TOX_{Bio} and TOX_{Fos}). TOX_{Bio} and TOX_{Fos} generally need to be defined only once (e.g., before a routine application). After these data are known, only TOX_{RDF} needs to be determined analytically for any RDF sample in order to apply the aBM. As TOX_{Bio} and TOX_{Fos} are crucial input parameter for the aBM, the presented paper aims to assess the most suitable and practical way for their reliable determination. Within this study, 6 different solid RDFs are investigated and the aBM is applied, whereby the suitability of literature values is compared to own analysis data for TOX_{Bio} and TOX_{Fos}. The potential utilization of literature data could save the initial workload when applying the aBM and could make the method even more economical and practical compared to other methods. Altogether, seven aBM results are compared utilizing seven different methods for generating input values of TOX_{Bio} and TOX_{Fos}: using generic values, literature values only, analyses results only, or combinations of literature and analyses data. The study results suggest that the usage of analysis data together with information from literature is the best option to derive reliable input data (TOX_{Bio} and TOX_{Fos}) for the aBM (mean deviation from standardized methods of below 2%). The findings further suggest that there is a typical composition of the biogenic and fossil organic matter present in RDFs produced out of commercial and industrial waste. Thus, the initial workload for conducting RDF-specific analyses could be significantly reduced when some more data about different types of RDFs are collected (e.g in a database).

1. INTRODUCTION

Within the European Union, in 2012 already 34% of primary energy carriers in cement kilns are substituted by mixed wastes and refuse-derived fuels (RDF). In Austria an average substitution rate of 75% is reported (VÖZ, 2015). The utilization of alternative fuels in energy-intensive industry branches is a means to reduce the exploitation of natural resources, reduce costs and to lower fossil carbon dioxide emissions (Aranda Usón et al., 2013; Pomberger and Sarc, 2014). For the appraisal on the fossil CO_2-savings (and therewith connected economic savings for emission certificates), it is required to know on the one hand the carbon content originating from materials of fossil origin in the RDF and on the other hand the overall calorific value of the RDF. Both depend, among others (e.g., water content), on the shares of fossil and of biogenic materials. In solid RDFs fossil materials comprise mainly plastics and synthetic textiles; biogenic materials could be paper, natural fibers, wood, etc. The shares of the compounds in solid RDFs are usually not known and may vary significantly depending on the material used for RDF production. The material used is usually pre-treated municipal solid waste (MSW), commercial waste (CW) and industrial waste (IW). The composition thus, strongly depends on the input material and hence, on the collection and sorting schemes, and also on local industries. Further, the shares of fossil and biogenic materials present in the RDF are decided by

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the production process (processing units used) which is designed based on the target application (e.g., cement kiln, waste-to-energy plant) (Lorber et al., 2012; Nasrullah et al., 2014a,b; Nasrullah et al., 2015; Sarc et al., 2014).

To distinguish between CO_2-emissions originating from fossil (climate relevant) and biogenic (carbon neutral) sources in solid RDFs, different methods are available: The Selective Dissolution Method (SDM), the Radiocarbon Method (^{14}C-method), the Manual Sorting (MS), and the Balance Method (BM). The former three are described in the standard for solid recovered fuels EN 15440:2011; the Balance Method has recently been published in the standard ISO 18466:2016. The methods all have critical limitations, such as high and hardly quantifiable uncertainties (SDM, MS), high workload (MS), high chemical demand (SDM), high analytical costs (^{14}C-method) and solely post-combustion application (BM) (Jones et al., 2013; Schwarzböck et al., 2018; Schwarzböck et al., 2016b; Staber et al., 2008).

A practical, and cost-efficient approach to determine the climate-relevant (fossil) CO_2-emissions from the utilization of solid RDF is provided by the so-called "adapted Balance Method" (aBM) (Fellner et al., 2011). This method combines data about the elemental composition of the RDF – TOX_{RDF} (C, H, N, S, O-content on water-and-ash free basis) – with the RDF-specific elemental composition of pure biogenic and pure fossil organic matter – TOX_{Bio} and TOX_{Fos} (C, H, N, S, O-content on water-and-ash free basis). By setting up mass balances for each element, the fossil carbon content in the RDF can be derived. The aBM has recently been demonstrated to produce robust results which are in good agreement with the radiocarbon method (usually regarded as reference method) (Schwarzböck et al., 2016a; Schwarzböck et al., 2016b).

The necessary data for the application of the aBM are TOX_{RDF}, TOX_{Bio} and TOX_{Fos}. The determination of TOX_{RDF} always requires elemental analysis in the laboratory. To validate the method, in previous works TOX_{Bio} and TOX_{Fos} have also been analytically appraised for each investigated RDF (initially). This initial investigation requires sorting of the RDF into its different compounds (paper, wood, plastics, etc.) and elemental analyses of each compound. In order to reduce the initial workload and costs, available data on the elemental composition of different materials present in the RDF could be used. Potential sources for these data are literature values, information from industries or theoretical considerations (e.g., theoretical chemical structure of cellulose, polyethylene, etc.).

For example in Fellner et al. (2011) data for TOX_{Bio} and TOX_{Fos} in RDF are provided, which were collected from various literature sources and production statistics in Austria. Monte Carlo simulations were applied to derive TOX_{Bio} and TOX_{Fos} which should be valid for RDF. However, it is not clear if these values are generally valid for RDF or maybe only for a certain type of RDF (e.g., for RDF produced out of MSW).

In Schwarzböck et al. (2017) it is shown, that the probable range of TOX_{Bio} only varies to a minor extend as the different compounds (paper, wood, garden waste, natural fibers) have a very similar chemical composition on water-and ash-free basis. Values for TOX_{Fos} depend on the shares of the different polymers. Schwarzböck et al. (2017) assume that polyethylene and polypropylene represent the major part (around 80wt%) in mixed wastes which are fed into waste-to-energy plants in Austria. Kost (2001) conducted an extensive study on the characterization of waste compounds and collected literature data and own analysis data on the fractional and elemental composition of MSW.

Due to varying origins of the RDF production material (MSW, CW, IW), the composition of RDF may vary, depending on the characteristics of the waste catchment area (urban, rural, types of industry and businesses, etc.) and waste collection and treatment schemes.

The objective of the present paper is to evaluate the suitability of literature values for the application of the aBM in comparison to RDF-specific values (derived via extensive analyses) to determine the fossil carbon content in RDF. Therefore, the following parameter necessary to derive TOX_{Bio} and TOX_{Fos} in the RDF are varied, either based on literature or own analyses:

- Fossil and biogenic mass share in each compound k of the RDF (e.g., fossil share in composite materials); $x_{mF,k}$, $x_{mB,k}$
- Chemical composition of fossil and biogenic matter in each sorted compound k; TOX_k^{Fos} and TOX_k^{Bio}

Additionally the so generated results are compared to TOX_{Bio} and TOX_{Fos} given in Fellner et al. (2011) to estimate the validity of the previously collected literature data.

2. MATERIAL AND METHODS

2.1 Adapted Balance Method (aBM)

The aBM combines data about the elemental composition of the RDF (carbon, hydrogen, oxygen, nitrogen, and sulphur-content on water-and-ash free basis) with the theoretical composition of pure biogenic and fossil organic matter present in the RDF (C, H, O, N, S-content on water-and-ash free basis) – see Figure 1. For each element (C, H, O, N, S) a mass balance equation is set up which contains the unknown mass fractions of fossil (m_F) and biogenic matter (m_B). As illustrated in Figure 1, the elemental composition of the biogenic and fossil matter is significantly different. This allows the 5 balance equations to be solved by data reconciliation and the mass fractions of fossil and biogenic matter to be derived. By inserting the results (m_B, m_F) into the carbon balance, the fraction of fossil carbon as percentage of the total carbon can be determined.

The aBM requires the following input data:

1. Elemental composition of the water-and-ash-free RDF (TOX_{RDF}): Determination by CHNSO-analyses and ash content determination
2. Information on the elemental composition of the water-and-ash-free biogenic and fossil organic matter present in the RDF (TOX_{Bio}, TOX_{Fos}). The following options are available:
    - Collection of necessary information from literature
    - Determination by manual sorting together with CHNSO-analyses and ash content determination of the sorted compounds

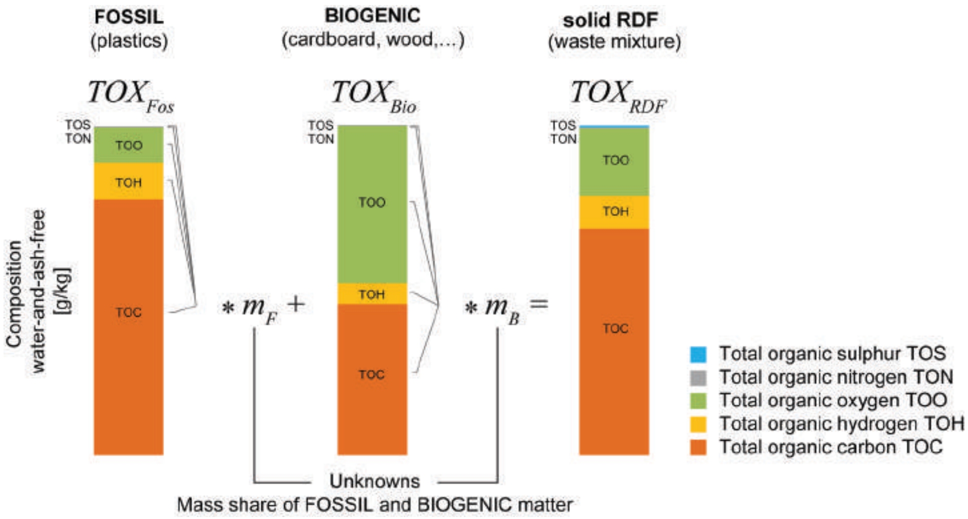

FIGURE 1: Schematic illustration of the adapted Balance Method, showing the relation between the elemental composition of the water-and-ash-free biogenic and fossil organic matter present in the RDF (TOX_{Bio}, TOX_{Fos}), the elemental composition of the water-and-ash-free RDF (TOX_{RDF}) and the respective mass shares of fossil and biogenic matter (mF and mB), which are unknown and determined by the aBM.

- Combination of information from literature and analyses

Within this study, the different options to gather the necessary information for the aBM are compared. The different variants used are described in Table 2.

2.2 Samples and sample preparation

Six different solid refuse-derived fuels (RDF) are subject of the study. Table 1 lists the different samples and their origin and indicates which methods to determine the fossil share, besides the aBM, are applied to the samples.

As only a few hundred milligrams of sample are necessary for the CHNSO-elemental analysis, the samples are comminuted down to a grain size of below 0.2 mm. The sample preparation is carried out in agreement with EN 15413:2011 and based on findings of Schwarzböck et al., 2016a. Two cutting mills (Essa CM 1000 and Retsch SM 2000), an ultracentrifugal mill (Retsch ZM 200), and a cryogenic mixer mill (Cryomill, Retsch) are used to produce representative analysis samples. Additionally riffle divider and rotary divider are used to reduce the sample mass between the grinding steps.

2.3 Analyses

2.3.1 CHNSO-elemental analysis

CHNSO elemental analysis is used to derive the elemental composition in the water-and-ash-free RDF samples, which is necessary in order to apply the aBM.

The water free (dried at 105°C for 24 hours) analysis samples undergo a CHNSO-elemental analysis using an Elementar Vario Macro instrument (for CHNS-analysis) and an Elementar Vario El instrument (for O-analysis, based on pyrolysis) (Elementar Analysensysteme GmbH, Hanau, Germany). At a combustion temperature of 1,150°C, the total carbon (TC), total hydrogen (TH), total nitrogen (TN), total sulphur (TS), and total oxygen (TO) content is determined according to EN 15407:2011. Additionally the ash content of each test sample is determined according to EN 15403:2011 and analyzed for its elemental composition to appraise the total inorganic content of C, H, N, S, and O.

TABLE 1: Investigated refuse-derived fuels and number of samples analyzed

Name	Origin / type of RDF	No of samples N	Sorting analysis	^{14}C-method[1]
Paper Reject	Residues of paper & board industry	15	✓	✓
RDF MSW+C&I	RDF prepared from pre-processed municipal solid waste and commercial & industrial waste (RDF production plant A)	8	✓	✓
RDF C&I (1)	RDF prepared from mainly commercial & industrial waste (RDF production plant A)	8	✓	✓
RDF C&I (2)	RDF prepared from mainly commercial & industrial waste (RDF production plant B)	3	✓	-
RDF C&I (3)	RDF prepared from mainly commercial & industrial waste (RDF production plant B)	2	✓	-
RDF C&I (4)	RDF prepared from mainly commercial & industrial waste (RDF production plant B)	3	✓	-

[1] *... tick indicates that Radiocarbon analyses (^{14}C-method) are additionally conducted according to EN 15440:2011 for selected samples to support the findings by aBM and sorting.*

Five measurements per sample, each of them comprising 40 mg of sample material, are carried out for C, H, N, and S. For the analysis of O, sample specimens of only 4 to 6 mg are used and 7 measurements per sample are conducted.

The measured values are converted according to Formula (1) in order to determine the elemental composition on a water-and-ash-free reference base.

$$TOX = \frac{TX - TIX * A}{(1 - A)} \tag{1}$$

TOX: total organic fraction of C, H, O, N and S in the water-and-ash-free sample [g/kg$_{waf}$]
TX: total fraction of C, H, O, N and S in the water free sample [g/kg$_{wf}$]
TIX: total fraction of C, H, O, N, and S in the water free ash (inorganic) [g/kg$_{wf}$]
A: ash content [kg/kg$_{wf}$]

The thereby obtained values for total organic carbon (TOC), total organic hydrogen (TOH), total organic nitrogen (TON), total organic sulphur (TOS), and total organic oxygen represent the input data required for the adapted Balance Method.

2.3.2 Radiocarbon method (^{14}C-method) to determine the share of fossil carbon present in RDF

The Radiocarbon method (^{14}C-method) is based on the distinctly different concentration of ^{14}C isotope in fossil carbon sources (where ^{14}C is completely decayed) and in modern (biogenic) carbon sources, which exhibit in a first approximation the current ^{14}C atmospheric levels. Thus, the ^{14}C-concentration in the emitted CO_2 when a waste mixture is combusted is directly proportional to the fraction of biogenic carbon in the combusted sample (Mohn et al., 2008). However, owing to anthropogenic activities the background level of ^{14}C levels in the atmosphere was altered, which complicates the calculation as it requires reference basis to be recalculated for each grow year of biomass (Fellner and Rechberger, 2009). Yet, the method is regarded as very reliable method for the determination of the biomass content in secondary fuels as it has the lowest analytical uncertainty (accelerator mass spectrometry <1% relative; Mohn et al., 2008).

Within the presented study, the ^{14}C-method is applied according to EN 15440:2011 utilizing accelerator mass spectrometry (AMS). Around 10 mg of 16 RDF samples at a grain size of < 0.2 mm are combusted at 900°C and the carbon isotope ^{14}C is separated from the stable carbon isotopes ^{12}C and ^{13}C before the mass analysis (for details see Szidat et al., 2014). The biogenic content in the sample is given as percent of modern carbon, allowing the fossil carbon to be calculated. A fossil mass share cannot be provided by this method.

2.3.3 Manual sorting

In order to determine the elemental composition of biogenic and fossil organic matter in the RDFs (see section 2.4), manual sorting analyses are conducted. Samples of each RDF are sorted into the following compounds:

- Paper
- Wood
- Plastics
- Composites & unrecognizable materials
- Textiles
- Rubber – only further considered for RDF C&I (B3); for other RDFs share is below 0.8wt%
- Fine fraction (around < 1-2 cm)
- Metals and inert materials – not further considered as this fraction is neither considered biogenic nor fossil

300 to 500 g (for Paper Reject up to 3.000 g) per sample are sorted. For Paper Reject and RDF from producer A – RDF MSW+C&I and RDF C&I (1) – a part of the fine fraction is further sorted into the above listed compounds. The so determined mass shares of the fine fraction are accounted for in order to estimate the total share of each compound in the RDF.

The results of the sorting analysis are on the one hand used to determine the respective input parameter of the aBM, and on the other hand the outcomes are utilized to estimate the overall share of the fossil mass present in the RDF. The latter are finally compared to the results generated by the aBM. However, some deviations from the procedure given in EN 15440:2011 for the manual sorting method are considered in order to generate more reliable results: the fine fraction is partly further sorted and the fossil and biogenic share in each compound are estimated analytically instead of relying on the information given in the standard. Yet, the manual sorting method is connected with high uncertainties, especially when high shares of mixed compounds (fine fraction, composites, textiles) are present in the RDF.

2.4 Determination of the elemental composition of biogenic and fossil organic matter present in RDF (TOX_{Bio}, TOX_{Fos})

Besides the elemental composition of the water-and-ash-free RDF (TOX_{RDF}), the application of the aBM requires information on the elemental composition of the water-and-ash-free biogenic and fossil organic matter (TOX_{Bio}, TOX_{Fos}). Seven different alternatives (Variants) are chosen in order to derive these data sets (description see Table 2).

The following assumptions are applied for all variants:

1. No analyses are available for the composition of fossil (synthetic) and biogenic textiles (cotton, wool) – $TOX_{textile}^{Fos}$ and TOX_{texile}^{Bio} Thus, for all applied variants, the chemical composition of textiles is based on data published by Kost (2001) and on theoretical considerations.
2. The biogenic share in the composite & unrecognizable materials represents paper and the fossil share represents plastics (all on water-and-ash-free basis).

The first variant (Variant L) only relies on the data given in Fellner et al. (2011), thus no analyses are considered for this option. For the other variants, at least information on the mass share of the different compounds ($x_{m,k}$) is necessary, which can be derived from manual sorting. Besides $x_{m,k}$, three further parameter are necessary to arrive at TOX_{Bio} and TOX_{Fos}: The share of fossil or biogenic matter in in each sorted compound ($x_{mF,k}$ or $x_{mB,k}$), and the chemical composition of fossil and biogenic matter of each sorted compound (TOX_k^{Fos} and TOX_k^{Bio}). In the second variant, these parameters are all derived from literature (Variant LL); where xmB,k is used as given in EN 15440:2011 and values for TOX_k^{Fos} and TOX_k^{Bio} are used as published by Kost (2001). In Variant AL and Variant LA either $x_{mB,k}$ or TOX_k^{Fos} and TOX_k^{Bio} are used as revealed by own analyses (see section 2.4.1. for details on how xmB,k is derived from analyses). Another option is to use all parameters, $x_{mB,k}$ and TOX_k^{Fos} and TOX_k^{Bio} as derived by own analyses (Variant AA).

However, assumption 2) implies that the composition of the sorted paper and plastics are representative for the composition of the paper and plastics contained in composite & unrecognizable materials. This is expected to be false, especially for plastics, as different polymers have a different chemical composition. Plastics in composite materials are presumably dominated by plastics foils (e.g., made out of polyethylene), but the sorted plastics also contain significant shares of other polymers (e.g., PET, foamed plastics, polyamide).

Thus, the sixth Variant AAL to determine TOX_{Bio}, TOX_{Fos} uses analyses results for $x_{mB,k}$, and considers both sources (analyses and literature) for TOX_k^{Fos} and TOX_k^{Bio}. For example, analysis results are used for the sorted plastics but the composition of the plastics within the fine fraction and the composite & unrecognizable materials fraction is assumed to correspond to a literature value for mixed plastics.

A sevenths variant (Variant Mean) is introduced which considers the mean value calculated of all values generated with "Variant AAL" of all RDF. This is to estimate the versatility and transferability of the generated data within this study.

Deviations for each variant from results of radiocarbon analyses or from manual sorting results are calculated as relative value (referred to the result of the standardized method). Negative deviations indicate that a lower value is found by aBM compared to the standardized method. Mean deviations between the aBM results and results of the radiocarbon/manual sorting are calculated by the mean aBM result of all RDFs compared to the mean radiocarbon/manual sorting result of all RDFs. Thus, no weighting according to the number of samples analyzed for the different RDFs is considered for the mean value.

Once one variant from Table 2 is chosen, the elemental composition of the water-and-ash-free biogenic and fossil organic matter ($TOX_{Bio,}$ TOX_{Fos}) is calculated. The equations (2) and (3) are given exemplary for the determination of the water-and-ash-free carbon content in fossil matter (TOC_{Fos}). First the relative share of each compound in the water-and-ash-free fossil matter is calculated by:

$$x_k^{Fos} = \frac{x_{m,k} * x_{mF,k}}{\sum_{k=1}^{k=n} x_{m,k} * x_{mF,k}} \tag{2}$$

- x_k^{Fos}: relative mass share of water-and-ash-free fossil compound k referred to total water-and-ash-free fossil matter in the RDF [kg_{waf}/kg_{waf}] (e.g., fossil matter is composed out of plastics, foamed plastics, synthetic fibers, plastics in compounds)
- $x_{m,k}$: relative mass share of water-and-ash-free compound k referred to total water-and-ash-free matter in the RDF [kg_{waf}/kg_{waf}] (e.g., share of plastics in the RDF)
- $x_{mF,k}$: relative mass share of water-and-ash-free fossil matter in compound k referred to total water-and-ash-free matter in compound k [kg_{waf}/kg_{waf}] (e.g., in pure plastics there is 100wt% of fossil matter if the sorting is precise; in composite materials there might be only 50wt% fossil and the rest biogenic).

Then the carbon content in the water-and-ash-free fossil matter is calculated by:

$$TOC_{Fos} = \frac{\sum_{k=1}^{k=n} TOC_k^{Fos} * x_k^{Fos}}{\sum_{k=1}^{k=n} x_k^{Fos}} \tag{3}$$

TOC_{Fos}: carbon content in the water-and-ash-free fossil matter [g/kg_{waf}]
TOC_k^{Fos}: carbon content in the water-and-ash-free fossil matter of compound k [g/kg_{waf}]

Formula (3) is likewise used for the determination of the hydrogen, oxygen, nitrogen, and sulphur content in the water-and-ash-free fossil matter (TOH_{Fos}, TOO_{Fos}, TON_{Fos}, TOS_{Fos}). Replacing the fossil share and fossil compounds by the biogenic share and biogenic compounds in formula (2)

TABLE 2: Variants within the study to derive information on the elemental composition of the water-and-ash-free biogenic and fossil organic matter present in the RDF (TOX_{Bio}, TOX_{Fos}; necessary input for aBM).

	Parameter	Mass share of compounds in RDF $x_{m,k}$	Fossil and biogenic mass share in each sorted compound $x_{mF,k}$ and $x_{mB,k}$ [1]		Chemical composition of fossil and biogenic matter in each sorted compound TOX_k^{Fos} and TOX_k^{Bio}	
				Source/analysis method		Source/analysis method
1	**Variant L**	not necessary	Literature	-	Literature	TOX_{Bio} and TOX_{Fos} given in Fellner et al. (2011)
2	**Variant LL**	Manual Sorting	Literature	EN 15440:2011	Literature	Kost (2001)
3	**Variant AL**	Manual Sorting	Analyses	aBM, SDM, ^{14}C	Literature	Kost (2001)
4	**Variant LA**	Manual Sorting	Literature	EN 15440:2011	Analyses	CHNSO-analyses + ash content
5	**Variant AA**	Manual Sorting	Analyses	aBM, SDM, ^{14}C	Analyses	CHNSO-analyses + ash content
6	**Variant AAL**	Manual Sorting	Analyses	aBM, SDM, ^{14}C	Analyses & Literature values	CHNSO-analyses + ash content; Kost (2001)
7	**Variant Mean**	mean of TOX_{Bio} and TOX_{Fos} of all 6 RDFs determined by **Variant AAL**				

[1] ... *Only $x_{mB,k}$ or $x_{mF,k}$ needs to be appraised as the other can be derived by $x_{mB,k} = 1 - x_{mF,k}$*

gives the relative share of each biogenic compound within the water-and-ash-free biogenic matter. Subsequently formula (3) can be applied to determine TOX_{Bio} in the same manner as TOX_{Fos}. Results are then the carbon, hydrogen, oxygen, nitrogen, and sulphur content in the water-and-ash-free biogenic matter (TOC_{Bio}, TOH_{Bio}, TOO_{Bio}, TON_{Bio}, TOS_{Bio}).

According to Table 2, different data sets for TOXBio, TOX_{Fos} are generated; using different sources of data (literature values and/or analysis results). Evaluations by means of the aBM are conducted for each RDF sample by combining the analysis results on the elemental composition of the water-and-ash-free RDF (TOX_{RDF}) with each generated data set of TOX_{Bio} and TOX_{Fos}. Thus, for each RDF seven different results on the fossil share are obtained.

In order to appraise if the results are in the range of the true value, they are compared to radiocarbon analyses (regarded as method with highest accuracy). For RDF from producer B – RDF C&I (2), (3), and (4) – no radiocarbon analyses are available. Thus, the results are compared to the sorting results only.

2.5 Estimation of mass share of fossil and biogenic matter in each sorted compound k ($x_{mF,k}$ and $x_{mB,k}$)

For each sorted compound, a fossil and biogenic share needs to be allocated. To do this, the standard EN 15440:2011 provides some guiding values. However, these generic values are assumed to not be valid for every RDF. For example, the fossil content of composite materials and of textiles can vary, depending on the type of RDF. Textiles from industry usually contain a much higher share of synthetic fibers than natural fibers. Similarly, the plastic content in the fine fraction can easily vary between RDFs. In addition, the sorted compounds of alleged "pure" materials (like plastics or paper) may be contaminated by other compounds as the sorting cannot be conducted precisely and is also prone to subjective assessments.

Thus, within this study, the fossil and biogenic mass shares for each sorted compound are appraised. This is done by utilizing the analysis results of the elemental composition of each compound and conducting a preliminary evaluation using the aBM. TOX_{Bio} and TOX_{Fos} for these preliminary aBM evaluations stem from literature (Kost, 2011, Fellner et al., 2011) and theoretical considerations (e.g., composition of cotton and wool to derive TOX_{Bio} in the textiles).

In addition, the radiocarbon method (according to EN 15440:2011) is conducted for selected compounds to support and confirm the aBM results.

3. RESULTS AND DISCUSSION

3.1 Composition of RDFs (sorting results) and assignment of fossil share for each compound

Table 3 presents the result of the sorting analyses of each investigated RDF. A share of plastics of at least 50wt% is found for all RDFs except Paper Reject (41wt%). The fine fraction has a considerable share with 24wt% to 65wt%; with the highest being found in Paper Reject. Table 4 shows that the fossil share (plastics content) in the fine fraction can vary significantly between the RDFs (between 40 and 78wt%). Thus, additional sorting of the this fraction are carried out for 3 RDFs to obtain more accurate results on the fractional composition of the RDFs.

The fine fraction, the composition of the composite & unrecognizable materials is uncertain, which represents between 3 and 22wt%. Fossil shares within this compound in the range of 32 and 66wt% are found (Table 4).

3.2 Elemental composition of sorted RDF compounds

Table 5 presents the average elemental composition on water-and-ash-free basis of the different compounds found by analyses within this study. Comparing the results to values published in Kost (2001) shows that they are generally in a similar range. The small differences in TOC, TOH, and TOO in plastics indicate the dependency on the

TABLE 3: Sorting results for the investigated RDFs – mass shares of compounds in each RDF.

	Mass share $x_{m,k}$ [wt% dry]								
	Paper Reject		RDF MSW+C&I		RDF C&I (1)		RDF C&I (2)	RDF C&I (3)	RDF C&I (4)
		incl. sorting of fine fraction		incl. sorting of fine fraction		incl. sorting of fine fraction			
Paper	5.9%	37.4%	4.1%	5.1%	2.8%	6.9%	5.0%	5.1%	8.5%
Wood	1.6%	2.5%	2.2%	1.9%	0.1%	0.2%	0.4%	0.3%%	0.2%
Plastics	20.4%	41.0%	47.1%	56.5%	32.5%	49.6%	54.0%	53.3%	50.1%
Composite & unrecognizable materials	3.0%	15.0%	8.4%	11.7%	21.8%	40.7%	9.4%	4.4%	4.7%
Textiles	1.1%	1.1%	12.1%	22.6%	0.7%	1.4%	1.3%	0.9%	7.3%
Rubber	0.7%	0.7%	0.8%	0.9%	0.1%	0.4%	0.4%	0.3%	1.2%
Fine fraction (around <1-2 cm)	65.1%	-	24.1%	-	41.3%	-	28.4%	32.9%	27.1%
Metals & inert materials [1]	2.2%	2.3%	1.2%	1.3%	0.7%	0.8%	1.1%	2.8%	0.9%

[1] *Metals & inert materials are not further considered in the study (these compounds are neither of fossil nor of biogenic origin).*

TABLE 4: Allocated fossil share in each sorted compound in the RDF.

	Allocated fossil share $x_{mF,k}$ [wt%, dry]						
	Paper Reject	RDF MSW+C&I	RDF C&I (1)	RDF C&I (2)	RDF C&I (3)	RDF C&I (4)	EN 15440:2011 [1]
Paper	7%	8%	7%	7%	6%	4%	0%
Wood	1%	1%	1%	5%	2%	4%	0%
Plastics	95%	92%	96%	96%	98%	98%	100%
Composite & unrecognizable materials	66%	54%	42%	40%	32%	56%	NA [2]
Textiles	35%	55%	55%	40%	45%	98%	50%
Rubber	NA	NA	NA	84%	84%	75%	80%
Fine fraction (around <1-2 cm)	40%	62%	55%	72%	78%	77%	50%

[1] Used for Variant LL and Variant LA
[2] 50% were used as there is no guiding value in the EN 15440:2011

TABLE 5: Elemental composition of different waste compounds analyzed in this study, compared to data published in Kost (2001).

	Elemental composition (water-and-ash-free)					
	N	TOC [g/kg$_{waf}$]	TOH [g/kg$_{waf}$]	TOO [g/kg$_{waf}$]	TON [g/kg$_{waf}$]	TOS [g/kg$_{waf}$]
Paper						
this study	21	476 ± 3	65 ± 1	497 ± 4	3.6 ± 0.5	5.1 ± 1.0
Kost (2001) [1]	43-62	467 ± 6	65 ± 1	443 ± 9	2 ± 1	1 ± 0
Wood						
this study	10	498 ± 2	62 ± 1	453 ± 2	6.3 ± 0.5	7.6 ± 1.2
Kost (2001) [1]	21-30	494 ± 2	60 ± 1	443 ± 4	1 ± 0	0 ± 0
Plastics (mixed)						
this study	24	771 ± 6	109 ± 1	103 ± 2	7.0 ± 0.5	8.4 ± 0.5
Kost (2001)1	11-15	790 ± 29	130 ± 9	30 ± 32	2 ± 2	1 ± 2
Composite & unrecognizable materials						
this study	16	620 ± 13	88 ± 2	286 ± 5	2.9 ± 0.5	9.5 ± 1.0
Kost (2001) [2]	1-8	560 ± 16	80 ± 3	320 ± 4	11 ±	4 ±
Textiles						
this study	15	564 ± 4	65 ± 1	355 ± 5	13.5 ± 0.6	11.2 ± 1.5
Kost (2001)	4-11	510 ± 16	70 ± 3	360 ± 7	27 ± 11	4 ± 2
Rubber						
this study	6-9	733 ± 4	90 ± 1	78 ± 4	14.6 ± 1.2	24.9 ± 2.7
Kost (2001)	4-5	860 ± 13	80 ± 9	60 ± 21	4 ± 1	17 ± 4

N Number of analyzed samples / number of literature values collected (by Kost, 2001).
[1] Used for Variant LL and Variant AL.
[2] Composite packaging.

plastics composition (shares of polyethylene, polyethylene terephthalate, polyurethane, etc.).

Further, differences of the analyzed values compared to the literature values are noticeable for rubber and textiles. For textiles this could be explained by high shares of synthetic fibers (with higher TOC, TOH and lower TOO content) which were observed for some RDF samples within this study.

Some differences in the oxygen content (TOO) are noticeable for almost all compounds. One factor explaining this phenomenon is assumed to be the chosen determination method. In literature, the O-content is often derived by subtracting all other elements from 1000 g/kg. Within this study, O-analyses are conducted by means of pyrolysis. It is assumed that the subtracting method holds higher uncertainties than actual analyses and can easily lead to different or wrong estimations of the O-content because the analytical uncertainty of the other elements is not considered.

3.3 Share of fossil carbon present in RDFs - aBM results based on literature values and based on analysis values

Figure 2 shows the outcomes of the aBM, namely the share of fossil carbon for each investigated RDF. In particular, the different results depending on the variant utilized to derive the input data (elemental composition of biogenic and fossil organic matter TOX_{Bio} and TOX_{Fos}) are presented.

The results for the fossil carbon share (which represents also the share of fossil CO_2-emissions when the RDF are thermally utilized) vary only in a small range for Paper Reject (52-59% fossil carbon); no significant difference can be detected when comparing results generated with literature values, analysis values or a combination of both (Variant LL, AL, LA, AA, AAL). This is different for the other RDFs: The determined fossil carbon shares range from 72 to 92%. A significant difference can be observed for values where the elemental composition of biogenic and fossil organic matter in the compounds (TOX_k^{Fos} and TOX_k^{Bio}) is derived from literature values only (blue bars in Figure 2) and where these values are determined by own analyses (green bars in Figure 2). The differences between the variants where the fossil mass share per compound ($x_{mF,k}$) was varied (according to EN 15440:2011 or own appraisal) appear to be minor (differences between Variant LL and AL and difference between Variant LA and AA). This suggests that the choice of the chemical composition of the different compounds (TOX_k^{Fos} and TOX_k^{Bio}) has a much higher influence on the aBM result than the estimated fossil and biogenic mass share in each sorted compound ($x_{mF,k}$ and $x_{mB,k}$). The small differences between Variant LL and AL and between Variant LA and AA are mainly caused by the allegedly false estimation of the fossil mass share in the fine fraction and textiles (as given in EN 15440:2011) and of composite & unrecognizable materials (50% were estimated) (see Table 4). If the shares of these "non-pure" compounds are high, then the result according to the standard are expected to be more prone to errors.

From Figure 2 it can be seen, that the estimated fossil shares are lowest when literature values are used for TOX_k^{Fos} and TOX_k^{Bio} (blue bars, Variant LL and AL). This implies that the generated TOX_{Bio} is closer to the measured values in the RDFs (TOX_{RDF}) than TOX_{Fos} for Variant LL and AL. This, in turn indicates that the actual carbon and hydrogen content in the TOX_{Fos} is lower than expected from literature values; and the oxygen content is higher. As the TOX_{Fos} depends on the different shares of polymers and their respective chemical composition, it can be expected that the polyethylene and polypropylene shares in the investigated RDF is lower than typically found in MSW. They may contain slightly higher shares of plastics with lower carbon and hydrogen and higher oxygen contents (e.g., polyamide, polyethylene terephthalate, or polyurethane). For example, in RDF MSW+C&I significant shares of foamed polymers could be observed, which are expected to account for around 16 to 26wt%. In RDF C&I (2), (3), and (4) shares of foamed plastics between 2 to 3wt% could be assessed by sorting, which corresponds to 4 to 5wt% in the plastic compound sorted out.

The fact that there are much smaller differences for Paper Reject (than for the other RDFs) when comparing results obtained using different TOX_k^{Fos} and TOX_k^{Bio} (green bars versus blue bars in Figure 2), suggests that the polymer composition in the Paper Reject is similar to the one typically found in MSW.

For all RDFs prepared from mainly C&I (4 of the 6 RDFs investigated), the results generated using values for TOX_{Bio} and TOX_{Fos} from Fellner et al. (2011) (Variant L, grey bar in Figure 2) are close to the results generated when using TOX_k^{Fos} and TOX_k^{Bio} from own analyses or combining own analyses with literature values (Variant LA, AA, and AAL).

The Variant AAL delivers aBM results on the fossil car-

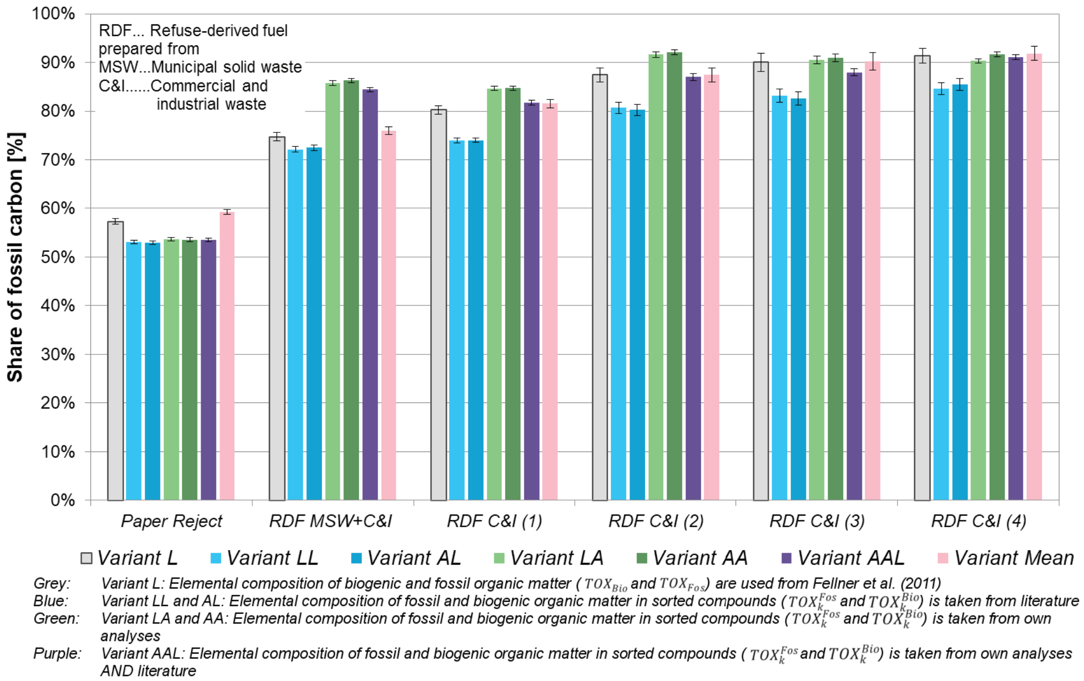

FIGURE 2: Share of fossil carbon for the different RDFs, determined by utilizing the aBM with different input values of TOX_{Fos} and TOX_{Bio} (estimated by using 7 different variants as listed in Table 2).

bon share which are slightly below the ones when using TOX_k^{Fos} and TOX_k^{Bio} from own analyses (Variant LA and AA, green bars), but are in most cases considerably above the ones when using literature values (Variant LL and AL). The Variant Mean, which uses the mean values of all investigated RDFs for TOX_{Bio} and TOX_{Fos} from Variant AAL, leads to results in a similar range as Variant AAL except for Paper Reject and RDF MSW+C&I.

In general, by trend a slightly higher fossil share can be detected for RDFs prepared from C&I (RDF C&I), compared to the RDF containing also compounds of MSW (RDF MSW+C&I).

3.4 Comparison of aBM results to alternative methods

In order to decide on the most probable result determined by aBM (to identify which variant delivers the lowest deviation from a comparable value), the outcomes are compared to values generated with alternative analysis methods.

Figure 3 shows the deviation of the aBM results from values generated by using the Radiocarbon method (^{14}C-method) for 3 of the investigated RDFs. Figure 4 presents the deviation of the aBM results from the manual sorting results for the other 3 RDFs (no Radiocarbon analyses are available for these RDFs). However, it must be kept in mind that the manual sorting results are connected with a significantly higher uncertainty than the results of Radiocarbon analyses. Even though the sorting is conducted very painstakingly, some errors (such as subjectiveness when sorting, wrong estimation of the composition of fine fraction or composite & unrecognizable materials) are hardly quantifiable.

For Paper Reject and RDF MSW+C&I using generic values or mean values for TOX_{Fos} and TOX_{Bio} (Variant L and Variant M) seems not to be suitable (+8 to +12% deviation for Paper Reject and −8 to −10% deviation for RDF MSW+C&I compared to Radiocarbon analyses).

For the other RDFs, which are the ones mainly produced out of C&I, a relatively good agreement of the aBM results to the results of alternative methods is observed when using Variant L or Variant M. Low deviations between -2.9% and +3.2% from results of alternative methods are found. This indicates that the usage of generic values is suitable and RDF produced out of C&I have typical compositions regarding the shares of the different synthetic polymers (within the fossil fraction) and regarding the composition within the biogenic materials (shares of paper, wood, natural fibers, etc.).

The results for all other variants than Variant L and Variants M, are in a close range when regarding Paper Reject. As Variant LL, AL, LA, AA, and AAL also show a low deviation from the alleged true value (0.5 to1.6%), it can be assumed that the most decisive factor when generating TOX_{Fos} and TOX_{Bio} is the usage of the actual shares of compounds present in Paper Reject. This means that an initial sorting campaign before applying the aBM to Paper Reject might results in a considerably higher reliability of the results.

A somehow different observation is made for the other RDFs: Significant underestimations of the fossil share can be observed when Variant LL and LA are used, where the information of the chemical composition of biogenic and fossil organic matter in the compounds stem from literature (mean deviation −9 to −13%). But the usage of the analyzed chemical composition of the compounds (Variant LA) or only analysis results (Variant AA) seems to rather overestimate the fossil carbon share for all RDFs. For example, for RDF C&I (2) the relative deviation when using Variant AA is considerably higher than when using Variant L where the input values for TOX_{Fos} and TOX_{Bio} are used without any analysis (values from Fellner et al., 2011). A possible explanation for this phenomenon is that the fossil and biogenic matter in the fine fraction, which represents

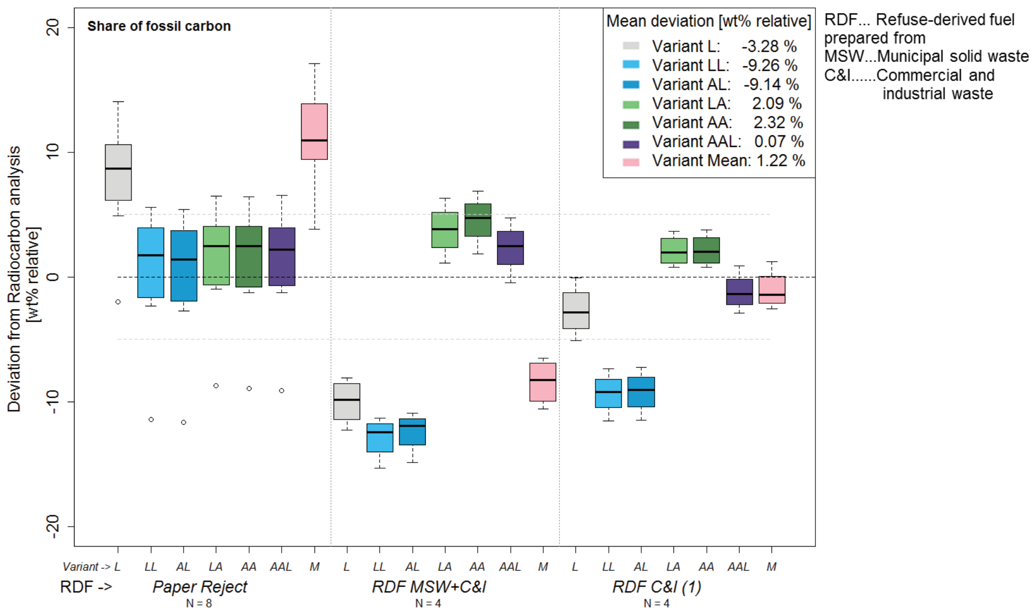

FIGURE 3: Deviations of the fossil carbon share as determined by the aBM from results determined by the Radiocarbon method; aBM results are generated by utilizing different input values of TOX_{Fos} and TOX_{Bio} (7 different variants as listed in Table 2 are used); negative deviations mean a lower value is found by aBM compared to the Radiocarbon method. One outlier is identified for Paper Reject.

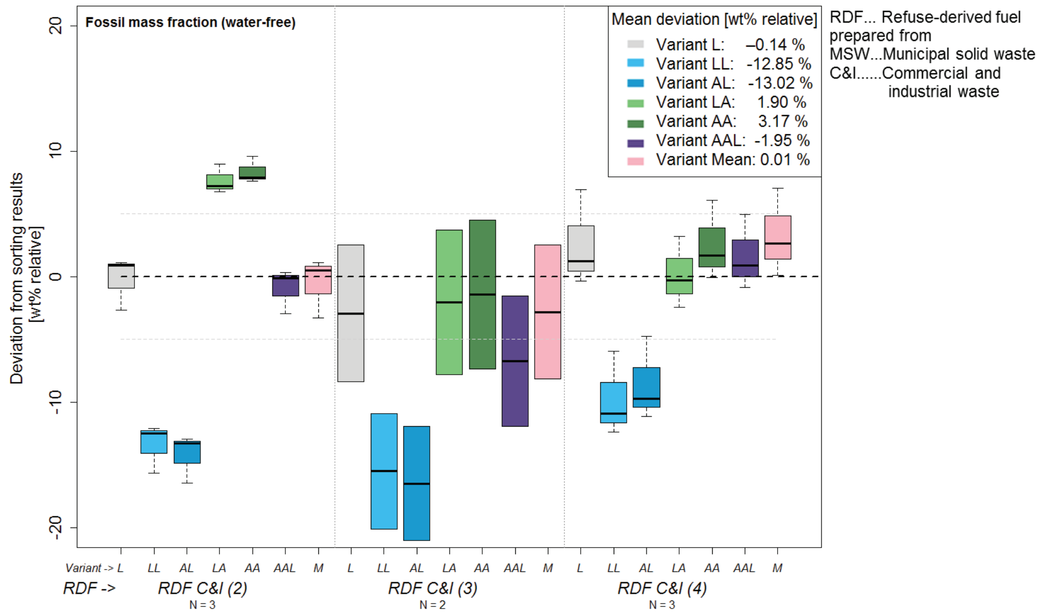

FIGURE 4: Deviations of the fossil mass fraction (water-free) as determined by the aBM from results determined by manual sorting; aBM results are generated by utilizing different input values of TOX_{Fos} and TOX_{Bio} (7 different variants as listed in Table 2 are used); negative deviations mean a lower value is found by aBM compared to manual sorting.

almost 1/3 of the mass of the RDF, has a significantly different chemical composition than the fossil and biogenic matter in the rest of the RDF. This would make the assignment of analysis results of, for example, plastics, and paper to the fine fraction incorrect (which is done for Variant LA and Variant AA). For Variant AAL, it is assumed that the fine fraction has a chemical composition similar to the one reported in the literature. This assumption appears to lead to more accurate results compared to the other variants. The boxplots per RDF in Figure 3 and Figure 4 show that the results of Variant AAL deviate from the result of the alternative method less than 2.5% (except for RDF C&I (3) where only 2 values are available and thus, one value can easily distort the average deviation).

A mean deviation for the Variant AAL of 0.07% is found when comparing the aBM results to the Radiocarbon results (see Figure 5a). When considering the sorting results as alleged target value (although the uncertainty of these values are relatively high) a mean deviation of -1.95% is found (Figure 5b). Although this mean deviation might be higher than found for other variants, a lower variation of results is visible from Figure 5b for this variant than for the

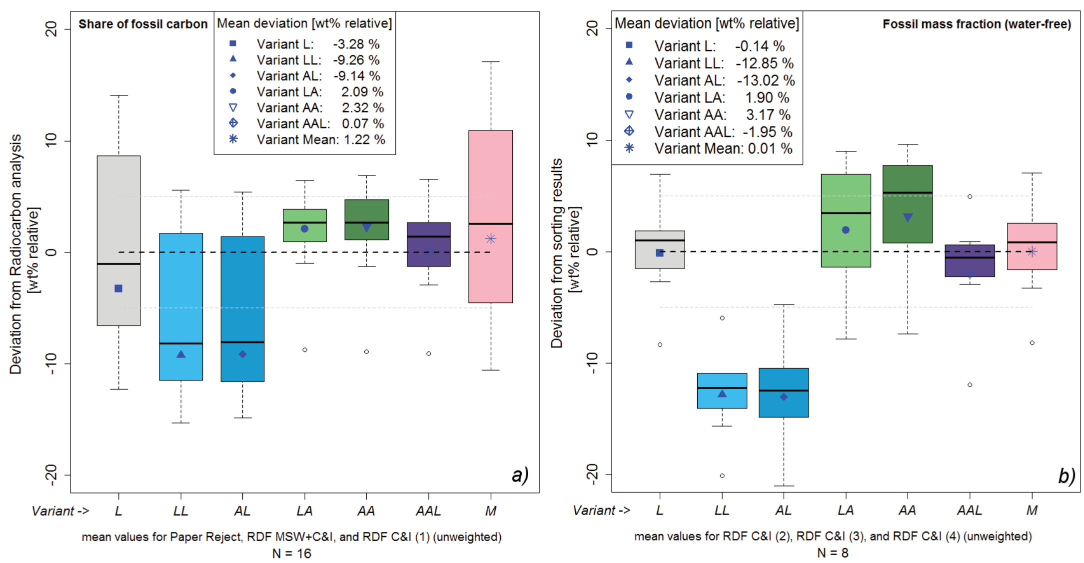

FIGURE 5: a) Deviations of the fossil carbon share as determined by the aBM from results determined by the Radiocarbon method. b) Deviations of the fossil mass fraction (water-free) as determined by the aBM from results determined by manual sorting. Negative deviations mean a lower value is found by aBM compared to the alternative method; Outliers in a) stem from one Paper Reject sample and in b) from one sample of RDF C&I (3) and from one sample of RDF C&I (4).

other options. Treating one of the values for RDF C&I (3) as outlier (only 2 values available, so no clear judgement can be made), results in a mean deviation of only -0.2% for Variant AAL when compared to manual sorting. The results which are obtained by applying Variant AAL are considered to result in a good agreement with the alternative methods.

3.5 Elemental composition of biogenic and fossil organic matter present in RDFs

Based on the results found in the previous sections, the Variant AAL is identified as the most suitable option to generate the input parameter (TOX_{Fos} and TOX_{Bio}) required for the aBM. In Figure 6 and Figure 7 the results for these parameters are presented and compared to values given in Fellner et al. (2011). Additionally the mean for each element is indicated together with the 95-% confidence interval determined for all the RDFs investigated in this study.

It can be seen that the values for all RDFs are in a relatively close range and also similarly to values found in Fellner et al. (2011) for most elements. The most varying value seems to be the total organic oxygen content (TOO). The TOO values generated within this study vary ± 21 g/kg on average in the fossil organic matter and ± 36 g/kg on average in the biogenic organic matter. This variation is partly due to the fact that conducting oxygen analyses is more difficult than for the other elements. Higher uncertainties need to be considered. Further, values reported in literature often only estimate the O-content based on the subtraction of the all other elements from 1000 g/kg.

The higher TOO content found in the fossil organic matter of RDF MSW+C&I indicates higher shares of polyamide, polyethylene terephthalate, or polyurethane. Whereas, the

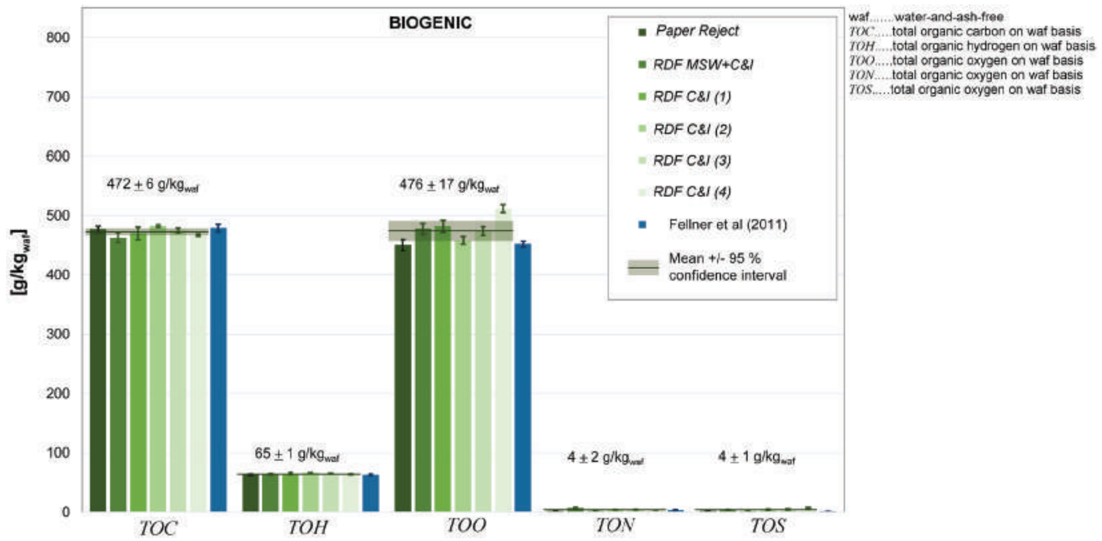

FIGURE 6: Elemental composition of biogenic organic matter (TOX_{Bio}) present in the RDFs (determined according to Variant AAL described in Table 2).

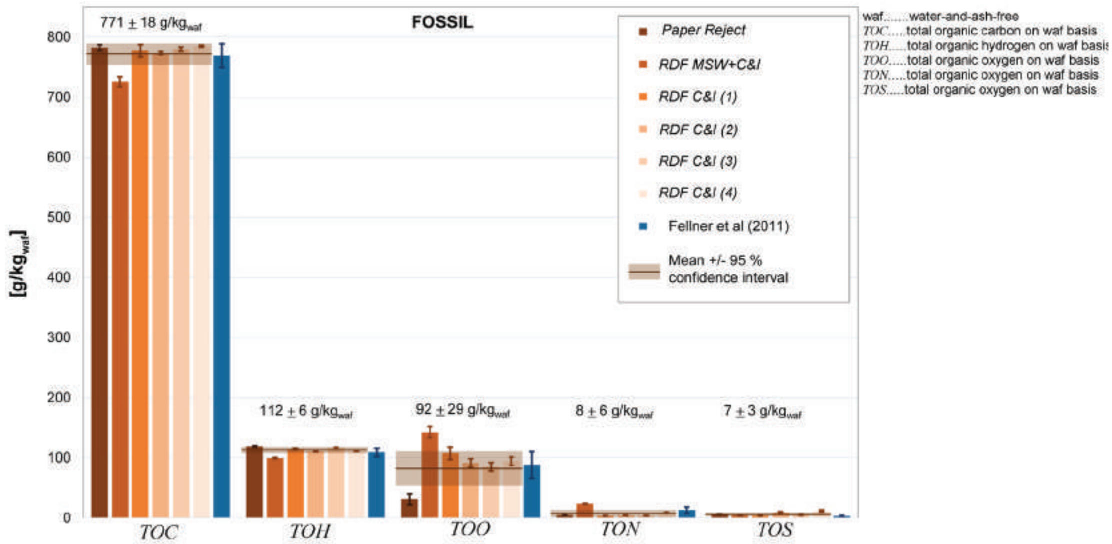

FIGURE 7: Elemental composition of fossil organic matter (TOX_{Fos}) present in the RDFs (determined according to Variant AAL described in Table 2).

Paper Reject can be estimated to contain high shares of polyethylene and polypropylene, which are characterized by a low or even zero oxygen content.

It can be seen that the values found for the RDFs C&I (1), (2), (3), (4) are in an even closer range than when all RDFs are regarded. Except for the TOC_{Bio} of RDF C&I (2) and TOO_{Bio} of RDF C&I (4) all values are within the 95-% confidence interval given in Figure 6 and 7.

4. CONCLUSION

The presented study shows that the shares of different compounds in RDFs (such as paper, plastics, textiles, etc.) can vary significantly, even when the RDFs are produced from allegedly similar waste material (such as C&I). Also the fossil shares in mixed compounds (mix of biogenic and fossil materials, such as in composite materials, textiles, fine fraction) can be considerably different between RDFs and compared to the suggested values in EN 15440:2011. However, for most RDFs these inaccuracies hardly influence the results of the adapted Balance method (aBM) and thus the determination of the fossil carbon share when applying this method.

The usage of generic values from literature or of an overall mean from this study leads to deviations from the allegedly true value of <3.5% when 4 different RDFs C&I are regarded. This finding suggests that there is a typical elemental composition of biogenic and fossil organic matter present in RDFs which are produced out of C&I (RDF C&I). It is expected that the values derived for this type of RDF on TOX_{Fos} and TOX_{Bio} can also be used for other RDFs of the same type. Yet, as the investigated samples stem from Austrian RDF producers, the question on the country-specific dependency remains unclear.

For RDFs where the input is more prone to strong variations (which might be expected from MSW), the initial generation of RDF-specific values on the chemical composition of biogenic and fossil organic matter (TOX_{Fos} and TOX_{Bio}) is recommended to increase the reliability of the aBM results. This also applies when special types of RDF are to be analyzed for their climate relevance (like Paper Reject).

The usage of previously collected literature values for MSW compounds seem not to be suitable for most of the investigated RDFs. Thus, if sorting analyses are conducted it makes sense to generate some data on the elemental composition of the different compounds. These could not only be used to increase the accuracy of the results but could also be collected in a database for different RDF types to be accessed for future investigations.

However, in the case of Paper Reject, the usage of the actual shares of the different compounds in the RDF appears to be the most decisive factor. Thus, if this type of RDF is to be analyzed for its fossil carbon content by means of the aBM, manual sorting campaigns for determining the share of the different compounds are sufficient; the chemical composition of the sorted compounds can be taken from previously analyzed Paper Reject (e.g., from this study).

Some workload can be saved for the determination of the fossil and biogenic share in each compound of the RDF ($x_{mF,k}$ and $x_{mB,k}$); this parameter seems to have a minor influence on the aBM result. Only when the share of textiles, composite & unrecognizable materials or of the fine fraction is significant (and compounds are expected to contain a significantly higher amount of plastics than 50wt%), then $x_{mF,k}$ and $x_{mB,k}$ should also be investigated specifically for these "mixed" compounds (mix of fossil and biogenic constituents).

In general, the results obtained by combining own analysis data with information from the literature are in a good agreement with the outcomes of alternative methods (relative mean deviation <2%). Moreover, the results are also less scattered when choosing this option. Thus, it is assumed Variant AAL represents the best option to generate the necessary input data for the aBM (TOX_{Fos} and TOX_{Bio}).

It can be concluded that if more data on TOX_{Fos} and TOX_{Bio} are collected in a database, generic values can be derived for different RDF types and the initial workload and costs for conducting sorting analyses of RDF before applying the aBM can be saved.

The aBM delivers information on the fossil and biogenic mass share, the fossil and biogenic carbon (CO_2) share and also the share of biogenic and fossil energy recovered from the RDF can be estimated. The heating value for the biogenic and fossil matter in the RDF can, for example, be estimated using an empirical equation based on the elemental composition (e.g., Garcés et al., 2016; Kost, 2001; Meraz et al., 2003). Thus, the aBM can be regarded as cost-efficient method which has been demonstrated to deliver reliable results.

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HOW DOES MUNICIPAL SOLID WASTE POLICY AFFECT HEAT AND ELECTRICITY PRODUCED BY INCINERATORS?

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ABSTRACT

This study examines the effects of municipal solid waste (MSW) policy interventions, specifically sorted collections and unit-based pricing of heat and electricity produced by incinerators in Japan, considering technological and demographic factors. The study shows that the technological factors such as incineration capacity and 24 hours operation affect the available heat energy and electricity. In addition, some sorted collections and unit-based pricing have also affected them. Sorted collections of organic waste can increase available heat energy. For plastics containers and packaging, no significant effects have been observed for both heat energy and electricity. In contrast, for sorted collections of paper containers and packaging, negative significant effects have been observed for both heat energy and electricity. This phenomenon indicates that other factors than a decrease in lower calorific values may affect the heat energy and electricity. Operating years has affected electricity negatively though it has affected heat energy positively. These findings indicate that proper make-decision of MSW policy and choice of incineration type depend on whether which option the municipalities focus on either material recycling or energy recovery (either heat energy or electricity).

1. INTRODUCTION

Sorted collection and recycling of municipal solid waste (hereafter, MSW) is widely practiced in most developed countries. In Japan, an increasing number of municipalities collect plastic and paper containers and packaging separately, since the entire implementation of the Containers and Packaging Recycling Law in 2000. Unit-based pricing is also widely introduced in most developed countries to promote 3R (reduce, reuse, recycle). In Japan, approximately 63% of municipalities implement unit-based pricing as of April 2017 (from the website of professor Yamaya, http://www2.toyo.ac.jp/~yamaya/survey.html - Japanese).

Further, it is expected that incinerators will play an important role as waste-to-energy (hereafter, WTE) plants, in order to produce energy in the form of heat and electricity in many countries (Persson and Munster, 2016; Psomopoulos et al., 2009; Tomic et al., 2016; Xing−gang et al., 2016). In Japan, there have been higher expectations on renewable energy including WTE since the major earthquake and tsunami on March 11, 2011. Although most MSW is burned in Japan, the amount of heat and electricity produced by incinerators is not substantial (Takaoka et al., 2011). As shown in Figure 1, two thirds of incinerators utilize heat and/or electricity produced by them. The figure shows that electricity utilization tends to increase recently. However, the scale is smaller than WTEs in the EU countries and the United States (ISWA 2015). In addition, off-site utilization is still limited. After the implementation of the feed−in tariff (hereafter, FIT) scheme in 2012, electricity originating from renewable energy sources such as biomass including waste is purchased at a fixed price and for a long-term period by existing electric utilities in Japan (METI 2012).

Waste with high calorific values and more waste are suitable for augmenting energy supplies. In that context, separation of organic waste could be superior to that of waste plastic and paper. On the other hand, Psomopoulos et al. (2009) showed that the WTE communities achieved a higher recycling rate than an average recycling rate, referring to the data by the US Environmental Protection Agency.

From another point of view, technological factors such as incineration capacity of incinerators and demographic factors such as population density can also affect energy supplies produced by incinerators. Therefore, it is important to examine the relationships between MSW policy, technology, demographic factors, and energy production for proper make-decision of MSW policy and choice of incineration type. However, no existing studies provide comprehensive empirical evidence as to the relationships between them.

Therefore, this study examines the effects of MSW policy interventions, specifically sorted collections and unit-

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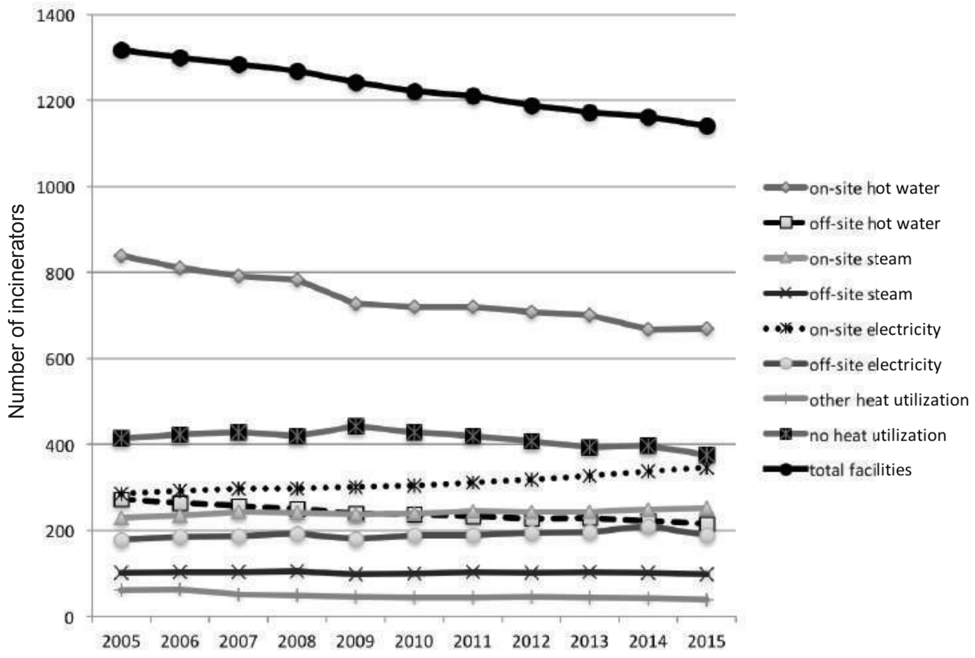

FIGURE 1: Trend of incinerators that produce heat and electricity in Japan. Note: The sum of the numbers in each item is not the same as the number of total facilities because some incinerators have more than one function. Source: Ministry of the Environment, Japan.

based pricing of heat and electricity produced by incinerators in Japan, considering technological and demographic factors.

2. LITERATURE REVIEW

Holmgren and Henning (2004) compared material recovery with waste incineration and subsequent energy recovery for an energy-efficiency case study focusing on two Swedish municipalities. They showed that paper and hard plastics should be materially recovered while cardboard and biodegradable waste was more suited for energy recovery. Calabro (2009) examined the relationship between carbon dioxide emissions released from incinerators and separate collection of MSW. He noted that separate collection of plastics is a key issue when residual waste is treated in WTE plants considering greenhouse gases emission. He showed that the potential increase in energy production due to co-incineration of waste plastics and other combustibles would not offset the increase in carbon dioxide emission from incinerators, though it increases with lower calorific values (hereafter, LCVs) of waste. Calabro (2010) evaluated the effect of separate collection on the characteristics of residual MSW in terms of LCVs and ash production in Italy. He showed that water content of residual waste and the share of combustible materials were affected by separate collection. However, while these existing studies considered the characteristics of the residual waste, they did not consider technological (e.g., incineration types and capacity) or demographic factors.

Nishitani et al. (2010) simulated the effects of sorted collection and recycling on the volume and composition of waste, based on the changes observed in two Japanese municipalities that had actually introduced sorted collection. They found that a decrease in calorific value would be limited though the share of combustibles in waste would decrease due to sorted collection and depopulation. However, they did not examine heat and electricity produced by incinerators while they focused on the future composition. Takaoka et al. (2011) examined various scenarios involving indirect reduction of carbon dioxide emissions by WTE plants in Japan. Although they focused on technical determinants such as enhancement of energy recovery capacity, they did not consider policy or demographic determinants. Additionally, none of these studies considered plant–variant or time-variant factors.

Tomic et al. (2016) indicated that changes in EU legislation (such as 99/31/EC on landfill of waste, 2008/98/EC on basic concepts of waste management, and 2012/27/EU on energy efficiency) affect the amount and composition of MSW, and WTE plants could improve their profitability by co-combusting other local wastes and introduction of area-wide waste management. However, no existing studies provide comprehensive empirical evidence as to the relationships between MSW policy, technology, demographic factors, and energy production.

3. METHODOLOGY AND DATA

3.1 Methodology

This study uses a parametric approach to estimate factors that can affect heat and electricity produced by incinerators. It invokes independent variables that capture the following three sets of determinants: MSW policy, technological factors and demographic characteristics. Further, models are delineated based on the following five dependent variables: (1) available heat energy; (2) utilized heat energy out of incinerators in (1); (3) available electricity; (4) utilized electricity out of incinerators in (3); and (5) LCVs. The last dependent variable (5) is considered to examine whether the independent variavles affect the characteristics of residual MSW in terms of LCVs or not. The actual (not nominal) values for these five variables are used on a priority basis in the study. When plants do not report the actual values, the study uses the nominal values. The study focuses on WTE plants for MSW that produce heat and/or electricity in Japan. However, some zero values are included in the dependent variable data arrays because some plants have either heat or power generation facilities. Of note, the number of the plants that supply heat and electricity outside of the plants is small. Some missing data are also included therein because a few plants do not report the actual data of utilized heat and electricity. In this case (censored data), ordinary least squares regression leads to inconsistent parameter estimates. Therefore, excluding (5) LCVs, the study applies a Tobit regression analysis to examine the foregoing determinants. As for (5), many plants have positive values of the actual or nominal values though a few plants do not report the both. Therefore, the study applies a normal panel regression for (5). Importantly, panel data are utilized which control for omitted variables, with consideration of time-variant factors.

The model structure of Tobit regression is as follows (Cameron and Trivedi, 2009):

$$WTE_i^{j*} = X_i^{j'}\beta_X + Y_i^{j'}\beta_Y + Z_i^{j'}\beta_Z + \varepsilon_i^j, \quad i = 1, \ldots, N, \; j = 1, 2 \ldots, 5$$

where $\varepsilon_i^j \sim N(0, \sigma^2)$, and WTE_i^{j*} is an unobserved latent variable for the i-th facility and for the j-th dependent variables - above (1) to (5). X_i^j, Y_i^j and Z_i^j denote the $(K \times 1)$, $(L \times 1)$ and $(M \times 1)$ vectors of exogenous and fully observed regressors for technological factors, MSW policy and demographic characteristics, respectively. The concrete elements of each factor are presented in the next subsection. K, L and M represent the number of independent variables in each determinant.

The observed variable WTE_i^j is related to the latent variable WTE_i^{j*} in the case of left−censored Tobit model as follows:

$$WTE_i^j = \begin{cases} WTE_i^{j*} & \text{if } 0 \leq WTE_i^{j*} \\ 0 & \text{if } WTE_i^{j*} < 0 \end{cases}$$

In actual estimation, the dependent variables are transformed by logarithms to capture (semi-)elasticity.

3.2 Data

635 incinerators (WTE plants) for MSW disposal in Japan were originally selected; they are the plants that were operating during 2007 to 2015. The data is an unbalanced panel because the plants are included if they operated in at least any two years during the period.

The study considers separation of plastic containers and packaging, paper containers and packaging, and organic waste, and unit-based pricing as the MSW policy. These are treated as dummy variables that take the value 1 if each policy is implemented in the municipality where the incinerator locates. Unit-based pricing is expected to promote waste separation because it generally increases the burden on burnable and unburnable waste other than recycling, and in contrast, it lightens the burden on recycled waste. In addition, it is expected to decrease the amount of waste disposal. With or without co-disposable industrial waste is also considered. It is treated as a dummy variable that takes the value 1 if co-disposable industrial waste is done in the incinerator. Industrial waste is disposed of separately from MSW, in principle, in Japan. Considering economic efficiency, however, some MSW incinerators dispose of industrial waste which consists of the same properties as MSW. Co-disposal of industrial waste and MSW may also increase energy efficiency.

Incineration type (whether melting treatment or not and 24 hours operation or not), incineration capacity and operating years are considered as the technological factors, referring to existing studies (e.g., Takaoka et al., 2011). Melting treatment is treated as a dummy variable that takes the value 1 if the incinerator adapts the incineration type. 24 hours operation is also treated as a dummy variable that takes the value 1 if the incinerator operates continuously for 24 hours per day. It is likely to produce more energy because waste is disposed of at higher temperatures and stably in case of melting treatment and 24 hours operation. If there are economies of scale, larger incineration capacity would produce more energy. Older plants are likely to be less energy efficient and produce less power, superseded because of technological innovation.

In addition, population density in the municipalities is considered as a demographic factor. It is likely that heat energy tends to be utilized in denser areas. Annual average outside temperature is also considered as a geographic factor. Kakuta (2010) shows that the lower outside temperature is, the less energy is produced. This is because some incinerators use heat for white vapor smoke prevention to mitigate residents' anxiety in Japan. Therefore, it is likely to decrease power generation and the outside supply of heat energy. On the other hand, the outside supply of heat energy may increase in colder areas because such areas can have a strong demand for that. Time trend is controlled using year dummy variables.

Waste management data pertaining to (1) to (5) above, as well as data pertaining to technological factors and MSW policy are taken from the website of the Ministry of the Environment, Japan (http://www.env.go.jp/recycle/waste_tech/index.html - Japanese). Population densities in each municipality are taken from the database hosted by the Asahi Shinbun Syuppan (a Japanese newspaper company) (2014) and the websites of the Ministry of Internal Affairs and Communications (http://www.stat.go.jp - Japanese). Outside temparature in each municipality are taken from the websites of the Japan Meteorological Agency (http://www.jma.go.jp/jma/index.html - Japanese).

Correlation coefficients indicate that the relationships between the explanatory variables are negligible. Descriptive statistics are shown in Table 1.

Focusing on the policy variables, the table shows that the occurrence rates for the separation of plastics and paper are high while those for the separation of organic waste and co-disposal of industrial waste are low. For the technological factors, the melting treatment has not been common yet.

There are two main formulations for extensions of the truncated regression model to panel data similar to the lin-

TABLE 1: Descriptive statistics.

Variables	Mean	P50	SD	Max	Min
Heat energy (MJ)	6.56E+07	712515	2.00E+08	3.24E+09	0
Outside supply of heat (MJ)	5924437	0	3.27E+07	6.99E+08	0
Electricity (MWh)	41797.18	15627	641757.3	2.74E+07	0
Outside supply of electricity (MWh)	89022.96	2185.5	2503995	9.72E+07	0
Lower calorific values (kJ/kg)	8566.47	8602	2154.38	22430	0
Incineration capacity (ton / day)	238.46	170	216.48	1800	7
24 hours operation (D)	0.78	1	0.42	1	0
Operating years	15.64	15	8.30	43	0
Melting treatment (D)	0.15	0	0.35	1	0
Separation of plastics containers and packaging (D)	0.66	1	0.47	1	0
Separation of paper containers and packaging (D)	0.68	1	0.46	1	0
Separation of organic waste (D)	0.08	0	0.26	1	0
Unit-based pricing (D)	0.40	0	0.49	1	0
Co-disposal of industrial waste (D)	0.18	0	0.38	1	0
Population density (100 person / km^2)	22.86	7.60	35.11	153.71	0.06
Average outside temperature (°C)	15.51	16.20	2.60	24.10	4.30

Note: (D) represents a dummy variable.

ear regression: fixed effects and random effects models. The study applies the random effects model for the following two reasons. First, the study includes time–invariant independent variables. Second, it does not seem to be unrealistic that the individual-specific effect is not correlated with the independent variables in the study because plant manufactures that manage all over the country build incinerators.

4. ESTIMATION RESULTS

The estimation results from a random effects Tobit regression are shown in Tables 2 and 3. Table 2 presents the results of heat energy and outside supply in it, and Table 3 presents those of electricity and outside supply in it. The results of the likelihood-ratio test and ρ, percent contribution to the total variance of the panel-level variance component, indicate that we should not use pooled data, but panel data. The estimation results of LCVs from a standard panel (random effects) regression are shown in Table 4. The results of the Hausman test and Breusch-Pagan test indicate that we should not use fixed effects, but random effects model. For each dependent variable, the estimation results for a case including all explanatory variables are shown in columns “Model 1”, and the results after elimination of insignificant variables are shown in the columns “Model 2”.

The parameters represent the effects of a change in each independent variable on the expected value of the latent variable WTE_i^{j*}, holding all other independent variables constant (Breen 1996). They also indicate the marginal effects in the mean of each variable among the uncensored observations because the heat energy and electricity including outside supply of them are transformed by logarithms, as noted Subsection 3.1. Positive values indicate more supply of heat energy or electricity in Table 2 and 3, and more LCVs in Table 4. Negative values indicate the opposite phenomena. The following are the estimation results of Model 2 for each energy utilization.

4.1 Heat energy

Significant variables that affect the available heat energy among the technological factors are incineration capacity, 24 hours operation (only outside supply) and operating years, which are significantly positive, and melting treatment (only available heat energy), which is significantly negative. One ton increase in incineration capacity increases heat energy and outside supply of it by 1.5% and 3.4%, respectively. 24 hours operation increases outside supply of heat energy by 245.0% though it does not affect the heat supply significantly. These findings are similar to the study’s a priori expectation. One year increase in operating years increases heat energy and outside supply of it by 9.0% and 26.0%. Positive effect of operating years does not accord with a priori expectation because it was expected that newer incinerators tended to generate more energy. This phenomenon will be discussed with the results of electricity in the next subsection. The results indicate that melting treatment decreases heat energy by approximately 404.9% though it does not affect the outside supply of heat significantly. Although this finding is contrary to a priori expectation, the plants with melting treatment are likely to have put a high priority on producing more electricity rather than heat energy to offset the increase of electricity with melting treatment . However, melting treatment is not significant for the electricity (though positive sign), as noted in the next subsection.

Significant variables that affect heat energy among the policy determinants are the separation of organic waste, which is significantly positive, and the separation of paper containers and packaging, which is significantly negative.

TABLE 2: Estimation results of heat energy.

	Heat energy		Outside supply of heat	
	Model 1	**Model 2**	**Model 1**	**Model 2**
Incineration capacity	0.0146	0.0147	0.0335	0.0341
	[6.59]***	[6.92]***	[8.03]***	[8.20]***
24 hours operation (D)	−0.2351	N.S.	2.3725	2.4503
	[−0.33]		[1.91]*	[2.02]**
Operating years	0.0891	0.0900	0.2284	0.2596
	[2.47]**	[2.50]**	[4.12]***	[5.29]***
Melting treatment (D)	−4.0731	−4.0493	−2.7671	N.S.
	[−3.79]***	[−3.81]***	[−1.66]*	
Separation of plastics containers and packaging (D)	0.0450	N.S.	0.5723	N.S.
	[0.10]		[0.85]	
Separation of paper containers and packaging (D)	−0.9626	−1.0254	−0.8328	N.S.
	[−2.43] **	[−2.61]**	[−1.41]	
Separation of organic waste (D)	1.1779	1.4053	−0.6372	N.S.
	[1.88]*	[2.25]**	[−0.68]	
Unit-based pricing (D)	−0.5045	N.S.	−1.8466	−1.9540
	[−1.07]		[−2.41]**	[−2.56]***
Co-disposal of industrial waste (D)	0.9488	N.S.	0.6816	N.S.
	[1.56]		[0.78]	
Population density	−0.0920	−0.0938	−0.0871	−0.0770
	[−6.47]***	[−6.83]***	[−3.82]***	[−3.49]***
Outside temparature	−0.1154	N.S.	0.5584	N.S.
	[−0.74]		[2.16]**	
Year 2008 (D)	−0.2610	N.S.	-0.3851	N.S.
	[−0.58]		[-0.57]	
Year 2009 (D)	3.4703	3.5443	1.9758	2.1266
	[8.02]***	[9.64]***	[3.01]***	[3.79]***
Year 2010 (D)	5.1546	5.0231	2.2958	3.5231
	[10.07]***	[13.48]***	[2.91]***	[6.27]***
Year 2011 (D)	4.7731	4.8843	2.9631	2.9905
	[10.51]***	[12.73]***	[4.31]***	[5.18]***
Year 2012 (D)	5.7226	5.8365	3.9437	3.8616
	[12.25]***	[14.89]***	[5.59]***	[6.61]***
Year 2013 (D)	6.1886	6.2401	4.0919	4.1191
	[12.93]***	[15.33]***	[5.71]***	[6.86]***
Year 2014 (D)	5.9722	6.0684	4.1431	3.9949
	[12.05]***	[14.39]***	[5.53]***	[6.45]***
Year 2015 (D)	5.1108	5.1210	3.0859	3.1995
	[10.02]***	[11.57]***	[4.04]***	[4.94]***
Constants	3.7603	1.7355	-37.1015	−29.9631
	[1.46]	[2.03]**	[-6.68]***	[−20.52]***
ρ	0.6957	0.6946	0.9139	0.9138
Num. of observations	5098		5098	
Num. of groups	635		635	
Log likelihood	−12826.52	−12847.67	−5990.74	−5995.10
Wald test	$\chi^2(19) = 640.02$***	$\chi^2(13) = 634.24$***	$\chi^2(19) = 291.08$***	$\chi^2(12) = 281.99$***
Likelihood-ratio test	3044.8***	3074.61***	3718.59***	3760.39***
Left−censored observations	1719		3624	
Uncensored observations	3379		1474	

Note: (D) represents a dummy variable. N.S. represents not significant. Values in square brackets represent z−statistics.
**** $p < 0.01$, ** $p < 0.05$, * $p < 0.1$.*

TABLE 3: Estimation results of electricity.

	Electricity		Outside supply of electricity	
	Model 1	**Model 2**	**Model 1**	**Model 2**
Incineration capacity	0.0219	0.0222	0.0203	0.0201
	[20.78]***	[21.01]***	[11.11]***	[11.09]***
24 hours operation (D)	1.0129	1.0371	10.7159	10.7882
	[5.01]***	[5.27]***	[5.17]***	[5.18]***
Operating years	−0.2701	−0.2782	−0.2581	−0.2665
	[−26.04]***	[−30.09]***	[−8.41]***	[−9.33]***
Melting treatment (D)	0.5777	N.S.	0.6980	N.S.
	[1.61]		[0.76]	
Separation of plastics containers and packaging (D)	0.0665	N.S.	−0.0792	N.S.
	[0.61]		[−0.21]	
Separation of paper containers and packaging (D)	−0.3799	−0.3811	0.6589	0.6372
	[−3.91]***	[−3.98]***	[1.78]*	[1.75]*
Separation of organic waste (D)	0.2736	N.S.	0.3641	N.S.
	[1.62]		[0.58]	
Unit-based pricing (D)	−0.1021	N.S.	0.7401	N.S.
	[−0.75]		[1.56]	
Co-disposal of industrial waste (D)	0.3084	0.3077	−0.2816	N.S.
	[2.14]**	[2.15]**	[−0.55]	
Population density	0.0138	0.0149	0.0377	0.0363
	[2.41]**	[2.73]***	[3.42]***	[3.48]***
Outside temparature	−0.0126	N.S.	0.1172	N.S.
	[−0.22]		[0.78]	
Year 2008 (D)	0.3520	0.3701	−13.2191	N.S.
	[3.25]***	[3.53]***	[−0.02]	
Year 2009 (D)	0.5931	0.5966	16.3349	17.1840
	[5.59]***	[5.76]***	[14.87]***	[16.63]***
Year 2010 (D)	0.8370	0.8326	16.8220	17.8756
	[5.901]***	[7.87]***	[14.91]***	[17.26]***
Year 2011 (D)	1.1542	1.1805	17.3675	18.1960
	[10.15]***	[10.93]***	[15.72]***	[17.52]***
Year 2012 (D)	1.4515	1.4894	18.2500	19.0855
	[12.20]***	[13.50]***	[16.44]***	[18.32]***
Year 2013 (D)	1.7529	1.7858	18.8392	19.7287
	[14.60]***	[15.73]***	[16.94]***	[18.87]***
Year 2014 (D)	2.0186	2.0608	17.5878	18.4600
	[15.84]***	[17.55]***	[15.72]***	[16.88]***
Year 2015 (D)	2.1120	2.1585	16.8327	17.7946
	[16.49]***	[17.75]***	[15.02]***	[16.88]***
Constants	−2.6329	−3.0530	−36.0411	−34.6020
	[−2.93]***	[−11.42]***	[−10.67]***	[−13.95]***
ρ	0.9662	0.9683	0.8413	0.8411
Num. of observations	5098		5098	
Num. of groups	635		635	
Log likelihood	−5627.11	−5623.87	−4230.96	−4236.30
Wald test	$\chi^2(19) = 1552.90$***	$\chi^2(19) = 1570.45$***	$\chi^2(19) = 607.48$***	$\chi^2(12) = 657.74$***
Likelihood-ratio test	7325.02***	7331.76***	2152.55***	2188.95***
Left−censored observations	2453		3820	
Uncensored observations	2645		1278	

Note: (D) represents a dummy variable. N.S. represents not significant. Values in square brackets represent z−statistics.
**** p < 0.01, ** p < 0.05, * p < 0.1.*

TABLE 4: Estimation results of lower calorific values (LCVs).

	Model 1	Model 2
Incineration capacity	0.0001	0.0001
	[1.78]*	[1.72]*
24 hours operation (D)	0.0998	0.0990
	[4.19]***	[4.30]***
Operating years	−0.0028	−0.0028
	[−2.91]***	[−1.99] **
Melting treatment (D)	0.0056	N.S.
	[0.25]	
Separation of plastics containers and packaging (D)	−0.0288	−0.0294
	[−2.37]**	[−2.47]**
Separation of paper containers and packaging (D)	−0.0278	−0.0273
	[−2.00]**	[−1.99]*
Separation of organic waste (D)	0.0026	N.S.
	[0.11]	
Unit-based pricing (D)	0.0176	N.S.
	[1.38]	
Co-disposal of industrial waste (D)	−0.0007	N.S.
	[−0.03]	
Population density	0.0012	0.0011
	[5.88]***	[5.77]***
Outside temparature	-0.0006	N.S.
	[−0.17]	
Year 2008 (D)	−0.0916	−0.0998
	[−5.79]***	[−7.06]***
Year 2009 (D)	0.0175	N.S.
	[1.16]	
Year 2010 (D)	0.0696	0.0602
	[4.09]***	[5.24]***
Year 2011 (D)	0.0637	0.0557
	[4.14]***	[4.54]***
Year 2012 (D)	0.0847	0.0772
	[5.94]***	[6.86]***
Year 2013 (D)	0.1178	0.1105
	[8.02]***	[9.44]***
Year 2014 (D)	0.1058	0.0988
	[6.81]***	[7.68]***
Year 2015 (D)	0.0934	0.0863
	[4.54]***	[4.70]***
Constants	8.9318	8.9427
	[148.05]***	[384.78]***
Num. of observations	4988	
Num. of groups	635	
Hausman test	$\chi^2(19) = 25.18$	$\chi^2(13) = 18.77$
	p = 0.154	p = 0.131
Breusch-Pagan test	$\chi^2(1) = 1677.02$	$\chi^2(1) = 1733.98$
	p = 0.000***	p = 0.000***

Note: (D) represents a dummy variable. N.S represents not significant. Values in square brackets represent z−statistics.
**** p < 0.01, ** p < 0.05, * p < 0.1.*

The results indicate that the separation of organic waste increases heat energy by approximately 140.5%, while the separation of paper containers and packaging decreases it by approximately 102.5%, which are similar to a priori expectation. A significant variable that affects the outside supply of heat energy among the policy determinants is the unit-based pricing, which is significantly negative. The unit-based pricing decreases the outside supply of heat energy by approximately 195.4% though it has not affected available heat energy significantly. This phenomenon will be also discussed with the results of electricity in Subsection 4.3. Co-disposal of industrial waste and MSW has not affected heat energy and the outside supply of it significantly so far.

Population density is negatively significant for heat energy. The results suggest that 100 people increase per km^2 in population density decreases heat energy and outside supply of it by 9.4% and 7.8%, respectively. This indicates that heat energy tends to be utilized in sparser areas. Agricultural utilization such as greenhouses may increase heat energy in sparser areas. Outside temparature has not affected heat energy and the outside supply of it significantly. The year dummies after 2009 are positively significant for heat energy (including the outside supply of it). Possible reasons of positive effects observed in the year dummies are as follows. The ministry of the environment has abolished state subsidy for siting incinerators without energy recovery, and raised the portion of state subsidy for siting WTEs that can produce heat and electricity highly efficiency since 2005. This incentive for siting WTEs may promote energy recovery with a time lag of a few years. In addition, the major earthquake and tsunami in Japan on March, 2011 is likely to increase outside supply of energy produced by incinerators because of making up for temporal loss of energy supply after the earthquake.

4.2 Electricity

Significant variables that affect electricity among the technological factors are 24 hours operation and incineration capacity, which are significantly positive, and operating years, which is significantly negative. These findings are similar to a priori expectation. The results indicate that one ton increase in incineration capacity increases electricity and outside supply of it by 2.2% and 2.0%, respectively. These findings indicate that 24 hours operation contributes to more electricity generation and there are economies of scale in production of both heat and electricity in the incinerators. 24 hours operation increases electricity and outside supply of it by approximately 103.7% and 1078.8%, respectively. Most of the plants that supply electricity outside the incinerators operate for 24 hours. This seems to bring significant differences of more than ten times between with and without 24 hours operation. One year decrease in operating years decreases electricity and outside supply of it by 27.8% and 26.7%, respectively. Old incinerators are likely to have put a high priority on producing more heat energy rather than electricity, considering the results in the previous subsection. This phenomenon seems to be caused by the implementation of the RPS (Renewables Portfolio Standard) that mandates electric utilities to use a fixed minimum amount of renewable energy sources in 2003, and FIT in 2012. On the other hand, melting treatment has not affected the electricity significantly.

Significant variables that affect the electricity among the policy factors are the separation of paper containers and packaging, which is significantly negative, and co-disposal of industrial waste and MSW, which is significantly positive. A significant variable that affects the ouside supply of electricity among the policy factors is the separation of paper containers and packaging, which is significantly positive. The results indicate that the separation of paper containers and packaging increases the outside supply of electricity by 63.7% though it decreases electricity by 38.1%. It should be noted that some incinerators purchase electricity from outside equal to or than the amount of electricity produced by the incinerator, as noted by Matsuto (2012). On the other hands, the results suggest that sorted collection of plastics containers and packaging and organic waste has not affected the electricity significantly. These findings will be further examined considering LCVs in the next subsection.

Population density is positively significant for electricity. The results suggest that 100 people increase per km^2 in population density increases electricity and outside supply of it by 1.5% and 3.6%, respectively. This indicates that electricity tends to be utilized in denser areas similarly to a priori expectation. It is likely that there is a stronger need for electricity in urban areas. This phenomenon is contrary to the results of heat energy noted in the previous subsection. Outside temparature has not affected electricity and the outside supply of it significantly. The year dummies after 2008 or 2009 (for the outside supply of electricity) are positively significant for electricity. Possible reasons of positive effects observed in the year dummies are similar to the points noted in the previous subsection. In addition, it is likely that the implementation of the RPS and FIT promotes more electricity generation.

4.3 Lower Calorific Values (LCVs)

Significant variables that affect LCVs among the technological factors are 24 hours operation and incineration capacity, which are significantly positive, and operating years, which is significantly negative. These findings are similar to a priori expectation. The results indicate that one ton increase in incineration capacity increases LCVs by 0.01%. 24 hours operation increases LCVs by approximately 9.9%. One year decrease in operating years decreases LCVs by 0.3%. These findings indicate that the fluctuation of LCVs affects heat energy and electricity.

Significant variables that affect LCVs among the policy factors are the separation of plastic and paper containers and packaging, which are significantly negative. The results indicate that the separation of plastic and paper containers and packaging decreases LCVs by 2.9 and 2.7%, respectively. These findings are similar to a priori expectation. However, there is no significant difference between the both rates though the calories of plastics are higher than those of paper in general. On the other hand, the separation of plastic containers and packaging has not decreased both heat energy and electricity significantly as mentioned in the previouse subsections, even though a decrease in LCVs was observed. This result is similar to the result by Nishitani et al. (2010). In contrast, the separation of paper containers and packaging has decreased both heat energy and electricity. In addition, a decrease in energy recovery is much larger than that in LCVs. This phenomenon indicates that other factors than LCVs may affect the heat energy and electricity. However, this study cannot clarify the factors. The results also suggest that unit-based pricing has not affected LCVs significantly so far. On the other hand, it has been negatively significant on the outside supply of electricity though it has not affectd the available electricity significantly, as shown in the previous subsection. This phenomenon also indicates that other factors than LCVs may affect the outside supply of electricity. A possible reason of negative effects observed in the unit-based pricing is as follows. Unit-based pricing was originally introduced in rural areas, which have weeker need for electricity than urban areas. Such a geological characteristic may affect the outside supply of electricity.

Population density is positively significant for LCVs. This result indicates that LCVs are higher in urban areas rather than rural areas. This phenomenon seems to be caused by the volume of business waste and the life-style in urban areas. Some business wastes are included in MSW in Japan. The rate of business waste tends to be higher in denser areas rather than sparser areas (from the website of the Ministry of the Environment, http://www.env.go.jp/recycle/waste_tech/index.html - Japanese). Business waste is likely to contain more calofric waste such as papers. In addition, the residents in urban areas seem to consume more plastics and paper containers and packagings than those in the rural areas. In contrast, outside temparature has not affected LCVs significantly. The year dummies after 2010 are positively significant for electricity.

5. CONCLUSIONS

It is technological factors such as 24 hours operation and incineration capacity that mainly affect heat and electricity produced by incinerators. However, some MSW policy interventions such as sorted collections and unit-based pricing have affected them. Sorted collections of organic waste can increase available heat energy. For plastics containers and packagings, no significant effects were observed for both heat energy and electricity. In contrast, for sorted collections of paper containers and packaging, negative significant effects were observed for both heat energy and electricity. However, a decline of in energy recovery was much larger than that in LCVs. This phenomenon indicates that other factors than the change in LCVs may affect the heat energy and electricity. Clarification of these factors will be a further research. For unit-based pricing, a negative significant effect was observed for the outside supply of electricity though it has not affected the available heat, electricity and LCVs.

The results that sorted collections have provided a limited impact on LCVs and energy recovery may suggest that segregation by residents has not been perfect. Unlike can, glasses and PET bottoles, it is difficult for residents to segregate plastics and paper containers and packaging. If the residents can segregate them more perfectly, LCVs would decline more and therefore less energy might be produced.

These findings indicate that proper make-decision of MSW policy and choice of incineration type depend on whether which option the municipalities focus on either material recycling or energy recovery (either heat energy or electricity). Although the study focuses on quantitive changes of energy recovery, the financial and environmental effects are also important for a more detailed examination of sustainable waste management. This will be a further research.

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SUSTAINABLE MANAGEMENT OF ENERGY SUPPLY INCLUDING THE USE OF WASTE-BASED BIOGAS PROCESSES

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ABSTRACT

In order to meet the goal of 50% renewables in the electrical energy mix till 2030 set by the German government the expansion must be continued. One of the biggest problems with the increasing share of renewable energy are the arising fluctuations. In this study it was investigated how the manageability of the biogas production from waste can be used to create a flexible production strategy in order to improve the reliability of a renewable energy supply. The current misbalances between energy demand and supply of wind turbines and PV systems are the basis for this new strategy. The simulation is based on a stand-alone 100% renewable energy supply of a model town with a population of 1,600 people. In order to investigate the impact of this new biogas management strategy using 100% renewable energy supply (including wind and solar energy) a simulation tool using MATLAB was designed. Furthermore, to receive preliminary real data regarding flexible biogas production tests were done with a plug-flow digester. Lastly, an economic analysis regarding the profitability was carried out. Simulations in this study have shown that using a flexible and demand adapted biogas management can reduce the required battery capacity by up to 60%. Nonetheless, the combination with feed-in management of wind and solar power has to be further investigated.

1. INTRODUCTION

While fossil fuel reserves run out, new energy sources are needed for maintaining the energy supply. From 2004 to 2015 the amount of renewable electrical energy in the gross final consumption of electricity has increased from 9.4% to 30.7% in Germany (Eurostat, 2017). In order to meet the goal set by the German government of 50% renewable electrical energy by 2030, the expansion must be continued (Energiekonzept für eine umweltschonende, zuverlässige und bezahlbare Energieversorgung, 2010). One of the biggest problems with increasing the share of renewable energy are the fluctuations of energy production by wind turbines and photovoltaic systems due to changing weather conditions. To avoid sudden drops of voltage, which could lead to brown-outs, and in worse cases to black-outs, conventional energy sources such as flexible gas turbines are currently used as back-up facilities.

Just as low production can cause problems, the over production of energy can lead to an overload of the energy grid. This can result from high wind speeds and increased solar radiation especially in times of low energy demands. To use this surplus in times of lower production rates, the energy needs to be stored, e.g. in battery parks. Not only do short term fluctuations but also long-term fluctuations resulting from seasonal variations have to be balanced.

An alternative renewable energy source that is not prone to fluctuations is biogas, which has a high market share in Germany (16.7% of electrical renewable energy in 2015 (AGEE-Stat, 2016)). The advantage of biogas is its better manageability compared to wind and solar power as it is independent from weather conditions and thus can be produced continuously.

This study investigated if the manageability of the biogas production can be used to create a flexible production strategy to improve the reliability of a renewable energy supply. In order to investigate the impact of this new biogas management strategy using 100% renewable energy supply (including wind and solar energy), a simulation tool using MATLAB was designed. To apply these results and collect real data to improve the tool in the future, preliminary tests were carried out so as to find a flexible operation mode protocol. For this reason a pilot scale plug-flow digester and several lab scale fermenters were fed with cow manure and organic waste (food and distillery waste) in

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different ratios and amounts. As a major result, the suitability of waste as basic substrate for the above described scenario was initially proven (according to several process-relevant parameters).

Lastly, an economic analysis was performed to make a statement about the profitability of this new biogas management concept.

2. ENERGY MANAGEMENT TOOL

The tool used for the following calculations was developed for the simulation of the energy supply of a model town based on 100% renewable energy including energy production by a wind turbine, photovoltaic system and an anaerobic digester. As the direct amount of the energy produced by renewable energy plants and the fraction of the energy consumption do not match over the time, strategies to manage resulting fluctuations are required. The most obvious strategy is to use batteries for balancing the mismatches but this requires not only material resources in terms of chemical substances but also financial funds. Another strategy is to adapt the temporary surplus of the energy production of the renewable plants. In this tool both strategies are combined and the impact on the supply security and the required battery capacity is analyzed including feed-in management referring to energy production by anaerobic digesters.

2.1 Energy production and load data

The main difference between conventional energy production based on fossil or nuclear power plants and an energy system which depends on 100% renewable energy is the fluctuating character of wind and solar energy which has to be compensated. These fluctuations can only be taken into account with high-resolution data in time not only for energy production but also for energy consumption which also varies over time.

The development of a strategy to optimize the renewable energy supply with a flexible biogas production and conversion into electricity was carried out for a small supply region representing 1,600 residents. Bigger regions with local differences in weather conditions would require consideration of additional effects which are out of the scope of this study. The load profile of this region has a resolution of one minute and is a synthesis of different load profiles, which were generated with the tool SynPro by the Fraunhofer Institute for Solar Energy Systems.

The Ostfalia University of Applied Sciences owns a small scale wind turbine and a photovoltaic system (Boggasch, 2016). The energy production of these plants is measured with a sampling rate smaller than one second but to match the energy production and consumption data, mean values of one minute were used and a scaling factor was set for aligning the dimensions of energy production and consumption. To consider the inertia of bigger wind turbines, the wind power data was also smoothed by generating mean values of five minutes. As the energy production of wind turbines and photovoltaic systems is weather-dependent, the extracted energy fluctuates heavily. In this paper the annual energy demand is covered from an energy mix shown based on 20% energy from the wind turbine, 30% from the photovoltaic plant and 50% energy from biogas. These shares represent the energy production of the plants divided by the energy consumption, but due to losses of the battery an energy balance of 100% is not necessarily achieved.

2.2 Development of a strategy for flexible biogas production

Biogas is the most flexible and controllable form of energy compared to wind and sun. This advantage is used for reducing differences between energy production and consumption. To quantify long term differences, the monthly balances of an energy supply based on 20% energy from wind and 30% energy from sun for the supply region of 1,600 residents were calculated. Figure 1 shows the monthly energy deficits between energy consumption and energy supply without the usage of biogas. The deficits correspond to the shown average power demand which was then adapted by trial and error to enhance the plant utilization. The result of this calculation, as shown in Figure 1, indicates a seasonal pattern of the energy deficit with lower shortfalls in summer time and up to three times bigger deficits in winter.

This seasonal variation was used to generate a demand adapted production of electrical energy from biogas. In this strategy two CHP units are used for the conversion of biogas to electricity. One CHP unit runs on a constant level which equals the energy need in summer and the other one is used to provide the additional energy. To increase the capacity utilization the needed power resulting from the monthly energy deficit was adapted. The total power of both CHP units is shown in Figure 1 by black columns. To enhance the supply security, the power level of the constantly running CHP unit can be adapted if the SOC (state of charge) of the battery is considered critical.

When planning the strategy for the biogas production in the context of this flexible energy production the microbiological limitations have to be taken into account. Each variation of the boundary conditions for the biogas process means a risk for the stability of the biogas production. Therefore, an adapted operating plan for the biogas plant has been developed which includes just two different production levels and two phases for the adjustment in the period of one year.

Two different types of substrates have been chosen for this biogas strategy. One type of substrate is used with a constant amount over the whole year on the level required in summer. The other type is used additionally during the higher production level in winter. The substrate for the constant feed is biowaste to lower the impact for the environment. As additional substrate storable material like corn silage has to be used because the needed amount varies over the year.

3. IMPACT ANALYSIS OF THE FLEXIBLE BIOGAS STRATEGY ON SUPPLY SECURITY

As the aim of the new strategy for biogas production and conversion is the enhancement of the supply securi-

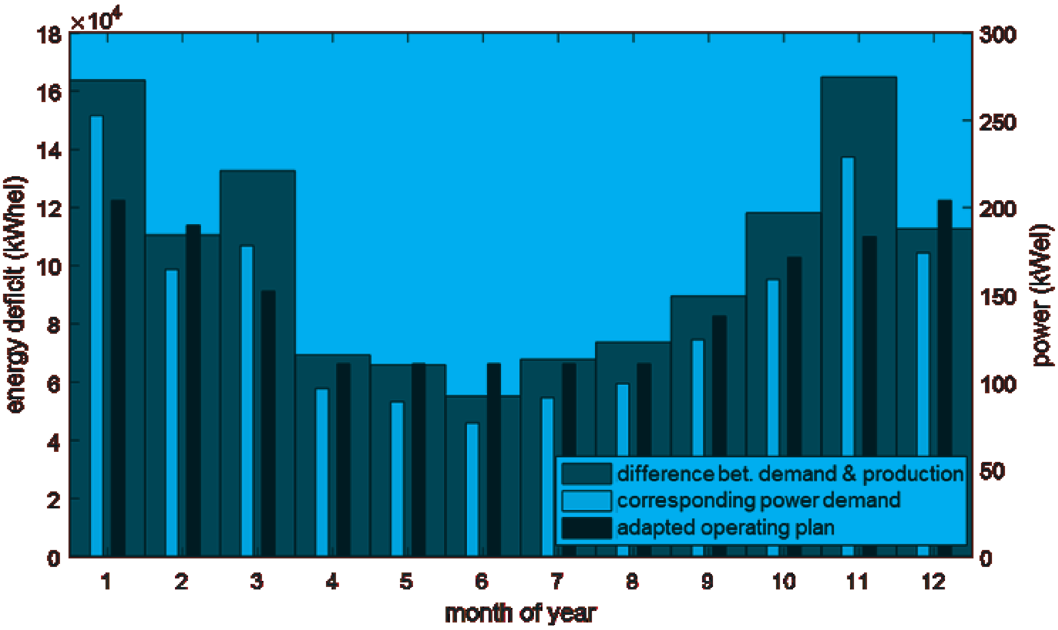

FIGURE 1: Monthly energy deficit between an energy consumption by 1,600 residents and an energy supply based on 20% energy from wind and 30% energy from sun in comparison to the resulting power demand and adapted power level.

ty, this impact was analyzed with the developed MATLAB tool by comparing this strategy with a biogas production and conversion on a constant level which equals the annual average of the planned CHP power levels. The chosen parameter for this comparison is the course of the battery SOC over the time because with this device fluctuations between energy production and consumption are balanced. The boundary conditions set for both calculations are the same as shown in Table 1.

Boundary conditions for the comparison of flexible and constant biogas production and conversion

Figure 2 shows the course of the battery SOC in the frame of a flexible biogas production and consumption. The CHP 1 is used to stabilize the SOC in the range between 20% and 80% and is adapted if the SOC is out of the range of 30% to 70%. For this stabilization the power level of the CHP unit can be varied between +/- 30%. In this scenario the minimum SOC is 22% and the maximum is 72% which is in the acceptable range of a Lithium-Ion battery (VDE, 2015).

The results presented in Figure 3 are calculated with a constant level of biogas production and conversion. Although the energy balance over the year is just slightly negative, the SOC drops below 20% and rises above 80% which damages a Lithium-Ion battery (VDE, 2015). Besides that, the SOC level even drops below 0% which would result in a breakdown of the energy supply (unlike real batteries the SOC of the simulated battery is not fixed to certain values to clearly show the effects of the operation strategies within a 100% renewable energy production).

This comparison with this configuration clearly indicates a positive effect of the flexible biogas production and conversion. The result of the simulation involving constant biogas production shows also that bigger battery capacities are necessary to balance the increased range between the minimum and maximum of surplus energy (147 MWh

TABLE 1: Boundary conditions for the comparison of flexible and constant biogas production and conversion.

Parameter	Configuration
Relation of energy from wind to sun	2:3
Relation of energy from wind and sun to biogas	1:1
Maximum power of the wind turbine	750 kWel
Maximum power of the photovoltaic system	600 kWel
Battery capacity	23 days of average consumption (105 MWh)
Efficiency of battery (filling / emptying)	0.9 / 0.9
Initial SOC of battery	40%
Efficiency of CHP units (electrical)	0.4
Calorific value of biogas	6 kWh/Nm³
Planned energy production from biogas per year	1182 MWhel
Energy demand per year	2103 MWhel (Private households, 1,600 residents)

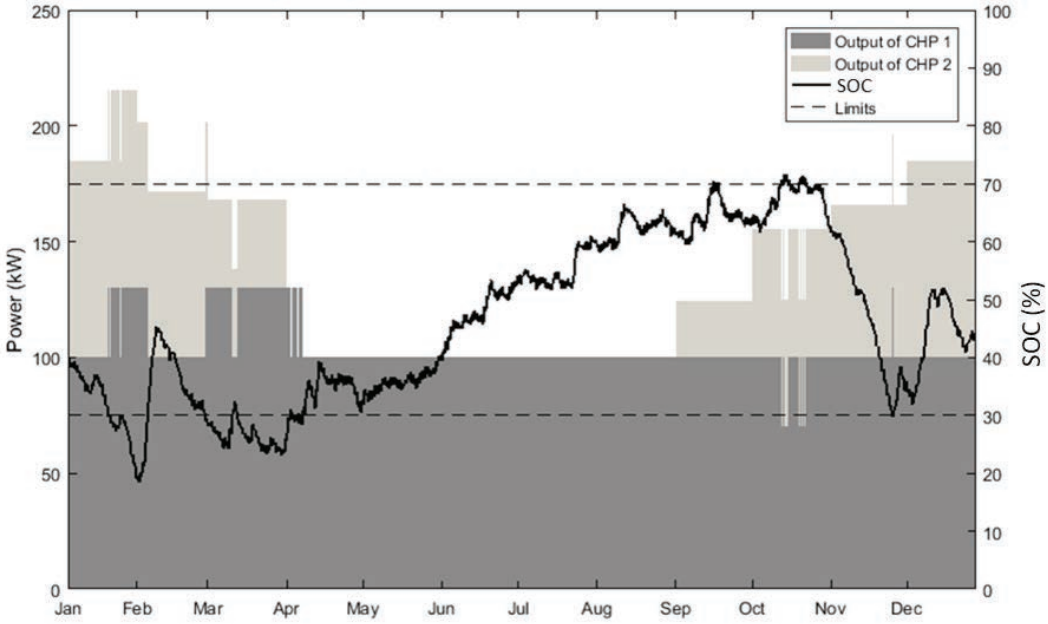

FIGURE 2: Course of the battery SOC and output of the two CHP units with flexible biogas production and conversion (resolution 1 minute).

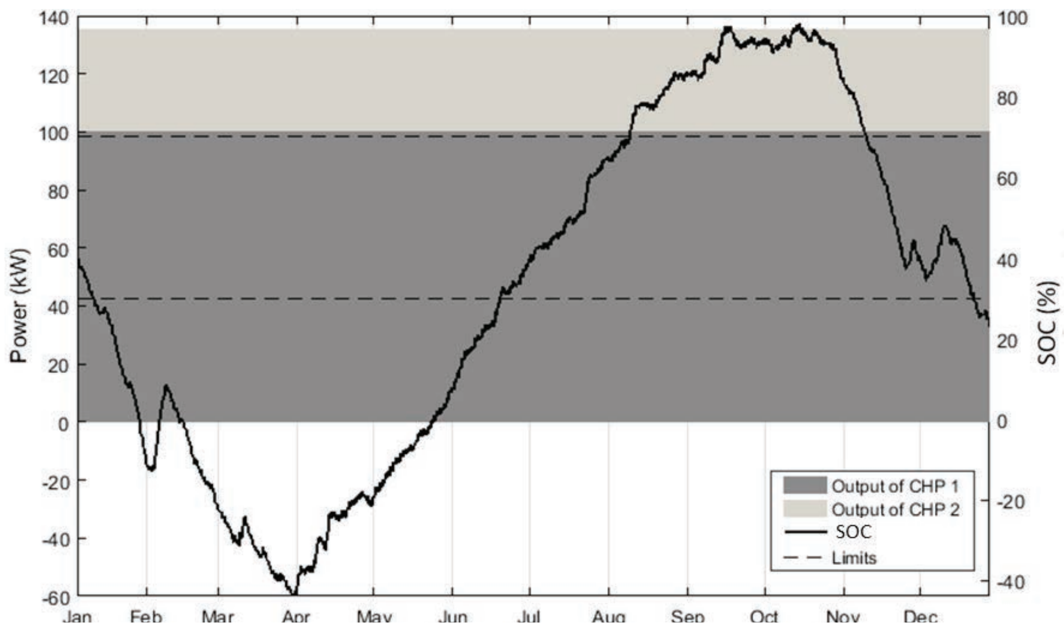

FIGURE 3: Course of the battery SOC and output of the two CHP units with constant biogas production and conversion (resolution 1 minute).

instead of 52.5 MWh).

Due to the operating range of a Lithium-Ion battery, a capacity of 245 GWh would be necessary to compensate the fluctuations which means an additional demand of 133%.

Another analysis that was done with the developed MATLAB tool is the impact of the variation of the share of biogas energy in the energy mix on the required battery capacity as shown in Figure 4. The flexible biogas strategy was used for this calculation. The required capacity drops significantly with rising shares of biogas but rises again after the optimum at 60% biogas in the mix has been reached. This development results on the one hand from the seasonal fluctuations of the deficit between energy consumption and energy production by the wind turbine and photovoltaic system and on the other hand from the biogas strategy based on this fluctuation instead on the fluctuation of the energy consumption. With regard to the dimension of the battery it has to be pointed out that the aim of this study was the analysis of the influence of a new biogas production approach on the battery capacity in a completely renewable energy system.

In order to test and evaluate the practicability of the described strategy, preliminary tests with regards to the flexible biogas production were performed in pilot scale. These are described in the next chapter.

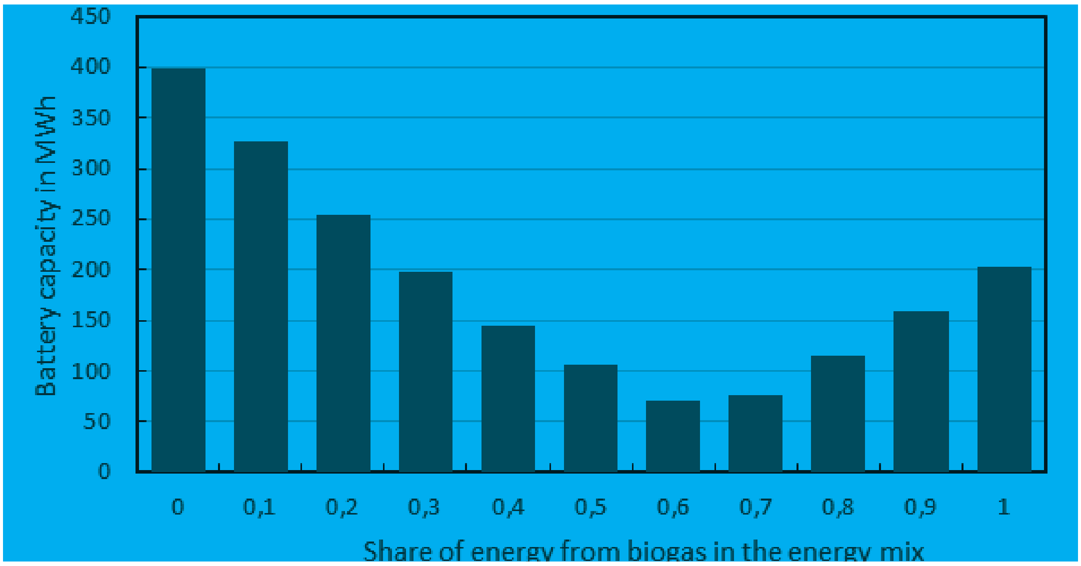

FIGURE 4: Minimum capacity of the Lithium-Ion battery in this scenario with different energy shares from biogas.

4. FLEXIBLE OPERATION MODE OF ANAEROBIC DIGESTION

The concept of a flexible biogas production adapted to the needs of renewable energy systems has to be verified not only on a simulative basis but also in practical systems. Mauky, Jacobi, Liebetrau, and Nelles (2015) showed that in lab scale a flexible daily feeding protocol did not influence the process negatively. As the focus of this study is not on daily fluctuations but rather on changes over a year, it is assumed that increasing or decreasing the biogas production over a month should not have a negative effect. Preliminary results are presented here.

4.1 Testing of flexible operation mode in a pilot scale plug-flow digester

The self-designed flow-plug digester Pilot B (Freidank, Daukšys, & Ahrens, 2013; Freidank, Drescher-Hartung et al., 2013) was used to study the effect of increased feeding over a relatively short time. In short, the reactor had a length of 2850 mm, width of 420 mm, height of 700 mm, and a volume of 500 l. Substrate was supplied by a supply hopper and fed in by means of a spiral conveyor. The entry point was slanted at an angle of 30°. An overflow box was built at the end of the reactor with an adjustable overflow level between 450 mm and 600 mm. The digest was mixed and moved through the digester by 4 horizontally arranged agitators. A ball valve was used for sampling. The produced biogas was collected in 3 air-tight bags with a volume of 250 l each. Using a drum-type gas meter (TG3-PP-PP Ritter) the volume was measured. The process was carried out under mesophilic conditions (42°C). A detailed description can be found in Freidank, Daukšys et al. (2013) and Freidank, Drescher-Hartung et al. (2013).

Substrates were chosen according to their availability within a Lithuanian model region. Cow manure was provided by a local farm, which did not use antibiotics in order to prevent a negative influence on the biogas process. Food waste was collected from three different day care facilities and mainly comprised of starchy foods, vegetables and small amounts of animal protein. Before use it was sterilized in a pressure cooker for 15 min. The distillery waste was provided by AB Biofuture. Algae were collected from the Curonian Lagoon's water surface and its coastal zone. The fresh algae were dried following collection (Freidank, Drescher-Hartung et al., 2013).

At the beginning of the studies the digestion was started out with cow manure and distillery waste. Due to the increase of the water content of the distillery waste, the use was discontinued. After five weeks food waste and algae biomass was added. During the last weeks only cow manure was used as substrate. Overall the amount of substrate was increased over time. The exact composition can be found in Figure 5.

In order to make a statement regarding the process stability VOA/TIC (VOA= volatile organic acids, TIC = total inorganic carbon) was measured according to Rieger and Weiland (2006).

4.2 Results of flexible operation mode

Results clearly showed that changing substrates within a few weeks does not have a negative influence on the process stability (Figure 6). The high VOA/TIC at the beginning showed that the initial amount of substrate was too high. Nonetheless, the value recovered over the next weeks. Indeed VOA/TIC results indicated that the process was underfed toward the end. In further studies the loading rate should be increased. Also, the focus should be more on different organic waste material as these are degraded more quickly which can lead to high VOA/TIC.

Furthermore, lab scale tests were run during the ABOWE project (Freidank, Drescher-Hartung et al., 2013). Substrates used here were cow manure, distillery waste, food waste and algae biomass. The weekly feed started out with 190 g cow manure which was increased to 1250 g within two weeks. During week 3 and 8 distillery waste was

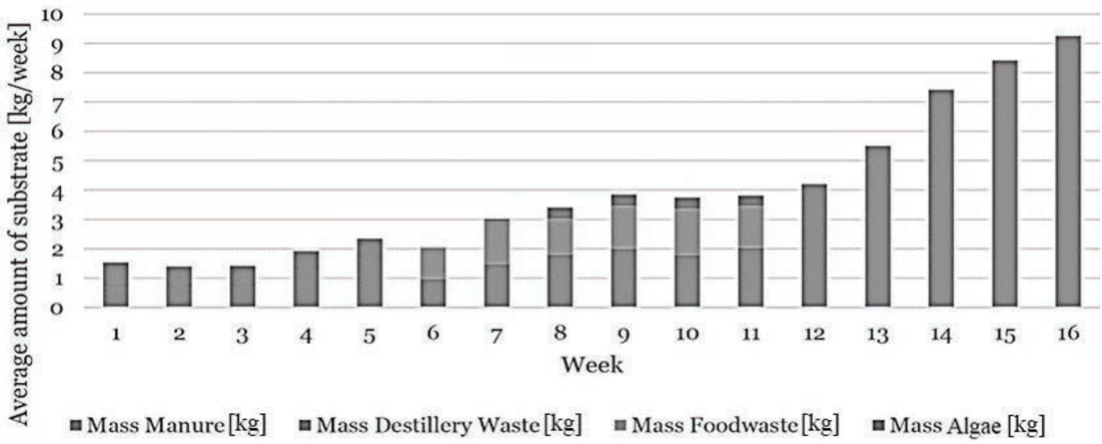

FIGURE 5: Feeding substrate composition of feed during testing of flexible operation mode. Taken with approval from Freidank, Drescher-Hartung et al. (2013)

added with varying quantities. The percentage of distillery waste was increased from approx. 15% to 50%. Following this, food waste was added as well. Approx. 10% were added with the rest being equally split between cow manure and distillery waste with a weekly feeding of around 1050 g. This was increased to a feeding amount of 1500 g from week 9 to 10. The weekly feeding was then kept at a steady level with equally split amounts between the three substrates between weeks 10 to 13 with a small amount of algae biomass during week 12 and 13. During the last week the weekly feeding amount was slightly reduced to 1350 g.

Results showed that once the methane production started after week 2, the amount slowly increased from approx. Vn = 0.020 m³ CH_4/week to Vn = 0.030 m³ CH_4/week within 4 weeks with small dips due to the change of substrate. Nonetheless, this showed that the process was kept stable. Even after adding food waste the process kept stable and the amount of produced biomethane increased to almost Vn = 0.070 m³ CH_4/week in week 10, thus showing that the production can be doubled within only one week. The biomethane amount kept on rising, reaching a maximum in week 12. This clearly indicates a stable process. Following the reduction of feed, the produced amount was reduced. Still, as there were no sudden drops in production rates, it can be said that adding organic waste and changing substrates within weeks does not bring the process to a halt.

These results will have to be applied to the pilot plant Pilot B by changing the previously used feeding strategy to show that this is also possible in a larger scale. Here the aim will also be to see whether the process can be kept stable over a longer period of time as the previous studies stopped after week 14. Furthermore, more organic waste types will be looked at thus creating a broader spectrum of the effects of different substrates on possible optimization strategies.

One of the main concerns of increasing the loading rate too fast and possibly using bio waste with unknown compositions is, besides over-acidification, foam formation. Extensive studies already exist (Lienen et al., 2013; L. Moeller, Goersch, Neuhaus, Zehnsdorf, & Mueller, 2012; M. Moeller, Görsch, Müller, & Zehnsdorf, 2012). Preliminary studies of the author and the research group have demonstrated that foaming became a serious problem in lab scale stirring-tank reactors with a load rate of 5.88 kg oDM /(m³*d) (oDM = organic dry matter). This resulted in the foam clogging up the tubing. As the biogas could not escape the reactor the sampling plug was pressed out under strong force.

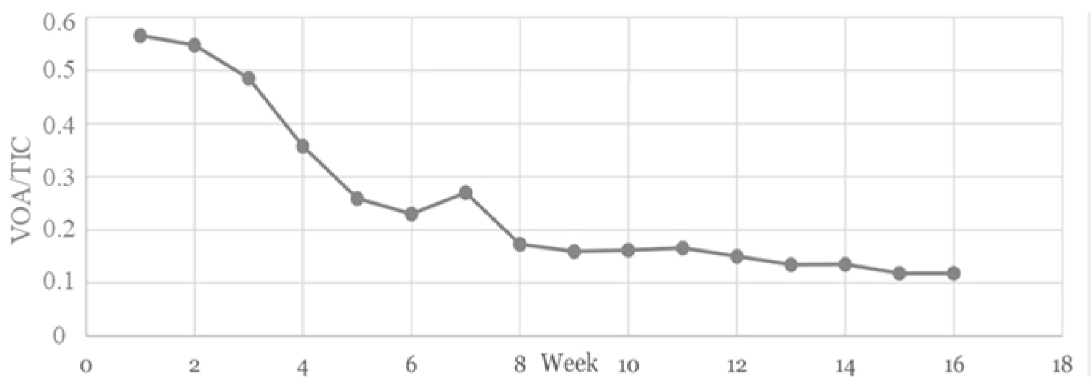

FIGURE 6: Results of VOA/TIC over the course of the investigation. VOA/TIC was high in the beginning due to too high about of starting feed. This recovered over time. Towards the end VOA/TIC values were too low indicating that the system was underfed. Taken with approval from Freidank, Drescher-Hartung et al. (2013)

In order to keep this from happening while still having high loading rates, anti-foaming solutions have to be found. One new approach will be to look into the effect of condensed tannins with focusing on sainfoin. Positive results on the reduction of bloating has been seen in ruminants (Patra & Saxena, 2011).

While studying the effects of a flexible operation mode on biomethane production and using the data to feed into the newly developed energy management tool is important, it is just as important to critically evaluate the economic aspects as well. This will be discussed in the next chapter.

5. CALCULATION OF THE BIOGAS PLANT PROFITABILITY

Consequently, the profitability of the biogas plant which would be necessary for the realization of the above described scenario has been evaluated. Therefore, the amounts of substrates which were determined in the simulation were used for calculating a simplified economic model of a biogas plant. Also, several important conditions for the operation of such a biogas plant, which have to be taken into account, were examined and determined (e.g. plant system, necessary reinvestments or discount rate), but will not be presented in detail.

In general and in the current situation, the implementation of biogas technology without any public governmental funding and under consideration of a demand based plant operation will result in new challenges concerning the profitability. Implementing biogas technology requires the assessment of many aspects concerning the economic efficiency.

The target parameter is an operative cash flow (OCF), whose calculation is based on the difference between annual revenues on the one hand and annual expenses plus investments on the other hand. For the design of the biogas plant the investment costs and operational plant costs must be considered.

For the calculation of these cash flows a model was developed at Ostfalia University of Applied Sciences, which allows estimating the profitability and economic development over a long time period on the basis of the cumulative discounted cash flow.

However, a detailed calculation and estimate of cash flows is only possible by defining real system models. Therefore, the model includes results of pilot plant operation in certain EU countries, lab tests at Ostfalia University and extensive collection of data (own investigations, literature sources and data from biogas plant operators).

The chosen scenarios and calculated model biogas plant constitute theoretical exemplary situations. The corresponding assumptions are not meant for a generalized use, as they depend on scenario-specific aspects and were only chosen for these scenarios.

Nevertheless, all variables that influence the profitability can easily be changed and adapted to other scenarios and regions. The resulting impacts can then be illustrated as described above.

For the biogas plant, which has to be implemented into the above-mentioned scenario, the following results concerning the profitability have been calculated (see case-study in Drescher-Hartung, Stasiškienė, and Ahrens (2018)).

With respect to the above described electricity supply scenario a theoretical production of about 250,000 m³ CH_4/year would be possible from a substrate amount of 2,004 tons of biowaste per year (which cannot be stored) and an adapted feeding of corn silage of 680 tons/year.

Table 2 shows the assumptions which were made for the calculation of the profitability.

Assuming that the produced electricity could be sold at a feed-in tariff of 0.247 €/kWhel (which could be the electricity rate currently to be paid by customers in Germany) (Stadtwerke Wolfenbüttel, 2017), a feed-in tariff for heat of 0.03 €/kWh (Herbes, Halbherr, & Braun, 2018), an income from the biowaste collection and utilization of assumed 40 €/ton and expenses for the corn silage of assumed 30 €/ton, the following cash flow would result (Figure 7). For this scenario the payback would be reached after about 6 years.

In comparison to that scenario a second one has been calculated for the case that the electricity producer has to pay grid fees and taxes (then the attainable tariff could be 0.078 €/kWhel) which means that there are no financial benefits. In that case the payback would not be achieved within the considered period (Drescher-Hartung et al., 2018).

As the results show, the payback period is fundamentally dependent on the earning which is achievable from the substrate and the sale of the electricity. That means the described model is strongly dependent on the tariffs for the electricity, the heat and possibly the digestate as fertilizer. These parameters have to be checked carefully before installing the biogas plant.

In addition, it can be seen from the two scenarios that the operation of the biogas plant is only possible with relevant benefits concerning the electricity feed-in tariff. Therefore, the advantages of the biogas production, which make it possible to balance the electricity production and therefore to provide e.g. the opportunity to reduce necessary energy storage, have to be considered very positive.

TABLE 2: Methane potential and quantities used for the calculation of the profitability

Substrate	Methane potential Vn (m³ CH_4/Mg FM) (FNR, 2017)	Quantity used (Mg/a) [1] (Drescher-Hartung et al., 2018)
Biowaste	102	2,004
Corn silage	104	680

[1] *Calculated with the simulation tool*

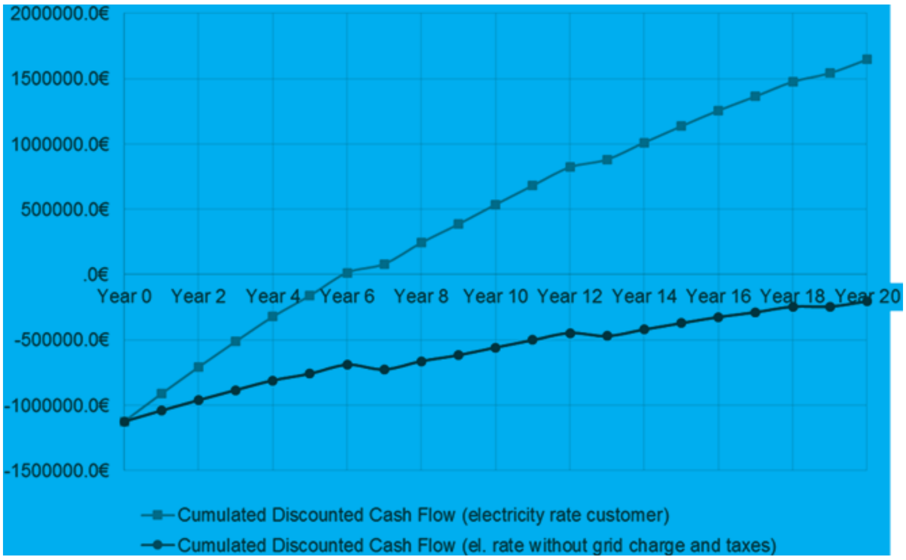

FIGURE 7: Cumulative discounted cash flow of two scenarios (Drescher-Hartung et al., 2018).

6. CONCLUSION

Simulations in this study have shown that using a flexible and demand-adapted biogas management can significantly reduce the required battery capacity. Nonetheless, the combination with feed-in management of wind and solar energy has to be investigated. Furthermore, the optimal technical design of storage systems will be taken into account.

In order to apply the results of the simulation to real life scenarios, flexible biogas production was studied. Results showed that increased feed with different compositions of substrates did not have a negative effect on the biogas process stability and resulted in higher biomethane production. Future studies will focus on increasing the loading rate as well as looking at a wider range of substrates where the emphasis should be on different organic wastes. Currently, studies are being carried out looking into the positive effects of sanfoin, as a co-substrate, on the reduction of foaming. This is a known problem that can occur during the biogas production especially when using easily degradable substrates such as biowaste at high loading rates. In order to get a first insight into the economic feasibility an economic analysis was performed. The results have shown that the concept can be profitable. Nonetheless, it is still in a developmental stage.

Although extensive studies are still needed this paper shows that biogas can play an important role in the transition from conventional energy production to a future where the energy demand is completely covered by renewable energy.

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EVALUATING THE CRUCIAL FACTORS AFFECTING HYDROGEN GAS GENERATION FROM MUNICIPAL SOLID WASTE INCINERATION BOTTOM ASH (MSWIBA)

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ABSTRACT

In this study, we examined the factors influencing hydrogen gas generation from municipal solid waste incineration bottom ash and methods to improve this process. A series of mixing and stirring experiments using bottom ash and water were conducted. The reaction temperature, liquid-solid ratio, stirring rate, and presence or absence of a grinding treatment were set as the experimental parameters. According to the results obtained in the present study, the optimum temperature for efficient recovery of hydrogen gas was 50°C. When the liquid-solid ratio was 5, or exceeded 3, more hydrogen gas was generated. When the stirring rate was 600 rpm, or exceeded 400 rpm, more hydrogen gas was produced. When bottom ash was crushed, the initial gradient of hydrogen gas generation dramatically increased.

1. INTRODUCTION

Approximately 44.0 million tons of municipal solid waste (MSW) was generated in Japan in 2015, 80% of which was incinerated. This resulted in the generation of approximately 3.2 million tons of MSW incineration residue, which was disposed of mainly in sanitary landfills (Ministry of the Environment of Japan, 2017). MSW incineration bottom ash (MSWIBA) contains an appreciable amount of metal aluminum that is distributed throughout various particle fractions of bottom ash. (Saffarzadeh et al., 2016; Arumugam, 2016). Hydrogen gas is generated by an aluminum-assisted water splitting reaction, which can be given by the following equation (Armstrong and Braham, 1996; Takatsuki, 1994; Toyofuku, 1989):

$$2Al + 6H_2O \rightarrow 2Al\,(OH)_3 + 3H_2\uparrow \quad (1)$$

Bottom ash imparts alkalinity to the reacting solution because it contains a large amount of Ca compounds. Hydrogen gas is generated through the reaction between water and essentially the metal aluminum in the bottom ash. When the hydration reaction proceeds in MSWIBA, a layer of hydrate is formed on the surface of the metal aluminum (Saffarzadeh et al., 2016), which prevents the aluminum from coming into further contact with the water. This is one of the reasons why hydrogen gas generation decreases over time.

Hydrogen gas is the only carbon-free energy source and its energy potential is higher than that of other known fuels, such as methane, ethane, and gasoline (Marbán & Valdés-Solís, 2006; Marbán et al., 2006; Shinnar, 2003; Granovskii et al, 2006). A newly developed energy source can replace fossil fuels. When hydrogen in fuel cells is subject to direct combustion, it is possible to generate energy without the production of CO_2, because water is the only by-product formed (DeLuchi, 1989; Momirlan & Veziroglu, 2002; Momirlan & Veziroglu, 2005).

If the generated hydrogen gas can be efficiently recovered, MSWIBA may be considered a source of hydrogen for energy production. To make the system of hydrogen gas generation from bottom ash and water practically feasible, it is necessary to reduce the production unit cost. Additionally, it is necessary to understand the most suitable conditions for efficient collection of the hydrogen gas. The most critical parameters are reaction temperature, stirring rate, and liquid-solid ratio. If the amount of water (l/s) required for hydrogen gas generation is reduced, it would be possible to reduce the amount of waste liquid after collection

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of the hydrogen gas. Equipment for mixing and stirring of bottom ash with water is also necessary. The size of the equipment depends on the amount of bottom ash and the time until completion of hydrogen gas generation with mixing and stirring. For these reasons, a shorter mixing and stirring time is important, and it is also necessary to consider accelerating the hydrogen gas generation process.

The purpose of this study was to elucidate the factors influencing the hydrogen gas generation from MSWIBA and to determine whether hydrogen gas generation can be promoted by crushing the bottom ash. A series of mixing and stirring experiments using bottom ash and water were conducted by regulating the reaction temperature, liquid-solid ratio, stirring rate, and the presence or absence of grinding treatment as the experimental parameters.

2. MATERIAL AND METHODS

2.1 Sampling

Bottom ash sample was collected from the incineration facilities S (Stoker-type, 750 t/day) and R (Stoker-type, 900 t/day) located in the F. city in Japan. Bottom ash from the S plant is hereinafter referred to as S-BA and that from the R plant as R-BA. The S-BA was sieved to less than 9.5 mm in diameter. Ferrous metals were manually removed from R-BA by a magnet and subsequently sieved to less than 30 mm in diameter. The R-BA was only used to evaluate the promotion of hydrogen gas generation. The samples were air-dried and then oven-dried at 65°C for 24 hours to remove moisture and minimize the effect of weathering. The samples were dried at 105°C, crushed to less than 0.5 mm, boiled in HCl (1 mol/l) for 30 min and then metal aluminum contents in the samples were measured by an atomic absorption spectrophotometer (UV-1230, Japan, SHIMADZU CORPORATION). The S-BA and R-BA contained 3.9 and 3.8% of metal aluminum by mass percentage, respectively.

To enhance hydrogen gas generation, S-BA was crushed to less than 45 μm by a vibration mill and R-BA was crushed to 4.2 μm in average particle size by a cutting mill and a jet mill. A cutting mill that was a crusher using blades, was used to reduce sample particle size to be suitable for the jet mill. The jet mill was a crusher using a high speed jet of compressed air. Sample particles were crushed by mutual contact. Particles that could not be crushed during the milling process were excluded from the experiments because it was presumed that the crushed and non-crushed bottom ash samples had the same metal aluminum contents.

2.2 Stirring and mixing experiment using bottom ash and water

Figure 1 shows a schematic diagram of the experimental setup, which included a reactor, liquid collection bottles, hydrogen gas concentration sensor, and flowmeter, all of which were connected by the PVC tubes. The bottom ash and water were mixed and stirred in the reactor, which could maintain a constant temperature and change the stirring rate. Hydrogen gas, generated in the reactor, flowed through the tube and passed through a liquid collection bottle to cool it and condense water vapor. The gas passed

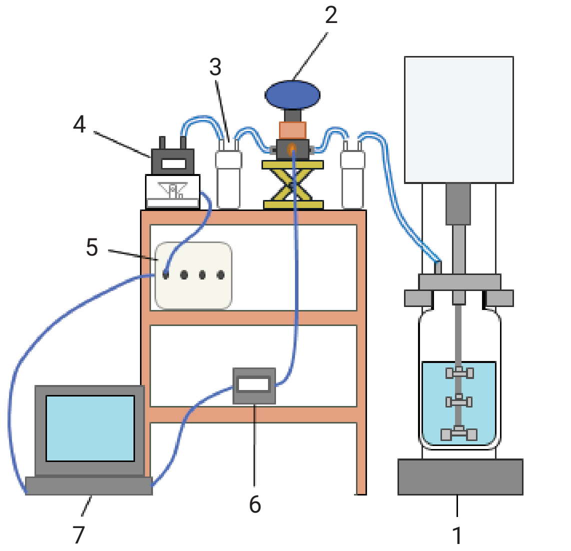

FIGURE 1: Schematic diagram of the experimental setup. 1) Reactor, 2) Sensor of hydrogen gas concentration, 3) Liquid collection bottle, 4) Flowmeter, 5) Data logger, 6) Display of hydrogen gas concentration, 7) PC.

through the sensor, followed by the liquid collection bottle, which prevented the breakdown of the sensor from the backflow of the flowmeter oil, and finally the flowmeter. Hydrogen gas concentration was measured at intervals of 1 minute during the experiment. The amount of gas generated was measured every 3.24 mL which was the collection capacity of the flowmeter. In the experiments, 500 g of the samples were used. Before the experiment began, the gas in the entire setup was substituted with nitrogen by purging 1 L/min of nitrogen gas for 5 min. Table 1 shows the experimental conditions.

3. RESULTS AND DISCUSSION

3.1 Consideration of the factors that influence hydrogen gas generation

3.1.1 Influence by reaction temperature

Figure 2 shows the relationship between hydrogen gas generation and time. At 50°C, the highest amount of hydrogen generated was 11.4 m^3/t-ash, and the shortest experiment time of 13.0 days was achieved. At 40°C, 10.4 m^3/t-ash of hydrogen gas was generated over 21.0 days. At 60°C, the amount of hydrogen gas generated was 10.2 m^3/t-ash over 20.5 days. According to the research of Zhao, Z et al., 2011, the chemical reaction rate increases when temperature increase. However, when temperature increase, also the concentration of OH- is decrease lead to the reduce of hydrogen generated. And the chemical reaction is heat dissipation reaction. For these reasons, the same phenomenon occurs in this study, at 50°C the amount of hydrogen generated was the highest, compare to 40°C and 60°C.

3.1.2 Influence of the liquid-solid ratio (l/s)

According to Figure 3, at l/s = 3, 7.9 m^3/t-ash of hydrogen gas was generated over 14.6 days. This was about

TABLE 1: Experimental conditions.

Case	Sample	Reaction Temperature (°C)	l/s	Stirring rate (rpm)	Crushing process
1		40			
2		50	5	600	
3		60			Non-crushed
4	S-BA		3		
5				400	
6		50		800	
7			5		Crushed
8				600	Non-crushed
9	R-BA				Crushed

3.5 m³/t-ash lower than that produced at l/s = 5. At l/s = 3, the precipitation of bottom ash particles was identified in the reactor. This could be because the mixing of bottom ash with water was insufficient at lower l/s. Therefore, l/s should be greater than 3.

3.1.3 Influence of stirring rate

As displayed in Figure 4, the highest amount of hydrogen gas generated was 11.4 m³/t-ash at 600 rpm. The second highest amount of hydrogen gas generated was 11.0 m³/t-ash at 800 rpm. The lowest amount of hydrogen gas generated was 6.8 m³/t-ash at 400 rpm. At 400 rpm, bottom ash and water were not sufficiently stirred and mixed in the reactor, as was the case when l/s = 3 (50°C, 600 rpm). The amount of bottom ash precipitated in the solution increased, which is why hydrogen gas generation decreased. The stirring rate should, therefore, be greater than 400 rpm. It is possible to achieve a practical hydrogen gas generation volume and speed even at 400 rpm or less if the stirring efficiency of bottom ash and water increases by optimizing the shapes of the stirring blades and reactor.

3.2 Promotion of hydrogen gas generation by crushing

In the case of S-BA, 6.6 m³/t-ash of hydrogen gas was generated from crushed S-BA and water. This was approximately 4.8 m³/t-ash lower than that for the non-crushed bottom ash (Figure 5). In the case of R-BA, 5.1 m³/t-ash of hydrogen gas was generated from crushed R-BA and water. This was approximately 7.6 m³/t-ash less than for the non-crushed bottom ash (Figure 6).

Conversely, the initial gradient of hydrogen gas generation (i.e. from 0 to 20%) increased. In the case of S-BA, this was from 3.8 m³/t-ash/day when non-crushed S-BA was used to 274.5 m³/t-ash/day when crushed S-BA was used. In the case of R-BA, this decreased from 8.2 m³/t-ash/day for non-crushed R-BA to 135.1 m³/t-ash/day for crushed R-BA. The initial gradient of hydrogen gas generation increased significantly by crushing; however, the cumulative amount of hydrogen gas generation decreased. Because of the rigorous crushing of BA, metal aluminum that is presumed to trigger hydrogen generation has turned to very fine particulates with significantly large surface areas. As a

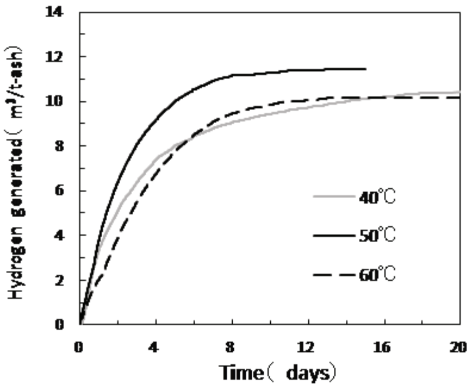

FIGURE 2: Influence of reaction temperature.

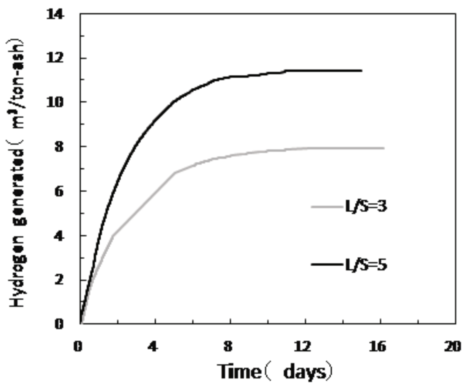

FIGURE 3: Influence of liquid solid ratio.

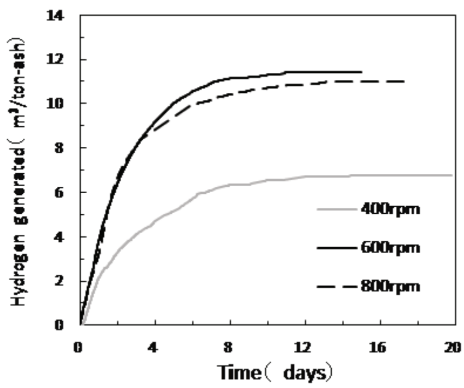

FIGURE 4: Influence of stirring rate.

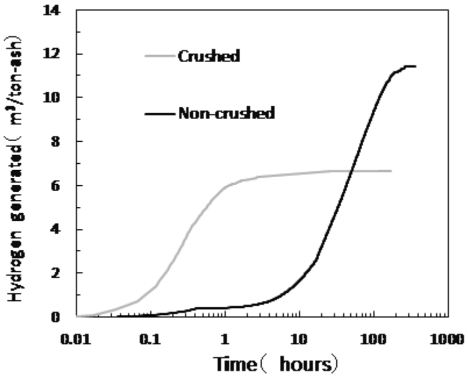

FIGURE 5: Promotion effect by crushing process (S-BA).

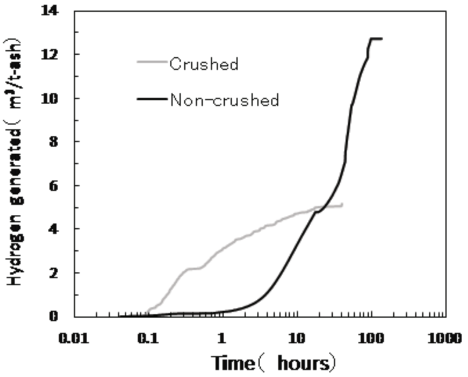

FIGURE 6: Promotion effect by crushing process (R-BA).

result, the corrosion rate of such fine metallic particulates has considerably accelerated and they rapidly consumed at the beginning of the reactions. Therefore, almost no or little reactive metal aluminum remained for the advanced steps of the experiment that resulted in lower cumulative hydrogen in the crushed BA.

3.3 Relationship between hydrogen gas generation and pH

Table 2 shows the volume of hydrogen gas generated and the pH of the liquid phase when the experiment completed. At the end of the experiment, the pH was within 10.9–12.0 when S-BA and R-BA were used. There is no noticeable relationship with reaction temperature or stirring rate. Mixing and stirring were insufficient for cases 4 and 5, which had comparatively low pH values of 11.5 and 11.4. When S-BA was crushed, the pH was 12.0, which is comparatively high. Because S-BA had a higher pH compared with R-BA, hydrogen generation was strongly influenced by factors other than pH.

3.4 Comparison with theoretical hydrogen gas generation and utility value by engineering

The S-BA and R-BA contained 3.9% and 3.8% of metal aluminum by mass percentage, respectively – if all of the aluminum in the bottom ash reacted then this would correspondingly result in 48.5 and 47.3 m³/t-ash of hydrogen gas. However, the highest volume of hydrogen gas generated in this study was 12.7 m³/t-ash (Case 8). Comparing the experimental values with theoretical values, hydrogen gas yield of about 26% was achieved. According to Macanas et al. (2011), an $NaBO_2$ solution is more effective for hydrogen gas generation than other solutions. If a small amount of 0.01 M NaF, MgCl, or $Fe_2(SO_4)_3$ is added to metal aluminum and water, the amount of hydrogen gas generated increases. Thus, an increase in hydrogen gas generation is expected from those additives.

By crushing S-BA, 6.6 m³/t-ash of hydrogen gas was generated from S-BA and water within 1 day. Approximately 15,600 tons of bottom and fly ash is generated annually from the S Incineration plant of F city, i.e., we could obtain about 25 kg (282 m³) of hydrogen gas in a day. Assuming the mileage of a fuel-cell car is 83 km/kg-H_2, a fuel-cell car could travel for over 2000 km using the recovered hydrogen gas. This distance is sufficient for fueling a garbage truck, for example. Owing to these reasons, a hydrogen gas recovery system using crushed bottom ash and water is valuable from an engineering standpoint.

4. CONCLUSIONS

1) In these experiments, the highest volume of hydrogen generated was 12.7 m³/t-ash at case 8 (a reaction temperature of 50°C, liquid-solid ratio of 5, stirring rate of 600 rpm, and with non-crushed R-BA).
2) The optimum temperature for hydrogen generation from MSWIBA exists. In this study, the highest volume

TABLE 2: Amount of hydrogen gas generated and pH after the experiment completed. Cases1-7 represent S-BA and 8-9 represent R-BA.

Case	Hydrogen gas generation time (days)	Hydrogen gas generation amount (m³/t)	pH after experiment
1	21.0	10.4	11.4
2	13.0	11.4	11.8
3	20.5	10.2	11.7
4	13.1	7.9	11.5
5	15.0	6.8	11.4
6	15.1	11.0	11.4
7	1.1	6.6	12.0
8	4.1	12.7	11.0
9	1.8	5.1	10.9

of hydrogen gas was generated at 50°C.

3) When l/s = 5 more hydrogen gas was generated than when l/s = 3 scenario. The reason for the lower amount of hydrogen gas generated could have been that mixing between bottom ash and water was insufficient due to a water shortage at l/s = 3.
4) At stirring rates of 600 and 800 rpm more hydrogen gas was generated than at a stirring rate of 400 rpm. This could be because bottom ash and water were not well mixed at 400 rpm.
5) By crushing bottom ash, the initial gradient of hydrogen gas generation dramatically increased, but the cumulative volume produced did not increase. In this study, the initial gradient of hydrogen gas generation increased from 3.8 to 274.5 m^3/t-ash/day in the case of S-BA, and from 8.2 to 135.1 m^3/t-ash/day in the case of R-BA.
6) In this study, the maximum amount of hydrogen gas yield was about 26% of the theoretical hydrogen gas generation amount.

From the above, to generate a large volume of hydrogen gas within a short period of time by mixing and stirring bottom ash with water, it is desirable to set the reaction temperature to at least 50°C, and the minimum stirring rate of 600 rpm. If the shape of the stirring blade and the reactor are optimized, however, even if the liquid-solid ratio is reduced to less than 5, or the stirring rate is reduced to less than 600 rpm, the amount of hydrogen gas generation would not decrease as it did in this study.

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RECREATIONAL DEVELOPMENT OF OLD LANDFILL: THE CASE STUDY OF GÓRKA ROGOWSKA LANDFILL IN ŁÓDŹ CITY, POLAND

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ABSTRACT

In recent years in Europe and the United States, there has been a noticeable increase in urban green areas, some of which are built on post-industrial areas, particularly those affected by unemployment and requiring broad structural change (Pancewicz, 2011). Decisions on the location of parks in degraded areas are mainly due to the lack of free space in the city (esp. in downtown zone). Therefore, they are beginning to be designed in suburban areas, often within larger complexes like forests, to protect open areas in suburbs of cities. These open areas are subjected to various activities (renovation, revitalization, reclamation, etc.), with the purpose of assigning them new functions in terms of development. Nowadays they are seen as an barren reserves that can contribute to improving the quality of life in the city and enhancing the attractiveness of urban areas and neighboring areas through the implementation of new investments. Among the degraded areas are landfill sites, which, due to the urban sprowl, are entering into conflict with neighboring areas (housing estate areas, open spaces like forests and arable areas), making it difficult for the environment and its inhabitants. One of the examples of such sites is Górka Rogowska in Łódź (Poland), the subject of presented case study. Unfortunately, because of many wrong attempt decisions this area has not been properly reclaimed and developed, and currently is still a social and environmental problem and a threat to the safety of residents. However, the above-mentioned landfill site has a valuable natural potential, which due to its convenient location can be used for tourism, recreation and therapeutic purposes. Thus, the proposed lanscape development can be a challenge to improve the condition of urban environment.

1. INTRODUCTION

Examination of waste dumping grounds and other degraded areas only in the context of environmental hazards seems to be inappropriate. In the course of time, these areas become not only a place of life for many plants and animals, but also they forge readable and original landscape forms, which affect significantly their environment (Kurowski, 1998; Siciński, 2001; Markuszewska, 2009; Pancewicz 2011; Długoński, 2012).

Among the publications devoted to technical methods of disposal such as biological utilization (Selivanovskaya et al., 2006; Das et al., 2015), energy and electric utilization (Rotheut and Quicker 2017), waste gasification / methane production (Behera et al., 2010; Zhen et al., 2016; Kormi et al., 2017; Zhao et al., 2017), characterization of landfill temperatures (Jafari et al., 2017), or waste management of urban and landfill areas in technical way (Teira-Esmatges and Flotats, 2003; Hui et al., 2006; Seadon, 2006; Weng ed., 2011; Slagstad and Brattebø, 2012; Josimović et al., 2015; Silva ed., 2017), there is a short list of publications on residential aspects (Joos et al.1999; Kontos et al., 2005) as well as ecological landuse of the landfills (Siuta et al., 1983; Maciak, 1996; Gasidło, 1998; Koda et al., 1999; 1000 x Landscape Architecture, 2009; Pancewicz, 2011; Długoński, 2012; Pluta, 2014; Chen et al., 2017).

Nowadays, many of the old landfills are becoming increasingly important parts of significant sports and leisure complexes (Pancewicz, 2011; Długoński, 2012). Sometimes they consist of multisection areas as hotel, club, restaurant, sports hall, indoor swimming pool (e.g. Górka Rogowska landfill in Lodz, Poland), or horror park, fantasy park, thrill park, action studio, western city, colosseum, indian village, panoramic restaurant with terrace, military camp, jungle adventure, lunapark, water sports harbor,

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gravity jumping, mobile surfing installation (e.g. Górka Retkińska landfill in Lodz, Poland) and ice and roller skating, ski slope (e.g. Alpincenter in Bottrop, Germany), which often require high financial decisions and make such investments questionable. However, some of the undeveloped areas are also devoted to less costly facilities, which are often accomplished through the large and selfless involvement of the public. Thus, because the landfill sites are often ideally located in city's suburbies near main roads junction, the idea of transforming them only into recreational and smart sport parks, seems to be justified. They can offer residents a variety of activities such as horseback riding, cycling, golfing (e.g. Stockley Park in London, Great Britain) or walking in contact with the surrounding nature (San Giuliano metropolitan park in Venice, Italy). Another aspect is also an usage there selected ecological technologies with a low budget for their implementation (e.g. rockery with selected cover plants), for example protection of site slopes of constructional waste and sourrounding groundwaters (Górka Bemowska landfill design concept in Warsaw, Poland).

Eventually, the landfill areas are important for local residents and fulfill a variety of ecological services for users (Baur ed., 2014; Haase, 2014; Długoński and Szumański 2016a). Moreover, the presence of these objects in the landscape also provides cultural identity of the area and becomes a landmark enabling the simple location of other landscape elements (Lynch, 1960).

The subject of a paper is constructional waste named Górka Rogowska, located in the city of Łódź, Poland. The aim of the paper is to present the ecological ways of old landfill development. The scope of the paper is proper selection of urban plants and spatial land use to transform old landfill of constructional waste into recreational area. The presented paper is a continuation of the author's research on green infrastructure (Długoński, 2014, 2016, 2017; Długoński and Szumański, 2015, 2016a, 2016b) and degraded urban areas (Długoński, 2011, 2012) that require revitalization processes.

1.1 Characteristic of study area

The study area is landifill named Górka Rogowska, with previous waste dumping ground in Łódź city, numerous defective concrete plates used to built skyscrapers in Lodz city. They were stored there from the 60's to the 80's of 20 C. In addition, waste materials derived from old buildings demolition, containing large amounts of debris, were stored. Waste building storage was completed in the early 80s. Due to the nearest housing estate area and a location of primary school, it was decided to cover the landfill with an anthropogenic soil.

1.1.1 Study area location

The Górka Rogowska ladfill is located in the north-eastern part of Łódź city (central Poland) in the suburban zone (Figure 1). The area of the landfill is 7 hectares with the sourrounding area of 20 hectares of open space sourronding the constructional waste (in total 27 hectares). The neighborhood (1-5 km) of the case study site are Łagiewnicki forest (1200 ha), service buildings (primary school, hospice and church) and single family housing estates. The area located near the expressway (road DK14: Sochaczew-Sieradz) and motorway junction (A1: Ostrava-Gdańsk / A2: Berlin-Warszawa).

1.1.2 Study area land use

The Górka Rogowska landfill is an open public space with three distinct inclined slopes, pedestrian-bicycle roads, and a viewing point. The sourrounding area under constructional waste consists of several meadows with natural trees and picnic places for outdoor games during the spring and summer season. The site is partly managed by the Municipality of Łódź city (landfill site). The sourrounding area of the landfill is mostly private or State Treasury property or it has unsettled legal status (APA Projekt Pracownia Architektoniczna 2007, InterSIT, 2017c). The study of conditions and directions of spatial development of the Łódź city (Studium, 2010) dedicates Górka Rogowska landfill as a strategic area, which is important element of functional structure of the city for creating sport and recreation city complex in the future. According to APA Projekt Pracownia Architektoniczna (2007) design concept and Długoński (2012) research, in November and December 2011 the minor renovation work of the landifll site was carried out. The surface of the paths has been replaced, and conrete slopes dehydration have been installed. At the top of the landfill small architectural elements (several benches, rubbish bins, wood barriers with a bicycle rack) has been built. It was decided to give up with rain protecting construction and sports fields under the landfill site. In addition, some of the wild vegetation that limit the view from the hill, has been cut. However, due to the limited budget, it was unable to complete all planned works. Actually, the main problem are still the eroding landfill slopes. Moreover, mainly in the north and east part of the landfill, the ground of slopes is unstable due to pedestrians and cyclists activities. Finally, many behavioral factors negatively affect the

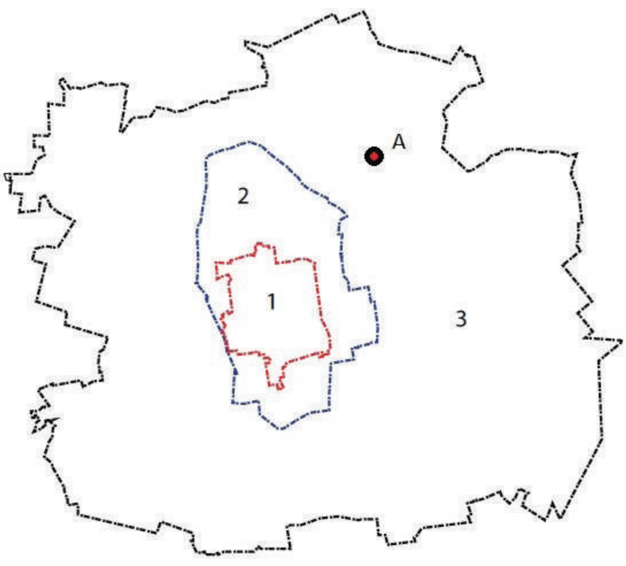

FIGURE 1: Location of Górka Rogowska landfill (A) in Łódź, Poland, 1- downtown zone, 2- residential zone, 3- suburban zone (own elaboration based on: Janiak, 2012; InterSIT, 2017b).

technical condition of existing trees, small architectural elements and stability all the landfill.

2. METHODOLOGY

The methodology consists of several steps. The first was studio research based on literature review.on degraded areas land development and their reclamation (Siuta ed., 1983; Maciak 1996; Pancewicz 2011; Długoński 2012) to select the most suitable solution for recreation of the case study. During this review objects located in cities suburban zone with different ways of their early function and landuse, were choosen.. Then, comparative analysis of development of old landfills was made (Długoński, 2012). It showed a large diversity of landfill sites in terms of their area, waste depositon type as well as the ways of their recreational development and activities for users (Table 1).

Transforming old landfill sites into new functional areas involves an introduction of recreational, sports and cultural activities and should take into consideration the social needs and the individual approach to each case.

The second step was field research with interview with the local community and scientists. The interview with residents of a nearby housing estate area (APA Projekt Pracownia Architektoniczna, 2007) showed their interest in creating a multifunctional park in the study area through practicing sports and active recreation (pro-health role); developing and expanding intellectual and cognitive horizons (cultural development role);and shaping civic attitudes, interest development and ecological education (pro-democracy role).

An additional interview with scientist Dr. Jan Ziomek (geologist) from the Faculty of Geography and Earth Sciences at University of Łódź, one of the few experts remembering the time of its building was conducted in November 2011 (Ziomek, 2011). Due to lack of materials and literature about the landfill, the scientit added most important information about the landfill formation and its development. According to this interview, numerous defective concrete plates, used to build skyscrapers in Lodz city, were stored from the 60's to the 80's of 20th Century. Waste materials derived from old buildings demolition, containing large

TABLE 1: Landfills development of selected landfill sites in Europe (Gasidło, 1998; Koda et al., 1999; 1000 x Landscape Architecture, 2009; Pancewicz, 2011; Długoński 2012; Pluta, 2014).

No.	Landfill	Year of build	Total area (ha)	City's localization	Waste depositon type	Leasure traffiic type	Land use development	Land use elements	Users activities
1	Górka Rogowska in Łódź, Poland	70' of 20 C (previous design concept of another design studio)	27	Suburbies	Constructional waste	Weakly/ yearly	sport and recreational park complex	Hotel, club, indoor swimming pool, sports hall, sports fields, recreation ponds, playgrounds, lookout terraces, bike trails, rainforests, barbecue squares	Cycling, ball games, walking, swimming, leisure, fun and games
2	Górka Retkińska in Łódź, Poland	70' of 20 C (design concept 2010)	35	Suburbies	Constructional and compost waste	Weakly/ yearly	amusement, recreational and sport park complex	Horror park, fantasy park, thrill park, action studio, western city, colosseum, indian village, panoramic restaurant with terrace, military camp, jungle adventure, lunapark, water sports harbor, gravity jumping, mobile surfing installation, alleys with trees	Extremal and professional sports or hobbies, fun, cycling, walking didactics
3	Górka Bemowska in Warsaw, Poland	1961 (design concept 2011)	32	Suburbies	Industrial and compost waste	Weakly/ yearly	sport and recreational park	Ski slope with downhill runs, sports and walking routes, rockery with selected cover plants	Skiing, cycling, walking
4	Stockley Park in London, Great Britain	1985	450	Suburbies	Municipal waste and gravel	Weakly/ yearly	recreational park (landcape)	Water reservoirs, horse riding routes, cycle tracks, gym area, meadow for golf practice, walking and health paths, groups of trees and shrubs	Horseback riding, cycling, golfing, walking
5	Alpincenter in Bottrop, Germany	N.A.	30	Suburbies	Constructional waste	Weakly/ winter season	ski center	Ski slopes, toboggan runk, groups of trees and shrubs	Ice and rolller skiing, skateboarding
6	San Giuliano Metropolitan Park in Venice, Italy	N.A.	N.A.	Suburbies	Muninipal waste	Weakly/ yearly	recreational park (landcape)	Water reservoirs and channels, recreational meadows, squares, groups of trees and shrubs	Walking and contemplation

amounts of debris, were also delivered. Waste building storage was completed in the early 80s. Due to the nearest housing estate area and a location of primary school, it was decided to cover the landfill with an anthropogenic soil..

Then the vegetation inventory (Długoński, 2012; APA Projekt Pracownia Architektoniczna 2007) shows that the area needs to be supplemented with cover vegetation and predisposition to strengthen the slope of the landfill. The colncusion is the new native species of trees and shrubs located in groups at the bottom of the landfill (urban park designing idea, is needed).

The storage stability analysis of Computer Thermography Laboratory DMCS (2011) by thermal imager, was made. The DMCS report confimed data obtained during the interview (Ziomek, 2011) that the landfill was constructed of buildup materials and debris. Because no hazardous materials were found, the facility does not pose a threat to the natural environment.

The landscape connections and terrain shaping analysis showed that the site is valuable view point of Łódź city and the Łódzkie region. Therefore, an important aspect is an arrangement of existing views from the top of the hill (panorama of Łódź city and Łagiewnicki forest) as well as inside the study area.

The users activities analysis showed that the landfill area is mainly used on weekends in the spring and summer. The users are mainly cyclists (downhill cyclists activities) and local residents (walking activity) as well as families and children from nearest school (outdor games). The conclusion is the facility should be diversified with the emphasis on the expansion of sports and recreational activities that could relieve the neighboring Łagiewniki forest from excessive usage (Długoński, 2012). The acoustic climate analysis showed that noise standards (over 65 dB) were permissible (InterSIT, 2017a). For this reason, it is necessary to rebuild the buffer zone with insulating vegetation on the side of high traffic streets.

The final step was sudio design scenarios base on five different scenarios of future landifill development (Długoński 2012). The scenarios diifer spatial arrangement and landscape eleemnts location. Also the programme for each scenario was preprared. The selecged programme based on recreational purposes with properly selected urban vegetation used in difficult urban conditons (like near roads, city centre and deep slopes).

3. RESULTS AND DISCUSSION

3.1 Future landuse of Górka Rogowska landfill

The concept of Górka Rogowska development is a result of author's comprehensive research on functional use of reclaimed (previously degraded) areas and could be included into general green infrastructure concept of Łódź city.The aim of Górka Rogowska landfill design concept is to maintain the original forest and recreational character of the study area in spatial layout composition by creating urban park for different user's activities (Figure 2).

Due to the area location in suburban zone near Łagiewnicki forest, it was decided to maintain the landscape and nature forest character of the area and creating there forest recreation centre. At the bottom of the landfill, the park design was aimed at creating many recreational meadows with natural groups of trees and shrubs, which provides more favorable conditions for the vegetation development, richer plastic effects displaying natural colour plants, wavy terrain as well as greater sense of security and comfort for park users. Because of the fact that the landfill is currently the only viewing point in suburban zone of the Łódź city, it is important to introduce the viewpoints (wooden terraces). For this reason, it was decided to create or maintain distant views (sight axes), which ended with characteristic management elements (i.e. ponds, church, school, hotel located in landfill sourrounding area), or group of high colour trees. The interior views of the study area are shaped by removing less valuable plants showing views and exposing particularly attractive spaces of the site (meadows). In addition, it is expected to gradually replenish the aging trees (*Acer negundo, Populs sp., Fraxinus excelsior, Prunus cerasifera*) with more valuable and long-lasting plant species referring to local environmental conditions (*Pinus sp., Tilia sp., Quercus sp.*) in the future. It is also envisaged to shape the top of the landfill with sufficiently stabilized soil (originating from the excavation of the proposed ponds in sourrounding area) and consolidation and correction of landfill slopes and usage of selected medium-sized bushes and cover plants used in slope constructions in urban green areas These are well-drained, quickly growing and long-lived plants with extensive root system and low care requirements, such as: *Berberis thunbergii, Aeuga reptans, Asarum europaeum, Cornus mas, Corylus avellana, Forsythia x intermedia, Hedera helix, Lonicera xylosteum, Pachysandra terminalis, Sambucus nigra,* and *Vinca minor*. It is worth emphasizing here that another important aspect of the design is creation of suitable drainage system of the landfill that consists of a living beaver drain, especially in the places of the largest runoff on the site slopes. Fascias are attached to healthy woody plant cuttings, slanting diagonally into the soil through the fascia. Such drainage system has a far broader and deeper impact, especially on difficult to dehydrate slopes than ordinary concrete gutters, currently used along sloping parks paths. Additionally it should be noted that the existing landfill slopes can serve as a great area for cycling in spring, summer and autumn season.

The concept assumes to build of three bicycle tracks at the landfill. The tracks, due to the high incline of the elevation (20-50 degrees), will only be made of anthropogenic soil, strewn with gravel. These will consist of properly formed embankments and muld, which vary difficulty level of the routes. The routes will also be completed with profiled bends due to the need to brake the bikes at high speed. However, in winter season the tracks are designing for skiing and toboggan runs.

What is more, under the landfill, the concept is to introduce rehabilitation trails for sport activities (health path, gym area, and meadow for golf practice) with dense plants to make comfort zones.

The design idea is also to introduce an educational path with information boards about fauna and flora adjacent to

FIGURE 2: Land use design concept of Górka Rogowska landfill, as urban park (Długoński 2012).

the Łagiewnicki forest along with stopping points equipped with seats. This site will be able to serve as a point of field lesson for school children.

Becuse of the deficiency of water reservoirs in Łódzkie region the idea of two ponds connected by a geometric channel with a boulevard and usage of water plant species to cleaning the environment was considered. These facilities are able to gather rainwater from the site and landfill and fulfill an ornamental and recreational functions with the extension of the sport function (canoeing, fishing, water bicycles etc).

Due to the observed exceeded noise standards of the area near expressway, creating an buffer zone (groups of trees and shrubs) to protect the area from excessive noise is proposed. These plants are also important in air purification of the planned site (phytoremediation).

As a result, the proposed design concept of recreational development of previous landfill area, expose natural values of the site, protect the environment, and make the object more attractive for different users.

3.2 Development of old landfill as a design challenge

The analysis of Górka Rogowska landfill has shown reclaimed area with high natural potential, however, not properly utilized. The improveing of the visual layout of the site by ecological utilization and adaptation of the site to users needs by appropriate spatial development seems to be very important there.

A literature review of the landfill development approaches shows the wide variety of proposed program elements (Gasidło, 1998; Koda et al., 1999; 1000 x Landscape Architecture, 2009; Pancewicz, 2011; Długoński 2012; Pluta, 2014). However, only a part of them (e.g. forest nature character of the site, recreational facilities for various types of recreation, terraces for observation, bicycle paths, and walking paths) was managed to introduce in the Górka Rogowska landfill design described in this work. On the one hand, economic development seems to be optimal and do not involve high financial investment, such as thrill park, action studio, western city, colosseum, indian village, military camp etc. presented in Górka Retkińska landfill future development. For the same reason, in spite of the favorable shape of landfill, it was not decided to adapt it fully to the skiing areas (e.g. Alpincenter). However, on the basis of these examples, it was managed to adapt the area to cross-country skiing and toboggan runs in the winter. Besides, the ideas used e.g. in Stockley Park were the inspiration for introducing places to practice various types of sport (e.g. cycle tracks, health path, gym area, and meadow for golf practice). These elements proved to be valuable in terms of landfill site program components, like level difficulty of cycling routes. Furthermore, due to existing location of the Górka Rogowska landfill area, they could find a solution in the presented design concept (landfill as urban park area). In addition to the recreational and sport development of the site, it was decided to preserve existing valuable plants and create group of vegetations as well as introduce missing cover plants assuming that the vegetation in this area would be an important element of the stability and purification of the whole system. Introduction of buffer zone in the form of vegetation on the boundary allows the area to be quieted and cleaner for park users.

According to presented methodology, it is possible to draw conclusions about current needs of the local inhabitants (society) and to formulate the program objectives of site development. Thus, based on it a general idea of the concept is: the site should operate yearly and offer many opportunities for spending free time by introducing different elements of recreational development tailored to the needs of different users. Moreover, interviews with experts are a valuable addition to the literature review in the absence of essential information or can confirm the credibility of other information sources or field studies (e.g. Computer Thermography Laboratory DMCS, 2011).

During the studio research and design process, various possibilities of land development were taken into account. An interesting approach, very common in Western Europe is vegetable gardens known also as pocket parks, where the residents of the estate can plant vegetables together. It is good toll used to integrate residents. There is the question, is the landfill site a good place for growing vegetables? If the soil and environment is not hazardous the anser is yes. However, the usage of the study area for vegetable production seems to be unjustified. Firstly, the area is surrounded by a single-family residential housing, where both residents and landfill users have their own backyard gardens. In addition, the idea of common vegetable gardens in the Polish landscape is problematic and such designs often do not work well. On the one hand, plants are often destroyed and not respected by local communities, especially neglected people, which are many in Lodz city. On the other hand, there is a grat opportunity of employment poor people from the city center and suburbies to handle the holiday development area rather than inducing them to run vegetable gardens. Another variant of area design focused on the usage of ornamental plants resistant to difficult urban conditions, including slopes, which seems to be a priority in such a case study, bearing in mind that the area is neglected and overgrown with plants, which poses a health risk to users.

The final concept of Górka Rogowska is principally based on transforming the area into a forest recreation center, with numerous recreational meadows for various recreation types, sports facilities and nature trails, which seems optimal and do not involve the excessive financial expenditures. For the same reason and having regard to the hilly terrain, it was proposed to adapt it partly to skiing (e.g. ski center in Botropp in Germany and Górka Bemowska landfill in Warsaw, Poland) and partly to another sports: golf, jogging, fishing, sailing etc. (e.g. Stockley Park in London, Great Britain), downhill (e.g. Poniatowski Park in Łódź). In terms of spatial aspects it was decided to preserve existing natural structures assuming that largely growing urban plants on this area will be an important element in the stability of the ecological system.

What is also important, soil research seems to be an important issue especially in the context of the selection of the species to pollution issue of the Lodz city and it should certainly be the subject of further research. This should be accompanied by a further step of technical design, with dimensioning and arranging of specific plant species. However, the aim of the research, was to assume the possibility of land development through the usage of plants and recreational elements that can improve the properties of study area by fully usage of its natural potential. It seems that the presented concept of old landfill development can be used as an example in other degraded areas for recreational purposes. Therefore, spatial and ecological solutions presented in this research may find a wider application in other case studies, also those strictly created from urban waste.

4. CONCLUSIONS

The article presents selected ecological solutions for constructional waste landfills recreational development (so called urban park) as an impotant future element of the city's green infrastructure. The ecological development way of municipal landfill development through by selection of plant species to constructional waste stability protection fulfills a role in purification of the cities natural environment. Moreover, the introduction of adequate land use elements in landfill's spatial management through proper landscape development of municipal landfills improves difficult inhabitants living conditions and prevents adjacent urban open areas from progressive degradation caused by unorganized recreational activities. The presented analyses of landfill sites may be helpful in defining ecosystem services of cities degraded areas. The research on ecological utilization of landfill sites, developed on the example of Górka Rogowska landfill, can inspire further research on other objects, and support interdisciplinary research in the field of ecological engineering and waste management, as well as, urban landscape planning. Finally, the research may also fulfiill an important role in supporting and developing tasks of local social and ecological organizations by identifying the importance and benefits of the cities degraded areas as a places for sport, recreation and social well-being.

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ARCHITECTURE AS A DEVICE: THE DESIGN OF WASTE RECYCLING COLLECTION CENTRES

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ABSTRACT

Thinking about the architectural project as a broader process first requires reflecting on the role of architecture in urban transformation. The current debate's main actors are theoreticians and architects, demonstrating that it is possible to direct one's gaze to a new way of conceiving architecture as a device. The proposed topic is related to research conducted on the architecture and design of public spaces for waste recycling collection centres in the Department of Architecture of the University Federico II in Naples, Italy. The main idea is to conceive waste recycling collection centres not only for their service function but for public spaces in the city. The waste collection centres can have new associated functions, such as laboratories, markets, exhibition areas and rooms for educational activities. According to the guidelines of the Campania Region, a recycling collection centre is an area located in the city, and the number of centres is proportional to the number of citizens. According to these guidelines, the project wants to demonstrate that it is possible to avoid the NIMBY (Not In My Backyard) effect and to integrate this inconvenient function within the city centre, connecting these centres with other activities intended for community use that can be integrated with the size and the function of the recycling collection centre. This functional contamination does not mask an inconvenient function but increases the social value of these places. The case studies for designing waste recycling collection centres were developed according to the guidelines of this research and are described in this article.

1. INTRODUCTION

1.1 Architecture and Waste

As cities are rapidly expanding and their populations increasing, the topic of waste and energy is becoming urgent in terms of defining new innovative and sustainable solutions to manage and reduce the amount of waste, particularly through improving citizens' education and habits. To fight the problems connected with global climate change and increasing pollution, the proposed research focuses on one aspect of this problem: to define the role of the architecture and design discipline in waste management as one of the crucial aspects in finding solutions to improve the relation between city users and the city's main infrastructure.

In recent years, the relationship between architecture and waste collection centres has emerged from the partnership between architecture and industry that achieved prominence during the 20th century.

Many architects, such as Walter Gropius, Eero Saarinen and Norman Foster, have developed important buildings that transformed an 'ugly' space into an opportunity, making factories more human (Kara, 2017). The idea that architecture can add value in the design of waste management centres, particularly those in urban cities, started with my 2004 research in Napoli during the last phase of my studies in architecture at the Department of Architecture in the University Federico II of Naples, and in 2016, the idea became the subject for my post-doctoral research on the design of waste recycling collection centres.

Considering everyone's daily lives depend upon industrial and infrastructure facilities and the increasing problems connected with urban design in the Anthropocene era, architects must consider the relation between the community (the users of architecture) and planning and design strategies.

1.2 Architectural Design as an Integration Tool for Different Subjects

Citizens, the economy and the environment are essential elements in an urban project. Architecture must be regarded as an instrument of convergence for the implementation of social, economic and environmental practices, in which the architect's role cannot be separated from the

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social and economic aspects.

Today, planning processes are increasingly linked to, and sometimes dependent upon, economic and political factors, placing the morphological and social aspects in the background. Cities are complex scenarios where it is increasingly difficult to distinguish responsibilities and roles in the management and development of urban regeneration strategies.

From 2004 to 2016, the proposed research had several occasions to reflect on the relation between architecture and community. In 2010, the International Congress of EURAU '10 (European Symposium on Architecture and Urban Research), organised by the Department of Architecture of the University of Naples Federico II, focused on the relationship between the quality in architecture (venustas) and democracy. The Congress, which completed its work in June 2010, wanted to propose a way of building a 'community', contemplating state-of-the-art methods in the European areas of education, research and architectural culture and providing inspiration for the National Research PRIN, which in subsequent years (2012-2013) produced reflections on the role of research in architecture and its 'utility' (PRIN "Research projects of national interest", funded by the the Italian Ministry of Education: "Architettura-mercato-democrazia: come si valuta la "venustas" dell'architettura?").

The 12th Architecture Biennale of 2010, 'People Meets in Architecture', dealt with the same theme and focused on the idea that architecture can be the mirror of collective awareness in a rapidly changing society. Imagining the city of the future means assuming as a fundamental premise that:

> *Architecture is an art that helps building the* res publica, *the spaces in which we live and organize our civilization in which we recognize, that we own without being owners, but that is part of our being men and society. People meet in architecture also means that we become people in architecture; It is in the* res publica *that man crowns his own effort to build society* (Sejima 2010).

The recent 2016 Architecture Biennale, 'Reporting from the Front', underscored and confirmed what was said in the 2010 edition, and it focused on the integration of architecture and other disciplines. The architect's role was deepened in all its different aspects, and architecture worked together with other disciplines in planning and managing the territories, in which the inhabitants and citizens play a central role (and mostly focused on the connection between architecture and community), with respect to the logic architecture had used to conduct its studies during the previous 50 years (Aravena 2016).

Today, an architectural project represents a tool for the convergence of disciplines, where architecture, along with other skills, evolve a city, reorganising the activities and involving citizens in the construction of a society that strives for the proper utilisation of resources and for achieving the common good.

1.3 The Architect's Role in the Climate Change Era

The past 50 years have shown that rapid urban growth, in some cases worrying and uninterrupted, has contributed to drastically reducing the resources available to humans. This uncontrolled growth has led to a series of problems that have contributed to climate change, and these problems will only worsen without a constant and intense reduction in human consumption. Climate change has raised public awareness, changing how one thinks of a city. Now, a city must be conceived as a place of experimentation, where citizens, politicians, investors, administrators and technicians, working together, can help in improving the living conditions of the inhabitants and in achieving a sustainable future.

Climate change studies have considered architecture one of the most important disciplines. Such studies are desperately trying to intervene through the introduction of legal protocols, international agreements and policy strategies, putting the theme of global consumption reduction at the centre of world attention; however, there is no going back, and society must adapt its knowledge to a new way of interpreting current climate issues (Figure 1).

The Urban Age conference, organised by the London School of Economics and Alfred Herrhausen Gesellschaft during the 14th Architecture Biennial of 2016, featured the major players currently involved in the city development debate. The conference was a way to consider some of the questions that concern the modern city (conceived as a set of citizens living in a given environment) both in the present and in the near future. Joan Clos, Director of Habitat III and UN Undersecretary, participated in the conference's concluding session, which anticipated and proposed guidelines for formulating a New Urban Agenda that will help shape the 21st century (Clos, 2016).

The Urban Age conference in Venice dealt with six different themes that includedthe following questions:

1. What are the economic and political forces that shape the urban society?
2. Impact of project design on city expansion: is it possible to control urban growth through the planning process?
3. Adapting and social integration in the city: how can the spaces promote interactions between different cultures?
4. What is the role of architecture? How can architecture respond to social challenges and daily environmental problems? What are the design limits in facing the human condition and nature?

Starting from these considerations, the research focused on the architect's role in managing and organising the city and stressed one of the conference's main interests: the cycle of waste and the management of spaces designed for the collection of the materials to be recycled, or waste recycling collection centres.

Thinking about the architectural project as a broader process first requires a different way of perceiving architecture's role in urban transformation. The current debate's main actors are theoreticians and architects, demonstrating that it is possible to direct one's gaze to a new way of conceiving architecture as a device (MVRDV, 2005). Archi-

FIGURE 1: Studies and researches about climate changes, waste, ecology and architecture (graphic by M.L.Nobile).

tecture becomes not only the design of a new space but a device, the process that drives the action during different phases.

The research presented in this paper focuses on the project's role in the use of public space and, thus, the project's role in its overall development, from the idea to the result, as a line tool applicable in different situations.

2. THE DESIGN OF WASTE RECYCLING COLLECTION CENTRES

2.1 The Architectural Project as a Tool for Collective Practices in the Design of Public 'Infrastructures'

The relationship between 'usefulness' and 'quality' in architecture is clearly evident in some large waste disposal projects (recycling plants and large waste incineration plants), not only for the urban value and the architectural interest of the object itself but particularly for the social and anthropological urban value that this way of 'making architecture' represents for the development of modern cities. Architecture becomes the instrument for communicating to citizens for 'didactic' purposes, teaching it is possible to treat 'waste'.

Usually, the character of such waste management plants is ugly and unwelcoming. Being only functional places, like factories, they are conceived as disconnected from the public, both from the social and visual points of view. In the last year, a Harvard University research entitled 'Architecture and Waste' (Kara, 2017) highlighted several examples and case studies of waste-to-energy plants that integrate public functions to engage communities beyond their core function. This research demonstrated how architecture can help in creating new opportunities. Recent experiments in hybrid architecture integrated recreational activities into the heart of previously uninhabited and largely forbidden public areas. The sky slope of the waste incinerator in Copenhagen, designed by BIG, and the educational centres in Brooklyn recycling facilities are just a few examples of architecture's contribution to the design of these infrastructures: 'These facilities spotlight how architects can create value in otherwise dull and alienated industrial environments, infusing them with a sense of community and purpose. Beyond their aesthetic qualities, these buildings also exceed established standards of environmental performance' (Kara, 2015).

An example of this kind of infrastructure is the Naka Waste Incineration Plant of Hiroshima, which the Japanese call a 'fusion design', marking the collaboration among architects, civil engineers, landscape designers, industrial hygienists and other figures. The Naka plant represents not only one of the most interesting and innovative infrastructures in Japan but also a structure that is capable of connecting the city with the sea (Figure 2).

The aim of the architect, Yoshio Tanigushi, was to bring the visitors (more than 200.000 a year) directly to the seaside, crossing the central corridor, the 'Ecorium', that has a double function: first, to enter and show what happens inside the plant and, second, to cross the enormous terrace that overlooks the sea. The architecture, conceived as a device and as a public space that belongs to the city,

FIGURE 2: The Naka Waste Incineration Plant of Hiroshima, architecture as a device (on the top-left and bottom-right ph. M.L.Nobile, on the top-right and bottom-left ph. A.Guarino).

improves the communication and the didactic aim. The visitors' path, which represents an extension of Yoshijima Street, ends outdoors on this platform cantilevered over a newly incorporated public park, providing a wonderful view of the harbour.

Therefore, waste-to-energy plants are one of the most interesting and challenging examples of the integration between architecture and other disciplines.

2.2 Research on the Design of Public Spaces for Waste Recycling Collection Centres

The more general topic of the interaction between architecture and waste plants (incinerator and recycling plants) is the starting point of the research presented in this paper.

The research has been developed in the Department of Architecture of the University Federico II in Naples and focuses on a specific topic: how the architecture and design of public space has a role in the planning of waste recycling collection centres (Amirante, 2006). The main idea is to conceive theseplaces not only as services but as facilities with new functions, such as laboratories, markets, exhibition areas and rooms for educational activities.

This research is based on the knowledge that these places work well only if they are inside the city and near the places where citizens live. Thus, the value of waste recycling collection centres is not only in the quality of each element but in the form of the place and in the relationship that each centre builds with others and with the land. The research investigates the possibility of considering waste collection centres not as a 'problem to solve' but as 'an opportunity to play'.

In Italy, waste is managed at a municipal level in accordance with national legislation, and the management practices differ widely from area to area, as the general law is applied at local levels through detailed regional guidelines. In the 14 years since the garbage crisis, municipalities have been increasing access to recycling services and domestic collections.

According to the Campania Region's guidelines, a waste recycling collection centre is an area that is located in the city centre, and the number of collecting centres in the city is proportional to the number of citizens. These guidelines fix the size of the centre and the general requirements: a space for collecting the different materials (plastic, glass, paper, textiles, WEEE, oil), a ramp, a small cabin for the guardian and an enclosure. In the guidelines, the quality of those spaces is not mentioned. If thinking about waste recycling collection centres as a local service, one cannot imagine them as impenetrable enclosures; their locations must be chosen not only for economic and functional principles but for the possibility of demonstrating that it is possible to avoid the NIMBY effect and to integrate this inconvenient function in the city centre, connecting these facilities with other activities intended for community use.

This functional contamination does not mask an inconvenient function but shows the centre's social value is greater than it may appear. The integration of public functions and service functions represents how the 'standard collecting centre' specifies its own characteristics in relation to each place. The 'associated function' started with the place's position in the centre of the city, i.e. from its 'urban rule', and this is used to train citizens to recycle. Thus, reflecting on each element of the centre's architecture is essential to show their representative components, ranging from the semantic to the iconic, from the virtual to the possible, from the communication to the information. Urban design has converted waste recycling collection centres, as an architectural theme, integrating them with the context and mending the split in the city's fabric, representing a new function centre that contributes to the citizens' knowledge and education.

3. APPLICATION AND PROJECTS

3.1 The Research: The Design of the Waste Collection Centres in Nocera Inferiore and in the Metropolitan area of Naples 2004/2016

The research has been the occasion to reflect on architecture's useful dimension, particularly regarding the design of waste collection areas. For many years, numerous architects have confronted the theme of designing large facilities for the collection and disposal of solid waste (mentioned above); however, few current researches are studying the areas closer to citizens, places within the urban centres, which are destined for the separate collection of waste to be recycled(plastic, glass, paper, textiles, WEEE, oil). Public administrations throughout Europe have implemented several strategies and have introduced regulations since the early 2000s to favour the gradual reduction of urban solid waste production. The Campania Region published guidelines for the drafting of eco-design projects in Law Order No. 11 on September 13, 2000, when the waste management topic became an emergency.

This research consists of two steps. The first step was the agreement in 2004/2006 between the Department of Architecture of the University of Naples and the Municipality of Nocera Inferiore, a city of 40.000 inhabitants in the Campania Region, south of Italy.

The research was developed with 25 students from the final-year course in Architecture, who started to design several projects in different parts of the city. The aim was to demonstrate how to design a waste collection centre as a public space, integrating the service function with a collateral function (educational laboratories, public areas, children's playgrounds, gardens). In 2006, the final projects of the master's degree in Architecture students delved deeper into the topic. I developed my Final Project in 2006 on the topic of Waste Facility Infrastructures as an Architectural topic: A design project for a Waste collecting centre in Nocera Inferiore (Tutor prof. Roberta Amirante, co-tutor Paola Scala, Orfina Fatigato, Department of Architecture University Federico II of Napoli).

This virtuous process produced a public tender for the design of a waste recycling collection centre: in 2009, the Municipality of Nocera Inferiore decided to build a waste recycling collection centre following the guidelines of the Campania Region but integrating those guidelines with the results of the research held in collaboration with the University of Naples. The design team (coordinated by Prof. Architect Sergio Pone) conceived this infrastructure as a 'hybrid' architecture, a place that could be used as a public space for events and concerts, a public square and a community area (Figure 3).

Every year, this structure hosts a festival called the 'Campania Eco Festival', and the event started with the building, which is one of the most interesting examples in Italy of a waste recycling collection centre that hosts a different 'hybrid' function, as conceived in its architectural design concept. The waste recycling collection centre in Nocera Inferiore is a public space that at different times of the year becomes a meeting place where the typical functions of living together blend into functional use, and this helps raise awareness and educate the citizens on the topic of waste management. During the day, the facility serves as a normal collection centre. The different setting and the disposal of the waste containers makes it possible to use the ramp as a stage, and the shelter hosts the area for the public audience.

The research's second phase, designing the waste collection centres in the metropolitan area of Naples 2016/2017, is currently ongoing. The research project, funded by the Campania Region, started in 2016 and considers the possibility of creating a network of waste recycling collection centres in the metropolitan area of Naples using the regional highway (research group: R. Amirante - coordinator, P. Miano, C. Piscopo, S. Pone, P. Scala, M. Rus-

FIGURE 3: Waste recycling collection centre in Nocera Inferiore (drawing by CMMKM, ph. Campania Eco Festival).

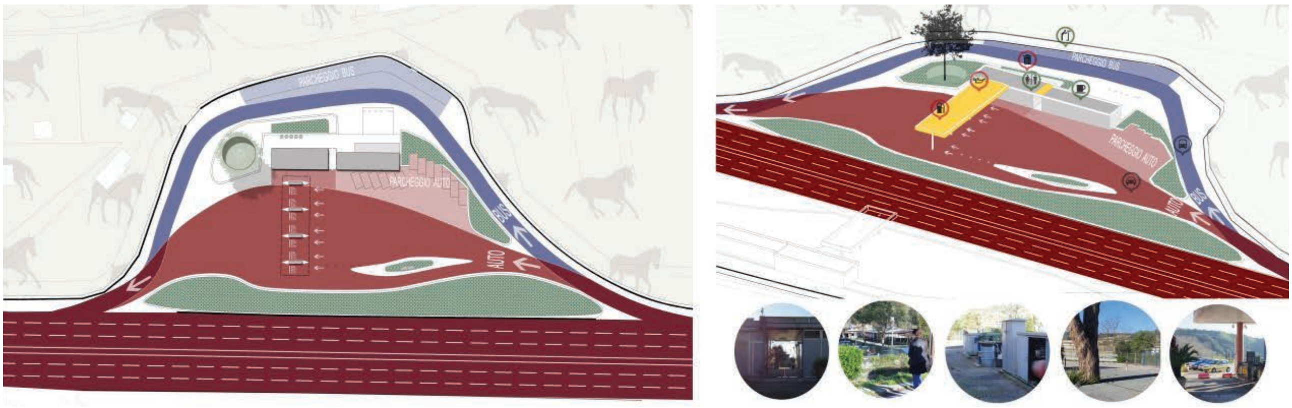

FIGURE 4: The reuse of regional highway's service areas in metropolitan Naples (drawing by Alessandra Acampora, Veronica De Falco, Maria Luna Nobile, Bianca Parenti).

so, A. Acampora, M. Castigliano, V. De Falco, M. L. Nobile, B. Parenti - Department of Architecture, University Federico II of Naples) - Figure 4.

Based on the considerations highlighted in the first part of this paper, and considering that during the 10 years (from building the Ecological centre in Nocera Inferiore in 2007 to the approval of the research project in 2017) the situation in the Campania Region has been changing—Campania's municipalities are currently equipped with waste recycling collection centres—the research has considered the current legislation and the actions provided for in the revised Programming Addresses of the Urban Waste Management Plan in the Campania Region through the addition of the C.I.R.O, Integrated Centres for the Optimal Reuse of Durable Goods, particularly the reuse of WEEE - Wood - Metal - Bulk - Textile.

This consideration was used to initiate a discussion on the hypothesis of designing a collection space for the modification of such types of 'waste', giving them a new life. This project hypothesis is not only architectural but initiates a virtuous process that blends the logic of space organisation into the management and sustainability of the proposed actions. The logic of 'utility' becomes the main theme for reconsidering 'waste' spaces, urban areas and the architectural objects of the regional highway's service areas in metropolitan Naples (disused shelter, parking areas, etc.) - Figure 5.

Considering the main aim of the research project, this article focused on integrating different functions into the design of waste management collection centres, highlighting the 'upcycling' process. Introduced in the research of architect William McDonough and chemist Michael Braungart, upcycling is the specific moment that defines the end of one cycle and the starting point of a new cycle.

The term upcycling was coined in Cradle to Cradle, a book on ecologically intelligent design. In its simplest terms, upcycling is the practice of taking something that is disposable and transforming it into something of greater use and value. Upcycling repurposes materials that are not easily recycled into something useful or artistic. Therefore, the action is intended to return a new quality, breathing new life into discarded materials.

The research investigates the possibility of considering waste collection centres not as a 'problem to solve' but as 'an opportunity to play'. Designing the waste management centres on the highway in Naples was an opportunity to

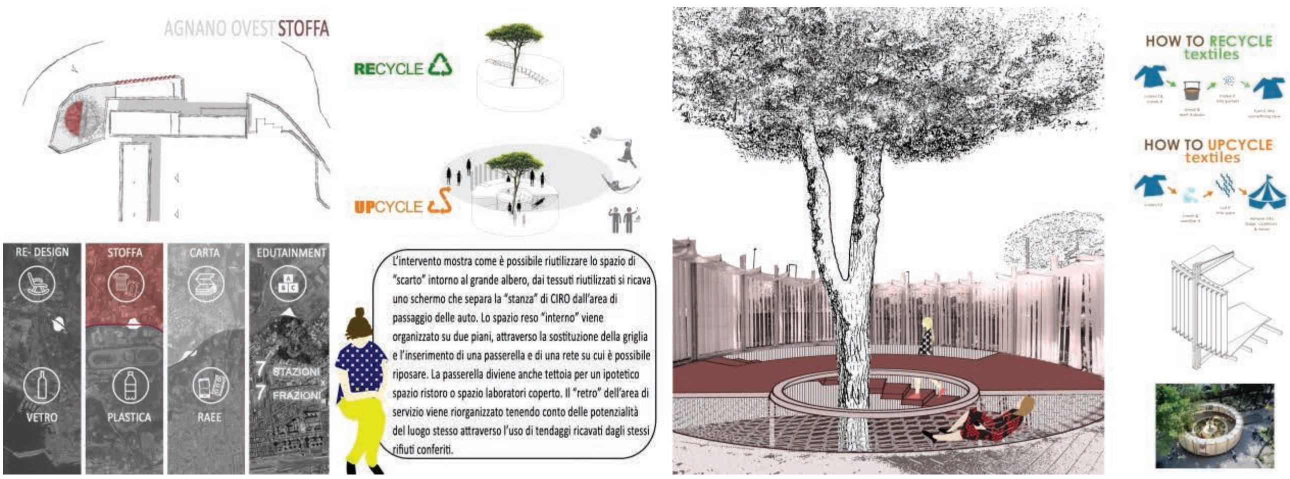

FIGURE 5: Upcycling the regional highway's service areas in Naples: the Integrated Centres for the Optimal Reuse of Durable Goods (drawing by Alessandra Acampora, Veronica De Falco, Maria Luna Nobile, Bianca Parenti).

experiment with the upcycle concept in the abandoned or 'wasted' regions that are included in the highway's service areas. This concept that is still a work in progress could be improved and applied to other similar areas.

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ENHANCED GEOMORPHIC DESIGN FOR RECLAMATION OF RURAL WASTE-SCAPES

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ABSTRACT

Many inventive concepts for the adaptive re-use of waste landscapes, or waste-scapes, have been proposed and constructed in the last decade. These are often located near or within large, urban populations, which provide much of the incentive for adaptive re-use. A different challenge presents itself when a waste-scape is rurally located, near a small - though equally important - population. How do we address complex socio-cultural, economic, and environmental objectives without the economic incentive provided by a large nearby population? This project looks at the mineable oil sands region of northern Alberta, Canada: a rural waste-scape covering 895 km^2 in Canada's boreal forest. Specifically, this project discusses the geomorphology and native substrate of northern Alberta, juxtaposed with the traditional design of waste storage landforms, in order to show that there are no natural analogues in the region. A geomorphic approach to the design of waste-scapes in this region has been developed using a Landscape Evolution Model (LEM) for long-term projections, and is being tested in the region. This project sheds new light on the rarely acknowledged issue of waste design in rural areas and the wide range of benefits achieved through use of an enhanced geomorphic design approach.

1. INTRODUCTION

How can we create value from what is traditionally considered a waste landscape or "un-usable" environment? This question has been answered through unique, inventive, and costly approaches where waste landscapes, or waste-scapes, are located near large populations and urban environments: architect Bjarke Ingles' artificial ski hill atop Copenhagen's 'Amager Bakke' waste incinerator, and Lusatia, Germany's new lake district are two popular examples. However, as NIMBY-ism proliferates waste-scapes are shifted increasingly towards sensitive rural environments with smaller communities. How do we address complex social and environmental objectives without the economic incentive provided by a large nearby population?

The resource sector, specifically mining, is one with experience in managing waste in rural areas. The waste management and landfilling industry has many overlapping concerns, impacts, and similarities to the mining industry:

- Required life cycle design and planning is a relatively new concept to the industry;
- Topography is altered over long timeframes and large areas;
- Expected and potential environmental impacts demand thorough environmental assessment;
- Possible ground and surface water contamination are long-term risks requiring management;
- High greenhouse gas emissions make carbon taxes significant;
- Regulatory issues are complex and intertwined across disciplines, generating "wicked" problems (Rittel & Webber 1973);
- New technologies seek to utilize waste as a resource;
- Working with NGO's, managing public perception and involvement is increasingly important.

Waste management is at the core of the mining industry, producing fractions of an ounce to a few ounces of profitable ore per ton of waste generated. In terms of spatial area, the extent of mineral extraction is often dwarfed by the area used to hold mine waste resulting from extraction and mineral processing. These holding areas consist of tailings ponds (where aboveground dams are constructed of mine waste to retain slurried mine waste in a pond), as well as aboveground "dumps" or piles comprised of unprofitable waste rock. The landscape resulting from surface mining - mainly a mined pit, aboveground waste storages, and a processing area - all require reclamation such that the land not pose hazards to human or environmental health.

Mines are frequently located in rural areas with small populations, providing welcome employment opportu-

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nities yet unwelcome residual landscape impacts. It has been estimated that rural communities in Canada rely on resource industries (which includes mining) for greater than 25 percent of their employment (Stedman et al. 2004). Mine waste is often physically and chemically unstable, making reclamation a challenge and frequently leading to prolonged disturbance. As mineralized portions of the substrata are removed from the ground, they are broken into smaller pieces, increasing the surface area exposed and available to react with the environment. Contaminated surface and groundwater discharge as well as excessive and/or contaminated dust are two challenges created through this process. Surface settlement of the waste rock or tailings ponds can also be a long-term challenge for reclamation.

The Athabasca oil sands (AOS) surface mining region in Alberta, Canada has been criticized for its widespread disturbance of the land through mining. In reality, mining occurs over a relatively short time frame (+/- 50 years), and the majority of this 895 square kilometre landscape will be used for waste storage in perpetuity once mining ceases. Reclamation is proposed to take another 10 - 20 years according to closure plans filed in the region, although when reclamation of other smaller mines with less complex waste is considered, this reclamation timeline is likely to be much longer.

For example, Brenda Mine in Penticton, British Columbia, Canada was a small copper and molybdenum mine with an 87 hectare tailngs pond reclaimed from 1988 to 1997. Tailings here do not present the same consolidation problems that exist in oil sands mines. 20 years later (in 2018) the tailings pond presently contains an unreclaimed portion with a water surface, requires that earthen areas be fertilized every two years to maintain vegetation, and will have staffed water treatment facilities long-term. This site is not self-sustaining despite reclamation efforts. In contrast, tailings ponds in the oil sands typically cover 10 times more area than that at Brenda, with more challenging tailings to reclaim. In order to lessen the burden of reclamation regulators, engineers, ecologists, hydrogeologists, and landscape architects have been working collaboratively to develop an economic method of restoring the environment for local populations in an expedited fashion.

The negative consequences typically attributed to mining are more specifically a result of waste management practices throughout the operational phase and in perpetuity thereafter. This paper describes how professionals currently design topography for this waste-scape, as well as an enhanced method of geomorphic design for long-term stability and adaptive re-use.

2. FORM AS A BASIS FOR LONG-TERM WASTE-SCAPE SUCCESS

The glossary of geology defines a landscape as "a distinct association of landforms, as operated on by geologic processes (exo- or endogenic), that can be seen in a single view." The concept that a view defines a landscape suggests that it is being inhabited or used by a person (or animal), and that there is a symbiotic exchange occurring between the user and the land as a result. While the AOS region in northern Alberta is certainly rural, it is also the location of historic migratory as well as hunting, trapping, and medicine gathering territory of indigenous Cree and Dene people (Joly et al. 2018; Baker & Westman, 2018). Today Fort McKay First Nation, a community comprising five Indian reserves and a population of approximately 600, sits on the shores of the Athabasca River at the center of oil sands development in the region.

A larger community that was originally built around servicing the oil sands mines, their employees, and their families, exists south of the mines in Fort McMurray and currently has a population of just over 60,000 people. Figure 1 illustrates the location of the mines and these communities.

2.1 Local community

First Nations have inhabited the Fort McKay area since the early 1800's. They have lived off the land by hunting, fishing, gathering, and also by nurturing the local environment where possible to ensure their sustainable way of life can be continued into the future (Fort McKay website, 2017; Baker & Westman, 2018; Joly et al. 2018). In recent decades, residents of Fort McKay have worked with industry towards mutually beneficial land and employment agreements (Baker & Westman, 2018). However, boom-bust cycles and economic dynamics occassionally leave the community at risk of low employment rates, as well as a full spectrum of positive and negative ramifications. Included in this spectrum are health impacts due to job insecurity stress and environmental degradation, compounded by limited access to health services due to their rural location (Shandro et al. 2011). The return of productive landscape in-line with sustainable indigenous land uses is currently thought to be the most desirable method of supporting local First Nations into the future.

2.2 The impact associated with geomorphology

Ideally, local communities pre-existing the mines will be able to continue their way of life within the future reclaimed waste-scape. While the success of the re-designed waste-scape is dependent upon many diverse aspects, the foundation for success can largely be attributed to designing the appropriate landform upon which ecological, chemical, and hydrological, processes may occur. For many years, abandoned mines have been shown to be subject to excess erosion and slope failure, thereby hindering vegetation productivity, local water quality, and return of native fauna (Martin Duque et al. 2015; Toy & Hadley 1987).

Landforms are altered slowly over time by physical and chemical erosional processes. This paper focuses on physical processes common to a boreal environment: erosion by wind and predominantly water, thereby changing the shape of a landform (Richards & Clifford 2011). For example, a sand castle may be indistinguishable from the rest of the beach following a storm with high winds and heavy rainfall. The same general processes of wind and water erosion act on the surface of our earth to create mature and increasingly erosion-resistant landforms over time, growing nearer to a state of equilibrium with increasing maturity.

In the design of waste-scapes, particularly those generated through mining, efficiency is a primary consideration with respect to the operational life span of the mine. Geotechnical and mining engineers often design the footprint using straight lines and the downstream dykesusing maximum slopes in order to optimize the waste-holding capacity. This engineered form may be easily maintained while the mine is in operation and staff are present; however, on mine closure this form will be more easily degraded by wind and water erosion until it reaches an equilibrium with the forces acting on it - similar to the sand castle during a storm (see Figure 2).

When a landform is eroding due to water runoff or "overland flow", eroded sediment is transported and deposited downslope, impacting the vegetation on both erosional and depositional portions of the slope. On the eroded portion vegetation cannot establish or be maintained, and on the depositional portion, vegetation is buried. In some instances eroded sediment is transported into adjacent water bodies, increasing the sediment load and decreasing water quality. Toy and Osterkamp (1995) go so far as to call human-accelerated erosion a "serious global problem" and "the pre-eminent environmental problem in the United States". Design using a mature form encourages the success of revegetation, minimizes the need for repeated seeding/ planting, and helps to preserve adjacent water quality. Regionally appropriate geomorphic landforms aim to provide topography that would occur after millennia of weathering, therefore the foundation they provide is more stable, and likely to generate microclimates that support local flora and fauna. This is aesthetically beneficial, but it is also important for future land productivity and a return to traditional land uses.

3. THE ALBERTA OIL SANDS REGION

3.1 The native landscape

In terms of topography, the landscape is naturally low lying around 260 - 350 meters above sea level (m.a.s.l.) and

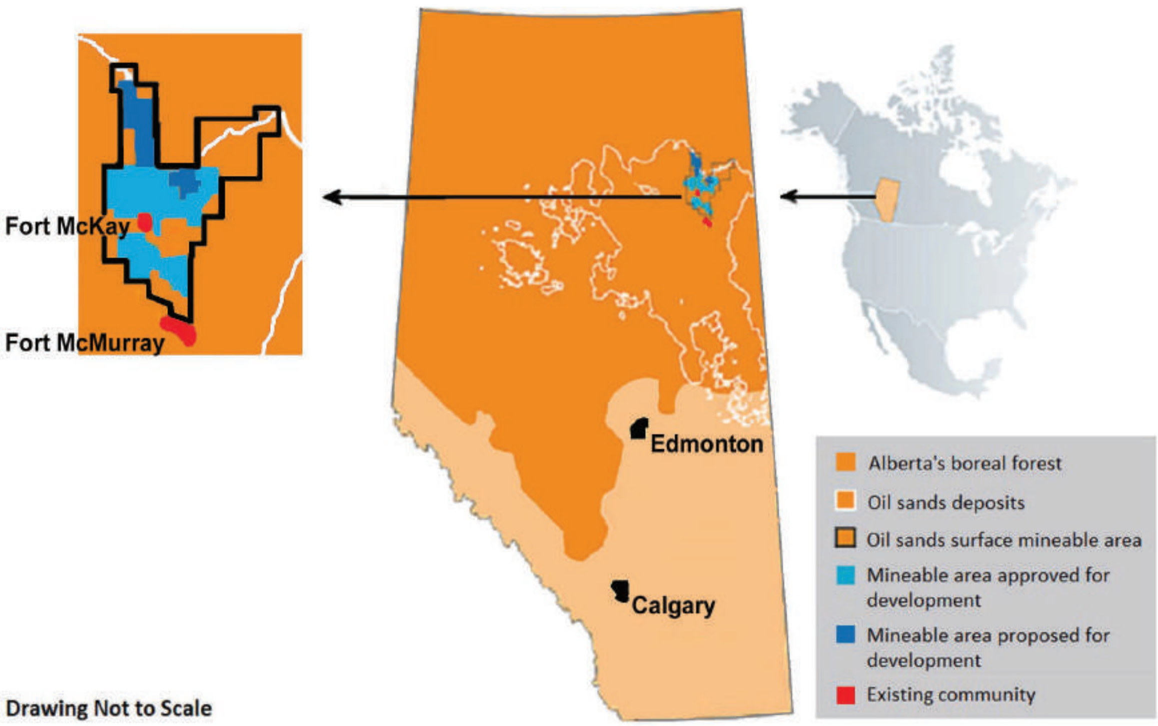

FIGURE 1: Location of mineable oil sands and nearby communities within Alberta, Canada. Adapted from AER (https://www.aer.ca/about-aer/spotlight-on/oil-sands).

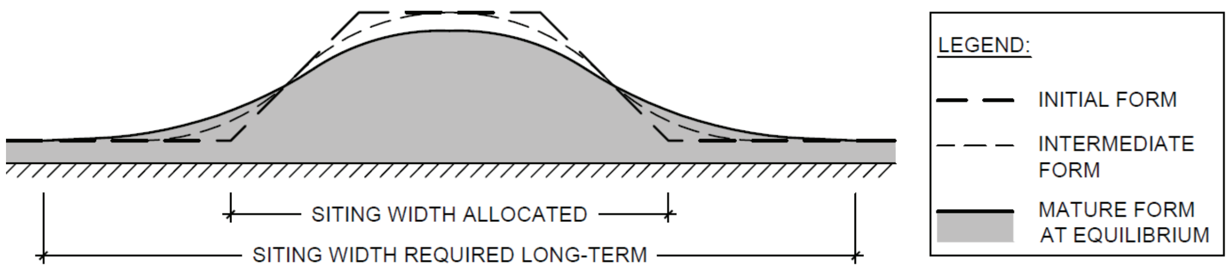

FIGURE 2: Generalized degradation of sand landforms due to physical erosion over time.

composed of dense boreal forest and muskeg wetlands (McPherson & Kathol, 1977). The Athabasca River flows northwards, cutting the surface minable area in two: a west and an east side, as seen in Figure 3. Native geomorphology and substrates are described in detail below.

3.1.1 Geomorphology

The glossary of geology defines a landform as "any physical, recognizable form or feature on the earth's surface, having a characteristic shape, and produced by natural causes; it includes major forms such as a plain, plateau, or mountain, and minor forms such as a hill, valley, slope, esker, or dune. Taken together, the landforms make up the surface configuration of the earth" (CEMA 2006).

The geomophology of the oil sands region is dictated by the last major glaciation, which occurred approximately 10,000 years ago and subsequent glaciofluvial events during melting. Glacial, glaciofluvial, and glaciolacustrine landforms and features cover the oil sands region, including drumlins, eskers, ground and hummocky moraines, meltwater channels, and proglacial lakes (Smith & Fisher 1993; Fisher & Smith 1994).

3.1.2 Substrate and material type

Surficial soil in the mineable oil sands region is of glacial origin: predominantly outwash sands and gravels, and glacial till. Glacial till is an unsorted mixture of grain sizes ranging from clay to coarse gravel that were originally eroded, entrained, transported, and subsequently deposited by glaciers. The glacial till present in ground moraines in this region tend to be high in clay content giving it a relatively low permeability. Eskers and drumlins are composed of soils ranging in texture from fine sand to coarse gravel, while proglacial lakes tend to be finer in texture, from clay to fine sand (Catto 1995). Outwash sands and gravels are widespread, particularly on the west side of the Athabasca River. It is important to note that surficial soils and their respective characteristic grain sizes are consistent over short distances and changes occur abruptly, making the landscape soils as a whole widely variable (McPherson & Kathol, 1977).

As nature has acted on these soils, they have been altered in form. Glaciofluvial sands have been mobilized and re-worked by the wind, creating aeolian sand dunes before being re-stabilized by vegetation. Glacial till has been weathered less due to the range in particle sizes present. Wind tends to transport silt and fine sand only, and particles transported by water can find refuge in the shadow of larger particles within the till matrix. In effect, the particle size range found in the till provided an inherent resistance against small-scale erosion due to wind and rain. Naturally occuring surficial soils have been covered by thick layers (less than 10 feet to a maximum of 30 feet deep) of peat and muskeg across more than 50% of the landscape, further limiting their physical erosion (McPherson & Kathol, 1977).

3.2 The Anthropogenic Landscape

The end goal of reclamation within the surface mineable AOS region is to produce a landscape that has a self-sustaining ecosystem including vegetation and wildlife common to the region (Natural Resources Canada 2015). This broad goal is stated and expanded upon to various degrees in each of the oil sands min operators' closure and reclamation plans submitted since they were first mandated seven years ago. For example, all presently active mines indicated their reclamation goal was "to achieve a maintenance-free, self-sustaining ecosystem with a capability equivalent to pre-development conditions" (CNRL 2011; Golder Associates 2011; SCE 2011 and 2012; SEI 2011); however the report for the Kearl Oil Sands Mine specifies a "locally common boreal forest ecosystem", while those for the Mildred Lake Mine, Jackpine Mine, Fort Hills Mine, and Suncor Base Mine also included the goal of recieving reclamation certification by the Alberta Government. Syn-

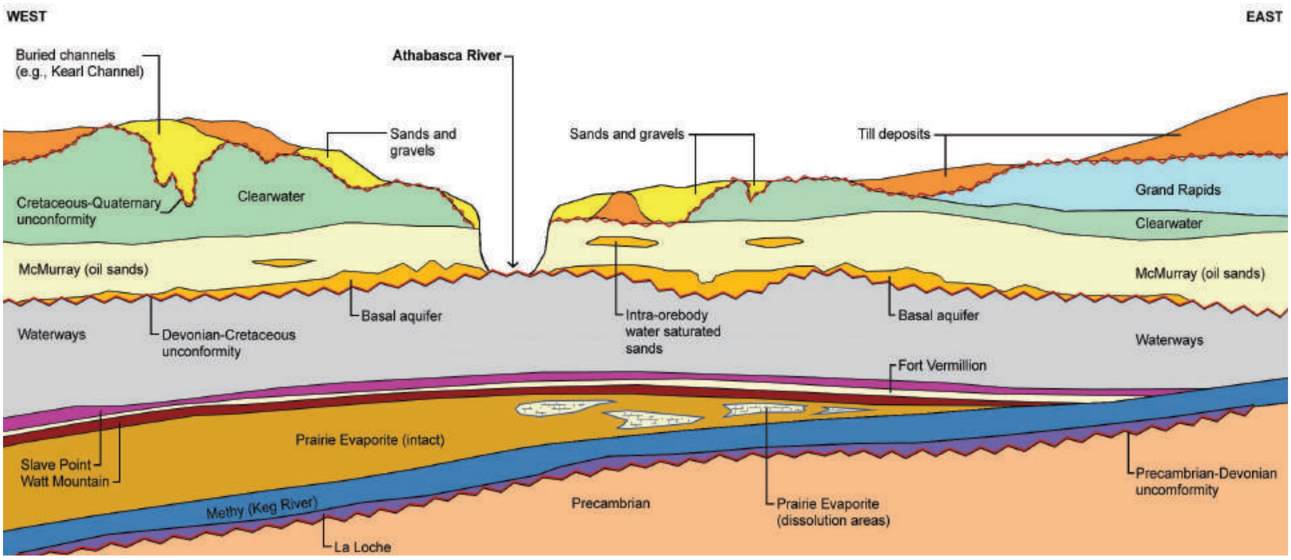

FIGURE 3: Conceptual geologic cross-section orientation from west to east in the mineable oil sands region of northern Alberta. Adapted from Shell Canada Energy, 2016.

crude Canada Ltd, who operates two active mines in the region and one in development, went one step further to state that the closure landscape will be planned in consultation with local stakeholders and will be integrated with surrounding areas, will include boreal forest lowland and upland communities, and will yeild water suitable for release to the natural environment (SCL 2011). These reclamation goals use simple language, however, the target is rather difficult to achieve, requiring extensive collaboration among diverse professionals in order to achieve goals that have never before been achieved. Once reclamation works have taken place, an extended period of active monitoring will take place, conducting occasional maintenence as necessary. It is presently thought that once maintenance is not required for an extended period of time, and the regulator determines that the landscape poses no threat to the environment or the public in excess of naturally occuring terrain, a reclamation certificate will be provided and the post-mining landscape will be returned to the Crown (OSTDC 2014).

The following section outlines what the post-mining, pre-reclamation landscape consists of, and provides a general overview of the current approach to topographic design.

3.2.1 Current topographic and drainage design

While the natural topography of the oil sands mining area is quite flat, the post-mining waste-scape topography will be quite the opposite. Due to this pre-existing flat topography, dams are built around all edges with tailings placed in the middle. Closure plans indicate that above ground tailings ponds are up to 100 m in height with dam side slopes from 3:1 to 20:1. Tailings ponds are several hundred hectares in area, and have a relatively flat surface between the dam edges. Overburden dumps are up to 100 m in height as well, but tend to have steeper side slopes and cover less surface area. This topography is in stark contrast to that which naturally occurred in the AOS. In addition to aboveground waste landforms, mined out pits are proposed to be backfilled with waste. A hierarchy of constructed wetlands, lakes, and streams are propsed to transport water from high elevations to low elevations while also allowing for contaminants to be naturally removed and concentrations decreased to background levels. This will be done with targeted vegetation planting, water treatment systems, and dilution.

The topographic design of this waste-scape is in its infancy. Conventional reclamation landscapes are designed to hold a specific, pre-determined flooding interval (1 in 200 year, or 1 in 1000 year flood, for example): using this criteria given that future storm intensity is predicted to increase with climate change means that failure (exceedance of design capacity) is inevitable (Golder Associates Ltd. 2008). Overburden dumps are beginning to be designed in such a way that they mimic more natural environments, as opposed to the square pyramids more typical of historic practice. Tailings pond surfaces are beginning to be designed using a geomorphic approach once the structures near their full capacity (Ade et al 2006). Design components for pond surfaces include ridges and swales, creation of multiple watersheds, and a dendritic drainage pattern.

The perimeter dams retaining tailings ponds are presently designed prior to the start of mining, and are done so strictly with geotechnical and mining constraints in mind for the duration of infilling (Golder Associates Ltd. 2008). Changes to these designs are made continuously as the structure is raised, and as the mine plan and waste management practices change (McRoberts 2008); however long term degradation or change in shape due to geomorphic forces are not yet considered in design. As such the sides of these tailings dams have regular slopes broken up at even intervals by terraces. This is called a platform-bank topography and is illustrated in Figure 4. According to the most recent proposed mine closure plans, there are no topographic changes proposed after mining to make these slopes more natural in their shape, appearance, or in their ability to self-heal following extreme storm events.

Ideally, geomorphic design for the closure of waste structures is intended to replace the rigid uniformity and straight-line slopes used during historic mining days with a variable topography similar to an 'S-curve' that is permitted to adapt to natural environmental changes (Beersing et al. 2004, Sawatsky & Beersing 2014). Geomorphic design in the oil sands presently imploys slope characteristics from regionally surveyed stable alluvial and vegetated channels (Golder Associates Ltd. 2008) and is applied to the reclaimed surface of a tailings pond, not to the dykes that confine the pond. The challenge in applying slopes of local-

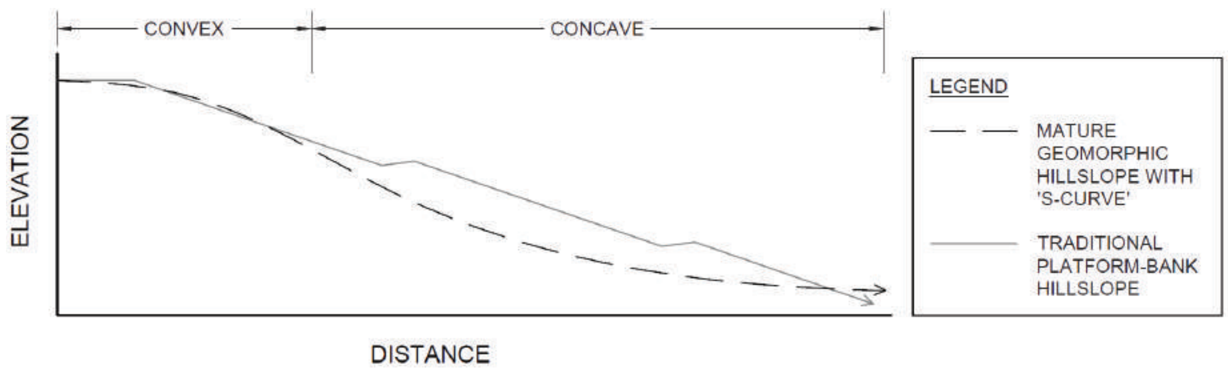

FIGURE 4: Mature slope with convex and concave curve components ('S-curve') compared to a traditional platform-bank slope used on tailings dams. Adapted from Slingerland & Beier 2016.

ly stable terrain to the waste-scape is that the construction materials have drastically different composition, and will therefore have different stable geometry at maturity.

3.2.2 Material and substrate types

The terrain of the waste-scape has man-made surficial geology generated from the waste products of mining. Depending on the waste materials (tailings, overburden) to be covered, various capping materials will be used. Over 40% of the finished waste-scape (mostly tailings ponds and some in-filled pits) will be capped with coarse sand tailings (CST). CST is a fine silty sand waste product of bitumen processing. The CST cap is then proposed to be topped with 30 cm of peat-mineral reclamation material for vegetation to take root.

Grain size distributions (GSDs) for man-made surficial geology (CST) and similar native surficial geologies are shown in Figure 5. The range of grain sizes in CST is highly erodible by wind and water, and hence topographic design is important in protecting these landforms and their surrounding environment. Fine mineral grains tend to hide in the wind or water “flow shadow” of larger particle sizes, meaning that soils with broader GSDs tend to erode less.

It is important to note that while Figure 5 shows that similar grain size distributions exist in sand dunes and CST, the height of local sand dunes ranges from 5 to 25 meters: a fraction of the height of tailings dams and overburden waste landforms. Additionally, these local sand dunes are stabilized by more than 5 meters of dense organic matter and vegetation (McPherson & Kathol, 1977). The greater topography of these anthropogenic structures means that they are more exposed to wind, and water has more fall height over which it can potentially erode the surface. As such, no true natural analogue for these landforms exist in the natural AOS landscape. It follows that the natural landscape may not be an appropriate tool, when used in an isolated manner, to inform design decisions for resultant mining waste landforms.

With the downstream dam topograhy unchanged at closure, an opportunity exists to estimate using empirical methods the average annual soil loss due to erosion on the downstream slope of a tailings dam. The Universal Soil Loss Equation (USLE) was first introduced by the United States Department of Agriculture (USDA) in the 1970’s and subsequently revised for various applications such as construction sites (Wischmeier & Smith 1978). The Revised Universal Soil Loss Equation For Application in Canada (RUSLEFAC) was published for use primarily in agricultural and conservation planning applications in 2002 and draws heavily upon previous work by Wischmeier & Smith (1978) as well as new calibrations. Although RUSLEFAC does not account for erosion due to wind or gully erosion, its primary goal is to provide a long-term average annual soil loss rate for planning purposes (Wall et al. 2002). RUSLEFAC was therefore applied to a typical tailings dam downstream slope, with all the necesary parameters found to be publicly available. Parameters include rainfall and runoff (R), a soil erodibility factor (K), a slope factor (LS), a vegetation management factor (C), and a supporting practice factor (P), each of which was tailored to conditions across northern Alberta. The average annual soil loss calculated using RUSLEFAC was found to be 50 Mg/ha/yr, which is

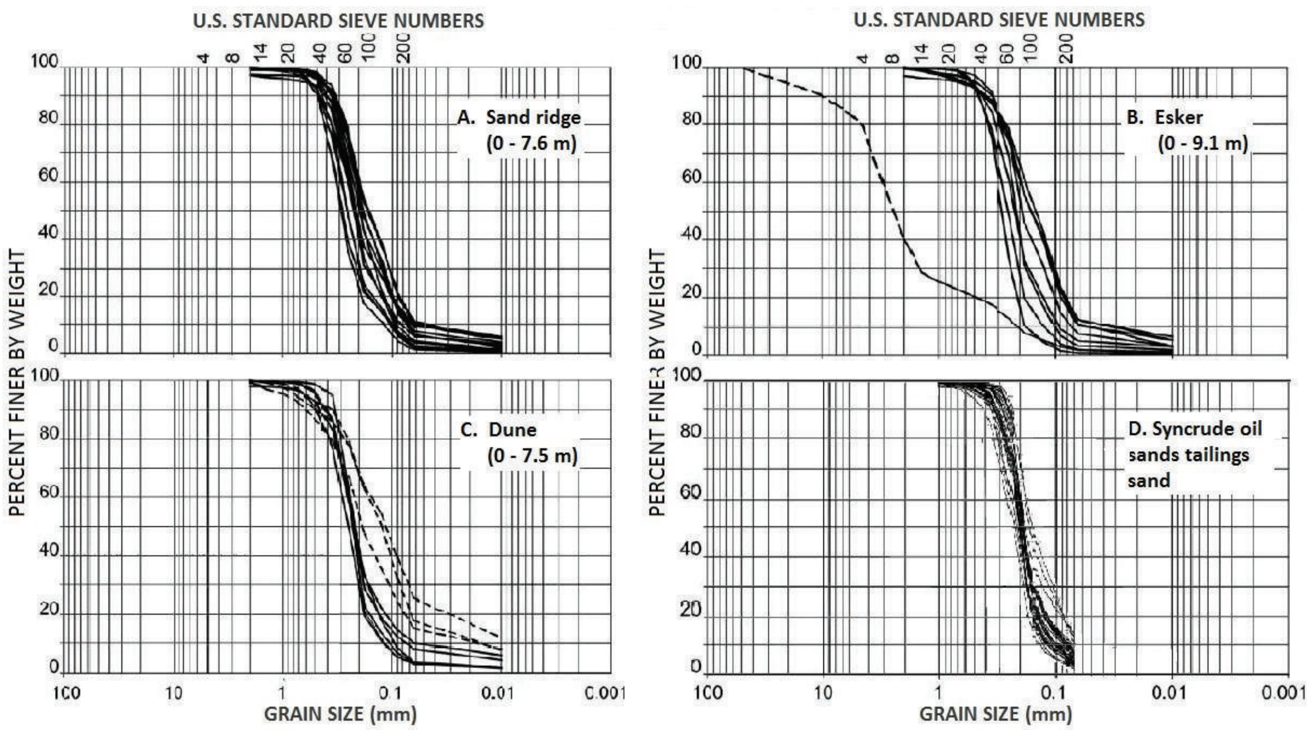

FIGURE 5: Grain size distributions for natural and artificial surficial geology in northern Alberta. Natural features in the central Boreal Plains of northern Alberta that are most similar to CST: (A) sand ridge, (B) esker, (C) sand dune, adapted from Smerdon et al. (2005). Grain size distribution for sand tailings from the Athabasca oil sands (D), adapted from McKenna (2002).

well in excess of the canadian tollerable upper limit of 6 Mg/ha/yr (Wall et al. 2002). Post-mining slopes that have not been reclaimed have measured erosion rates ranging from 93 Mg/ha/yr at the Waunafon coal mine in Wales, UK (Haig 1979) to over 400 Mg/ha/yr at the Kidston gold mine in north-eastern Australia (So et al 2002). Reclamation of these downstream slopes is therefore important to the surrounding environment and long-term stability of the structures.

4. AN ENHANCED GEOMORPHIC APPROACH FOR WASTE-SCAPE DESIGN

While the principles of geomorphology are well understood, well documented, and selectively applied with success (Bugosh 2009), thus far the mining industry as a whole has failed to consistently design large scale landforms that are geomorphically stable. Basic erosion resistant slope profiles have been outlined (Schor & Gray 2007; Toy & Hadley 1987), but how do we determine specific design criteria of the mature form associated with a particular climate and surficial geology so that we might construct it?

The landscape evolution model (LEM) developed as a result of increased computing power, beginning in the 1980's in combination with advanced understanding of factors influencing erosion and sedimentation (Coulthard 2001). SIBERIA (Willgoose 1989) was the first widely used LEM making use of a digital elevation model (DEM) for topographic information. The immediate benefit of this was that an entire landform could be modelled as it was exposed to precipitation over long time frames (10 - 10,000 years). Previous empirical formulas allowed erosion from a single simplified profile to be estimated quantitatively for one year only (Wischmeier & Smith 1978). Alternatively, one could use fallout radionuclide dating (^{137}Cs) to determine erosion rates (Hancock 2009; Hancock et al. 2011; Martinez et al. 2009).

LEM's have been tested in a range of environments to predict how the landform will degrade over time due to water-based erosion processes (Welsh et al. 2009; Verrdon-Kidd et al.2017); Welsh et al. (2009) studied erosion in the French Alps using historical records since 1826 to recreate current conditions. Verdon-Kidd et al. (2017) used a reconstructed rainfall and runoff record in order to accurately represent landform response to climate in Monsoonal north-west Australia. LEMs have been verified successfully on erosion-prone environments where end-of-construction topography is known and where the landscape has remained maintenance-free since this time (Hancock et al 2016). There are now several LEM's that predict erosion and deposition at the hillslope and catchment scale as a response to calculated runoff rates and spatially distinct soil and precipitation parameters (Coulthard 2001, Hancock et al. 2011). Most recently, fluvial erosion models have been combined with an aeolian morphology model to account for the erosive and morphological forces of wind and water together in China (Liu & Coulthard 2015, 2017). While LEMs have been used to understand the morphology of anthropogenic waste-scapes in the past, to the authors' knowledge they have not been used proactively in order to design a geomorphically stable landform.

The enhanced geomorphic approach (outlined graphically in Figure 6) is currently being applied to a tailings pond and surrounding environment in the AOS. This approach uses the current method of applying stable slope characteristics found in locally stable environments as a first step. The resulting topographic design is then used to generate a digital elevation model (DEM) that includes the tailings pond, dams, and surrounding waste-scape within receiving watershed(s). A LEM is subsequently used to mimic degradation of the designed DEM due to fluvial erosion over extended time frames until an equilibrium topography is found. This equilibrium topography is further analysized, however the topography generated by the LEM guides surface design of the waste-scape for long-term geomorphic stability.

This approach differs from the existing approach in recognizing that waste-scapes change over time and applied geomorphic tools in a new, proactive manner to inform design. The failure mechanisms at play during active waste-scape construction are different from those that dominate after reclamation: the LEM helps to determine how the waste-scape (including perimeter dams, central ponds, and overburden dumps) will morph over time. This has not been previously considered in detail.

5. RESULTS AND DISCUSSION

The enhanced geomorphic design approach is currently being tested in the AOS, with some unexpected and far-reaching peripheral benefits. Many of these benefits are recognized during the design stage, as the use of landscape evolution modelling forces the engineer or architect to consider the site more holistically than previous methods.

Currently adopted practices that use a geomorphic ap-

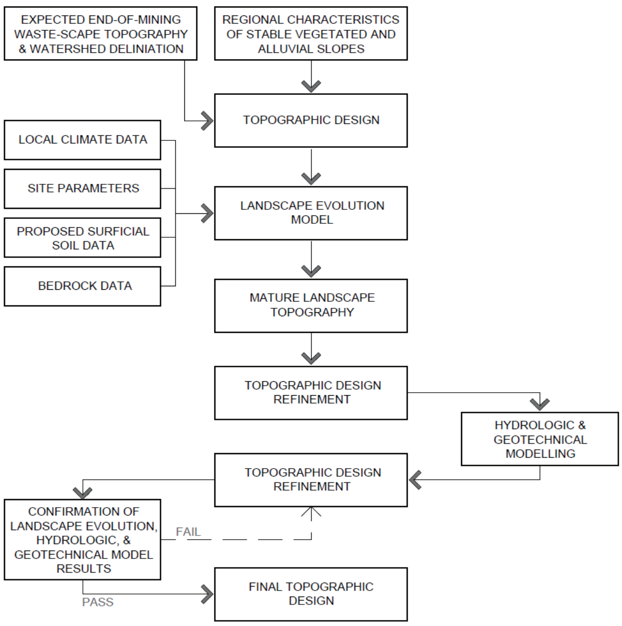

FIGURE 6: Enhanced geomorphic approach for waste-scape design.

proach on tailings pond surfaces but not on tailings dam side slopes may lead to increased environmental loading of the surrounding environment due to sediment erosion from dams. Regardless of whether cross-sectional differences are subtle or extreme (Figure 7), given the vast 895 km^2 waste-scape to be re-shaped (220 km^2 are aboveground tailings ponds), cumulative effects are likely to be significant (Government of Alberta - Alberta Energy website).

One benefit of this process is the realization amongst landscape architects and engineers that additional space allowances are necessary for long-term stability of aboveground features compared to what was used during planning and operation: this has implications for the siting and layout of future waste-holding facilities, and their future planning for inevitable natural deterioration as opposed to its prevention.

This concept has peripheral impacts to current structures that are not presently affected by erosion and sedimentation, dune migration, or water runoff, but may be in the future (Figure 7). Additionally, the process predicts whether ongoing maintenance will be necessary to preserve the slopes that were designed without regard for long-term degradation, and which will affect the income potential of local communities, owner/ operators, or regulators tasked with monitoring and maintaining sites into the future. Where this approach is used early in the mine life, the enhanced geomorphic approach is anticipated to remove the corresponding economic strain of ongoing staffing, materials, and equipment on site post-closure.

The enhanced geomorphic approach indicates where future drainage pathways and sediment concentrations may shift to, and where additional drainage pathways are likely to develop on the landscape. While the LEM, and any model, should not be used to directly generate a design, it does inform the architect or engineer as to where additional drainage capacity, erosion control measures, spatial allowances, and buffers can be best applied. Figure 7 illustrates our current approach (top left) and what that cross section might look like after modelling with a LEM (bottom left). In this example, a stream adjacent to the waste structure has been filled in/ diverted due to sediment deposition on the bottom of the slope, and soil has encroached upon a building. If LEM modelling were conducted on this waste-scape design prior to construction these impacts would be understood and proactive measures could be taken to alter the design and construction.

When topography is designed with the surficial geology in mind, the resultant erosion resistance over long time frames provides stable ground for vegetation to establish and further stabilize terrain at greater depth. This means that unexpected impacts to surrounding environments are minimized. Target end landscapes can be a design challenge in mining: The enhanced geomorphic aproach encourages stakeholder coordination, reduces uncertainty regarding natural spatial projections, and provides a starting point to design from.

In the case where existing waste-scapes have been placed too close to site boundaries - whether they be natural (rivers, streams) or man-made (highways, infrastructure) - the use of a LEM over short time intervals can inform the scheduled return period for maintenance. Maintenance over extended time frames is best avoided where safe to do so; however, in many instances long-term maintenance is required and the accuracy of financial planning by corporations and governments can be improved with this knowledge. This is a long-term benefit to the companies involved, the province of Alberta and it's regulatory body that holds ultimate liability if a mining company were to go bancrupt, and tax-paying residents of Alberta.

Lastly, the geomorphic and environmental performance

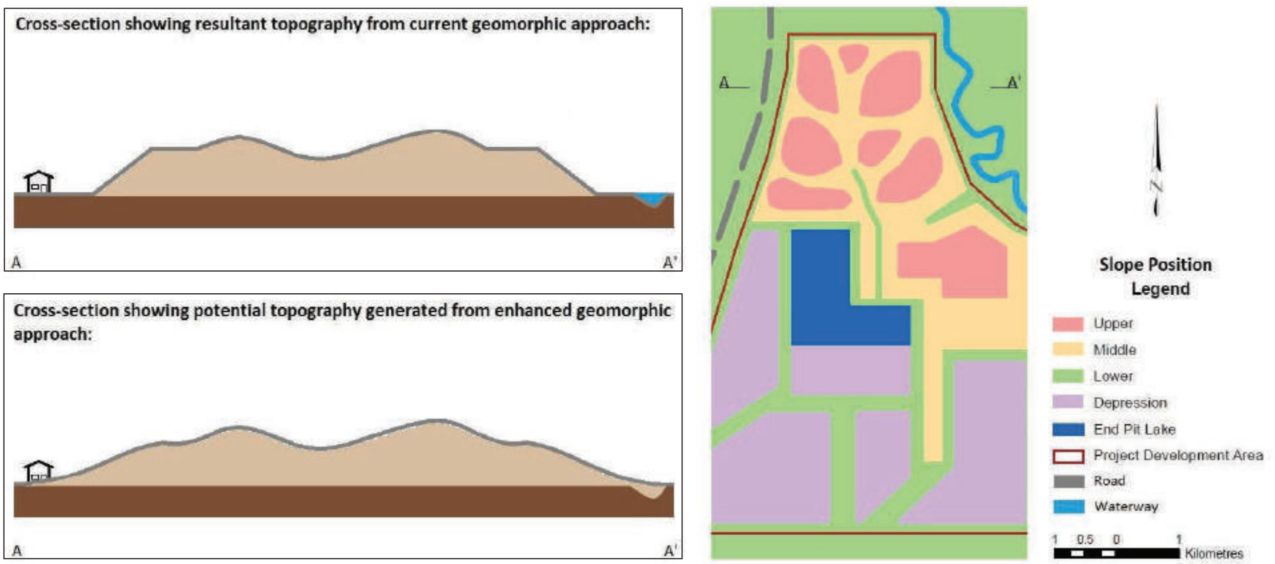

FIGURE 7: Currently adopted geomorphic approach to topographic design of waste landforms at a fictional oil sands mine on the right. Cross-sections (A - A') of the current geomorphic approach (top left) as compared to the same cross-section modelled using the enhanced geomorphic approach (bottom left). Note the enhanced approach provides insight as to potential long term form after decades of wind and water erosion, and impacts to adjacent infrastructure or natural features represented by the building and river.

of waste structures in Alberta's north has a profound impact on local communities. In a 2014 study of mine closure goals and outcomes involving 143 mine closure professionals and practitioners, it was found that geotechncial factors had a cumulative impact on other reclamation goals (Baida et al 2014). Where slope stability was not optimally achieved other goals categorized under themes of 'biological', 'land-use', and 'socio-cultural' were also more likely to be unsuccessful (Baida et al. 2014; Slingerland et al. 2016). Traditional practices are inherently sustainable in their nature (Baker & Westman 2018), and are more likely to be successfully returned to when the landscape itself is stable, and hospitable to flora and fauna.

6. CONCLUSIONS

Waste-scapes have been re-invented in recent years by creating adaptive re-uses at significant expense. The drive for re-purposing typically comes from large local populations such that there is economic incentive in terms of land costs/value, owner reputation, and potentially revenue generation from the waste-scape. In rural locations, smaller communities do not have the same luxury.

Waste management and design is at the core of mining: mining waste-scapes are often rurally located with small populations nearby. The historical track-record of mining companies internationally has been inconsistent with respect to closure and community relations; however, the enhanced geomorphic approach to topographic design of landforms provides a holistic design method for rural waste-scapes and a strong basis for successful future land uses.

While similar soils exist in the northern Alberta landscape, similar landforms do not: there are no natural analogues to the waste landforms generated from mining in this region. Landscape evolution models may be used as a tool to determine what these waste-scapes may eventually degrade into., An opportunity therefore exists to proactively design with the local climate and geology of the landform in question using the methodology proposed herein. An equal opportunity exists at existing rural waste-scapes that have not been proactively geomorphically designed, but where refined estimates of long-term maintenance with respect to erosion are beneficial to owners/ operators for fiscal planning.

This enhanced geomorphic approach is currently being tested in the AOS region to provide insight as to the long-term geomorphology of aboveground tailings ponds and dams. Benefits are thus far diverse and holistic in nature, providing a greater capacity than previous approaches to achieve the regions' overall goal of constructing self-sustaining local ecosystems with equivalent land capability to pre-development conditions (CNRL 2011; Golder Associates 2011; SCE 2011 and 2012; SEI 2011).

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INFO FROM THE WORLD

PRELIMINARY EVALUATION OF THE PROGRAM GOALS FOR RECYCLED SOLID WASTE IN THE SÃO PAULO MUNICIPALITY, BRAZIL

Municipal solid waste recycling in developing countries, such as Brazil, is fundamental for methane gas emissions minimization from landfills and dumps, reduced extraction of natural resources, water and soil contamination reduction, and improvement of the socio-economic conditions of waste pickers.

In 2014, São Paulo published the Municipal Solid Waste Integrated Plan that defined the goals with an inclusive approach regarding the National Waste Pickers movement. The targets for 2016 were: 1. to recycle 3,000 tons per day from selective collection; 2. to cover 100% of the city area with selective collection.

The objective of this paper is to evaluate the achievements of the goals for recycled solid waste, and to identify those regions that mostly contributed to it.

The amount of generated and selectively collected waste per month and year by regions was obtained from the Municipal Solid Waste Control System (PMSP, 2014).

In 2016, São Paulo had a population of 12,038,175 inhabitants and produced 5.2 million tons of solid waste (8% of Brazilian total). 68% was household waste, more than 90% was landfilled, and 2.36% was selectively collected for recycling.

The public selective waste collection of recyclable residential waste (papers, plastics, glass, and metals) is executed by: (i) two outsourced firms that collect door to door; (ii) 21 sorting centers operated by associations of waste pickers that have an agreement with the municipal government, and 20 associations of waste pickers without an agreement with the municipality.

The selective collection of recycled waste in São Paulo is carried out separately from the regular collection and does not cover all households and streets. No specific law mandates citizens to sort their waste in the generation sites and the expenses with the cleaning services are computed into the urban property and territorial tax.

Results demonstrated that São Paulo did not reach the targets for 2016. Only 84,652 tons of waste (2.36% of the total) were collected and recycled in 2016. The highest percentage of recyclable solid waste recovery was 10% in the region of Vila Mariana that has the highest family income. In the peripheral areas of Sao Paulo, the percentages of household recyclable solid waste recovery were much lower.

Regions with the greatest generation of household waste were not necessarily the regions with the most significant recycling rates (Table 1). Therefore, the actions for improving the recycling rates of household dry waste could be: (i) to prioritize the top 10 regions that generate more recyclable household waste and (ii), under the socio-environmental perspective, to carry out priority actions in the peripheral areas of the city inhabited by families with lower income and near areas of environmental protection.

It will be also necessary to strengthen the associations and cooperatives of waste pickers and to build more central mechanized waste sorters.

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TABLE 1: Ranking of household waste generation per region, percentage of recycled waste, and income per capita in São Paulo in 2016.

RANKING OF HOUSEHOLD WASTE	REGIONAL OFFICE	TOTAL HOUSEHOLD WASTE in 2016 (t)	% RECYCLED	PER CAPITA INCOME (SEADE, 2010)
1º	SE	182,152	3.51%	1,970
2º	CAMPO LIMPO	179,294	2.25%	1,074
3º	CAPELA DO SOCORRO	173,736	3.60%	1,001
4º	MBOI MIRIM	158,783	0.18%	513
5º	BUTANTA	157,497	2.28%	2,018
6º	MOOCA	152,384	2.14%	1,442
7º	ITAQUERA	146,374	0.85%	638

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RANKING OF HOUSEHOLD WASTE	REGIONAL OFFICE	TOTAL HOUSEHOLD WASTE in 2016 (t)	% RECYCLED	PER CAPITA INCOME (SEADE, 2010)
8º	PENHA	145,636	1.07%	861
9º	IPIRANGA	143,680	3.53%	1,309
10º	PIRITUBA / JARAGUA	136,428	1.31%	818
11º	PINHEIROS	134,865	5.84%	4,138
12º	CIDADE ADEMAR	125,744	1.90%	586
13º	FREGUESIA/BRASILANDIA	125,601	0.81%	739
14º	SAO MATEUS	121,900	1.12%	495
15º	LAPA	121,345	5.91%	2,266
16º	VILA MARIANA	116,995	10.06%	3,765
17º	MARIA / GUILHERME	112,868	0.84%	919
18º	SANTANA / TUCURUVI	108,125	2.91%	1,523
19º	CASA VERDE	103,552	0.79%	924
20º	SAO MIGUEL PAULISTA	98,583	0.45%	518
21º	SANTO AMARO	94,236	5.21%	2,678
22º	ITAIM PAULISTA	92,125	0.36%	469
23º	ARICANDUVA/FORMOSA	90,615	1.25%	1,248
24º	JACANA/TREMEMBE	89,002	0.58%	714
25º	SAPOPEMBA	81,059	0.88%	Not available
26º	VILA PRUDENTE	71,986	2.58%	871
27º	GUAIANASES	68,505	0.36%	463
28º	JABAQUARA	66,831	3.80%	1,233
29º	ERMELINO MATARAZZO	61,524	2.93%	703
30º	PERUS	44,617	0.05%	500
31º	CIDADE TIRADENTES	43,307	0.37%	407
32º	PARELHEIROS	36,187	0.01%	375

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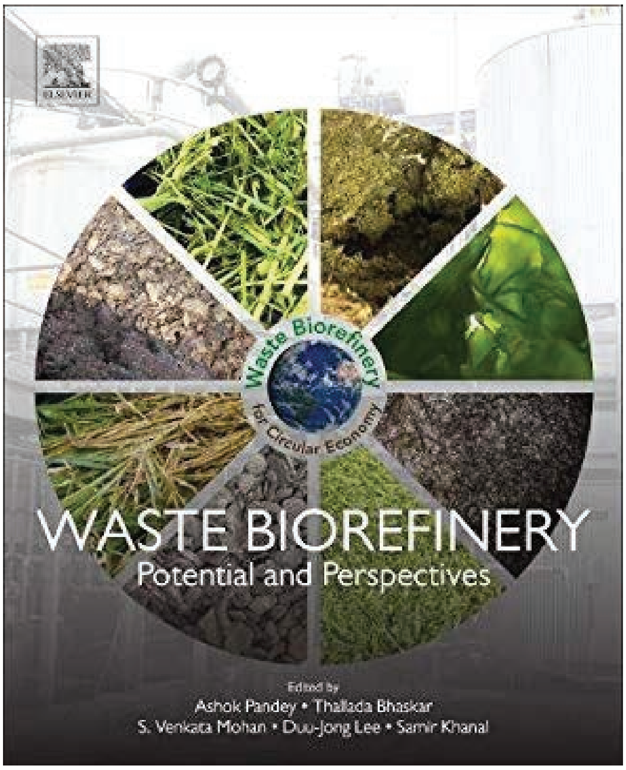
BOOKS REVIEW



WASTE BIOREFINERY: POTENTIALS AND PERSPECTIVES

by Thallada Bhaskar, Ashok Pandey, S.Venkata Mohan, Duu-Jong, and LeeSamir Kumar Khanal

The major drivers of bioenergy of this century include the improvement of energy security, addressing the issue of finite amounts of fossil fuels and natural resources and mitigation of the effects of climate change.

Consequently, due to the currently ongoing energy crisis, focus on the production of biofuels and bio-products has increased worldwide. Biofuels and bio-products can be produced from biomass by means of a conversion process known as biorefinery.

However, the key questions we should attempt to provide an answer to are: 'are biofuels and bio-products fully sustainable?' and 'how might we solve the biofuel/bio-product vs food and tank or table debates?'. Indeed, at times precious plant-based resources, obtained using fertile land and water, are shifted from the production of food and feed to the generation of bio-energy or bio-materials, thus resulting in the development of a sensitive social issue. This may only be solved by eliciting a change in mentality: indeed, wastes should no longer be regarded as residues to be treated and disposed of, but rather as a valuable resource that can be exploited as renewable feedstock for use in the production of bio-energy and chemicals. This is precisely the strategy adopted by the so-called "waste biorefinery" approach which is investigated in detail in the book "Waste biorefinery: potentials and perspectives". The latter incisively shows how true sustainability can be achieved by valorising what has until now been considered a useless residue, whilst at the same time helping to identify cost-effective strategies.

The book is divided into 26 Chapters grouped into 8 broad Sections. Section A relates to an analysis of the waste feedstocks suitable for use in biorefinery and the most recent technologies that contribute towards enhancing the sustainability and efficiency of the conversion process. In particular, it is highlighted how the waste biorefinery fits perfectly within the circular economy regenerative system. Section B provides a detailed insight into the advanced and innovative methods for biomass conversion, namely thermochemical and combined gasification-fermentation. Likewise, Sections C, D, and E focus on the wide range of possibilities related to the valorisation of food waste, municipal solid waste, and lignocellulosic waste, respectively. Food waste is preferentially used as a substrate in anaerobic digestion and acidogenic fermentation processes in order to gain biogas and biological monomers, respectively. Pyrolysis is the method investigated for use in the conversion of municipal solid waste into bio-energy products. Lignocellulosic resources are efficiently valorised in a number of ways, including hydrothermal treatment, pyrolysis, microwaves, and ultrasounds. Since lignin cell walls are particularly difficult to break down, specific pre-treatments, such as the use of ionic liquids, are also illustrated in the book. Furthermore, Section F explores the innovative field of water-based biorefinery relating to both the recovery of resources through bioelectrochemical systems and the potential of exploiting microalgae biomass. In the first part, nutrients, metals, energy, and chemical products are investigated and a series of case studies commented on. In the second part, in addition to considering microalgae cultivation for use in the production of bio-fuels, closing the loop of the microalgae biomass is the main goal, demonstrating an expanding market for food and feed additive production, and for high-value chemicals. Section G deals with biorefinery projects on the cutting edge of scientific research. At the beginning of the section, three types of biomass residues generated in arid/semiarid regions (palm tree residues, seawater biomass residues, and organic fraction of municipal solid waste) are reviewed. Subsequently, castor biorefinery is carefully explained with a special focus on castor oil and the different

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extraction methods applied. Insect-based bioconversion is another hot topic addressed in this section. Indeed, insects are capable of stabilizing organic waste while enabling the recovery of bio-fuels, fertilisers, food, and polymers. The last chapter describes how advanced thermochemical technologies, with particular focus on pyrolysis, are capable of converting low-value materials such as deinking residues from the paper industry into value-added products. Finally, Section H provides a discussion on integrated technologies and approaches associated with lignocellulosic biomass, including pre-treatments required, and describes a case study of a woody biomass biorefinery in Japan.

To conclude, through a balanced combination of different branches of science: chemistry, biology, engineering, and biotechnology, this book provides data-based information on the state of the art of biogenic waste utilisation within the field of biorefinery in which waste and resources are interchangeable.

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Professor Ashok Pandey is currently Distinguished Scientist at CSIR-Indian Institute for Toxicology Research, Lucknow, India and Honorary Executive Director at the Centre for Energy and Environmental Sustainability- India; he was the former Deputy Director for CSIR's National Institute for Interdisciplinary Science and Technology at Trivandrum, where he head the Centre for Biofuels and Biotechnology Division. Professor Pandey's research interests are on bio-based economy for the production of fuels and chemicals. He has over 1000 publications and communications, which include 14 patents and design copyright, 34 books, 99 book chapters, and 391 original and review papers.

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Duu-Jong Lee is currently a Life-Time Specially Appointed Professor of National Taiwan University (NTU) and Chair Professor and Dean of College of Engineering of National Taiwan University of Science and Technology (NTUST).

During the past 25 years of his career, he has completed several projects successfully and developed various processes with full-scale applications. The research carried out till-date has credited him with more than 1000 publications/communications. The citations in SCOPUS on Dr Lee's work has exceeded 8000 with h=42. Professor Lee is the recipient of many national and international awards and fellowships.

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Dr. Samir Kumar Khanal is an Associate Professor of Biological Engineering at the University of Hawai'i at Mānoa. Previously, he was a post-doctoral research associate and Research Assistant Professor at Iowa State University for 6 years.

Dr. Khanal is also an editorial board member of the highly prestigious international journal, Bioresource Technology and Korean Journal of Environmental Engineering.

He is also a recipient of CTAHR Dean's Award for Excellence in Research (2016), University of Hawaii at Manoa. Dr. Khanal is a professional engineer (environmental engineering) in the state of Iowa.

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PORTRAITS

Professor Dr. Ing. OKTAY TABASARAN
born in Neveshir (Turkey) 1938

Education

- 1957 study of civil engineering at the Technical University Stuttgart
- 1967 Doctoral. Degree (Dr.-Ing.) "Options for the acceleration of sludge digestion"

Academic activities

Assistant of Prof. Pöpel at the Institute for waste water, water supply and water quality management, University Stuttgart. 1974 Full Professor for Waste management, (the first one in Germany.

Education activities

Implementation of the topics of waste management (basics, landfilling, biological and thermal treatment and recycling) in a special course in the study of civil engineering. He was playing an important role in creating and establishing a master course "Infrastructure planning" and a Master course "Air Quality Control, Solid Waste and Waste Water Process Engineering (WASTE)" (in English) and was Study Dean of the interdisciplinary Diploma study Course "Environmental Engineering" He was the first professor who covered all aspects of waste management by integrating his team for special aspects. Hundreds of Diploma and Master Thesis were elaborated under his supervision. Eighteen Doctoral thesis were successfully finished under his guidance. From his school at least now international eight professors were born and nine more in the second generation.

Scientific activities

He established four areas of research at his Institute:

- Landfilling (development of gas production, gas migration, construction of gas and leachate collection systems);
- Biological treatment (technical development, product quality definition, odour emission control);
- Thermal treatment: test and optimisation of pyrolysis plants;
- Hazardous waste. Leachate treatment from hazardous waste landfills).

In these areas more than 100 research projects were carried out for the German research Ministry, the EU, the German environmental agency, the Environmental Ministries of nearly all German countries, and industry.

Next to that in his Institute analytical methods for waste analyses the design for waste management concepts were developed. More than 50 0f these concepts were elaborated.

Main achievement

He provided important contributions to the preparation of the first German waste law in 1972. And, in the following years, he was permanently an advisor to the government not only in Germany as well as in other countries.

Knowing that it is important for such an emerging scientific field that the results from research must transferred into practice, he organized seminars for people from the administration and consultants to promote the knowledge about waste engineering.

Especially a seminar with Poland and Turkey were a great international success. 2009 he was the Generel Secretary of the World Water Forum in Istanbul and leading member of the executive committee of the international 3 W conference in Istanbul in 2013.

Others

To disseminate the results of his research and to get a better feedback from the practice he was involved in many waste management concepts and planning processes for waste treatment plants.

He has more the 100 publications, was the editor of the publication series "Stuttgarter Berichte zur Abfallwirtschaft" with 80 editions under his supervision. One of the standard book for students in Germany about waste management was edited by him.

Personal remarks: Professor Tabasaran established "Waste Management" as an academic field in environmental engineering in Germany. He is the first and last generalist at a German university regarding waste management with the talent to familiarize himself deeply in special topics.

He has the talent to motivate his coworkers in a warm and familiarly way giving them freedom in their research and backstopping whenever it is necessary.

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A PHOTO, A FACT, AN EMOTION

"The "PIRANA" dumping site is used for the disposal of solid waste in the city of Ahmedabad, Gujarat, India. I visited the site, witnessed the situation and captured these images. It was completely beyond imagination; every 15-20 minutes loaded trucks entered the site to unload solid waste. On seeing this, it occurred to me that the population needs to learn how to reduce the generation of household waste. A multitude of small heaps of mixed solid waste were dotted over the site, the air was heavily polluted and the smell was intolerable; surprisingly hundreds of men, women and even children were working on the dump to scavenge wastes which they could then recycle and sell as a paltry source of income. They worked steadily, rummaging through the waste - it was a miserable situation.

The recycling process from this site is officially managed by the local authorities, however, this informal system often provides the main source of income for the scavengers. I realised that the health of these people could likely have been compromised, but that they needed to undertake these activities, often together with their children, to provide for their families. It was a truly shocking situation."

"IN SEARCH OF ..."

PIRANA - Ahmedabad of Gujarat, India

Prabha Jayesh Patel, India

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This photo had been selected to participate in the first edition of Waste to Photo in 2015, the photo contest connected to the Sardinia Symposium, International Waste Management and Landfill Symposium organised by IWWG.

The most significant shots were used to set up a photographic exhibition to illustrate the differences, the contradictions, the difficulties and progresses encountered by this complicated issue in a series of contexts throughout the world, ranging from the developing countries to the more industrialized nations.

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ABOUT THE AUTHOR

Prabha Jayesh Patel

Prabha Jayesh Patel, 44, lives in Ahmedabad, Gujarat, India. She took up photography five years ago. Since then, her awards have included being the winner of the 'Waste to Photo' award organised by the Sardinia Symposium and the "Clean & Green India", organised by ASSIST India/Ten Photos to Shake the World. Her documentary projects include "Salty Life" about the lives of salt workers in Little Rann of Kutch.

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