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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

THE POWER OF THE WRECK

THE NATURE OF WASTE (S. Antoniadis)

The second French edition of Marc-Antoine Laugier's (1713-1769) successful *Essai sur l'architecture* (Laugier, 1755) displays a rather well-known allegorical engraving by Charles-Dominique-Joseph Eisen (1720-1778). Besides giving a more visible explanation of his known theoretical approach (nature is the origin of everything, in a nut shell), the illustration, featuring Architecture as goddess seated on the ruins of a destroyed building showing a primitive hut to the genius of reason (a cupid), talks about landscape, nature and waste: three extraordinarily up-to-date items production-related debates still focus on. The picture is made up of closely-standing uncut trees supporting slightly-tamed branches that provide a roof among their partially-preserved boughs as model for possibly obsolence-proof building. Bypassing anthropomorphic, unreal and allegorically-charged suggestions, the illustration features an ambiguously anthropized landscape where nature blends with fragments resulting from the collapse of an arrogant (because irrespective of an "according to nature" praxis) building. Venturing a bold shortcut, we might subscribe to Laugier's tenet "nature generating artifice" as still enjoying large approval. It is a successful interpretative paradigm followed throughout the centuries, in various branches and various scales, in keeping with present-day results and applications both in techno-ecological fields, in the production of architecture and in land management.

Remaining faithful to the three-faced approach landscape-nature-waste, it is interesting to lay stress on the position allotted to each item – not indulging in arbitrary self-satisfying speculation, but accounting for the factual reality in which we nowadays work –, let's apply a few mutations: are we really certain that the rational relation of causality is to be univocally meant the way Laugier and other thinkers intended? What is more, are there preconditions nowadays to suggest their equation may be turned inside out into the "waste generating nature" formula?

The urban setting we live in is no longer the former, and above all we must admit that the presence of those remains merely occupying the bottom right corner of the French engraving has become much more cumbersome nowadays. Whereas in the abbot's mind that pile of ruins belonging to a decayed building was to have a merely symbolical meaning, our eyes and our awareness turn it into a real everyday experience. In the illustration the ruins are placed almost nicely at one side of a meadow, in our reality litter is massively present even in the inner space of our Earth. The increasing degree of obsolescence of (even architectural) products, the larger and larger amounts of abandoned areas and buildings and the recent resort to laying out untidy clusters of buildings dotting the country reveal the scattered (Rasmussen S.E., 1974) nature of our contemporary landscape.

Therefore, it is worth taking a different look at the artificial objects, potentially much more capable of supporting ecosystems, or even generating new ones, than we are led to believe. It's proved with simple – yet extraordinary – evidence when dealing with sea wreckage. Sometimes immense chunks of wreck on the bottom of the sea are at first seen as seriously impairing natural environment, yet later they prove to be the vital triggers of lush oases evincing a high degree of biodiversity. It would be wrong to interpret such evolution as the reappropriation of nature, as its winning back what was stolen. Biofouling operates in much more fascinating ways: not only does it restore, it upgrades. Man-made artefacts behave as effective trigger devices enacting more favourable conditions for "new natures" to develop.

In the wake of the above reflections, the choice has been to intentionally sink artificial objects with the aim to increase the biotic potentialities of certain areas. It is surprising to examine the range of objects used purposely in the various geographic-cultural areas in order to set up artificial reefs (Fabi et al., 2011): a sort of catalogue of unacknowledged objects, generally regarded as polluting garbage of our artificial world, from end-of-life New York subway train carriages, to hollow reinforced- concrete blocks, to the cumbersome tyres of lorries.

On the Earth's surface the same practice might be resorted to, involving even more discarded materials: segments of viaducts, portions of water-carrying infrastructures, frames of unfinished buildings, left-behind building-yard and temporary cranes is all wreckage impacting on man and landscape awaiting public opinion deliverance.

Laugier's allegorical illustration is to be re-interpreted, and the goddess' forefinger pointing at that artificial heap of materials deserves first of all to be seen in a new light; in this way the bases of real innovation can be laid, taking into account the huge and complex amount of artificial objects belonging to contemporary landscape.

THE FORM OF WASTE (L. Stendardo)

The power of the wreck is not only a matter of environmental opportunity, it is actually a matter of culture according to its widest meaning, and can be successfully

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FIGURE 1: The frontispiece for the *Essai sur l'architecture* (1st edition 1753) by Marc-Antoine Laugier (1713-1769). Engraving by Charles-Dominique-Joseph Eisen, 1755.

FIGURE 2: Wreck-scape, west trans-urban area of Padova. Photo by Stefanos Antoniadis, 2017.

dealt with from the point of view of architectural, urban and landscape design.

Architecture, as well as engineering artefacts, describe the route of civilization, make fundamental layers of material history, and sometimes represent peculiar events. These works are actually part of our memory and imaginary. They are a source for literature, art, cinema, but above all they do shape into form the physical space, the city and the landscape we live in. Beyond architecture (in the narrowest sense), all kind of construction (which includes ordinary buildings, infrastructure, equipment, machines…) may be considered part of this reservoir of formal and cultural resources, as long as their form is capable to overcome their obsolescence, which stands as their inescapable destiny.

A basic difference between architecture and ordinary construction, which may actually be held as a conceptual divide between what is architecture and what is not, is that the former is never obsolescent. Even when architecture is no longer able to cope with neither its original use or its eventual ones, when it gets wounded by time and neglect, when it is mutilated and dismembered, even when it is eventually transformed into ruins, it still is architecture, i.e. a form which is capable to generate space, further form, and landscape, a fragment that is still capable to interweave relationships with the context.

While dismissed, decommissioned, or abandoned architecture is headed to turn into ruins, obsolescent ordinary construction is headed to turn into debris. Ordinary construction – and especially infrastructure and machines – is always obsolescent. When some machines or infrastructure are obsolescent, broken in pieces, they become waste, scraps that may be recycled or, at best when it is worthy, exhibited as relics in a museum. This is why an ancient Roman aqueduct, even when it ceases supplying water to town, is not held as debris and no one would think of it as a waste management problem to cope with, but everybody would recognize it as an extraordinary landmark across landscape. On the contrary, a technologically advanced contemporary oil pipeline, a highly specialised device, is not likely to play such a significant role in the future. The smarter machines or infrastructure are, and the more technologically advanced a device is, the more rapidly obsolescent they become. This is clear enough, since planned obsolescence policies, along with disposable smart devices and machinery market, allowing no possibility to fix broken hardware, are actually flourishing, while the production of hazardous waste is over increasing, although we all eager to flaunt our environmental care worries.

Of course, we can easily see that there is a wide in-between range of artefacts. Architecture itself is getting smarter and smarter, sophisticated and high-tech, and the amount of technology that is some kind of added, though inalienable, value makes architecture potentially obsolescent. Yet while its technological endowment is bound to become debris, its formal core, since we are still talking about architecture, is going to be resilient to obsolescence. On the other side we may still recognize some formal remains in some ordinary construction wrecks, which is capable to make them survive as generators of form, space, memory, imaginary and so on, and finally acknowledge them as architecture in a broader sense.

Actually, the aptitude of a wreck to be acknowledged as architecture depends on its formal features; or rather we should say, on our skill to recognize its potential as formal and spatial material for architecture. It looks like this potential acknowledgment implies the complementarity of the human mind and the wreck, showing some relevant similarity with the concept of affordance as defined in environmental psychology by James J. Gibson (1966-1979). According to this acknowledgment the power of ruins, which is bound to the widely accepted concept of architecture, may be successfully shifted onto wrecks, so allowing not only rehabilitation and reuse of decayed built environment, through new functions, but a broader re-creation of architecture and space with strong cultural impacts.

These reflections can be implemented both in the recycling of built waste, such as infrastructural and built debris and scraps, and in a more aware attitude in architectural, urban and landscape design. An attitude that is not actually new, if we just recall that one of the most powerful images of the project for the Bank of England (1830), designed by Sir John Soane, was represented by its author as an imagined view of the building in ruins. Although nowadays the trend of architecture and civil engineering is to make artefacts based on such concepts as fitness and smartness, while ignoring any long-term anti-obsolescence resilience, trying to image one's project as ruins should be a must for today's architects as well.

The importance of form in the dichotomy between ruins and debris should finally be taken into account both for good design practices of new buildings, and for the acknowledgment of the wide asset of existing built objects that are spread throughout today's landscape.

Any effort in this direction is a step forward in the enlargement of our architectural and imaginative dictionary, and possibly a step forward towards a world that is richer in culture, resources, health and, why not, happiness.

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FIGURE 3: The Former Cattle Market of Padova (1967), by architect Giuseppe Davanzo (1921-2007). Photo by Stefanos Antoniadis, 2012.

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HOW CAN SUSTAINABLE CHEMISTRY CONTRIBUTE TO A CIRCULAR ECONOMY?

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ABSTRACT

The transformation from a linear to a circular economy and from a fossil oil-based to a biobased economy creates challenges that need to be solved. Challenges are associated with the introduction of biobased compounds, such as bioplastics, as new compounds, in existing material cycles and the difficulties to separate such compounds in a circular economy from conventionally used materials. The transformation, however, is necessary due to the expected limitation in fossil resources and associated climate and environmental issues. Sustainable chemistry aims on a simultaneous consideration of resource, production, product and recycling. The focus is not only on sustainable transformation of matter, but also on its origin and fate. Whenever biobased products are to be introduced in existing material cycles, following question might be considered beforehand: 1. Are renewable resources available to carry out production processes in order to meet the demand of certain products?, 2. Is the technology available to carry out recycling and production processes efficiently?, 3. How likely is the separate collection of products after use?, 4. Does the product eco-design allow a recycling of resources?, 5. Are additives as unwanted compounds circulated as well?, 6. Are recycled resources useable in repeatedly carried out production processes? and 7. Does society accept products based on recycled resources? Those questions can be addressed when totally new material cycles are generated. The challenge, however, is finding the beginning of an already existing cycle in a circular economy which allows an introduction of new materials and/or production as well as recycling processes.

1. INTRODUCTION

1.1 Sustainable chemistry and circular economy

Chemistry is generally defined as discipline dealing with the transformation of matter. Over time, several subdisciplines, such as inorganic chemistry, organic chemistry, physical chemistry and biochemistry, have been developed. Those disciplines deal with a narrowed but still complex aspect of chemistry. In order to address the challenges of sustainable development, a new approach – sustainable chemistry – has been emerged (Kümmerer 2017). Sustainable chemistry might be defined as discipline dealing with the sustainable transformation of matter, but this definition is incomplete. Sustainable chemistry does not only address the sustainable transformation of matter, but also its origin and fate. It further addresses social aspects, such as the demand and acceptance of products by society. Particularly the consideration of societal acceptance and demand for products goes beyond the 12 principles of green chemistry formulated by Anastas and Warner in 1998 (Anastas and Warner 1998).

In a circular economy, sustainable chemistry considers following four key elements: resource, production, product and recycling (Figure 1). The key elements are strongly influenced by resource availability, demand, eco-design and recyclability of products. The availability of resources influences the efficiency of production processes. The efficiency usually decreases when sufficient resources are available. The demand decides which product is formed. Product eco-design influences recycling and consequently recyclability is central to resource availability (Figure 1). The improvement of products and processes in order to sustainably address the four key elements is challenging. It is common to adjust production processes regarding the demand for products and to ignore the availability of resources, or to eco-design products regarding functionality and to ignore the need for recycling.

Even in a circular economy an increase in entropy and dissipative loss of materials cannot be avoided (Kümmerer 2017). Every approach to avoid a loss of materials would consume a disproportional amount of energy. Thus, a more realistic approach considering resource, production, prod-

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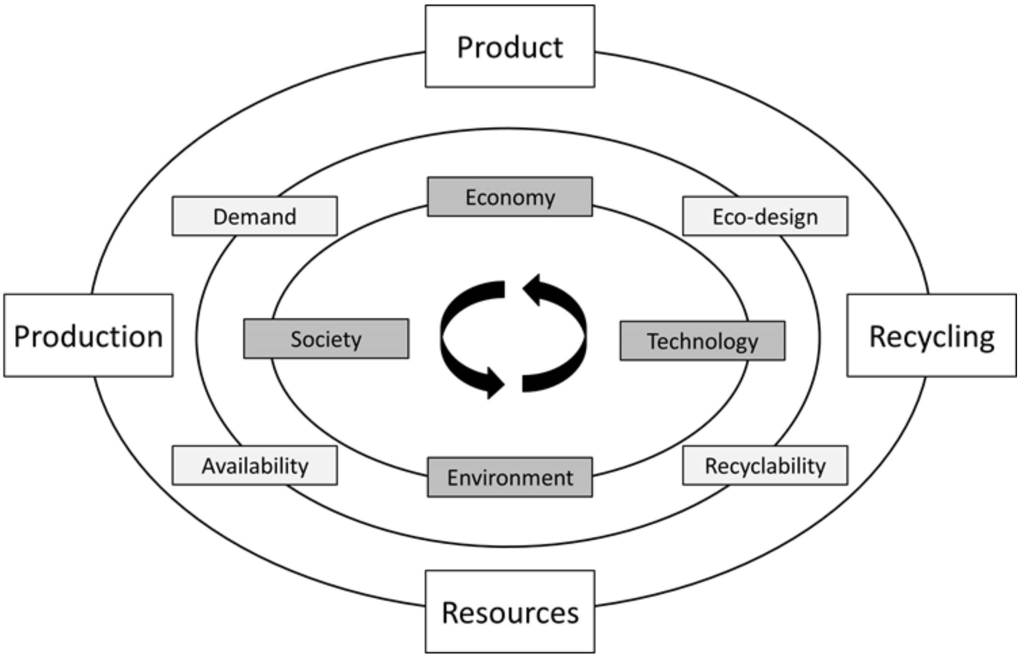

FIGURE 1: Sustainable chemistry and the circular economy. The key elements: Resources, production, product and recycling are dependent on availability of resources as well as demand, eco-design and recyclability of products. The two inner circles represent those aspects which are essential for a circular economy. The most inner circle is considered moving and it should be admitted that new approaches addressing the key elements may not be successful when societal acceptance, economic feasibility, technological possibilities and environmental impacts are considered inappropriately.

uct and recycling is needed which minimizes the loss of material. Recycling and production technologies are essential elements of a circular economy as resources provided by recycling are to be processed into new products. Failing to provide a sufficient amount of resources with a certain quality cannot result in new high quality products. Sustainable chemistry aims on considering the aforementioned elements simultaneously. The challenge, however, is finding the beginning of an existing cycle allowing an introduction of new materials and/ or processes which address resource availability as well as existing technologies with lowest environmental impact and highest social acceptance (Figure 1).

An example where key elements have not been addressed adequately is the introduction of bioplastic in the material cycle of conventional fossil oil-based plastics. Bioplastic, such as poly(lactic acid) (PLA), is often characterized as biodegradable. This stands as synonym for environmental benignity and sustainability. Biodegradability is an important aspect when plastic is released in the environment by accident or design. The eco-design of products, however, should not only focus on biodegradability, but also on recyclability. In Germany, the recycling of bioplastic is basically non-existent. PLA is not compostable fast enough to be mineralized in composting plants. Consequently, PLA is banned from being disposed with the organic waste. Due to difficulties to separate PLA from conventional plastic material it is also banned from being disposed with the so called recyclable material, a category where it actually should belong to. At the end, PLA is disposed with the residual waste and either incinerated or stored in landfills. This means not only a loss of functionalized material, but also a loss of all resources initially applied in biomass production (PLA is formed from lactic acid obtained after biotechnological conversion of sugars from biomass) and PLA formation.

The introduction of bioplastic was a reaction to the societal concern regarding the environmental impact of conventional fossil oil-based materials. The last couple of years industry has been using the commonly positive attitude of society regarding biomaterials. This created a pseudo-sustainability on the consumer's and producer's side, but the missing recyclability and material utilization at the end contributed to unsustainability on the side of waste management.

In a bioeconomy biobased products substitute fossil oil-based ones (Fitzgerald, 2017). This makes sense due to the concerns associated with the limitation in fossil resources and environmental impact of fossil-oil based products. Biobased products can be food and feed, chemicals and materials as well as fuels. At local scale, biobased products can be produced where needed. At global scale this involves a transport of biomass. The energy density of biomass, for instance, is low compared with fossil oil and more biomass is needed to be transported to reach the same amount of energy. Consumed food, feed and biofuels cannot be collected after use, but chemicals and materials basically can and should be considered for recycling. However in bioeconomy concepts the recycling of biobased products is usually not properly addressed and degradation is favored. Recycling, however, contributes to the preservation of resources applied in production.

Even though processes are available to convert biogenic resources into wanted products (Koutinas et al., 2014) it remains a challenge to meet resource availability and demand. Biogenic resources, such as biomass and organic residues, are literally available everywhere, and thus bioeconomy concepts can theoretically be applied everywhere. Quality and quantity, however, are not equally distributed due to differences in water, fertilizers and arable land availability around the globe. The different availability of resources makes bioeconomy concepts case specific for certain areas. Establishing a recycling of biobased products can contribute to resource efficiency. This is particularly of interest in areas which are limited in water and/ or land. Furthermore, it is contradicting when a bioeconomy starts with an extensive use of fossil resources, such as fertilizers.

The progressing interest in developing bioeconomy concepts worldwide makes it necessary to elaborate on following questions: How many material cycles (biobased and fossil oil-based) can be maintained in parallel? How can all the different streams be collected separately and recycled or can we even combine the cycles of biobased and fossil oil-based materials?

A couple of biobased materials, such as polyethylene and polyethylene terephthalate, are chemically identical to their fossil oil-based equivalents, and thus a mixing of materials is uncomplicated. However, as elaborated above, a mixing of PLA with fossil oil-based materials causes serious difficulties when it comes to separation.

The term recycling is well defined and describes the reuse of materials after disposal. For polymers, recycling can either occur by degradation to monomers and reuse of monomers as secondary raw materials or by maintaining the initiate structure and functionalization. Degradation thereby contributes to an increase in entropy. The higher the entropy the more energy is necessary to structure and reform new materials. Therefore, if possible, the target should be on maintaining the original structure and functionalization. The closure of the cycle of matter by recycling depends on the separate collection of materials in order to maintain the purity of material streams. In existing circular economy concepts this has already been shown to be challenging. Materials are either too distributed and/ or materials are mixed with other materials. A concentrated and pure material stream cannot be achieved without applying a disproportionally high amount of energy.

Whenever it comes to the introduction of new materials in an existing cycle, it seems appropriate in terms of circularity and resource efficiency when following questions are considered beforehand:

- Are renewable resources available to carry out production processes in order to meet the demand of certain products?
- Is the technology available to carry out recycling and production processes efficiently?
- How likely is the separate collection of products after use?
- Does the product eco-design allow a recycling of resources?
- Are additives as unwanted compounds circulated as well?
- Are recycled resources useable in repeatedly carried out production processes?
- Does society accept products based on recycled resources?

The answers to those questions are relevant for every circular approach. Resources and production processes are usually properly addressed due to economic planning. However, eco-design and recycling of products are not, but necessary to achieve a sustainable development.

2. CONCLUSIONS

Challenges associated to the transformation to a circular and biobased economy need to be overcome. It is rather unsustainable to introduce new products in form of biobased materials in existing cycles without aiming on a recovery and recycling after use. The aim of sustainable chemistry to address resources, production, product and recycling simultaneously is ambitious, challenging and difficult to achieve. Sustainable chemistry creates the disciplinary basis to discuss the sustainable transformation of matter, its origin and fate in a bioeconomy and is necessary to critically reflect and evaluate all the solutions and technical improvements that come along with a sustainable development.

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DEVELOPING A PRACTICAL FRAMEWORK FOR USE IN THE SEPARATION OF ALUMINIUM WASTE FROM RESTAURANTS AND ESTIMATION OF POTENTIAL BENEFITS

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ABSTRACT

Population growth, urbanization and technological progress have resulted in an increased generation of various types of wastes. However, a series of different techniques have been developed for use in the segregation of waste components, with waste being separated either at source or at the disposal site. In Iran, waste separation at the disposal site is more popular than source separation due to a lack of public collaboration. The city of Sari is located in the northern part of Iran and is the largest and the most populous city in Mazandaran province. Per capita waste generation in Sari city is 1 kg/person^{-1} day^{-1}. Due to a high population density but also to the presence of tourist attractions, a significant amount of waste is generated in the city, particularly by restaurants. Whilst the majority of wastes generated is constituted by aluminium drink cans, this study aims to develop a practical framework for the source separation of aluminium waste and to estimate the advantages represented by recovery. Data were collected from 25 restaurants by means of questionnaires and interviews. The results showed that the majority of restaurant owners and staff had been made aware of the need for source separation of waste via social media. However, no source separation had been implemented. The pilot study showed how approx. 320 million USD could be obtained through the recovery of aluminium waste produced by the restaurants studied per year.

1. INTRODUCTION

Population growth has resulted in the generation of a series of different types of wastes which require appropriate management (Abbasi and El Hanandeh, 2016). A key strategy in waste management is the source separation of waste (Meng et al., 2018; Rada et al., 2018; Rousta et al., 2017; Seyring et al., 2016). Nowadays, a wide series of different techniques are available for use in waste separation. In general, waste separation methods are classified into two groups including waste separation at source and waste separation at the disposal site (Seyring et al., 2016). Waste separation at source is of greater interest to developed countries than separation at the disposal site due to the lower cost involved, shorter time required, low contamination of recyclable waste, absence of mixing with other waste, and consequently lower treatment costs. In addition to the health and financial benefits, goods manufactured using the resulting recycled materials will be cleaner, and a large part of the cost of collecting and organizing waste and of washing and disinfection can be eliminated (Bartelings and Sterner, 1999; Rada et al., 2018; Watson, 1999). In Iran, 20% of total municipal waste is made up of recyclable materials such as paper, carton, plastic, glass, and metals, approx. 70% of which are compostable materials; therefore, the implementation of source separation would introduce a fundamental change in the management of solid waste (Hassanvand et al., 2008).

Although the advantages of waste separation are evident, few studies have addressed this issue to date. A pilot study was carried out by Zhuang et al. to separate garbage or household waste in the city of Hangzhou in the centre of Zhejiang state in China (Zhuang et al., 2008). They proposed a new source separation framework based on the classification of household waste into three groups including food, dry and harmful waste. In another study by Tai et al., a pilot program was conducted focusing on separation of municipal solid waste (MSW) from eight megacities in China (Tai et al., 2011). Moreover, they provided an analysis of collection of separated municipal solid waste in China. The results demonstrated that implementation of collecting separated MSW at source was relatively successful only in the cities of Beijing and Shanghai. According to the results obtained in this study, implementation of the

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proposed framework should be the main focus in China in order to encourage people to separate kitchen waste and recyclable materials at source. They also pointed out the main challenges of waste separation which included a lack of legislation, lack of inter-institutional coordination, and lack of public awareness. Branstad et. al. (2011) evaluated the life cycle assessment of separating solid household waste at source in Sweden. Since 2001, families have been able to separate waste into six categories in Sweden. The present condition of waste separation was compared with an ideal scenario in which families entirely separate all waste at source, and with a scenario with no waste separation. The results show that under the current conditions, the source separation of waste yielded significant environmental benefits in comparison with non-separation. Zhang et al. (2012) studied popular views of municipal waste separation in Shanghai, China. In this study, face-to-face interviews and questionnaires were used to collect information about waste separation. The main outcome of this study was that only limited numbers of people separated their wastes, although they were familiar with the environmental need for waste separation.

Nowadays, a large percentage of formerly bottled drinks are supplied in metal cans, generally made of aluminium or steel. According to statistics provided by industrial analysts, 80 percent of the market is devoted to drink cans, indicating a high consumption of these compounds compared to other goods worldwide (Pinkham, 2002). This trend is expected to continue in the future, with beverage industries using metal cans for packaging (Chen and Graedel, 2012). Zartabi (2008) investigated the possibility of separation and recycling of drink bottles at the food courts located in Tehran city council zones 1, 2, 5 & 6. The results showed that approx. 75% of bottles were not separated, largely due to the high volume of bottles. This pilot study was implemented in four regions. As a result, an integrated household waste management system, a recycling system, and a mechanical water absorption system for food waste were developed to promote the separation of waste at source. Geographically, Asian and European countries have represented the largest market in recent years, with Asia expected to dominate the metal can market in the future (Sahota, 2009). Indeed, the production of aluminium from recycled aluminium requires 90% less energy than production from ore, and the recovery of aluminium reduces contamination by up to 95% (Green, 2007). If one ton of aluminium is separated from waste and re-used, 400 tons of ore and 700 kilograms of coke and bitumen will be saved (Quinkertz et al., 2001). It is therefore possible to efficiently reduce the respective contaminations and costs by implementing complete separation of this type of waste. Since cans disposed of by restaurants, this would correspond to rendering all restaurants a source of minerals for can manufacturers. To date, no efforts have been made to separate waste cans at source.

The main aims of this study were to investigate the possibility of improving management of separation of aluminium packaging and addressing the challenges, and to determine the amount of waste aluminium cans generated by restaurants and examine the financial benefits to be gained from separation of this type of waste.

2. MATERIALS AND METHODS

2.1 Case Study

Sari, the capital of Mazandaran province, generates the highest amount of commercial waste in the province. Sari is the largest and most highly populated city in Mazandaran, Iran, with a population of 296,417 persons according to a 2011 census. The per capita waste generated in this city reaches an average of one kg person^{-1} day^{-1} waste (http://saricity.ir/En/HomeEn?OrgId=21). Currently, 250 to 300 tonnes of waste are produced per day in Sari. Due to its proximity to the sea, the forest and tourist attractions, Sari is served by a large number of restaurants which generate a considerable amount of waste aluminium cans.

2.2 Data collection and pilot study

The study was conducted at 25 restaurants located in Sari city. Both questionnaires and face-to-face interviews were used for data collection. Questionnaires A and B were prepared to understand the challenges of implementing can waste separation at source and to provide an estimate of the number of aluminium drink cans present in restaurants. The questionnaires used in this study are provided in the supplementary information.

To investigate the feasibility of implementing the separation plan, the pilot study was conducted in a restaurant located in Sari during 25 days. This pilot was implemented in agreement and coordination with the restaurant staff. Two restaurant staff were initially trained, and one waste bin equipped with a compactor was provided in which to place empty metal cans; finally a request was made to inform us once the waste bins were full for collection. During the study period these bins were collected seven times and the contents were sold. In conducting an economic analysis, we considered the average price of aluminium waste of 2.82$ per kg.

3. RESULTS AND DISCUSSIONS

3.1 Results obtained from questionnaire A

The first sections of the questionnaire contained information relating to awareness of the need for waste separation at source. The results derived from the first section of the questionnaire are shown in Figures 1 and 2. Most respondents were familiar with the need for waste separation at source, with media playing an effective role in informing the public. The results demonstrated that only 8% of respondents admitted to a lack of knowledge to this regard.

Figure 3 illustrates the results of facilities provided for waste separation at restaurants by the Sari city council. As seen in Figure 3, only 4% of respondents confirmed that the council provided the required facilities for waste separation, whilst the rest received almost no services. The satisfaction of restaurant owners with waste collections is shown in Figure 4. The information obtained revealed a relative degree of satisfaction with the current waste management system.

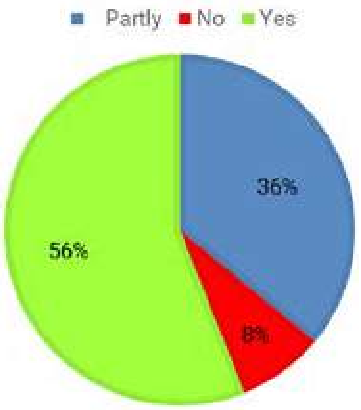

FIGURE 1: Public awareness of the importance of the source separation of wastes.

The next sections of the questionnaire examined the reasons underlying the lack of waste separation at restaurants. Inadequate facilities and the huge volumes of waste were indicated as the most important problems, with solutions identified by the restaurant owners relating to the provision of waste bins equipped with a compactor and an increased waste collection frequency. The majority of restaurant owners tended to obtain their information from brochures and staff training courses.

To implement a waste separation plan, the use of incentives was identified as the most successful approach aimed at increasing the participation of commercial sectors such as restaurants in waste separation programs.

64 percent of respondents agreed with the use of incentives, whilst 52 percent indicated punitive measures as an effective approach for the implementation of waste separation at source (Figure 5). Punitive measures included stopping waste collection services, the payment of fines, and tax increases. As shown in Figure 6, training was suggested as the best option to enhance the awareness and participation of restaurant staff in executing such a plan.

3.2 Results obtained from questionnaire B

An estimated 48 kg of waste cans is generated on average per day in the Sari restaurants. Maximum waste can generation was 140kg and the lowest 15kg (Figure 7).

Considering the positive correlation between the number of customers and generation of waste cans, the number of customers were measured at each restaurant. Figures 8 and 9 demonstrate daily generation of waste cans and the number of customers per season at each restaurant. It is estimated that 233g waste is generated per customer per day. The highest numbers of customers were observed in summer with an average of 535 persons per day. Subsequently, the highest amount of waste cans was generated in the summer, followed by the spring, corresponding to an estimated average of 529 customers (Figure 10). As shown in Figure 10, aluminium containers constituted the largest portion of the collected waste cans, corresponding

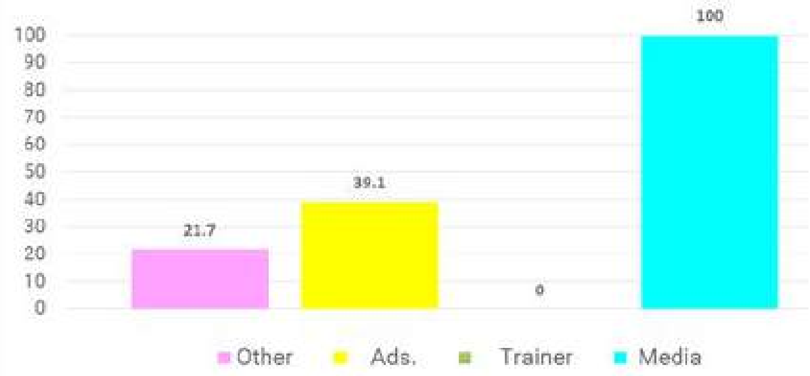

FIGURE 2: Means of increasing awareness of the need for source separation of wastes.

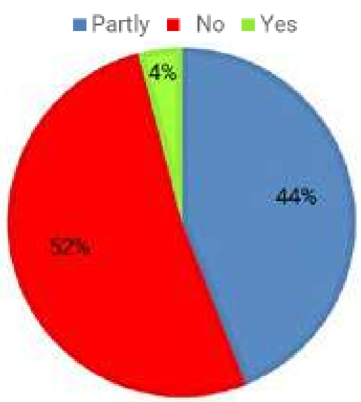

FIGURE 3: Restaurants equipped with facilities for waste separation.

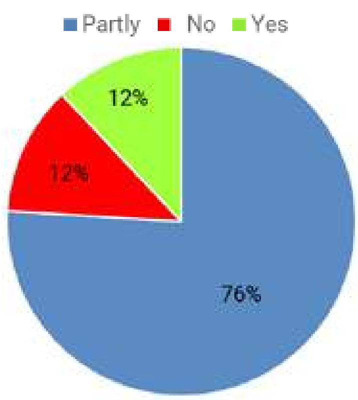

FIGURE 4: Satisfaction of restaurant owners with current waste collection.

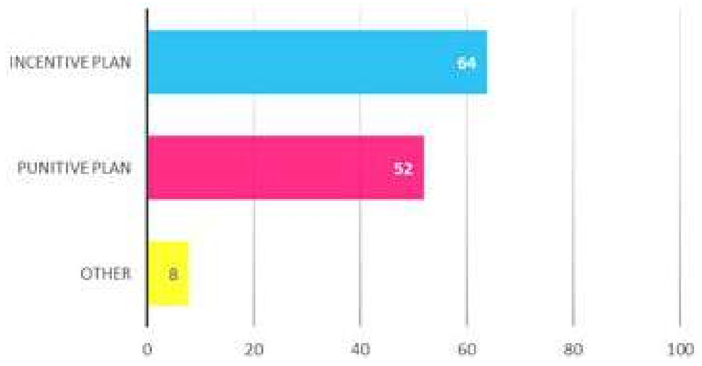

FIGURE 5: Legal solutions for implementation of a waste separation plan.

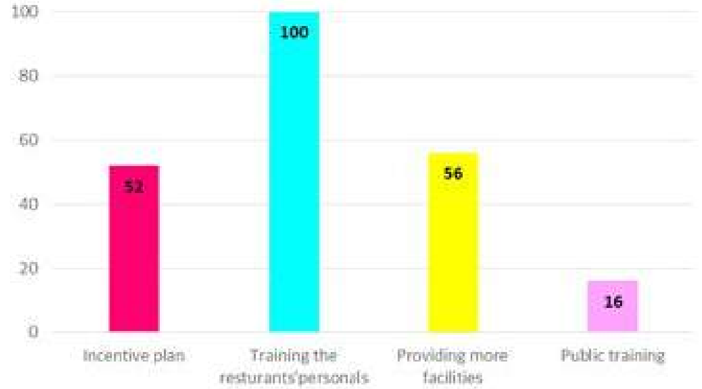

FIGURE 6: Suggested solutions to increase awareness and participation of restaurant staff.

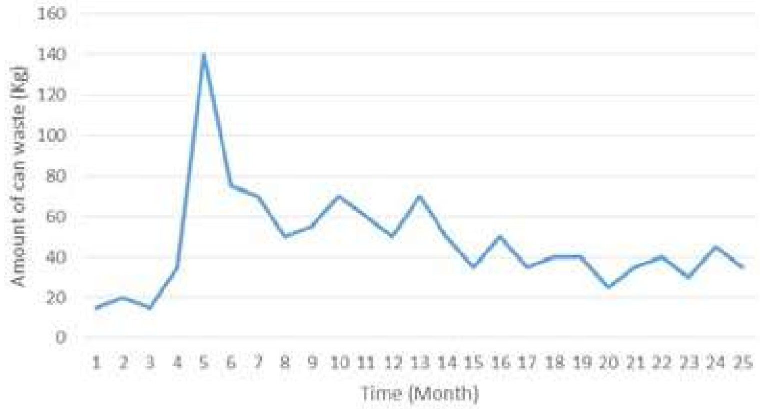

FIGURE 7: Daily generation of waste cans at the restaurants.

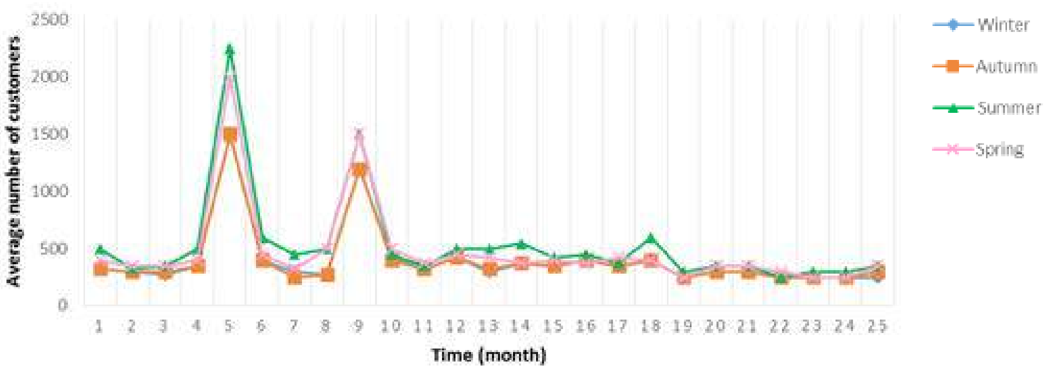

FIGURE 8: Number of customers per season at each restaurant.

to 52% of total container wastes. In addition, no separation of waste cans was conducted.

Statistical analysis revealed that the generation of waste cans peaked in the summer with an average rate of 8750 cans per day for a total weight of 140 Kg. The total amount of waste generated in the restaurants studied was estimated to be approx. 350 Kg per day.

3.3 Results of pilot study

The pilot study was carried out over a 25-day period from 27/04/2017 to 21/05/2017 at the Atishe restaurant in Sari. The results of this study are summarized in Table 1. By separating waste aluminium cans, 78.3 Kg of waste cans was collected and sold for 220$.

3.4 The challenges of the source waste separation program in Sari city

3.4.1 Lack of coordination between relevant organizations

In order to implement and carry out waste separation at source, an effective dialogue and coordination between organizations in charge of waste collection and separation, such as the government, the city council, and finally the private sectors should be facilitated.

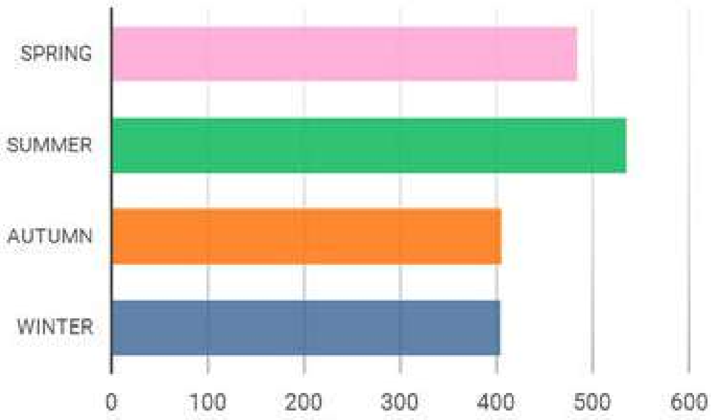

FIGURE 9: Average daily number of customers throughout the different seasons.

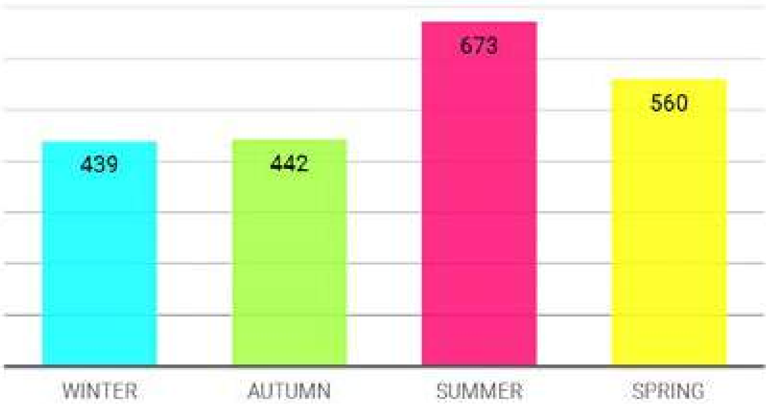

FIGURE 10: Daily average of waste cans generated at restaurants per season.

FIGURE 11: Types of drink bottles discarded at restaurants.

TABLE 1: Results of pilot study of separation of cans at Atishe restaurant in Sari city.

Date of collecting	Weight (Kg)	Selling price
28/04/2017	9.200	25.944
03/05/2017	11.700	32.994
05/05/2017	8.600	24.252
10/05/2017	14.600	41.172
12/05/2017	8.500	23.97
17/05/2017	13.800	38.916
21/05/2017	11.900	33.558
Total	78.300	220.806

3.4.2 Regulation and environmental policy

The government plays a major role in supporting and implementing comprehensive plans for waste management and waste recycling. In Iran, the waste management system has been heavily affected by the absence of environmental regulation, with legislation at times being so generic or obsolete that huge issues may be encountered in waste management programs. The promulgation and enforcement of new legislation would contribute towards fostering the implementation of source separation programs for waste aluminium cans .

In Sari, the city council is responsible for waste management services. Incentives and punitive measures might be applied by the council in an attempt to galvanize the commercial sectors into performing waste separation at source. This would contribute towards reducing the amount of collected waste, landfill size and the cost of waste management.

3.4.3 Lack of facilities

The effective provision of the necessary equipment would increase the participation of restaurant staff in separating waste cans. The use of designated trucks for the collection of separated waste, waste bins equipped with compactors, a suitable timetable for the collection of waste cans and training are the suggested main requirements of implementing any waste separation program.

4. CONCLUSIONS

The source separation of waste represents a priority strategies in any waste management system. Sari city, in the centre of Mazandaran province in Iran, generates the largest amounts of waste throughout the province. Due to the presence of tourist attractions, numerous restaurants are present in the city, and are responsible for generating huge quantities of waste aluminium cans. A pilot study was thus conducted to collect data relating to the amount of waste cans generated and to obtain information on waste separation and the challenges involved. Accordingly, data were collected from 25 restaurants by means of interview, field studies, and questionnaires. The results showed the generation of waste aluminium cans peaked in the summer with an average rate of 8750 cans per day. The total amount of waste generated in the restaurants studied was estimated in approx. 350 Kg per day. The pilot waste separation study collected 78.3 Kg of waste aluminium cans, which were then sold for 220$. A lack of coordination between the relevant organizations, insufficient legislation and lack of environmental policy, together with lack of facilities represented the main challenges to be overcome in implementing a waste separation program in the restaurants in Sari, Iran.

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FLOAT-SINK SEPARATION OF CONSTRUCTION AND DEMOLITION WASTE FINES

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ABSTRACT

Landfilling and waste incineration are two major waste management options. However, due to their carbon content, some wastes may be unsuitable for these systems. Therefore, methods capable of removing organic carbon from wastes should be identified. One of these wastes is represented by construction and demolition fines. In this paper, we investigate the use of water in separating the waste by density, to verify the suitability of this method in the separation of carbon-containing materials, both in lab and field scale tests. The results obtained show that half of the carbon (measured as volatile solids) can be separated. However, this method fails to reliably produce a sink fraction suitable for landfilling, as it continues to be characterized by an excessively high organic material content.

1. INTRODUCTION

Worldwide, landfilling continues to represent the main means of disposing of waste. In the EU, landfilling is the most commonly applied waste treatment method, with over one billion tons, or 48% of all waste, being landfilled in 2012 (Eurostat 2016). With the aim of reducing the amounts of waste forwarded to landfills, as well as minimizing landfill gas and leachate emissions - legislation in Europe and developed countries has created a situation in which waste with a higher than marginal organic content is being diverted from landfills towards thermal, biological and other treatment systems. As a consequence, the environmental impact and costs of treating large waste streams has risen sharply, particularly due to the presence of waste streams with properties that fit neither of the treatment systems. Construction and demolition waste (CDW) fines are an example of this type of waste. Construction and demolition waste is the largest waste stream in the EU, including mining and quarrying wastes, accounting for 33% of all waste produced (Eurostat 2016), with fines constituting a major portion of the CDW (Jang, Townsend 2001). Huang et al. (2002) reported fines <40 mm as representing 52% of CDW, and Montero et al. (2010) reported 37.5% as fines <8 mm. CDW fines from Nordic construction and demolition sites typically contain significant amounts of wood since houses in this region are frequently based on timber structures. This makes the waste heterogeneous in its physical properties (particle size, density etc.), rendering conventional mechanical separation complex and expensive. In previous CDW studies conducted in Japan (Montero et al. 2010), the feasibility of wet density-based separation of organic matter has been demonstrated, although a variety of separation steps implying an increasing complexity and higher costs was used. Di Maria et al. (2013) investigated the use of soil washing equipment for use in the wet separation of residual municipal solid waste (MSW) fines. However, the organic waste fraction was removed and not considered. Fines from an MSW landfill were treated in a wet jigger (Wanka et al. 2017), however, fines <10 mm were not studied. Float-sink devices are available in many countries, including the UK (Haith recycling group), Germany (Beyer), US (Hosokawa polymer systems) and Sweden (Norditek, since 2nd half of 2017). In this study, a residual CDW fine fraction (<40mm) was characterized with the aim of investigating a new treatment method. Float-sink separation was investigated, both in lab and field scale to verify whether this method was suitable as a single method for use in the separation of carbon-containing materials from CDW fines. The resulting sink and float fractions were then characterized to check their suitability for landfilling or incineration, respectively, without further treatment.

2. MATERIALS AND METHODS

2.1 Waste origin

A Swedish waste management company site provided unsorted CDW crushed using a Komptech Terminator 5000 crusher and sieved using two Komptech Nemus 2700 with 40 mm drum sieves. The resulting fine fraction contained a

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heterogenous mixture of wood, stones, mineral wool, plastics etc. (Figure 1), which was analyzed in both lab and field scale tests.

2.2 Sink float lab scale

From the waste processing site in the south of Sweden, three 60-liter samples were taken on 3 different days at intervals of approx.10 days in early 2016. In the lab, subsamples were taken using a method similar to that recommended in the Swedish waste association guidelines for waste analysis (Avfall Sverige 2013); an elongated loaf was formed by pouring the waste onto a long strip of construction plastic, and straight segments were taken randomly from across the loaf.

2000 g samples (wet weight) were obtained and added to 60 liters of water in a 90-liter plastic tub similar to a waste bin. The mixture was stirred vigorously and left to settle for 5 minutes. The floating parts were manually removed and put into a 200 and 250 micrometer sieve to let the water drain back. The settled parts were left in the water for the next addition. A second 2000 g sample was then taken from the loaf, added to the same water, and so on. This was repeated 10 times. For the first of the three samples, this was repeated 11 times with the last using 4600g of waste. The floating particles were put into an oven to dry at 70°C after each addition of waste. The sunk particles were left in the water for the next addition. Once all the waste had been sink-floated, the process water was removed using a 68-micrometer sieve. The settled materials were put into metal buckets and dried at 70°C for several days until dry.

2.2.1 Sink float field scale waste and sampling

Twenty cubic meters of waste was transported from the processing facility to a test site in the north of Sweden in the spring of 2017. From this pile, waste samples were initially taken using a front loader tractor, which was weighed at the plant vehicle scale, with an accuracy of ± 20 kg.

The resulting float and sink fractions were weighed using the same procedure. Fractions were sampled using the "loaf method" described above, by using the tractor to form an elongated string of approx.10 meters, and then obtaining three random samples.

FIGURE 1: Example of CDW fines <40 mm.

2.2.2 Experimental setup and procedure

In order to investigate the validity of the results from the lab scale test for use in a large-scale process, a batch field scale experiment was set up.

An open 20-m^3 bulk waste container was filled with 10 m3 water and used for density separation. Using a front loader, 2.5 tons of waste were dumped into the container and stirred with the tractor using the bucket scraping from the bottom. Subsequently, the floating particles were mechanically removed by the tractor, letting the water run off towards the side of the container. The final pieces were removed using a hand net. The water was pumped into a GT 500 D, 1000/0500 geotube, to which a flocculation agent, BASF ZETAG 8140, was added at 20 grams per m^3 to prevent clogging of the pores. Following removal of the water using a 2-inch heavy duty sludge pump, the settled particles were excavated using an excavator.

This process was repeated three times, although for the last two replicates the waste amount was changed to 1.25 tons due to the thickness of the floating material, corresponding to approx. 30 cm in 67 cm water, to prevent mixing of the float and sink fractions.

2.3 Characterization assays

Samples from the lab scale float and sink fractions were taken using a riffler and milled using cryogenic milling and/or a ring mill at an external laboratory (ALS Scandinavia, Luleå, Sweden). The untreated material and materials from the field scale tests were milled using a Blendtech xpress mixer to particle size <10 mm using the sample preparation method described in the EN 12457-4 leaching test standard.

Total solids (TS) and volatile solids (VS) of the milled samples were analyzed by first drying the samples for 24 hours at 105°C and then igniting them for 2 hours at 550°C according to Swedish standard SS 028113. TS was calculated by dividing the dry weight of the sample by the wet weight of the sample. VS was calculated by dividing the loss on ignition by the TS. Total organic carbon (TOC) was analyzed using a Shimadzu TOC-V SSM3 Total Organic Carbon Analyzer. The organic content of the sample was measured using the direct method, as described in the European standard EN 13137. Total carbon (TC) was analyzed by measuring the formed CO_2 after oxidation in oxygen at 900°C. TOC was measured in the same way, first removing carbonates through addition of HCl. These measurements were repeated five times. An analysis of the elemental, inorganic and organic carbon was performed by means of a Netch STA409 thermoanalyser using simultaneous Thermogravimetric analysis (TGA) and quadrupole mass spectrometer (QMS). Dried and milled material was used with a sample weight of 134.8 ± 2.31 mg. The heating rate used was 10°C min^{-1}, starting from room temperature up to 1000°C in argon and air atmospheres. The gas flow was 100ml min^{-1}. Elemental carbon content was calculated according to (Kumpiene, Robinson et al. 2011) using GNU Octave v. 4.2.1 to calculate the integrals.

Leaching of metals and metalloids was carried out in a one-step batch leaching test at L/S (liquid/solid ratio) 10 and performed according to European standard EN 12457-4. Water samples were analyzed using ICP-AES or ICP-SFMS for all elements with the exception of fluoride, which was analyzed according to ISO 10304-1. Sulfates (SO4) and chlorides were analyzed in the process water following final float separation. The biomethane potential (BMP), also known as GB21, was analyzed according to Chen et al. (1995) using a 3:1 waste/inoculum on a VS basis as adopted from Owen et al. (1979).

The respiration activity of the sink fraction was analyzed at an external university using a Sapromat respirometer (Comp. Voith, Germany) at 20°C. Large residues >10mm,including metal objects, stones, and glass, was sorted before analysis. Samples were taken using a riffler and watered to 70% of the water holding capacity (WHC) before analysis.

Elemental analysis was performed using a Thermo Scientific Niton XL3t XRF analyzer. The milled samples were placed in 100 ml LDPE plastic bags and sampled 3 times on each side at non-overlapping spots.

The calorific value was determined for the milled samples using an IKA c200 bomb calorimeter, using no support fuel, and oxygen at 30 bars of pressure.

The water holding capacity was measured in a similar way to that described by Bergman (1996), placing 1 liter of the saturated waste in plastic cylinders on geotextile, covering the top with plastic, letting the water run off for 2 hours, and then measuring the weight and comparing it to the dry mass.

Chlorides and sulfates were analyzed spectrophotometrically (AACE Quaatro, Bran + Luebbe, Germany).

All analyses were carried out in triplicate, at least. Unless otherwise specified, results are presented as “average value” ± “standard deviation”.

3. RESULTS AND DISCUSSION

Characteristics of the raw waste are shown in Table 1. A factor 3 variation of VS was observed, clearly showing the heterogenous nature of this material. This is also reflected in the mass balance for the float-sink procedure shown in Figure 2, where 2.8 times more material floats in the field scale tests. This underlines the need for a robust treatment method.

Mass balances for TS, VS and TOC are shown in Figure 2.

3.1 Characterization of sink fraction in lab scale tests

3.1.1 Biological activity

The carbon content of the sink fraction (Table 2) is too high (>6%) for landfilling as non-hazardous waste according to Swedish regulations. However, as shown by RA4 and GB21 analysis (Table 2), biological activity is low. German regulations for landfilling of mechanically-biologically treated wastes establish a limit for RA4 of 5 mg O_2 g TS^{-1}, more than 10 times the observed value of this waste. GB21 displays a gas potential approx. 20 times lower than the German limit for mechanically-biologically treated wastes of 20 ml/g TS. TGA measurements (Table 2) also show that a significant part of the carbon is elemental or inorganic; 18%±1% of the total carbon is organic carbon, 73%±8% elemental and 10%±7% is inorganic carbon. This means that the biological activity of this waste is overestimated when using only TOC or LOI analysis, as discussed in earlier studies (Kumpiene et al. 2011), and as supported by RA4 and GB21 analysis.

TABLE 1: Characteristics of the waste used in the two experiments

	Unit	Lab scale	Field scale
Total solids (TS)	% of wet weight	73 ± 3	75 ± 2
Volatile solids (VS)	% of TS	14 ± 4	42 ± 4

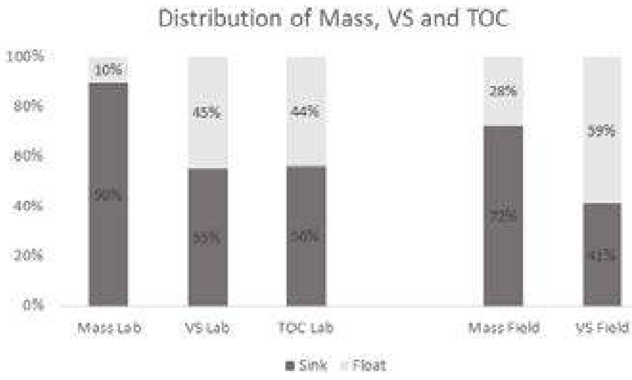

FIGURE 2: Distribution overview of lab and field scale sink-float separation tests. Average values shown, n = 3.

3.1.2 Leaching test

As shown in Figure 3, leaching from this material is low. Based on the Swedish regulation for landfilling, the majority of metals leach less than the limits for inert waste. The exceptions to this are antimony and fluoride. Antimony is used in paint, glass and ceramics (Weast 1982), and has been shown to leach from CDW in previous studies (Butera et al. 2014). Fluoride leaching is 14 mg/kg TS with a stan-

TABLE 2: Characterization data for the sink fraction of the lab scale tests.

	Unit	Value
VS	% of TS	10±2
TOC	% of TS	6.1±1.4
WHC	g water/g TS	80±10
RA4	mg O_2/g TS	0.4±0.09
GB21	ml gas/g TS	1.0±0.7
As	mg/kg TS	18.6 (17/18 measurements <LOD)
Pb	mg/kg TS	51 ± 18
Cd	mg/kg TS	<15 (LOD)
Cu	mg/kg TS	42 ± 6
Cr	mg/kg TS	86 ± 18
Hg	mg/kg TS	<15 (LOD)
Ni	mg/kg TS	76 ± 4
Zn	mg/kg TS	345 ± 109

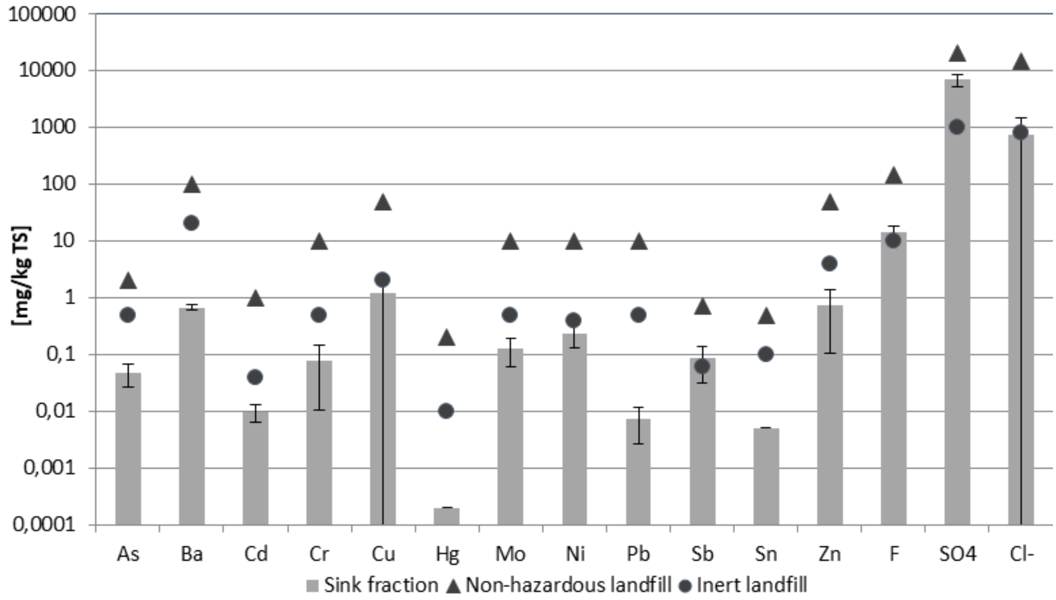

FIGURE 3: Leaching of sink fraction, n=9.

dard deviation of 3.8 mg/kg TS, which is close to the legal limit for landfilling of non-hazardous wastes of 10 mg/kg TS (NFS 2004a); a result similar to other studies (Saca et al. 2017). As for Sulfates and chlorides, these are measured in the process water after all the waste has been added, meaning that leaching of these compounds from the sink fraction will likely be lower, as the waste has already been washed in the float-sink process.

3.1.3 Landfilling of the sink fraction

In Sweden, landfilling is regulated by law NFS 2004a and subsequent amendments. Organic carbon is measured as TOC and determined using (EN 13 137 2001). The established limits in Sweden are 3%, 5% and 6% for inert, non-hazardous and hazardous wastes, respectively. Some exceptions are provided for, thus allowing homogenous wastes with less than 10% TOC to be landfilled (NFS 2004b).

In lab scale tests, the sink fraction was shown to have a TOC of 6%. If this waste could be considered as homogenous, it would be suitable for landfilling as provided for by the exception rules mentioned above. However, field scale experiments showed a VS content of 25% in the sink-fraction, which, in line with the VS/TOC ratio of lab scale tests, would imply a TOC of 15%, thus not suitable for landfilling. However, in other EU countries, such as Germany and Austria, other means may be applied to assess whether a waste features an appropriate organic carbon content for landfilling. Germany has established limits for respiration activity for wastes treated by mechanical-biological processes (Abfallablagerungsverordnung 2001) of 5 mg O_2 g^{-1}, for which the sink fraction from the abovementioned lab scale tests yielded values as illustrated in the table below. Assuming the same VS/RA4 ratio as in the lab scale test, this would provide an RA4 value for the field test of 1 mg O_2/g TS, still well within the German limit for landfilling.

As shown in Table 4 and Table 5, both the sink and float fraction absorbed significant amounts of water corresponding to three- and two-fold the dry weight, respectively. Accordingly, it would be necessary to dry the material following separation to avoid the landfilling of excessive amounts of water.

Due to the considerably wide variability between the different tests, this method cannot be considered adequate for reliably producing a sink fraction suitable for landfilling in Sweden.

3.2 Characterization of the float fraction in lab scale

Ocular characterization shows a high content of wood and organics in the float fraction, which is consistent with the VS of 69% (Table 3). For incineration, typically a VS content of 20% (Williams 2005) and 25% (Hulgaard, Vehlow

TABLE 3: Basic characteristics of the float fraction in the lab scale test.

	Unit	Value
VS	% of TS	69 ± 8
Ash	% of TS	31 ± 8
TOC	% of TS	42 ± 3
HHV	kJ/g TS	14.2 ± 1.38
As	mg/kg TS	17 ± 7
Pb	mg/kg TS	20 ± 8
Cd	mg/kg TS	<10 (LOD)
Cu	mg/kg TS	53 ± 14
Cr	mg/kg TS	130 ± 34
Hg	mg/kg TS	<11 (LOD)
Ni	mg/kg TS	<61 (LOD)
Zn	mg/kg TS	215 ± 39
Ca	g/kg TS	69 ± 20
S	g/kg TS	13 ± 6

2010) of wet weight is needed for a waste to be incinerated without using support fuel. Given that the VS to ash ratio is 2:1, the float fraction from the lab scale tests will be combustible at any moisture content below 70%. As shown in Table 4 and Table 5, after the sink-float separation the float fraction had a moisture content of approx. 50%, meaning there is no need for drying of the float material after separation.

The heating value of the float fraction was found to be 14.2 ± 1.38 MJ kg TS^{-1} (Table 3). Given a 75% TS content, this gives a heating value equal to or above that of municipal household waste of 9-13 MJ/kg (Williams 2005, Avfall Sverige 2014).

The sulphur content is high, 13 g kg^{-1} TS, likely from gypsum $CaSO_4$ $2H_2O$, as the waste also contains Ca. Typically, waste forwarded to waste incineration contains approx. 2 g kg^{-1} TS of sulphur (Williams 2005, Hulgaard, Vehlow 2010). Higher levels might cause problems with corrosion and SO_2 emissions. However, when incinerating alkali and silicate rich wastes such as industrial or municipal solid wastes (MSW), an addition of CaO and SO_x may help to reduce ash related problems and corrosion (Skoglund et al. 2016). As the majority of MSW incinerators are equipped with filters to reduce SO_2 emissions this fraction may be suitable for co-combustion with MSW.

3.3 Field scale experiments

1.25 to 2.5 tons of waste was subjected to sink/float separation. The outputs of the process are shown in Table 4 and Table 5. A considerable difference (up to 30%) was observed in the TS entering and exiting the process, likely due to sampling difficulties with the sink fraction, due to the high free phase water content, which produces erroneous TS measurements. The floating percentage is based on the TS of the raw waste and float fraction. Volatile solids in the float fraction were found to be 90% ± 2%, and 25% ± 7% in the sink fraction.

In addition, a total of 63.9 kg of solids were collected in the geotube, originating from the process water. As this amounts to about 20 kg per batch, this was considered negligible.

TABLE 4: Weights on entry to and exit from the process

	Raw waste	Float	Sink
Unit	**Kg**	**Kg**	**Kg**
Batch 1	2480	1000	2660
Batch 2	1280	540	1440
Batch 3	1220	540	1460

TABLE 5: Mass balance of the field scale test.

	Raw waste TS	Float TS	Sink TS	Floating	Difference In/out
Unit	Kg	Kg	Kg	%	%
Batch 1	1850	514	1127	28%	11%
Batch 2	955	252	401	26%	32%
Batch 3	910	264	520	29%	14%

3.3.1 Differences between lab and field scale tests

The difference in VS found in the sink fraction in lab and field scale tests may have been caused by the experimental procedure. In the lab scale, a more rigorous stirring was performed, including stirring of the whole water mass. Further, the wastes were added using different procedures: In the lab scale, waste was added in increments, whilst in the field scale the waste was added all at once and stirred using a machine. Due to the size of the tractor bucket, it is likely that the whole volume may not have been stirred as rigorously.

Another factor contributing to the difference is the variation in waste itself. With almost three times more material floating in the field scale experiment, and three times more VS in the raw waste, it is evident that the waste tested in the field scale trials contained more wood and organic materials. Using the field scale waste in a lab scale test would have likely produced a high VS sink fraction as well, meaning that the results from the field scale test would be in line with those of the lab scale test. This also implies that a lab scale test would have sufficed. Since the field scale test was carried out as a batch experiment without any special sink-float machinery, no extra information was provided with regard to practical applications. Any practical applications using sink-float should always be performed using a continuous process, as discussed also by Bilitewski (2010).

4. CONCLUSIONS

Density based separation using water was successfully applied to separate an organic material from inorganic. However, the resulting sink fraction may not always be suitable for landfilling, as the organic content may continue to be too high. Nevertheless, the use of analytical assays other than TOC to measure biological activity indicate that the organic content remaining in the sink fraction is not as biologically available as the TOC value may suggest.

In a practical Swedish scenario, sink-float separation fails to reduce the carbon content of the treated material in a reliable manner. In addition, since the material absorbs significant amounts of water, if density is to be used for separation this should be undertaken using a dry method.

4.1 Further research

Further research methods to be applied include the use of dry density separation methods, such as wind sifting, and combination treatments also including sieves or screens. Temporal variations of the waste should also be investigated further to better assess the appropriateness and feasibility of treatment methods. It is clear however that temporal variations may be considerable (up to a factor of three), thus a robust treatment method is needed.

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POTENTIAL AND MAIN TECHNOLOGICAL CHALLENGES FOR MATERIAL AND ENERGY RECOVERY FROM FINE FRACTIONS OF LANDFILL MINING: A CRITICAL REVIEW

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ABSTRACT

Multiple landfill mining investigations of municipal solid waste landfills have been carried out worldwide in the past decades. Some of these studies have led to the conclusion that landfill mining is not feasible and could represent more of a problem than a solution for old landfill sites. This is the case to a certain extent because, to this day, material and energy recovery in landfill mining has been restricted to the coarse fractions (>10 mm to >60 mm) in most projects, while the fine fractions (<10 mm to <60 mm) have been often re-directed to the landfill with poor or no treatment at all despite their recovery potential. The fine fractions account for 40-80 wt.% of the total amount of the landfill-mined material. Its material composition is characterized by about 40-80 wt.% decomposed organic matter or weathered mineral fractions which cannot be hand-sorted, followed by significant amounts of calorific fractions and a small amount of metals. The main chemical compound found in landfill mining fine fractions is SiO_2, mostly present as quartz and minor amounts of sheet silicates, followed by CaO, mostly present in carbonate minerals. MgO, Fe_2O_3 and Al_2O_3 represent minor components. Heavy metals are present in concentrations of few to several hundreds of mg/kg without a clear general trend of enrichment compared to the coarse fractions. In contrast, the net calorific value of the fine fractions (about 3-9 MJ/kg DM) can be several times lower than that of the coarse fractions (about 10-30 MJ/kg DM). These data clearly indicate that both a mineral fraction for waste-to-material and a calorific fraction for waste-to-energy might be recovered if suitable mechanical processing technologies can be employed. The potential of the fine fractions for material and energy recovery, as well as the main technological challenges to unlock it, are the main topics discussed in the present review article. This article has been elaborated within the framework of the EU Training Network for Resource Recovery through Enhanced Landfill Mining – NEW-MINE.

1. INTRODUCTION

Investigations on landfill mining (LFM) and enhanced landfill mining (ELFM) of municipal solid waste (MSW) landfills have shown that landfill-mined material is composed of a mixture of fine fractions (e.g. cover layer material, organic material and small particles of diverse materials) and inert materials (e.g. stones, glass, ceramics and construction & demolition waste (C&D)), as well as of a smaller amount of wood, leather, rubber, textiles, plastics, metals (ferrous and non-ferrous), paper & cardboard (P&C), among others (Hernández Parrodi, Höllen, & Pomberger, 2018).

Table 1 presents the average material composition of the standard landfill calculated by Van Vossen & Prent, 2011, from data of 60 LFM projects. The table shows that the fine fraction (<24 mm) accounts for about 55 wt.% (raw state) of the total mass of landfill-mined material.

However, it is important to highlight that the material composition varies between individual landfills and also within one landfill due to differences in the type of landfilled material, as well as due to the decay of organic and the weathering of inorganic matter, including the corrosion of metals. Nonetheless, the previous material composition presents a good average compared to other LFM studies (Quaghebeur et al., 2013; Wolfsberger et al., 2015).

The mixture of fine fractions is sometimes also referred to as "soil", "soil-like" or "soil-type" fraction in other studies, due to their appearance, organic matter and mineral contents and relatively homogeneous composition compared

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TABLE 1: Average material composition of landfill-mined material (Van Vossen & Prent, 2011).

Material fraction	Amount
Fine fractions (<24 mm)	54.8%
C&D	9.0%
Inert	5.8%
P&C	5.3%
Organic	5.3%
Plastics	4.7%
Wood	3.5%
Others	2.6%
Stones	2.5%
Total metals	2.0%
Textiles	1.6%
Leather	1.6%
Glass	1.1%
Non-MSW	0.3%

Note: Figures have wt. and raw state basis

to the coarser fractions. However, it is relevant to note that the different genesis of the fine fractions in landfills with respect to that of soils and the lack of separation of the fine fractions from other materials in the landfill, do not allow referring to the fine fractions from landfill-mined material, in a rigorous manner, as soil.

The fine fractions (commonly considered as the material with a particle size from <60 mm to <10 mm, depending on the author and investigation) have been identified as 40-80 wt.% of the total amount of landfill-mined material in several investigations (Hernández Parrodi et al., 2018), which is in agreement with the average amount of about 55 wt.% reported by Van Vossen & Prent, 2011.

Furthermore, the fine fractions tend to contain most of the moisture of the whole excavated material. This is the case because water is retained in a stronger way by the fine fractions than by the coarser fractions, which occurs mainly by physical absorption, chemisorption and capillary forces, together with the fact that the fine fractions present a higher specific surface area than the coarse fractions. A significant variation of the moisture content in the fine fractions has been identified from previous research, ranging from 16 wt.% to 54 wt.% (Hernández Parrodi et al., 2018).

Due to their quantity, composition and characteristics, the fine fractions are of utmost relevance to assess the feasibility of a (E)LFM project. This is partly the case because, to this day, material and energy recovery in (E)LFM has been restricted to the coarse fractions in most of the projects, while the fine fractions have been re-directed to the landfill with poor or no treatment at all beforehand (Bhatnagar et al., 2017; Münnich, Fricke, Wanka, & Zeiner, 2013). Moreover, it is important to add that the revenues to be obtained through land recovery play a crucial role for the economic feasibility of most (E)LFM projects, since without such revenues it is highly unlikely that the OPEX and CAPEX associated with the processing of the fine fractions can be covered (van der Zee, Achterkamp, & Visser, 2004).

A detailed study on the material composition of the fine fractions (<40 mm) of landfill-mined material from a MSW landfill in Austria (Wolfsberger et al., 2015) reveals the composition shown in Figure 1. The largest constituent of the fine fractions in that study accounts for the fraction "Sorting residue", which was in this case 65.6 wt.% (raw state). This fraction is hereafter referred to as "soil-like fraction". The three following, most abundant constituents of the fine fractions correspond to the sub-fractions: "Plastics", "Minerals" and "Wood, leather, rubber", with amounts of 11.6 wt.%, 6.6 wt.% and 5.9 wt.% (raw state), respectively; while amounts of 1.9 wt.% of metals and textiles (each) were reported.

This information suggests that the fine fractions can contain an interesting amount of materials that could be recovered and, therefore, to ignore their potential and keep on directing them to the re-disposal pathway is to be questioned.

Analyses on the chemical composition of the fine fractions (<10 mm) from the Remo landfill, Belgium, report a composition of 45 wt.% SiO_2, 9 wt.% CaO and 5 wt.% Fe_2O_3 (Quaghebeur et al., 2013). Mineralogically, few data on the composition of the fine fractions are available. One of these is the composition of the fine fractions (<40 mm) from an Austrian landfill which was investigated in the LAMIS project and showed amounts of 34 wt.% quartz (SiO_2), 30 wt.% calcite ($CaCO_3$), 16 wt.% dolomite ($CaMg(CO_3)_2$), 15 wt.% muscovite ($KAl_2[(OH,F)_2|AlSi_3O_{10}]$) and 5 wt.% kaolinite ($Al_4[(OH)_8|Si_4O_{10}]$). This confirms the presence of SiO_2 and CaO as main components and further suggests that also MgO and Al_2O_3 can be present in significant amounts.

It is important to reiterate that the composition of MSW changes according to geographic region, its development level, culture and many other factors (UNEP/Grid-Arendal, 2004). Additionally, the internal conditions to which the disposed waste in a landfill is exposed to (e.g. aerobic/anaerobic conditions, moisture, temperature and pressure) can vary significantly from site to site, as well as the operation procedures, local weather conditions and legislation, among many others. Even between landfills that appear to be very similar to each other (in terms of size, volume, region, received type of waste and climatic conditions), the straightforward application of information from one landfill to the other, without sampling, appears unfeasible (Sormunen, Laurila, & Rintala, 2013).

Moreover, previous research has emphasized that the costs and benefits in (E)LFM projects are always case-specific and cannot be generalized (Hogland, Marques, & Nimmermark, 2004; van der Zee et al., 2004). The specific conditions of a given landfill will determine, to a large extent, if landfill mining and land reclamation, which is an essential factor for the implementation of (E)LFM, are feasible for the site (Kurian, Esakku, Palanivelu, & Selvam, 2003; van der Zee et al., 2004). For instance, landfills and dumpsites without leachate and biogas collection systems could be appealing candidates for (E)LFM projects, since the economic and environmental assessments for the mitigation of their environmental pollution would not include investments made in such infrastructure, which might raise the feasibility of this kind of project.

Studies have also highlighted the importance of a prop-

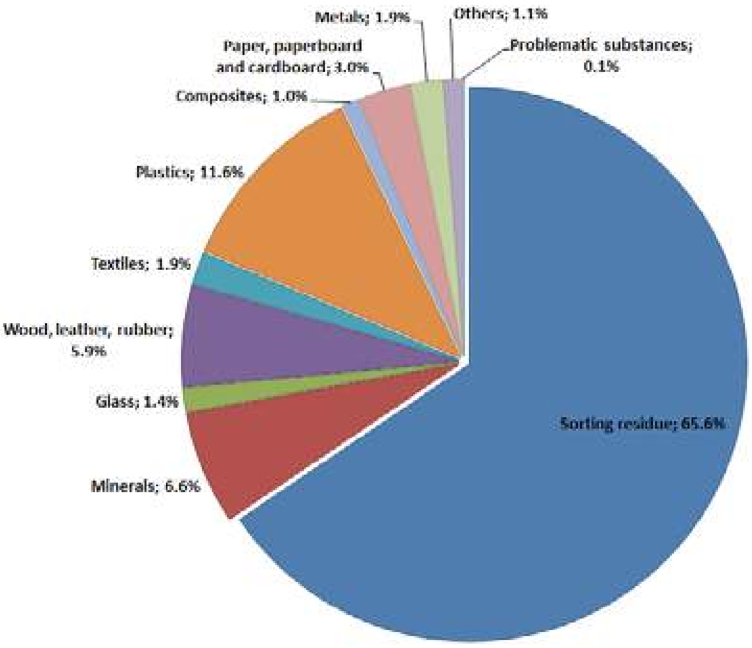

FIGURE 1: Composition of fraction <40 mm of landfill-mined material from a MSW landfill in Austria (modified from Wolfsberger et al., 2015).

er exploration of the landfill as one of the initial phases of a (E)LFM project (Cossu, Salieri, & Bisinella, 2012; Hull, Krogmann, & Strom, 2005; Quaghebeur et al., 2013). During the exploration phase of a (E)LFM project, test excavations or drillings into the landfill are necessary to assess the composition of the landfilled material (Krook, Svensson, & Eklund, 2012). The validation and utilization of non-invasive exploration methods, such as geophysical exploration, will play a critical role in (E)LFM projects.

Thus, in order to evaluate the material and energy recovery potentials of the fine fractions from a specific landfill, adequate and proper quantitative and qualitative characterization of the disposed waste is to be performed and several factors are to be taken into account.

Additionally, technological, legal and economic challenges are to be overcome and a new approach to process the fine fractions is to be implemented, so that a significant amount of the fine fractions can be directed towards material and energy recovery.

The potential of the fine fractions for waste-to-material (WtM) and waste-to-energy (WtE), as well as the main technological challenges to unlock it, are the main topics discussed in this review article.

2. MATERIALS AND METHODS

The present review article comprises the discussion of the potential for material and energy recovery from the fine fractions of landfill-mined material, as well as the main technological challenges that need to be overcome to unlock it. The arguments for the discussion are based on the analysis of several previous (E)LFM investigations found in scientific literature and their reported results. The scope envisages scientific articles published in international peer-reviewed journals, as well as review papers and international conference proceedings, books, guidelines, standards and legislation.

The search for literature was performed using different internet search engines and online scientific databases of peer-reviewed research articles, scientific journals, books and conference proceedings. Relevant references and citations from previous studies were also taken into account.

Keywords such as landfill mining, enhanced landfill mining, resource recovery, material recovery and energy recovery were used as baseline for the search of information; while terms such as fines, fine fractions, soil-like material, soil-type material and soil were employed to filter the search results and obtain more specific information.

The selection of the sources of information for this paper was made based on the relevance of their content regarding the material composition, properties and characteristics of the fine fractions from (E)LFM, as well as to provide a solid foundation for the discussed topics in this article.

3. RESULTS AND DISCUSSIONS

3.1 Potential for WtM and WtE

As already mentioned, previous (E)LFM research reveals that the fine fractions are mainly composed of a soil-like fraction, followed by a smaller amount of a mineral fraction, plastics, metals, textiles, leather, rubber, wood, P&C (Hernández Parrodi et al., 2018). This information suggests that some of these materials could be recovered from the fine fractions via further material processing.

One approach to achieve material recovery, which is the one proposed and discussed in the present article, is to separate the fine fractions (incl. those which cannot be separated by manual sorting) from each other, according to their physico-chemical properties, by mechanical processing. Once the LFM material has been separated into different material fractions, it would be possible to direct these to either material or energy recovery pathways. For this purpose, the segregation of individual particles is required. To accomplish an adequate segregation, a particle size classification of the excavated material is to be performed as an initial step (e.g. particle size fractions: >40 mm, 40-20 mm, 20-10 mm, 10-8 mm, 8-6 mm, 6-4 mm, 4-2 mm, 2-1 mm and <1 mm). This will raise the efficiency and effectiveness of the further mechanical processing regarding the disintegration of agglomerates and subsequent material classification (e.g. plastics, metals and inert material fractions). Some particle size ranges might require a wet processing (e.g. washing) to achieve quality material (e.g. for the recuperation of plastics and inert materials). Additionally, the amount of moisture contained in the different particle size fractions will play a significant role while selecting the most appropriate mechanical processing method.

However, it might be the case that a certain amount of these fractions is not suitable for any of the previously referred pathways. And as a result, this residual fraction could be re-stored, perhaps at the same landfill, till new technologies for its exploitation are available. An alternative approach, which has been already studied in previous (E)LFM projects, would be to thermally valorise the fine fractions as a whole. This would require additional fuel to compensate for its low calorific value, which has been found to be in the range of 0.4-9 MJ/kg in previous studies (Hernández Parrodi et al., 2018). Nevertheless, the calorific value of the fine fractions could be raised by reducing the amount of inert materials and the moisture content. Therefore, to separate the fine fractions into different material and particle size fractions, as proposed in this paper, could be the most adequate pathway to achieve a holistic valorisation.

In order to get a visual understanding of the approach proposed herein regarding the theoretical potential of the fine fractions, Figure 2 presents the material fractions that constitute the fine fractions grouped into three clusters, which are WtM, WtE and Re-storage.

The EU has employed a hierarchical concept for the management of waste (Directive [2008/98/EC] of the European Parliament and of the Council on waste) in order to minimise the overall impacts and improve the efficiency of the utilization of resources, in which waste management has been given five main priorities. These priorities are shown in Figure 3 from highest (top) to lowest priority (bottom).

Prevention targets the avoidance of waste, while preparing for re-use, recycling and recovery aim for the valorisation of waste materials. Disposal, as a last resort, targets the safe permanent storage of waste.

Therefore, according to the European waste management hierarchy, preparing for re-use and recycling are to be preferred, as far as they are feasible and represent a better environmental solution, to energy recovery from waste. In other words, WtM is, in general, to be considered before WtE.

The quality of the retained materials in the landfill and the WtM and WtE technologies available for material valorisation will, among others, determine the feasibility of (E)LFM (Quaghebeur et al., 2013).

3.2 Waste-to-Material

This concept refers to the recovery of materials from waste. These recovered materials are commonly referred to as secondary raw materials. In theory, these materials can be directly reused, recycled or processed in such a way that they can be reincorporated to the material´s life cycle.

In the case of (E)LFM, the quality of the recovered ma-

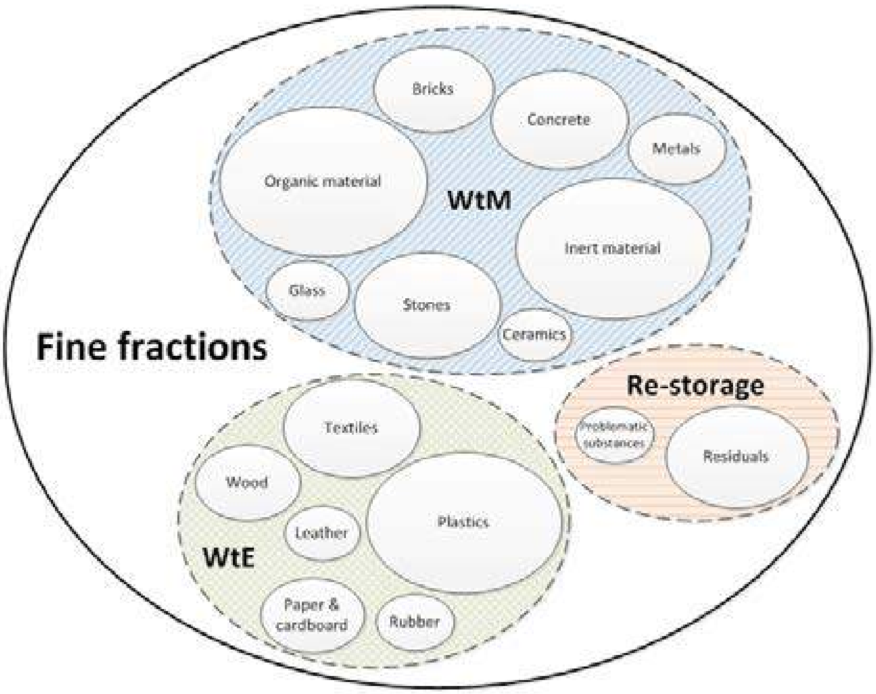

FIGURE 2: Potential of the fine fractions for WtE and WtM.

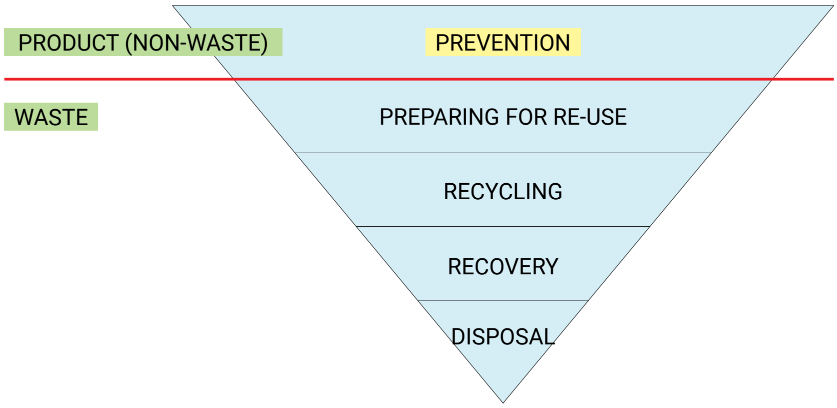

FIGURE 3: Waste management hierarchy (European Commission).

terials might impede its direct re-use and, as already stated, limit the recyclability of some of them. Nonetheless, previous (E)LFM investigations have revealed that interesting amounts of ferrous and non-ferrous metals could be recovered from the fine fractions for recycling (Hernández Parrodi et al., 2018).

Besides metals, two other highly interesting fractions from the fine fractions for material recovery are the soil-like and inert fractions, which could be used in various applications (e.g. soil-like fraction as ground substitute and inert fraction as construction aggregates) if the heavy metals and organic pollutants contents are low. These fractions are of paramount importance because they, together, account for most of the fine fractions and they are, to this day, the materials that are mainly sent back to the landfill for re-disposal, hindering the overall economic and environmental feasibility of (E)LFM projects.

It is known from previous research (Hernández Parrodi et al., 2018) that the soil-like fraction is, in some cases, composed of the material used to cover the waste (daily, intermediate or final cover material) during the operation of the landfill. In many cases, materials with a low permeability (e.g. clay) have been used for this purpose. Therefore, the presence of large amounts of fine fractions in landfill-mined material can be related to landfill sites, while a low amount could be related to open dumpsites (Mönkäre, Palmroth, & Rintala, 2016). The intermediate and daily cover materials usually consist of a 15-30 cm layer of e.g. soil, clay or compost (Tchobanoglous, Theisen, & Vigil, 1993).

Furthermore, it is not rare to find landfill sites where a variable amount of construction and demolition waste (C&D) was mixed with the cover material to give a better load capacity to the platforms for the transit of the trucks on the landfill area, as well as the use of other received materials in combination with the main cover material, such as soil, compost and dry sewage sludge, as daily cover materials.

A significant percentage of the fine fractions can also be formed through the weathering of mineral wastes and through the humification and mineralisation of biowaste (Hernández Parrodi et al., 2018).

Thus, it can be concluded that the soil-like fraction is mostly composed of organic and mineral materials, which could be separated from each other, up to a certain extent, by further mechanical processing.

As for the inert fraction, which has been identified as mainly composed of C&D, stones, minerals, glass and ceramics in previous studies, a relevant amount of organic matter could also be contained in it due to the presence of soil and waste mixtures.

The recovery of these organic and mineral materials could yield an organic material, which might be used, among others, as ground substitute or soil improver, and a mineral material, which might be suitable for the substitution of mineral aggregates for construction purposes (e.g. construction sand). This, provided that they comply with the corresponding quality and characteristics stipulated in the local regulations.

Figure 4 schematizes the recovery of metals, construction aggregates and a ground substitute from the fine fractions.

Using the material composition of the fine fractions reported by Wolfsberger et al., 2015 (Figure 1), as an example and assuming a hypothetical scenario, in which optimal material processing allows quality and proper segregation of the corresponding material fractions with a recovery amount of 60-80 wt.%; an amount of around 3-5 wt.% minerals, 1 wt.% metals and 1 wt.% glass could be recovered from the total amount of the fine fractions. This would represent that about 5-8 wt.% of the whole amount of the fine fractions could be recovered through the WtM pathway.

The previous range represents a small amount of the total excavated material and, thus, might not be very appealing to future (E)LFM projects. However, a very significant amount of about 60-70 wt.% of the total excavated material corresponded to the soil-like fraction (shown as "Sorting residue" in Figure 1), which depends on the implementation of an adequate material processing approach, as well as on the further development of technology and recovery techniques, in order that organic and mineral materials can be recovered from this fraction, as previously proposed.

Therefore, it could be said that, to this day, the main key-factors to divert a large amount of material from re-disposal to material recovery are: the implementation of an adequate material processing approach and the further de-

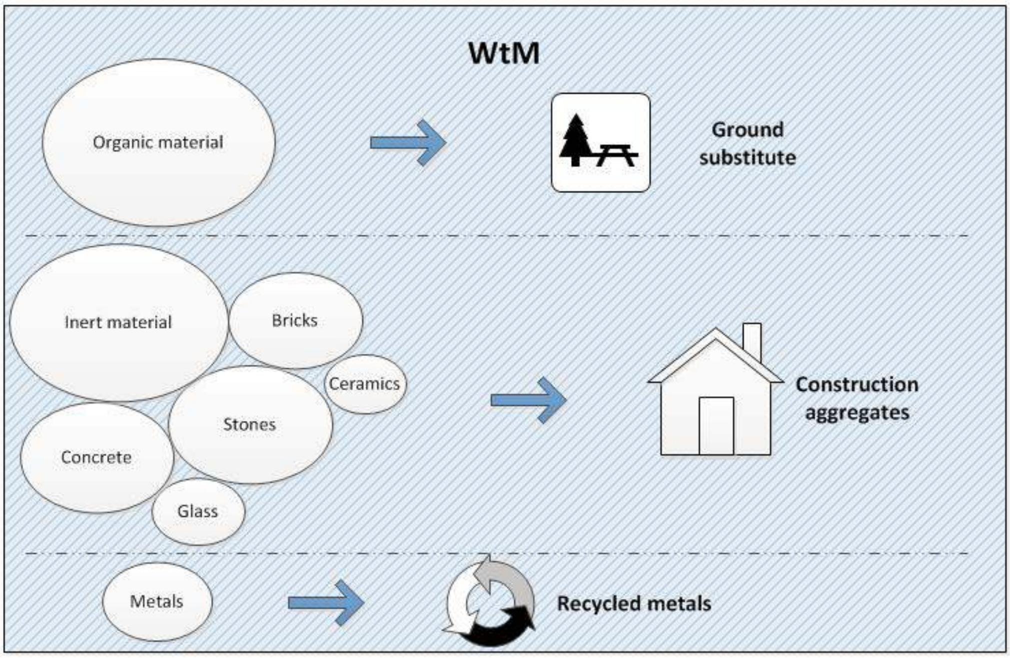

FIGURE 4: Material recovery from organic and inert fractions of LFM fine fractions.

velopment of technology and material recovery techniques. Given the previous key-factors, the amount of material recovery could increase significantly, as well as the economic and environmental feasibility of the whole project.

Moreover, the fine fractions have been also used as cover material in landfill sites to build a methane degradation layer (Kaczala et al., 2017). When the level of pollutants is low, the fine fractions could be used as future landfill cover (Bhatnagar et al., 2017). The material can be used as cover material after assessing the geotechnical suitability (Kurian et al., 2003).

One additional potential end-use for the fines excavated from landfill could be as clean fill off-site (Hull et al., 2005). The fine fractions of most recent landfilled MSW might even be able to be used as soil fertilizer or compost at green areas and gardens (Jones et al., 2013; Quaghebeur et al., 2013), provided that the pollutant concentrations meet the corresponding requirements for such use. Landfill-mined materials should be characterized for heavy metals of environmental concern before they are applied on land use (Jain, Kim, & Townsend, 2005). Amounts of inorganic pollutants, such as Cr, Cu, Pb and Zn, in the calorific fractions (i.e. plastics, textiles, rubber, leather, wood and P&C) of the same order of magnitude as in MSW have been found (Vesanto et al., 2008). However, besides the total contents, also the leachability and the mineralogical bonding of these possible contaminants have to be assessed.

According to a previous study by Kurian et al., 2003, the fine fractions complied, for most parameters of most samples, with the heavy metal limit values from the US EPA standards to be used as compost for non-edible crops. It has been reported that the concentrations of almost all heavy metals (except for Pb, Cd and As in some cases) in waste samples (<10 mm and <4 mm) met the pollutant ceiling concentrations set by the US EPA and the EU limits (Masi, Caniani, Grieco, Lioi, & Mancini, 2014). Fines from older disposed MSW might exceed pollutant concentrations and would then need further treatment to be used as soil fertilizer or compost (Quaghebeur et al., 2013).

Furthermore, landfills could be transformed into temporary storage sites (Jones & Tielemans, 2010) for, if the case, a fraction concentrated in problematic substances, such as heavy metals and asbestos. As conceptualized by Deltares independent institute for applied research, a temporary storage site is a structurally and environmentally safe deposit place that would enable in-situ material recovery from waste materials in the future, facilitating the access to the potential future resources, when the technology to recover certain materials is available. Such concept would also allow the implementation of improvements to the temporary storage sites, such as reshaping and volume reduction. For instance, German landfill mining and site remediation investigations reported reductions of 8-30 vol.% after re-landfilling and re-compacting the excavated MSW without recycle or reuse of the waste fractions (Collins, Brammer, & Harms-Krekeler, 2001). The compaction of re-landfilled MSW results in a considerable volume reduction due to the decrease of pore spaces and voids caused by the degradation of the organic waste fractions (Collins et al., 2001). The extent of the reduction depends on the degree of degradation of the organic fraction and the compaction of the MSW in the landfill before the excavation (Hull et al., 2005). Additional volume reductions can be expected if the fine fractions are reused or recycled (Hull et al., 2005).

Consequently, temporary storage would mean a step further towards circular economy, creating a connection between the past, present and future regarding resource availability (Bosmans, Vanderreydt, Geysen, & Helsen, 2013; Jones et al., 2013; Krook et al., 2012; Krook & Baas, 2013; Quaghebeur et al., 2013).

Some other end-uses might arise in the future, depending on available markets, material quality and regulatory framework for reuse (Jain et al., 2005). Both the increasing market prices for recovered materials and the legal framework will set the conditions to justify new waste processing technologies (Archer, Baddeley, Klein, Schwager, & Whiting, 2005; Forton, Harder, & Moles, 2006; Tachwali, Al-Assaf, & Al-Ali, 2007).

3.3 Waste-to-Energy

In general, energy recovery from waste refers to the generation of electricity and/or heat by processing waste materials, as well as to the production of energy carriers (e.g. refuse derived fuel (RDF) and syngas for the production of hydrogen and methane). RDF is an alternative fuel, produced from diverse kinds of waste materials, which can replace partially or completely the use of fossil fuels in various industrial applications (e.g. cement and power plants).

As already mentioned, relevant amounts of materials such as plastics, P&C, wood, textiles, leather and rubber, which could be suitable for the production of RDF, can be found in the fine fractions. If recovered, it is unlikely that these materials can meet the required quality criteria for material re-use and recycling, whilst recovered wood, textile, leather and rubber materials are hardly re-used or recycled. However, assuming that these materials could be recycled, their value on the recyclables market would most likely be very low with high recycling costs.

Moreover, these materials are composed of carbon to a major extent and they, in a dry state, possess high calorific values. Calorific values of 4.4-9 MJ/kg DM have been determined for the fine fractions (<20 mm) from two Austrian landfills (Wolfsberger et al., 2015), which can be significantly increased by reducing the amount of the inert fraction present in them (e.g. extracting mineral materials like stones, glass and ceramics).

Provided these circumstances, the recovery of calorific materials (i.e. plastics, P&C, textiles, rubber, leather and wood) in order to produce RDF and exploit its WtE potential can be suggested as an interesting option. Additionally, the added value of the calorific materials would be significantly larger in this manner.

Figure 5 displays the usage of the calorific fractions from the fine fractions for the production of energy.

Similarly as in section 3.2, using the material composition from LFM fine fractions reported by Wolfsberger et al., 2015 and assuming a hypothetical scenario to provide with an example of the amount of material that could be directed to WtE with recovery amounts of 60-80 wt.%: a total amount of about 15-20 wt.% of the whole quantity of the fine fractions could be used to produce RDF, from which around 8-10 wt.% would be conformed of plastics, 4-5 wt.% of wood, leather and rubber (all together), 2-3 wt.% of P&C and 1-2 wt.% of textiles.

It is important to note, that the quality requirements and, hence, the composition of the produced RDF will vary according to the thermal valorisation technique to be employed and, therefore, the total amount of RDF to be obtained will depend strongly on these requirements.

Mined waste from landfills may be used to improve combustion through co-incineration at MSW incineration plants; helping to avoid auxiliary fuel consumption and releasing landfill space (Chen, Guan, Liu, Zhou, & Zhu, 2010).

Thermo-chemical based technologies, such as gasification, pyrolysis and incineration, to process the fine fractions from landfill excavated waste materials have been tested to a certain extent in recent years (Bosmans et al., 2013). Incineration with energy recovery would be possible with the fines fraction (<18 mm) after the removal of

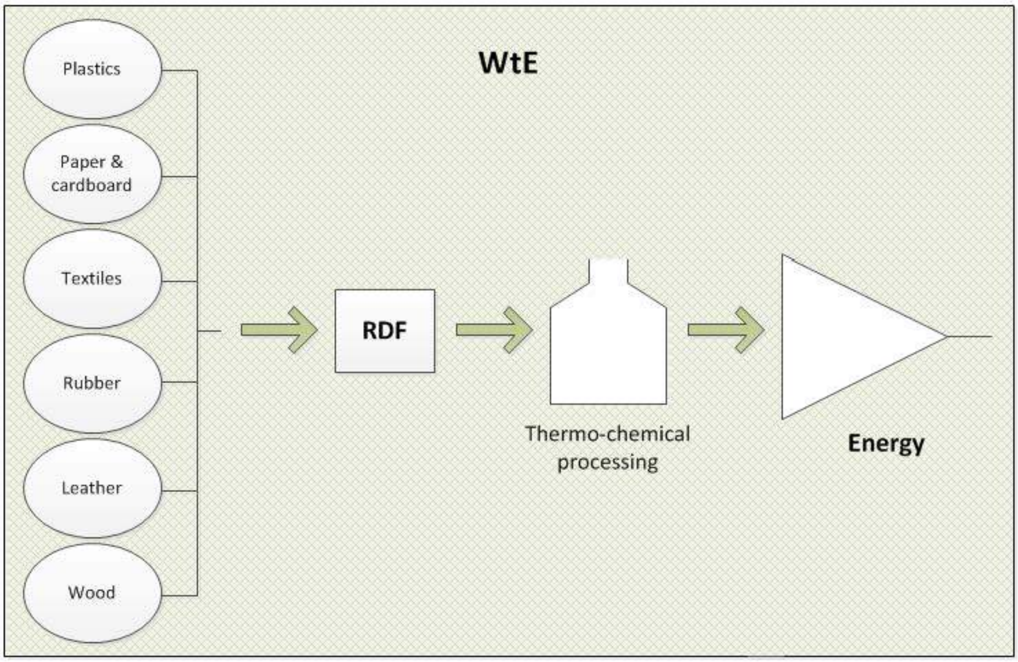

FIGURE 5: Energy recovery from calorific fractions of LFM fine fractions.

coarse inert material (Hogland et al., 2004). Research on plasma gasification and further upcycling of its by-products has been increasing in the last years (Bosmans et al., 2013; Danthurebandara, van Passel, Machiels, & van Acker, 2015; Danthurebandara, van Passel, Vanderreydt, & van Acker, 2015a; Danthurebandara, van Passel, Vanderreydt, & van Acker, 2015b). The outcomes of those studies have shown that, due to the robustness and flexibility of the process, plasma gasification might be used as an adequate WtE route for the calorific fractions from the fine fractions.

The EU standard that states the specifications and classes for solid recovered fuels (SRF), which is a type of RDF, is the BS EN 15359:2011, in which the net calorific value (linked to water content) and chlorine and mercury contents are among the most important parameters. A case study reported that the limit values for SRF usage in cement or power plants, according to the Austrian guideline BMLFUW 2002, were not exceeded by the fines fraction (<40 mm) from one LFM case study, but exceeded them for another one (Wolfsberger et al., 2015).

3.4 Main challenges to overcome

There are a large number of factors that play a very important role in a (E)LFM project (e.g. landfill site´s particularities, excavation and material processing procedure and utilized equipment, sampling and laboratory analysis procedures and followed guideline, among many others) and, therefore, attention must be paid to the singular characteristics of a site while analysing and comparing information between different projects.

The factors discussed below, together with the economic and legislative aspects, represent some of the main challenges in order to start full scale recovery of resources from landfills (Jones, Geysen, Rossy, & Bienge, 2010; Krook et al., 2012; Van Vossen & Prent, 2011).

3.4.1 Variations in composition and properties

In order to identify the material and energy recovery potentials and possible alternative uses of the fine fractions, and be able to design an appropriate material processing and final disposal method during the planning phase, the characterization of the fine fractions is an essential first step (Jani et al., 2016; Mönkäre et al., 2016). Some key aspects to be considered are: the material, chemical and mineralogical composition, size and volume of the site, type of the landfilled waste, location of the site, historic operation procedures of the site, extent of degradation of the disposed waste, types of markets and uses for the recovered materials and environmental and health risks (Frändegård, Krook, Svensson, & Eklund, 2013; Kurian et al., 2003; Quaghebeur et al., 2013). Additionally, the amount of moisture present in the fine fractions is a key parameter for the selection of an appropriate material processing, as the water content plays a decisive role regarding the efficiency and suitability of mechanical separation methods (e.g. sieving and density separation methods and sensor-based sorting).

Compaction and expansion of solid waste components, as well as the material´s contamination and degradation make excavated material more difficult to sort and characterize than fresh MSW (Hull et al., 2005).

As it has been reported in previous research, a variable quantity of problematic substances could be present in the fine fractions. These are substances that, due to their toxic or undesired characteristics, would hinder or limit to a great extent the further usage of the produced or recovered materials from the fine fractions. The presence of trace amounts of hazardous chemicals would most likely limit the quality of the fines fraction for further use (Reinhart & Townsend, 1997).

Some problematic elements that have been found in the fine fractions are, for example, heavy metals, chlorine and sulphur. Such elements can be toxic at certain concentrations and speciation and might form harmful compounds when released to the environment. In addition to that, they can damage the equipment with which the fine fractions are being handled or processed.

The risk due to the elevated pollutant concentrations should be evaluated before such material can be reused outside of a landfill (Jain et al., 2005). To reduce the concentration of metals such as Cr, Cu, Ni and Zn in the fine fractions might be needed to enable the use of this fraction for further purposes (Quaghebeur et al., 2013).

In general, metal concentrations, except those of As, Be and Cd, were found below EU, UK and US regulatory threshold values, for use in unrestricted settings, for the fine (<0.425 mm) and intermediate (>0.425 mm and <6.3 mm) fractions (Jain et al., 2005).

One first step to identify an adequate processing of the fine fractions from landfill-mined material is to determine the leaching properties at laboratory scale (Mahmoudkhani, Wilewska-Bien, Steenari, & Theliander, 2008). These tests can bring valuable information about the compliance with existing standards and norms.

Hence, the mechanical processing of the fine fractions is to be aimed to remove problematic materials (e.g. using sensor-based sorting equipment to sort out materials containing chlorine) to produce a RDF with the adequate properties for the corresponding thermo-chemical processing technology. This together with the recovery of an organic and a mineral fraction, whilst concentrating the undesired elements and compounds in a residual fraction, which might be suitable for further processing for the recovery of certain elements (e.g. heavy metals) in the near future.

3.4.2 Surface defilements and material agglomerates

During disposal time, fines adhere to the surface of other materials (Maul & Pretz, 2016), leading to limitations in the final sorting outputs due to decreased sorting performance of the sensor-based sorting units. This has also been reported in other investigations (Wolfsberger et al., 2015), in which the fines adhered to other waste fractions as impurities, contaminating the rest of the waste fractions and decreasing their quality and value. Results from a previous study show that all manually sorted size categories contained impurities of the other sorted fractions (Kaartinen, Sormunen, & Rintala, 2013). Contamination of all fractions with fines (adherent “soil”) showed an increasing trend with age, which in high levels will likely prove to be an insurmountable obstacle to recycling most of the excavated waste fractions, unless further processing is conducted (Hull et al., 2005).

This adhered layer, also known as surface defilements, can lead to efficiency losses of sensor-based sorting (Maul & Pretz, 2016). If the surface defilements can be removed, it would be easier to use plastics from LFM as a secondary resource (Maul & Pretz, 2016). Further analyses on the sorted plastics show that the mass share of the surface defilements in the final sorted products can be as high as 7.5 wt.% (Maul & Pretz, 2016).

Apart from the above, the presence of moisture in the fine fractions favours the formation of material agglomerates during the mechanical processing, especially in the sieving steps. Material agglomerates are a mixture of water and fine particles (mainly material <1 mm) that stick together to form slumps of fine fractions. These slumps might encapsulate other material fractions contained in the fine fractions as well, such as plastics, P&C, metals, etc. This can hinder the performance of the mechanical processing (in particular the size and density separation methods) and, hence, the material recovery from the fine fractions as well.

The drying of the material might increase the amount of the fines, as in moist conditions some fine particles tend to stay attached to bigger particles (Kaartinen et al., 2013) and avoid the formation of material agglomerates. This could improve the quality of the coarse fractions and raise the overall efficiency of the material processing. Composting (aerobic biodrying) has been suggested to dry the excavated waste prior to thermal valorisation; this would improve the removal of the material contamination due to adhered fines, the efficiency of the sieving steps and reduce the ash generation during the thermal processing (Collins et al., 2001).

In contrast, the implementation of a wet processing (e.g. wet sieving and washing units) in order to decrease the amount of surface defilements and eliminate material agglomerates might also result in a high quality of the recovered calorific fractions and an efficient separation of the different material fractions. Nevertheless, the feasibility of a wet processing is yet to be fully assessed in the context of LFM fine fractions, since it is a complex treatment that might require sophisticated processing, as well as additional energy to reduce the moisture content of the products afterwards.

3.4.3 Application range of available mechanical processing technologies

The particle size is a very important factor for an optimum separation process; though, conventional waste sorting techniques (e.g. metals separation, density classification and sensor-based sorting equipment) cannot be applied below a certain particle size of the material (Spooren, van den Bergh, Nielsen, & Quaghebeur, 2013).

Also, the removal of ferrous materials from the fine fractions slows down separation processes and requires a relatively dry material (Bhatnagar et al., 2017); the latter would mean the addition of a certain amount of energy to the process and could negatively affect the economic feasibility of (E)LFM projects.

Therefore, the application range of the technologies for processing the fine fractions, with respect to particle size, needs to be extended in order that smaller particle sizes (<3 mm to <1 mm) can be reached. This will play an essential role regarding the separation of organic and mineral materials and the recovery of non-ferrous metals from the soil-like fraction of LFM fine fractions, since these materials are mainly found in small particle size ranges.

Concurrently, material processing technologies and techniques are to be developed further in such a way that LFM material can be processed in an efficient way without the need of a drying step.

The planning of a suitable treatment process for recovering waste fractions in (E)LFM projects requires not only knowledge on the composition of the landfilled waste, but on the treatability of the different fractions as well (Kaartinen et al., 2013).

One of the main technological aspects of ELFM is the development of a processing plant that enables maximum resource recovery (Quaghebeur et al., 2013).

4. CONCLUSIONS

The primary recoverable waste fractions from LFM fine fractions are complementary materials for RDF production, soil-like and inert materials and metals. In this respect, the specific conditions of a given landfill will determine if landfill mining and land reclamation are feasible for the site. One critical factor for the implementation of (E)LFM is the recovery of land, since the revenues to be obtained from land reclamation can be the main driver of the project´s business case. The quality of the retained materials in the landfill and the WtM and WtE technologies available for material valorisation will also, among others, determine the feasibility of (E)LFM. Landfills and dumpsites without leachate and biogas collection systems could be appealing candidates for (E)LFM projects, since the economic and environmental assessments for the mitigation of their environmental pollution would not include investments made in such infrastructure, which might raise the feasibility of this kind of project.

The organic fraction recovered from the fine fractions could have potential as ground substitute, such as cover material for operational landfills, soil for non-edible crops and formation of bio-soils to be used in environmental remediation activities. This fraction could, theoretically, be used as fertilizer at green areas and gardens, the latter given that the material complies with all applicable regulations for such purpose. Particle size and nutrients content are relevant parameters to evaluate the use of the fine fractions in soil applications. When the level of contamination of P&C, plastics, textiles and wood (calorific fractions in general) recovered from a landfill is too high or their quality is too low, WtE could be the most suitable valorisation route. Material properties such as moisture and ash contents, calorific value, organic and total carbon amounts and hydrogen and nitrogen contents are needed to assess the efficiency for WtE applications.

For metals, glass, ceramics, stones and other inert materials, WtM might be possible if the materials can be separated adequately from each other and the applicable limit values for pollutant substances can be met.

The planning of a suitable treatment process for recovering material fractions in (E)LFM projects requires not only knowledge on the composition of the landfilled waste, but on the treatability of the different fractions as well. In order to identify the material and energy recovery potentials, possible alternative uses of the fine fractions and to be able to design an appropriate material processing and final disposal method during the planning phase, the characterization of the fine fractions is an essential first step. Some key conditions to be considered are: the composition and type of the landfilled waste, location of the site, historic operation procedures of the site, extent of degradation of the disposed waste, types of markets and uses for the recovered materials and environmental and health risks.

To this day, predominant factors to divert a large amount of material from re-disposal to material recovery, raise the amount of material recovery and, hence, the economic and environmental feasibility of (E)LFM projects, are: the implementation of an adequate material processing approach and the further development of technology and material recovery techniques. These, together with the economical and legislative aspects, represent some of the main challenges in order to start full-scale recovery of resources from landfills.

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MORPHOLOGICAL AND STRUCTURAL CHANGES IN PADDY STRAW INFLUENCED BY ALKALI AND MICROBIAL PRETREATMENT

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Microbial

ABSTRACT

Paddy straw is a lignocellulosic waste rich in holocellulose (cellulose+hemicellulose) content. It can be used as a good substrate for biogas, bioethanol and biodiesel production. However, the recalcitrant cell wall components i.e. lignin and silica are the main deterrent to efficient utilization of paddy straw. This stringent sheath of lignin and silica does not allow fermenting microbes to access holocellulose. Pretreatment of paddy straw is, therefore, crucial to get rid of lignin and silica. In this context, paddy straw was pretreated with various alkalis viz. NH_3, Na_2SO_3, Na_2CO_3 and NaOH in the current study. All the alkali pretreatments were supplemented with microwave irradiations (720 W, 18°C) for 30 minutes. Paddy straw was also pretreated microbially with *Pleurotus florida* for 15 days via spawning. Morphological and structural changes in the pretreated paddy straw were visualized via Scanning Electron Microscopy (SEM). The straw turned remarkably fragile with enhanced bulk density and surface wettability after 4% NaOH-30 min microwave pretreatment. The pretreated straw was also found to lack silicified cuticle layer and lignin sheath which exposes straw sugars (cellulose and hemicellulose) to fermenting microbes.

1. INTRODUCTION

Cereal straws are abundant, cheap and potential feedstock for biofuel production (Sun, 2010). Rice crop generates straw as much as 23% of its total weight, which is usually burnt to clear the fields for the next crop. In India, open field straw burning contributes up to 0.05% of the total green-house gases emission (Gadde et al., 2009). On that account, utilization of paddy straw for biofuel production will help slow down climate change resulting from straw burning besides providing renewable fuel.

However, the main deterrent to paddy straw utilization is the structure of paddy straw which is composed of cellulose, hemicellulose, lignin and silica. Accessing holo-cellulose (cellulose+ hemicellulose) for biofuel production is difficult due to the presence of recalcitrant compounds like lignin and silica (Phutela et al., 2012; Kaur and Phutela, 2016). Hence, pretreatment of paddy straw is essential to access the holocellulose component.

Pretreatment methods are broadly classified as physical, chemical, physico-chemical and biological. In physical pretreatment, biomass is chopped, ground or milled to reduce particle size, exposed to irradiations or subjected to high temperatures or pressure (Fan et al., 1982) to alter the physical and/ or chemical properties. Milling increases methane yield (Delgenes et al., 2002) and hydrolysis rate at the same time. Since there are no inhibitors like furfural and hydroxyl methyl furfural (HMF) formed during milling, it is a good pretreatment method for methane and ethanol production (Hendriks and Zeeman, 2009). However, due to high energy requirements (Ramos, 2003) and continuous rise of energy prices, physical pretreatment methods are not cost effective on their own (Fan et al., 1982) but should be used in combination with other pretreatment methods.

Chemical pretreatment primarily includes chemicals like dilute acid/ alkali, urea etc. Acid pretreatment solubilizes, condensates and precipitates lignin (Liu and Wyman, 2003; Shevchenko et al., 1999). It also causes hemi-cellulose hydrolysis and solubilized hemi-cellulose is then subjected to hydrolytic reactions producing monomers, furfural, HMF and other (volatile) products (Fengel and Wegener, 1984). Due to the formation of these inhibitors, acid pretreatment is more attractive for methane production than ethanol production because methanogens are not affected by compounds like furfural and HMF to a certain concentration and within an acclimatization period. During alkali pretreatment, solubilization, redistribution and condensation of lignin occurs. It also modifies the crystalline state

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of cellulose (Gregg and Saddler, 1996). Change of cellulose structure to a denser and thermodynamically more stable form as compared to the native cellulose is an important aspect of alkaline pretreatment (Pettersen, 1984). Again, loss of fermentable sugars and production of inhibitory compounds make the alkaline pretreatment less attractive for ethanol production but the effect is less severe to methanogens as compared to yeasts (Pavlostathis and Gossett, 1985; Hendriks and Zeeman, 2009). Chemical pretreatments are less energy intensive than the physical methods. However, chemical residues left after pretreatment pose a threat to the environment if not disposed of cautiously.

Physico-chemical methods are a combination of physical and chemical techniques e.g. Ammonia fibre/ freeze explosion (AFEX). In AFEX, an amalgamation of physical and chemical effects induce the cleavage of lignin-carbohydrate complex (Chundawat et al., 2007), hemi-cellulose hydrolysis and cellulose de-crystallization which increases the surface area. These modifications enable complete enzymatic conversion of cellulose and hemi-cellulose to fermentable sugars at low to moderate enzyme loadings (Teymouri et al., 2005). Alizadeh et al. (2005) reported 2.5 fold increase in ethanol yield after AFEX pretreatment of switchgrass. In physico-chemical methods, physical pretreatment primes the substrate for better penetration of chemical. These methods, therefore, have merits over the independent physical and chemical methods (Liu and Wyman, 2005; Kaur and Phutela, 2014).

Biological pretreatment involves the use of lignocellulolytic micro-organisms or enzymes. White-rot, brown-rot and soft-rot fungi are chiefly used for this purpose. Brown rots mainly attack cellulose, whereas white and soft rots attack both cellulose and lignin (Ward et al., 2004). Kirk (1984) stated that if lignin is even partially removed by the chemical or biological means or even if its relationship with the polysaccharides is modified well, the polysaccharide become much more susceptible to the enzymatic degradation. Biological pretreatment is favored over the other pretreatment methods because it is more eco-friendly and does not generate any kind of toxic waste. The slow rate of hydrolysis, substrate and temperature specificity of fungi and enzymes, however, reduce the overall fascination of this method.

In this context, paddy straw was pretreated with different alkalis (viz. NH_3, Na_2SO_3, Na_2CO_3 and NaOH) in combination with microwaves and a lignocelluloytic fungus viz. *Pleurotus florida*. In order to enhance the effect of various alkalis, the alkali suspended paddy straw was irradiated with microwaves (720 W, 18°C) for 30 minutes. Alkaline pretreatment (NH_3, Na_2SO_3, Na_2CO_3 and NaOH) causes swelling of biomass via solvation and swelling, rendering it more accessible to fermenting microbes and enzymes. Microwave irradiations degrade lignin with their high heating efficiency. Microwaves heat the target object directly by applying an electromagnetic field to the dielectric molecules as compared to conduction/convection heating which is based on intramolecular heat transfer (Newnham et al., 1991). Then, the alkali + microwave pretreatment is compared with biological pretreatment (*Pleurotus florida*) method, the least residue generating method, to assess their efficiencies.

2. MATERIALS AND METHODS

2.1 Alkali pretreatment of paddy straw

Paddy straw was suspended in various alkalis viz. 4% NH_3 (v/ v), 4% Na_2SO_3 (w/ v), 4% Na_2CO_3 (w/v) and 4% NaOH (w/v) and irradiated with microwaves (720 W, 18°C) for 30 minutes in a 1000ml glass beaker placed inside a microwave oven (MC-2681DS LG Electronics). The solid to solvent ratio was kept constant at 1:10. The pretreated paddy straw was washed with tap water till the washings were clean, colorless and neutral to the pH paper. The straw was then air-dried and analyzed for change in bulk density, proximate and chemical composition, and surface characteristics.

2.2 Microbial pretreatment of paddy straw

Paddy straw was pretreated with lignocellulolytic fungus viz. *Pleurotus florida* via spawning. To develop the spawn of *Pleurotus florida*, wheat grains were boiled till tender and excess water was drained off. The boiled grains were allowed to cool and then mixed with 2% $CaSO_4$ and 4% $CaCO_3$ powder (per cent w.r.t mass of grains) to avoid clumping of grains. The grains were dispensed into glass bottles (250g/ bottle). The bottles were then cotton plugged and autoclaved for 90 minutes at 15psi. After cooling, the bottles were inoculated with 10^7 spores of *Pleurotus florida* and incubated at 27±2°C. To pretreat paddy straw, moist paddy straw (65-70% moisture) was inoculated with 10% (w/w) mycelium impregnated grains (per cent w.r.t mass of paddy straw). The inoculated paddy straw was pretreated at 27±2°C for 15 days. After 15 days, pretreated paddy straw was dried and analyzed for change in bulk density, proximate and chemical composition, and surface characteristics.

2.3 Determination of bulk density of paddy straw

The density of paddy straw is defined as the mass of paddy straw that can be accommodated in a known volume of a container. The bulk density of dried paddy straw was determined as follow:

$$D = \frac{M}{V} \tag{1}$$

where D = density of dried paddy straw
M = mass of dried paddy straw
V = volume of the empty beaker

The mass of dried paddy straw was determined by the formula:

$$M = M_1 - M_0 \tag{2}$$

where M = Mass of dried paddy straw
M_1 = Mass of dried paddy straw occupying beaker + Mass of beaker
M_0 = Mass of the empty beaker

2.4 Determination of proximate composition of paddy straw

Proximate composition provides information on the combustion characteristics of biomass. It is a measure of total solids (TS), volatile solids (VS), total organic carbon (TOC) and ash. Proximate analysis was done by the standard methods of AOAC (Thiex, 2000). The total solids in the sample (paddy straw) were determined by drying the sample overnight in hot air oven at 70°C. The dried sample was ignited in a tarred silica crucible in a muffle furnace at 650°C for 2h to determine ash which is the mineral content in the biomass that remain in oxidized form after combustion. While volatile solids comprise that fraction of biomass that is driven off by heating the sample at a specific temperature for a specific time leaving behind ash. Thus, VS was determined by the formula:

$$\text{VS}\ (\%) = 100 - \text{Ash}\% \tag{3}$$

Total Organic Carbon (TOC) was calculated from volatile solids by the formula:

$$\text{TOC} = \frac{\text{Volatile Solids}}{1.8} \tag{4}$$

2.5 Determination of chemical composition of paddy straw

Chemical composition gives information about the chemical components like cellulose, hemicellulose, lignin, silica and alcohol/ benzene extractives of biomass. Chemical analysis was done by the standard methods of AOAC (Thiex, 2000) using Fibretech Analyser (FibraPlus-FES08 AS, Pelican). The sequential determination of these components involves the determination of Neutral Detergent Fibre (NDF) and Acid Detergent Fibre (ADF) of biomass.

The concept behind the detergent fibre analysis is that plant cells can be divided into less digestible cell walls (containing hemicellulose, cellulose and lignin) and highly digestible cell contents (containing soluble sugars) (Figure 1). Van Soest (1963) separated these two components by the use of two detergents: a neutral detergent [Sodium Lauryl Sulfate (SLS) + Ethylene Diamine Tetra Acetic acid (EDTA) and an acid detergent - Cetyl Trimethyl Ammonium Bromide (CTAB) in 1M H_2SO_4.

Neutral detergent fibre (NDF) includes those components of biomass which are extractable (not soluble) with SLS+EDTA. Cellulose, hemicellulose, lignin and silica compose the neutral detergent fibre content of paddy straw.

$$\text{NDF}\ (\%) = \text{Cellulose} + \text{Hemicellulose} + \text{Lignin} + \text{Silica} \tag{5}$$

While acid detergent fibre (ADF) consists of those components which are extractable (not soluble) with CTAB in 1M H_2SO_4. Because hemicellulose gets solubilized in acid detergent fibre while extraction, so acid detergent fibre is composed of cellulose, lignin and silica only.

$$\text{ADF}\ (\%) = \text{Cellulose}+\text{Lignin}+\text{Silica} \tag{6}$$

Thus, hemicellulose is calculated as the difference between neutral detergent fibre and acid detergent fibre.

$$\text{Hemicellulose}\ (\%) = \text{NDF}\ (\%) - \text{ADF}\ (\%) \tag{7}$$

2.6 Morphological characterization of paddy straw through Scanning Electron Microscope (SEM)

The morphological and structural changes in pretreated paddy straw were recorded with scanning electron microscope (Hitachi S-3400N) at Electron Microscopy and Nanoscience Laboratory (EMNL) in PAU, Ludhiana, India. Paddy straw was conditioned and processed to view under SEM. The straw (untreated and pretreated) was dried overnight in a hot air-oven at 70°C. Then, the oven dried paddy straw was processed for SEM imaging.

The processing involves fixation of the sample (untreated and pretreated paddy straw) with glutaraldehyde, dehydration with increasing alcohol series followed by mounting of the sample on a stainless steel round stub using carbon-tape. Subsequently, the sample is sputter coated with gold (Au) nanoparticles in an ion-sputter coater before being viewed under the Scanning Electron Microscope. The samples were viewed at an accelerating voltage: 15000 Volts, emission current: 123000 nA, vacuum: 15 kV, magnification: X 1.50 K, working distance: 12.7 mm.

3. RESULTS AND DISCUSSION

Chemical and microbial pretreatment efficacies were evaluated in terms of the changes in bulk density of paddy straw; cellulose, hemicellulose, lignin and silica content; and change in surface structural properties of paddy straw.

3.1 Change in bulk density of pretreated paddy straw

The bulk density of paddy straw increased after various pretreatments. Paddy straw became remarkably fragile and lighter in color in the case of 4% NaOH-30 min microwave pretreatment which has also been reported in one of our previous studies (Kaur and Phutela, 2016a). The increased fragility enhanced the surface wettability of paddy straw. There was ~60% decrease in the mass of 4% NaOH-30 min microwave pretreated paddy straw. The bulk density of paddy straw after various pretreatments was found to fall in the range of 85.0 kg/m^3 (NH_3 pretreated paddy straw) to 210.0 kg/m^3 (NaOH pretreated paddy straw). Pretreated paddy straw with enhanced bulk density can be accommodated in a lesser space, if and when needed, which can help storage of huge amount of paddy straw in a limited space (Table 1).

3.2 Changes in proximate composition of pretreated paddy straw

The proximate composition of paddy straw remained unaffected after various pretreatments. As shown in Table 2, total solids varied non-significantly, if at all, after various pretreatments. While a little increase in volatile solids was observed in Na_2CO_3 and NaOH pretreated paddy straw, ash content decreased on the contrary. In point of fact, high volatile solids are good for methanogenesis as more volatile fatty acids are then generated which further enhance the biogas production.

3.3 Change in chemical composition of pretreated paddy straw

As shown in Table 3, least NDF (55.9%) and ADF (44.2%) were recorded in NaOH pretreated paddy straw which indicates that NaOH dissolved the insoluble fibres

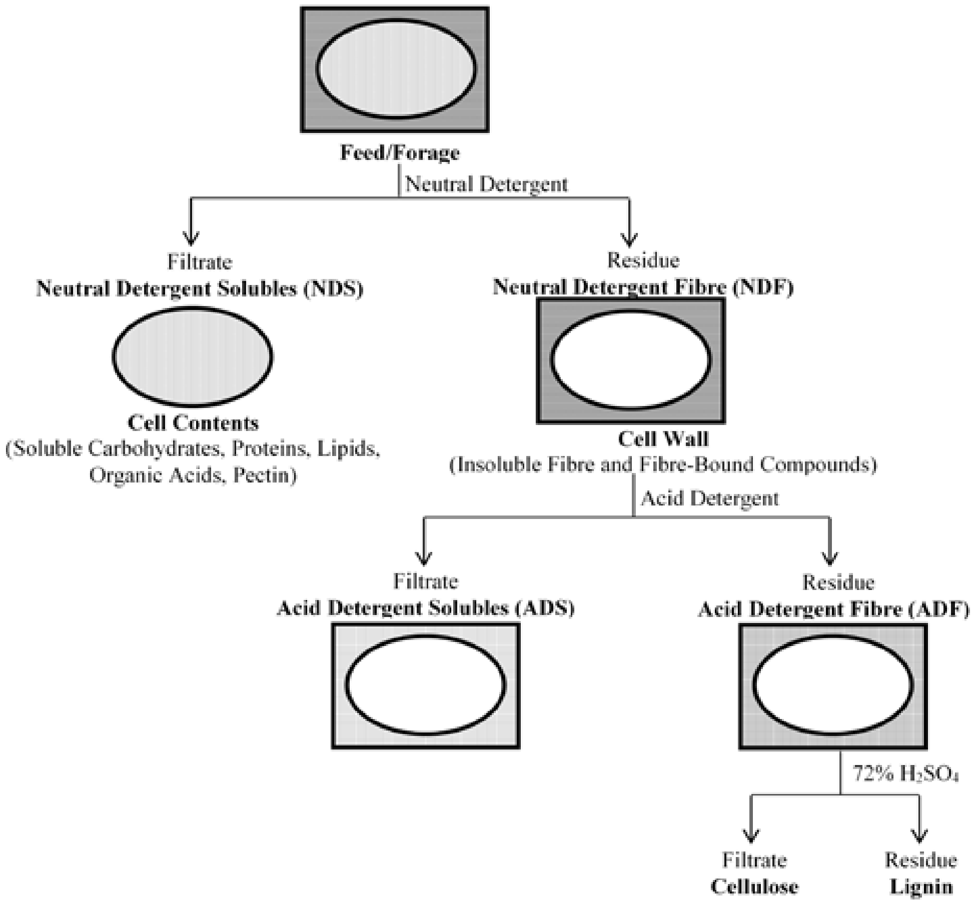

FIGURE 1: Detergent partitioning of fibre fraction according to Van Soest (adapted from Analytical Techniques in Aquaculture Research by Laboratory of Aquaculture and Artemia Reference Center).

and fibre-bound compounds in the cell wall of paddy straw. This then led to the decrease in hemi-cellulose, lignin and silica in the pretreated straw. Interestingly, cellulose content increased to 65.0% from 45.5% (control) in NaOH pretreated paddy straw which is attributed to the removal of recalcitrant components (Kaur and Phutela, 2016a) that makes it freely available and more accessible to fermenting microbes. In NaOH pretreated paddy straw, hemicellulose, lignin and silica content, respectively, decreased to 11.7, 2.9 and 1.0% from 23.0, 8.2 and 9.2% in control. Zhu et al. (2015) reported 92% decrease in lignin after 10% sulphuric acid pretreatment of rice straw for 2h with no significant loss of cellulose. *Pleurotus florida* pretreated paddy straw could not surpass the enhanced digestibility that was achieved with NaOH pretreatment. However, the hydrolyzed straw was still significantly more digestible than the control. Seven fold increase in hydrolysis was observed in fungal consortium pretreated wheat, rice, sugarcane and pea straw (Taha et al., 2015). Cianchetta et al. (2014) noticed minimal cellulose loss in *Ceriporiopsis subvermispora* pretreated wheat straw. Amongst all the pretreatments, maximum decrease in recalcitrance was recorded in case of NaOH pretreated paddy straw. The significant and appreciable decrease in silica content makes the pretreated paddy straw a more favored substrate for pyrolysis because slag formation due to melting and crackling of silica will no longer occur in gasifier filled with pretreated paddy straw. At the same time, the NaOH pretreated highly digestible paddy straw is a great substrate for biogas too as reported in our previous study (Kaur and Phutela, 2016a). The current method worked equally well when upscaled to 100 kg paddy straw and 1m^3 biogas digester.

TABLE 1: Change in bulk density of paddy straw after pretreatment.

Pretreatment	Bulk density of dried paddy straw (kg/m³)
Control	81.5±1.18
30 min microwave	81.2±1.57
4% NH_3-30 min microwave	85.0±1.29
4% Na_2SO_3-30 min microwave	98.0±1.43
4% Na_2CO_3-30 min microwave	175.5±1.65
4% NaOH-30 min microwave	210.0±1.78
Pleurotus florida (15d)	88.0±1.51

Control: Untreated paddy straw; Values depicted are mean ± standard error for n=3.

3.4 Morphological and structural changes in pretreated paddy straw

Several changes were recorded in the morphological

TABLE 2: Change in proximate composition of paddy straw after pretreatment.

Pretreatment	Proximate composition of paddy straw (%)			
	Total Solids	Volatile Solids	Ash	Total Organic Carbon
Control	85.4±1.23	75.6±1.16	24.4±1.05	42.0±0.98
30 min microwave	85.4±1.15	75.5±1.22	24.5±1.34	41.9±0.93
4% NH_3-30 min microwave	85.4±1.54	76.0±1.27	24.0±1.13	42.2±1.06
4% Na_2SO_3-30 min microwave	84.9±1.42	75.9±1.31	24.1±1.20	42.2±1.11
4% Na_2CO_3-30 min microwave	85.4±1.33	79.1±1.19	20.9±1.16	43.9±0.93
4% NaOH-30 min microwave	85.7±1.27	79.9±1.05	20.1±1.27	44.4±1.20
Pleurotus florida (15d)	86.1±1.38	75.9±1.24	24.1±1.35	42.2±1.03

Control: Untreated paddy straw; Values depicted are mean ± standard error for n=3.

TABLE 3: Change in chemical composition of paddy straw after pretreatment.

Pretreatment	Chemical composition of paddy straw (%)					
	NDF	ADF	Cellulose	Hemi-cellulose	Lignin	Silica
Control	74.5±1.35	51.5±1.12	45.5±1.02	23.0±0.87	8.2±0.45	9.2±0.62
30 min microwave	74.1±1.40	50.9±1.09	45.5±1.31	23.2±1.02	7.7±0.64	9.0±0.41
4% NH_3-30 min microwave	73.9±1.42	50.1±1.34	45.8±1.26	23.8±1.04	6.3±0.76	10.0±0.43
4% Na_2SO_3-30 min microwave	72.2±1.27	48.7±1.48	47.6±1.17	23.5±1.21	4.8±0.57	8.3±0.58
4% Na_2CO_3-30 min microwave	65.8±1.33	45.5±1.24	61.8±1.35	20.3±1.16	7.2±0.63	1.9±0.39
4% NaOH-30 min microwave	55.9±1.57	44.2±1.31	65.0±1.42	11.7±0.94	2.9±0.32	1.0±0.18
Pleurotus florida (15d)	69.1±1.61	50.6±1.53	49.5±1.47	19.5±1.09	5.6±0.49	8.1±0.51

Control: Untreated paddy straw; NDF: Neutral Detergent Fibre; ADF: Acid Detergent Fibre.
Values depicted are mean ± standard error for n=3.

and surface structural properties of paddy straw. Different pretreatments alter the surface properties of paddy straw in different ways (Figure 2).

In Figure 2-a, the surface of untreated paddy straw was seen intact with phytoliths (P- shown by arrow) and wart-like (W- shown by arrow) structures occupying the surface in an ordered fashion. These well-arranged phytoliths make the straw surface less wettable and add resistance against microbial attack. This makes straw less digestible and non-preferred substrate for feed and fuel. Sarkar and Aikat (2012) have also noticed similar rigidity in the surface of untreated rice straw.

In Figure 2-b, the effect of 4% NH_3-30 min microwave pretreatment is shown on the surface of paddy straw. The waxy cuticular silica layer was found breached. Cracking in the cuticular layer was observed due to the reduction in number of cross-linkages involving ester bonds between the wall polymers (Lam et al., 1992). Wang et al. (2007) have also observed that NH_4HCO_3 crack off the cuticular waxy layer of rice stem epidermis resulting in degradation by rumen micro-organisms. Bae et al. (1997) observed that the increased digestibility of NH_3 pretreated paddy straw was not due to the degradation of external epidermis of leaf sheath, but was attributed to the weakening of the adhesion between the cuticle and underlying tissues.

As shown in Figure 2-c, tearing of paddy straw surface was recorded in case of 4% Na_2SO_3-30 min microwave pretreatment which results in the removal of lignin (via degradation) from the torn epidermis. The surface degradation in Na_2SO_3 pretreatment was more than NH_3 pretreatment. Oxygen-sodium sulphite pretreatment has been reported for 95% delignification with retention of both cellulose and hemi-cellulose (Park et al., 2000).

Partial separation of waxy cuticular layer of paddy straw from the underlying tissues was noticed in 4% Na_2CO_3-30 min microwave pretreatment (Figure 2-d). This exposes the holocellulose content of pretreated straw making it more accessible to the fermenting microbes. Harbers et al. (1982) have also recorded substantial rupture of cuticular surface and its separation from adjacent ground parenchyma in the pretreated wheat straw.

In case of 4% NaOH-30 min microwave pretreatment (Figure 2-e), paddy straw surface was seen completely torn and tattered due to the dissolution of waxy cuticular silica layer, hence, exposing the inner more digestible components (cellulose and hemicellulose) of paddy straw. NaOH pretreatment was markedly effective in increasing paddy straw digestibility. After pretreatment, paddy straw turned lighter in color, smoother in texture with increased bulk density and surface wettability. These structural changes were in agreement with the changes in chemical composition of paddy straw whereby nearly 90% silica removal was recorded. NaOH cleaves the esterified bonds within the plant cell wall structure thereby reducing the physical enmeshment of cellulose (Chesson, 1981). Van Soest (2006) reported that NaOH dissolves silicified cuticular layer of paddy straw. Complete delignification of paddy straw along with appreciable silica removal can be achieved using NaOH (Park et al., 1999) which precipitates silica as sodalite in pulping (Jan and Alexandra, 2006). Papillae, wart-like structures and micro-hairs on the cuticular layer of epidermis were found crimped by NaOH pretreatment

(Wang et al., 2007).

In *Pleurotus florida* pretreated paddy straw (Figure 2-f) fungal hyphae (FH- shown by arrow) were found penetrating into various layers of cell wall of paddy straw which resulted in the degradation of lignin and enhanced straw digestibility. Degradation of cell wall by fungi depends upon the cell wall composition. Some fungi can colonize the entire tissue (like *Cyathus stercoreus*) whereas others are localized in poor lignified areas like mesophyll (leaf parenchyma) (Karunanandaa et al., 1995).

SEM studies of pretreated paddy straw were in accordance with the chemical make-up of the treated straw which showed enhanced cellulose content in pretreated paddy straw as compared to untreated paddy straw. However, lignin, hemicellulose, and silica, were recorded to reduce significantly.

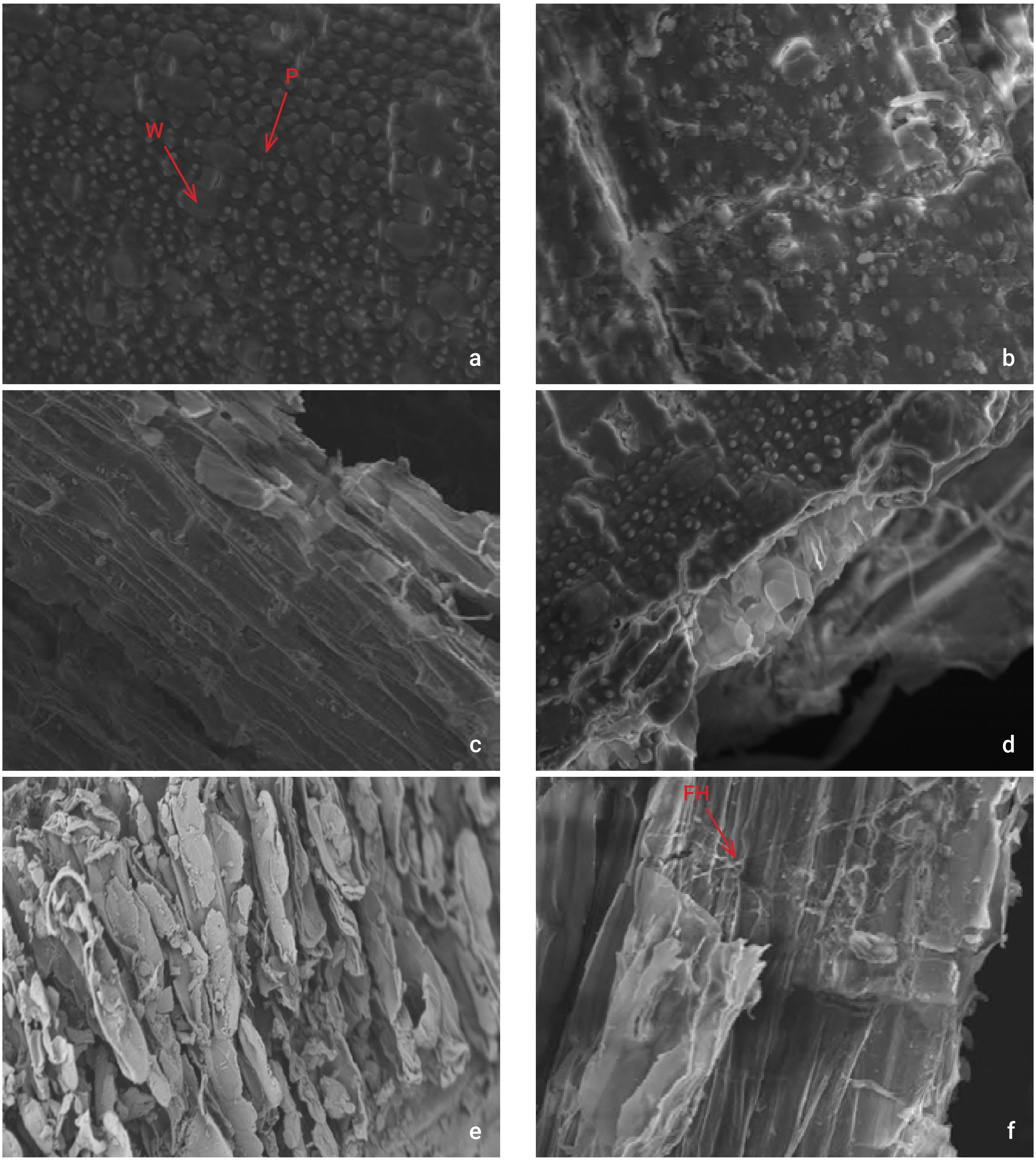

FIGURE 2: Morphological and structural changes in pretreated paddy straw after various pretreatments. a. Control; b. 4% Ammonia-30 min microwave; c. 4% Sodium Sulphite-30 min microwave; d. 4% Sodium Carbonate-30 min microwave; e. 4% Sodium Hydroxide-30 min microwave; f. *Pleurotus florida* - P: Phytolith; W: Wart-like structures; FH: Fungal Hypha.

4. CONCLUSIONS

Enhanced bulk density, decreased lignin and silica content, and tattered paddy straw structure indicates that microwave assisted NaOH pretreatment method is the best amongst the various pretreatment methods investigated in the current study. This pretreatment technique removes more than 90% silica besides reducing lignin. It is lot quicker with shorter pretreatment period and is straightforward. This method is equally effective when upscaled to 100 kg paddy straw. However, the wash water generated while washing the pretreated straw is a huge concern which needs to be addressed in future.

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SUPPLY AND SUBSTITUTION OPTIONS FOR SELECTED CRITICAL RAW MATERIALS: COBALT, NIOBIUM, TUNGSTEN, YTTRIUM AND RARE EARTHS ELEMENTS

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ABSTRACT

European industry is dependent on the import of raw materials. The European Commission has recognized that some raw materials are crucial for the function of the European economy and show a high risk of supply shortage. This communication addresses supply and substitution options for selected critical raw materials: cobalt, niobium, tungsten, yttrium, and the rare earth elements. For each element, the most relevant data concerning mining, abundance, recycling rates and possible substitutes are summarized and discussed.

1. INTRODUCTION

The availability of certain raw materials is crucial to Europe's economy (EC 2014). The COST Action CA15102, Solutions for Critical Raw Materials (CRM) Under Extreme Conditions (www.crm-extreme.eu), focuses on the substitution of CRMs in high value alloys and metal-matrix composites used under extreme conditions of temperature, loading, friction, wear, corrosion, in energy, transportation and machinery manufacturing industries. Presently, the European Commission identifies 26 raw materials or groups of raw materials of strategic importance; these materials exhibit both a high supply risk and important economic impact (EU 2017). The present communication reviews the current situation for a subset of this list: cobalt, niobium, tungsten, yttrium, and the rare earth elements (REE). It is evident that a strategy should be developed for the identified materials to close the loop and minimize the demand for virgin resources.

2. STATE-OF-THE-ART

2.1 Cobalt

Cobalt (Co) belongs to group 9 of the periodic table. The interest in Co is due to its industrially useful properties including ductility, malleability and magnetizability. These characteristics, combined with heat resistance (melting point 1495°C and boiling point 2870°C) and strength, make cobalt suitable for a wide variety of industrial and military applications (Minerals UK 2009).

Co has been known since ancient times. The first evidence dates to 2600 B.C., when blue glazed pottery was found in Egyptian tombs. Co-containing materials have been used as pigments for decades. The pure metal was isolated by Georg Brandt in 1735 (Donaldson and Beyersmann 2005).

The vast majority of Co is mined in Congo, which accounted for 54% of mine production in 2016. Furthermore, about half of the global reserves of Co are estimated to be in Congo. The importance of other countries is limited, with the individual share of other countries not exceeding 6%. Table 1 gives an overview of the geographical distribution of Co mining and reserves.

Typically, Co is used for metallurgical applications, as a component of superalloys, for the building of turbine engines for aircrafts, in the chemical sector (catalysts, adhesives, pigments, agriculture, and medicine), for the

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TABLE 1: World Mine Production and estimated reserves of Co (Shedd 2017a).

	Mine production 2016		Estimated reserves	
	[t]	Share	[1000 t]	Share
Congo	66,000	54%	3,400	49%
China	7,700	6%	80	1%
Canada	7,300	6%	270	4%
Russia	6,200	5%	250	4%
Australia	5,100	4%	1,000	14%
Zambia	4,600	4%	270	4%
Cuba	4,200	3%	500	7%
Philippines	3,500	3%	290	4%
Madagascar	3,300	3%	130	2%
New Caledonia	3,300	3%	64	1%
South Africa	3,000	2%	29	0%
United States	690	1%	21	0%
Other countries	8,300	7%	690	10%
World total (rounded)	**123,000**		**7,000**	

production of cemented carbides, and for the ceramics and enamels industry (CDI 2006). Nevertheless, the most common application is the manufacture of lithium-ion batteries, used for the power supply of electronic equipment. China is the leading consumer of cobalt, with nearly 80% of its consumption being used by the rechargeable battery industry (Shedd 2017a).

The recycling of Co is massively dependent on the application. Co-containing alloys are reprocessed into similar alloys and do not require a specific recycling technology. Hardmetal scrap is commonly recovered within the metal carbide sector. As lithium-ion batteries are the most common application, several recycling procedures have been developed for this area. The process commonly starts with reductive leaching (e.g. H_2SO_4, H_2O_2) followed by extraction and cobalt precipitation (Chen et al 2011, Pagnanelli et al 2016, Jian et al 2012). Cobalt recycling from applications in pigments, glass, paints, etc. is not readily possible as these usages are dissipative (EU 2016).

Table 2 summarizes possible substitutes for Co. For some applications, however, Co is essential as substitution would lead to a loss of product performance. This is in particular the case for the application with the highest share, lithium-ion batteries (25 %). Even though intensive research is being conducted in this area, a short-term breakthrough cannot be expected (Nayak 2017).

Considering the many uses, the recent Co demand has grown and it is essential to counteract the increased production of waste with increased recovery efforts (Cheang and Mohamed, 2016). According to EC 1014b, the end-of-life recycling input rate in the European Union in 2014 was 16%. For the USA, a recycling rate of 32% was reported in 1998 (Shedd 2004). In a more recent document, however, the EU Commission estimates the end-of-life recycling input rate to be zero (EC 2017).

2.2 Niobium

Niobium is a transition element of group 5. Due to its properties, it belongs to the group of refractory metals (Bauccio 1993). A Nb-containing oxide was first described by Charles Hatchett in 1801 who proposed the name Columbium (Hatchett 1802). Due to its similar properties, Nb could not be distinguished from Tantalum until 1865. Even if the official IUPAC name is Niobium (Nb), the name Columbium (Cb) is still widely used in North America.

Nb reserves are virtually inexhaustible (Schulz and Papp 2014), but are classified as critical due to the high production and deposit concentration in Brazil, as shown in Table 3.

Ferroniobium is by far the most important application for Nb and consumes almost 90% of the market (TIC 2016). Ferroniobium itself is used almost exclusively as an alloying element for steels containing Nb. In particular steel numbers starting with 1.45 or 1.46 may contain Nb, even if the concentration is below 1% (DIN 2014). Other end-uses are Nb chemicals, vacuum-grade Nb master alloys, pure Nb metal and Nb alloys such as NbTi (TIC 2016).

Commonly, Nb is not recycled as pure element but Nb-containing steels and superalloys are recycled for the same alloy. Thus, Nb recycling is not a question of technology, but of logistics. According to Papp (2017), the amount of recycled Nb is not available, but it may be as high as 20%. However, other sources report recycling rate of 56% (Birat and Sibley 2011). As Nb is used in relatively low concentrations (< 1%) in alloys (DIN 2014), separate handling of Nb is often not worthwhile. Therefore, the element is strongly diluted in iron scrap, where it no longer has any function. Only recently, the European Commission has claimed that the end-of-life recycling input rate is as low as 0.3% (EU 2017).

It is reported that Nb can be substituted by other materials, as summarized in Table 4. In any case, a loss of performance or higher cost accompany the substitutes (Papp 2017). It should also be noted that the possible substitutes themselves (e.g. W) are critical or mine production is much lower than for Nb (e.g. Ta). Therefore, it is essential to reintroduce Nb into the product cycle. Demand for new ore could be reduced through improved scrap management.

TABLE 2: Possible substitutes for Co (Shedd 2017a).

Application	Possible substitutes
Magnets	Barium or strontium ferrites, neodymium-iron-boron, nickel-iron alloys
Paints	Cerium, iron, lead, manganese, vanadium
For curing unsaturated polyester resins	Cobalt-iron-copper or iron-copper in diamond tools; copper-iron-manganese
Cutting and wear-resistant materials	Iron, iron-cobalt-nickel, nickel, cermets, ceramics
Lithium-ion batteries;	Iron-phosphorous, manganese, nickel-cobalt-aluminum, nickel-cobalt-manganese
Jet engines	Nickel-based alloys, ceramics
Petroleum catalysts	Nickel

TABLE 3: World Mine Production and estimated reserves of Nb (Papp 2017).

	Mine production 2016		Estimated reserves	
	[t]	Share	[1000 t]	Share
Brazil	58,000	90%	4,100	95%
Canada	5,750	9%	200	5%
Other countries	570	1%	n.a.	n.a.
World total (rounded)	**64,300**		**4,300**	

TABLE 4: Possible substitutes for Nb (Papp 2017).

Application	Possible substitutes
Alloying elements in high-strength low-alloy steels	Molybdenum and vanadium
Alloying elements in stainless - and high-strength steels	Tantalum and titanium
High-temperature applications	Ceramics, molybdenum, tantalum, and tungsten

Nb-containing steel grades should not be mixed with other steel grades, but rather should be remelted for similar alloys.

2.3 Tungsten

Tungsten (W) has the highest melting point of the pure metals and is irreplaceable in special industrial applications (BGS 2011). The name tungsten is derived from the Swedish words tung (heavy) and sten (stone) and goes back to Frederik Cronstedt, who described a high-density mineral in 1757 (ITIA 2011). Juan José de D´Elhuyar is considered to be the discoverer of tungsten. In 1783, he reduced tungsten oxide with charcoal (ITIA 2011).

Cemented carbides, also known as hardmetals, are the main use of tungsten and cover 56% of the market, followed by steel/alloys (20%), mill products (17%) and others (7%) (Somerley 2011). Other applications include catalysts, pigments, lubricants, electronics and electrical applications, solar power, medical and dental applications (Christian et al. 2011). Special attention is paid to W applications in materials under extreme conditions (Schubert et al. 2008).

As Table 5 shows, China is of paramount importance for tungsten production. In 2016, the country accounted for 82% of mine production. Vietnam, the second largest producer, is lagging behind and has a share of 7%. No data are available for the USA, but it has been reported that a new tungsten mine was opened in northwest Utah in 2016 (Shedd 2017b). In 2016, however, 76% of the tungsten imported into Europe came from Russia (EC 2014a).

According to Shedd (2011), the recycling rate for tungsten in the USA was 46% in 2000. A recent study (Zeiler et al 2018) shows that on a global scale the end-of-life recycling rate of tungsten (i.e. ratio of old scrap fed back) is 30% by 2016 and the recycling input rate (i.e. ratio of new and old scrap fed back) is 35%.

Possibilities for W-containing waste materials are described by Testa et al. (2014) and Shishkin et al. (2010), for example. Potential substitutes for W are summarized in Table 6. In some applications, however, substitution would lead to higher costs or loss of product performance (Shedd 2017b). Although depleted uranium or lead are not classified critical, their use is extremely problematic due to its toxicity. It should also be noted that tungsten carbide has unique properties which cannot be met by the suggested substitutes. For instance, Mohs harness of WC is 9.5, while MoC lags far behind (5.5). It must be concluded that tungsten is indispensable for certain applications at the moment.

2.4 Yttrium

Yttrium (Y) is a transition metal but is also considered to be a rare earth element (REE) along with scandium and the lanthanoids (Connelly et al. 2005). Y is mainly consumed in the form of high-purity oxide compounds for phosphors, in ceramics, electronic devices, lasers, and metallurgical applications (Gambogi 2016).

World production of Y came almost exclusively from China, as Table 7 shows. Minor amounts of mine production are reported for Brazil, India and Malaysia. However, the estimated reserves are quite large (more than 0.5 Million t) and far exceed mine production, which was estimated at 8,000 to 10,000 t in 2015 (Gambogi 2016). In contrast to mine production, China's dominance of global reserves is less pronounced. As shown in Table 7, only 41% of reserves are estimated in China followed by the USA, Australia and India. The reserves of Y are linked to those of rare earths (Gambogi 2016).

In many cases, Y is irreplaceable, as substitutes are generally much less effective. Especially in electronics,

TABLE 5: World Mine Production and estimated reserves of W (Shedd 2017b).

	Mine production 2016		Estimated reserves	
	[t]	Share	[1000 t]	Share
China	71,000	82%	1,900	61%
Vietnam	6,000	7%	95	3%
Russia	2,600	3%	83	3%
Other countries	1,700	2%	680	22%
Canada	1,680 *	2%*	290	9%
Bolivia	1,400	2%	n.a.	n.a.
Austria	860	1%	10	0.3%
Spain	800	1%	32	1%
Rwanda	770	1%	n.a.	n.a.
United Kingdom	700	1%	51	2%
Portugal	570	1%	3	0.1%
United States	n.a.	n.a.	n.a.	n.a.
World total (rounded)	**86,400**		**3,100**	

** Data for 2015*

TABLE 6: Possible substitutes for Co (Shedd 2017b).

Application	Possible substitutes
Cemented tungsten carbides	Carbides based on molybdenum carbide and titanium carbide, ceramics, ceramic-metallic composites (cermets), tool steel
Tungsten mill products	Molybdenum
Tungsten steels	Molybdenum steels
Lighting	Carbon nanotube filaments, induction technology, light-emitting diodes
Applications requiring high-density or the ability to shield radiation	Depleted uranium or lead
Armor-piercing projectiles	Depleted uranium alloys or hardened steel

TABLE 7: World Mine Production and estimated reserves of Y (Cordier 2012).

	Mine production 2011		Estimated reserves	
	[t]	Share	[1000 t]	Share
China	8,800	99%	220	41%
India	55	0.6%	72	13%
Brazil	15	0.2%	2.2	0.41%
Malaysia	4	0.04%	13	2.4%
USA	n.a.	n.a.	120	22%
Australia	n.a.	n.a.	100	19%
Sri Lanka	n.a.	n.a.	0.24	0.04%
Other countries	n.a.	n.a.	17	3%
World total (rounded)	**8,900**		**540**	

lasers, and phosphors, Y cannot be replaced by other elements. Yttrium oxide could be substituted by CaO or MgO as stabilizer in zirconia ceramics, but a lower toughness has to be accepted (Gambogi 2016).

Yttrium can be extracted from secondary resources preferably by hydrometallurgical processes, as they are also used for primary ores (Innocenzi 2014). Currently, no large scale Y recycling facility is documented (UNEP 2011), but progress is being made, including investigations into the recovery of Y from flat panel displays, spent optical glass and ceramic dusts.

2.5 Rare Earth Elements

The rare earth elements (REE) comprise the group of 14 lanthanides, of which promethium exhibits the lowest natural abundance. In addition to the 14 lanthanides, scandium and yttrium also belong to the REE group (Connelly et al. 2005), since these elements have chemical and physical similarities with the lanthanides.

REE are considered to be of critical importance in sustainable applications. REE and their compounds also find a multitude of applications in various branches of industry. Their demand is due to their use in various high-technology applications, for example, phosphors for fluorescent lamps, high strength permanent magnets, metallurgy, and applications in a number of green energy technologies. The main applications of REE are catalysts, metallurgy, magnets, electronics and in optical, medical, and nuclear technologies (Long et al. 2010).

China plays a dominate role in the production of REE. As shown in Table 8, China accounted 80% of mine production in 2016, followed by Australia with an 11% share. Other producers are of inferior importance. Global mine production in 2016 was around 132,000 t.

REE are relatively abundant in the earth's crust, and there are significant deposits outside China. Even if China hold 80% of mine production, only 37% of the estimated reserves are in China. Relevant deposits are located in Brazil, Thailand, Russia and India. As summarized in Table 8, minor REE deposits are estimated in several other countries.

Despite their highly fragmented applications, viable recycling technologies are already available today. In reality, however, less than 1% of REE are currently returned to the production cycle. (UNEP 2011, Tunsu et al. 2015). It is estimated that improvements in recycling can be achieved, particularly in the area of magnets, fluorescent lamps, batteries and catalysts (Jowitt 2018).

3. CONCLUSIONS

The present paper elucidates the availability, critical nature, and analysis of production value chains and downstream processes of for selected critical elements: cobalt, niobium, tungsten, yttrium, and the rare earth elements. The European share of reserves and mine production of these crucial elements is very low or even zero. Mine production is often concentrated in a single or very few countries. For Yttrium, 99% of mine production is in China. As the selected elements are crucial for the European industry, actions to reduce the dependency are strongly encouraged. On the one hand, the COST Action CA15102 evaluates the possibilities of replacing these critical materials with common materials without significant loss of performance. On the other hand, the demand for critical materials can be reduced by substituting new ores by secondary raw materials. It is evident that recycling needs to be significantly increased, as current recycling rates fall to zero (e.g. for Co).

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TABLE 8: World Mine Production and estimated reserves of REE (Gambogi 2017).

	Mine production 2016		Estimated reserves	
	[t]	**Share**	**[1000 t]**	**Share**
China	105,000	80%	44,000	37%
Australia	14,000	11%	3,400	3%
United States	5,900*	4%	1,400	1%
Russia	3,000	2%	18,000	15%
India	1,700	1%	6,900	6%
Brazil	1,100	1%	22,000	18%
Thailand	800	1%	22,000	18%
Malaysia	300	0.2%	30	0.03%
Vietnam	300	0.2%	n.a.	n.a.
South Africa	n.a.	n.a.	860	1%
Canada	n.a.	n.a.	830	1%
Greenland	n.a.	n.a.	1,500	1%
Malawi	n.a.	n.a.	136	0.1%
World total (rounded)	**132,000**		**120,000**	

** Data for 2015*

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INDIUM EXTRACTION FROM LCD SCREENS

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ABSTRACT

Liquid crystal display (LCD) screens are present in a variety of electronic devices including televisions, computers, cell phones, global positioning system (GPS) devices, and others. On a vitreous layer of their inner surface these screens contain the chemical element indium. The presence of this element, considered a critical raw material due to its economic importance and scarce availability, renders the recycling of these screens increasingly attractive. The present study therefore was undertaken with the aim of extracting indium present in LCD screens. Damaged or obsolete monitors with LCD screens were collected and dismantled manually to remove the glass layer containing indium, and subsequently, the glass layer was ground in a ball mill. After grinding, leaching tests for indium extraction were performed. Hydrochloric acid (HCl), at different temperatures and concentrations, was tested as a leaching agent at solid/liquid ratios of 1/100 and 1/10. The results obtained reveal the possibility of extracting indium, with the best result being obtained with HCl 6 M, 60°C, s/l ratio 1/100, with 298 mg In/kg.

1. INTRODUCTION

The electrical and electronic equipment (EEE) sector is one of the fastest growing in the world. However, due to rapid and constant innovation EEE tend to rapidly become obsolete. Once these products become obsolete or are no longer functional, they are considered as waste electrical and electronic products (WEEE). The main WEEE products include TVs, computers, cell phones and home appliances. Recycling of WEEE products involves a wide range of techniques and processes, largely aimed at the recovery of different materials, particularly metals (Gramatyka P. et al., 2007; Schaik A.V. and Reuter M.A., 2010; Wang X., Lu X. and Zhang S., 2013).

Television and computer monitors are characterized by a complex structure consisting mainly of glass, polymers, various metals and printed circuit boards, all of which add value to these materials (Gabriel et al., 2014). In addition, these wastes may contain precious metals such as gold and silver, and also indium, which are suitable for recovery in view of their potential value.

Since their invention, LCDs (Figure 1) have become one of the main types of screens used in televisions, computers and cell phones. This type of equipment has a short life cycle and contains considerable quantities of valuable materials.

The composition of LCDs comprises materials such as polymers, metals and ceramics (glass), thus rendering the recycling of this type of product extremely complex. Among the metals present, indium (In), a rare and highly versatile metal used in LCDs in the form of indium-tin oxide (ITO) can be detected.

ITO is a mixture of indium (III) oxide and tin (IV) oxide (indium-tin oxide) with a typical composition of 90 wt% In2O3 and 10 wt% SnO_2 in LCDs (Swain et al., 2016). ITO is a transparent and conductive material frequently used in the manufacture of thin-film transistors (TFT) used in liquid crystal displays (Virolainen et al., 2014; Chou and Huang, 2009).

The basic structure of the LCD screen as described by Juchneski et al. (2013) is illustrated in Figure 2. Item 1 represents the vertically polarizing film; item 2 the layer of glass with ITO; item 3 the liquid crystal; item 4 another glass layer with ITO; item 5 is the horizontal polarizing film; and item 6 the diffuser sheet.

Generally, the LCD panel consists of a glass substrate and a backlight module. The surface of the glass substrate is attached to a polarizing film, and the inner side is coated with functional films (ITO). As major functional units of LCD, the glass substrate accounts for 40-50 wt%, and the backlight module for another 35-40 wt% (including light guide plate and backlight). The light guide plate consists of polymethyl methacrylate (PMMA), polyethylene terephthalate (PET) or polycarbonate (PC) and a small printed circuit board (PCB) (Fisher, 2004). A cold cathode fluorescent lamp (CCFL) used as a backlight is parallelly

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FIGURE 1: Image of a monitor with LCD screen.

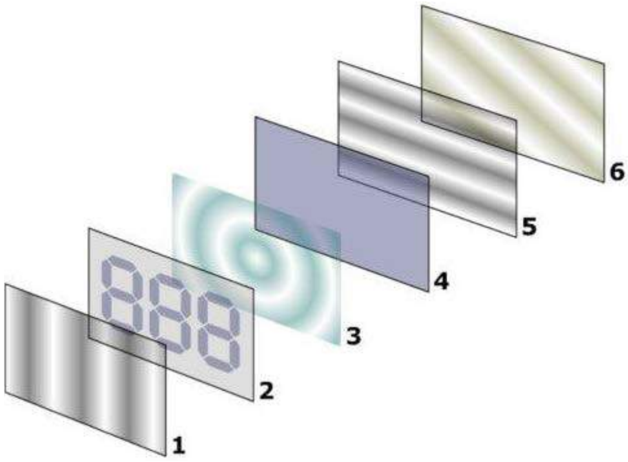

FIGURE 2: Structure of an LCD screen.

loaded along one or both sides. A metal frame (made of steel or aluminum) is usually mounted over the glass substrate and backlight module to protect the structure (Li et al., 2009).

Leaching is a widely used method in the extraction and recovery of metals from electrical and electronic waste (Bernardes and Veit, 2015). This process may also represent an effective method for use in the extraction of indium present in monitors.

Indium is a fundamental element in electronic devices, thus underlining the importance of developing and improving new recycling methods. Accordingly, the present study was aimed at extracting indium from LCD screens using hydrochloric acid and testing variations of the acid concentration, leaching temperature and solid/liquid ratio.

The variations applied in leaching tests (temperature, concentration and solid/liquid ratio) highlight the innovative nature of this work, which differs considerably from other studies reported in the literature. Several articles have reported the use of a series of different leaching agents or concentrations, others varying temperatures; however, to date no previous studies have been conducted to compare all these parameters. The only comparison available was performed by Yang et al. (2013) who compared a series of different concentrations.

2. MATERIALS AND METHODS

2.1 Manual Disassembly

LCD monitors of different brands and year of manufacture were initially collected. The monitors were then weighed and manually disassembled to segregate the components.

Monitors were essentially separated into four parts: liquid crystal screen, polymer sheets, polymer frames and printed circuit boards, as shown in Figure 3.

For the purpose of the present study the screens alone were used, whilst all remaining components were forwarded to recycling companies.

2.2 Comminution of the screens

The glass layers were manually fragmented into smaller pieces (3 to 5 cm), before milling the screens for 6 hours in a Servitech model CT-242 alumina ball mill. Particle size is of particular importance with an aim to improving the leaching process, because the larger the surface area the better the contact between the material and the leaching agent. Figure 4 shows the ball mill with the comminuted screens.

2.3 Chemical Characterization

X-ray fluorescence (XRF) analysis (Thermo Scientific Niton XL3t portable analyzer) was used to verify the pres-

(a) (b) (c)

FIGURE 3: Disassembled LCD monitors: (a) screen and polymer sheets, (b) polymer frames, (c) printed circuit board.

ence of indium in the powder obtained by milling. To obtain a better result, the powder obtained from milling was sieved in a mesh (#325). The particle size was selected on the basis of data from the literature (Dias et al., 2018 and M. del C. Ruiz et al., 2004).

2.4 Leaching

Samples of the powder obtained from the milled screens were separated for use in the leaching tests. For each test, specific amounts of materials were used, as shown in Table 1.

For this step an acidic leachate was used (hydrochloric acid) at a concentration of 38% and subsequent dilution with deionized water until the desired molarity was reached, as shown in Table 1.

The concentrations of metals obtained in the leaching tests were determined by means of an ICP OES, Agilent model 5110. In the ICP-OES analyses, a calibration curve was made between 0.02 ppm and 6 ppm with linearity of 99.9% from a standard solution of In with 100 ppm of brand SpecSol. The results presented correspond to an average of three wavelengths (325.609, 230.606 and 410.176).

3. RESULTS AND DISCUSSION

3.1 Manual Disassembling

Table 2 shows the total weight of two monitors and the weight of the screens.

3.2 Comminution of the screens

Figure 5 illustrates the material obtained after 6 hours of milling. The screens containing the layer of ITO were milled repeatedly through collisions between the balls or the balls and the mill container, thus facilitating the obtaining of extremely small particles.

3.3 Chemical characterization of screens

Table 3 describes the main components obtained by the FRX test, in powder after comminution. The main elements detected are typical of the standard formulation used for glass production, although, additionally, indium and tin were also detected.

Values obtained for indium and tin in the FRX were very low, indicating the presence of minimal quantities of these elements in the screens.

3.4 Results of Screen leaching

The results of the leaching tests are presented in Table

FIGURE 4: Material inside the ball mill container.

FIGURE 5: Material obtained after milling.

4. The best result was obtained using 6 M HCl at a temperature of 60°C and a solid/liquid ratio 1/100, with 298 mg In/kg, although in general the results were all comparable. On taking into account the economic and environmental aspects, the most interesting result was obtained at the lowest acid concentration (1 M) and solid/liquid ratio of 1/100 and 60°C, with 272 mg In/kg.

Using 5 g of material, no significant differences were observed when comparing temperature and concentration; moreover, molarity of the acid scarcely influenced the results obtained. However, using both low and high concentrations, better results were yielded when using 0.5g material.

Yang et al. (2013) studied ITO leaching using a series of

TABLE 1: Leaching tests parameters with hydrochloric acid.

Concentration of HCl (M)	Leaching Time (hours)	Temperature (°C)	Amount of Milled Screens (g)	Volume of HCl (ml)
1	4	28	0.5 and 5	50
		60	0.5 and 5	50
4	4	28	0.5 and 5	50
		60	0.5 and 5	50
6	4	28	0.5 and 5	50
		60	0.5 and 5	50

TABLE 2: Mass of samples collected.

Brand	Year of manufacture	Monitor total weight (kg)	Screen weight (kg)
A	2005	4.3	0.46
B	2006	4.0	0.37

TABLE 3: Components found by FRX in the screens.

Component	Sample (%)
SiO_2	68.58 ± 4.34
Al_2O_3	12.93 ± 0.82
CaO	7.36 ± 0.49
As_2O_3	1.45 ± 0.22
SrO	1.13 ± 0.17
K_2O	0.41 ± 0.38
SnO_2	0.04 ± 0.02
InO_2	0.02 ± 0.02

TABLE 4: Results obtained in the leaching test with HCl to samples after 6 hours of milling.

Concentration	Temperature (°C)	Result 5 g (mg/kg)	Result 0.5 g (mg/kg)
1M	28	193	208
	60	200	272
4M	28	189	285
	60	218	278
6M	28	204	259
	60	193	298

reagents, including HCl, and concluded that with 1 M HCl indium is leached over a period of less than 8 hours. These authors found approximately 260 mg In per kg of glass. Savvilotidou et al. (2015) studied leaching using hydrochloric acid and water at a ratio of 3:2, obtaining a result of 317 mg In per kg of LCD at 80 °C. These results are comparabale to the findings of the present study.

The indium contents detected both in the present study and in studies conducted elsewhere are similar, being higher than the average contents found in ores (1 to 870 mg/kg when obtained from the processing of zinc minerals). This finding justifies the use of research resources and investment in technologies for the extraction of this mineral from secondary sources.

It should also be highlighted how in this study the monitors had been previously disassembled manually, and only the component (screen) containing indium was comminuted, in this way concentrating and thus facilitating detection of the element.

On an industrial scale the monitors would likely be comminuted whole, which would hamper the concentration and extraction of indium, thus rendering the process more complex, involving more process steps and adding to the costs.

4. CONCLUSIONS

Following manual disassembly, monitors were found to be constituted largely by polymers, printed circuit boards and the screen, representing an average of 10% of total weight of a monitor.

FRX analysis revealed how the glass used in LCD screens contained standard components of glasses, in addition to indium, in the form of the ITO layer.

In leaching tests, the best condition was obtained using hydrochloric acid 6 M, 60°C, with 298 mg In/kg. However, other conditions should also be taken into account due to the finding of largely comparable results using lower acid concentrations.

Further studies are currently being carried out using different leaching agents, concentrations and temperatures in order to enhance technical comparison and shape future economic design.

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A CRITICAL TAXONOMY OF SOCIO-ECONOMIC STUDIES AROUND BIOMASS AND BIO-WASTE TO ENERGY PROJECTS

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ABSTRACT

Since biomass and bio-waste to energy systems condense activities that have important socio-economic and environmental sustainability effects, it is important that viability and impact studies have a socio-economic dimension, beyond the techno-economic and institutional aspects. This is necessitated in particular, by the limited and scattered availability of biomass or its residues, links to agricultural and forestry activities and associated socio-economic sustainability issues like land use, harvesting, transporting and economic conversion plant supplies. Such socio-economic studies, done prior to the project, can reflect a lot on the feasibility of projects, likely impacts and even help to optimize facility locations, network configurations or fleet management at various points or on the whole the supply chain. When the studies are done in retrospect of the project, as impact studies, they show how bioenergy projects can transform societies. The impact studies can then be useful precursors to similar projects within the same country/region or other similar regions. This review classifies socio-economic study literature into 'viability' studies- done prior to the project; and 'impact' studies- usually done after the project, except for 'projected impact studies'. The studies are also classified as 'quantitative and systematic' or 'qualitative'. Nonetheless, there are occasional overlaps between these study classes. Intentionally designed integrated approaches could actually give more comprehensive results, although in most cases, they result in complex models. This classification can guide researchers to make the right choice of the socio-economic study to carry out based on their objectives.

1. INTRODUCTION

The need for comprehensive socio-economic studies around bioenergy projects has been emphasized by different authors for various reasons (Deenanath, Iyuke, & Rumbold, 2012; Gasparatos et al., 2015; Ji & Long, 2016; Nogueira, Antonio de Souza, Cortez, & Leal, 2017; Pradhan & Mbohwa, 2014). Bioenergy projects in this study are inclined towards 2nd generation (2G) biofuel production from biomass and its residues or waste, although some cases of 1st generation studies and conversions to heat or power will be used occasionally as examples. Socio-economic studies around such projects can serve different purposes, depending on their focus. For instance country-wide surveys help to quantify and contextualise biomass distributions in various regions, while feasibility studies have been carried out to assess the potential viability of setting up a particular project in specific regions, taking socio-economic and ecological factors into consideration (Iakovou, Karagiannidis, Vlachos, Toka, & Malamakis, 2010). Socio-economic impact studies, on the other hand, help to quantify the socio-economic effect of bioenergy projects; and such case studies can serve as precursors of similar projects within the same region or in parallel regions (Pradhan & Mbohwa, 2014). The importance of prior socio-economic surveys has been demonstrated by the results of the Jatropha hype in Africa. More than 40% of the projects failed because that were started on the premises of projected assumptions, with no proper validation through socio-economic studies (Gasparatos et al., 2015). The studies could have exposed the plant's actual climatic, soil, water and labour requirements; the willingness and ability of communities to meet the demand; potential competition with agricultural inputs and/or infrastructure and sustainability of the supplies (Econergy, 2008).

Such socio-economic studies are pertinent and relevant because bioenergy projects represent a convergence of many socio-economic activities. In their analysis of socio-economic studies on biofuels, especially in the developing world, Nogueira et al. (2017) noted that bioenergy

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systems are strongly linked to activities with important socio-economic and environmental sustainability effects (Nogueira et al., 2017). This is because they lie at the intersection of energy and agricultural/forestry activities. In light of this, socio-economic viability or impact analyses of bioenergy systems are vital and should also include 'agricultural, environmental, economic and social aspects in addition to technological and institutional factors' (Nogueira et al., 2017).

An analysis of the mix of factors that will determine the biomass and therefore, bioenergy potential of a country or region (Figure 1), shows that more than 70% of these factors are socio-economic; spanning land, agriculture, logistics, policies, skilled labor availability and demographics (Gasparatos et al., 2015), (Von Maltitz & Setzkorn, 2013), (Econergy, 2008), (Friends of the earth, 2009). Inevitably, the success of a biofuel project for instance will, to a large extent, depend on these socio-economic factors, assuming technical and commercial viability. Already, the large demand for biofuels globally makes for a strong case for commercial viability (Econergy, 2008). Therefore, beyond technical R&D, site specific studies of regions with a high potential for biofuel projects should also be made to assess the biofuel potential, given the external socio-economic constraints (Iakovou et al., 2010).

To date, most research has focused on the techno-economic feasibility of producing biofuels like ethanol through various routes, but not much has been done on socio-economic side; especially in creating universal and/or integrated models and solutions (Nogueira et al., 2017). For instance, Pradhan and Mbohwa (2014) assert that comprehensive studies will be required to identify suitable feed stocks and technologies to establish a successful biofuel industry in South Africa (Pradhan & Mbohwa, 2014). Studies like feasibility assessments and supply chain optimizations can help assess and ascertain viability of such bioenergy projects, while impact studies like Life Cycle Analysis (LCA) can help evaluate socio-economic and environmental impacts through projections or in retrospect. Integrated socio-economic studies that cover the whole supply chain and use a combination of approaches, with various combinations of feed stocks and technologies, would be ideal, although they are complex to build. Pradhan and Mbohwa (2014) asserted that localized LCA studies could therefore help select the right feedstock and technologies best suited to the nation(s) and advise policy makers accordingly (Pradhan & Mbohwa, 2014).

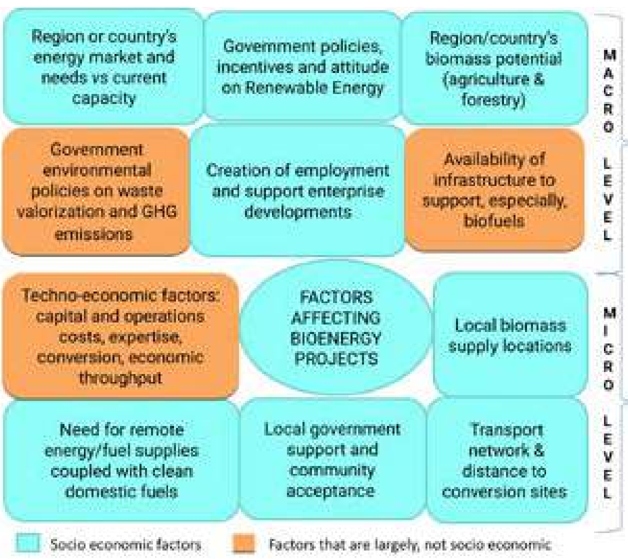

FIGURE 1: Factors that affect bioenergy projects.

2. TAXONOMY AND NOMENCLATURE: VIABILITY CONSIDERATIONS AND IMPACTS

This section reviews literature on socio-economic studies done prior to biofuel projects (viability considerations, including projected impact studies) and retrospective to biofuel projects as summarized in Figure 2. In most cases, these studies exclude detailed techno-economic study of the conversion plant, but span the upstream supply chain, value chain and impacts on relevant stakeholders.

The socio-economic studies will be further classified as follows:

- Quantitative and systematic socio-economic studies: incorporating computational (mathematical or heuristic) models to measure certain results, potential outcomes or compare methods or routes (Ba, Prins, & Prodhon, 2016). The basis for conclusions and recommendations is evidence from systematically drawn facts, though accuracy depends on the reliability of the model and its inherent assumptions. Examples are supply chain optimization through use of mathematical, heuristic or simulation models. In retrospective studies, some quantitative and systematic models have also been used to evaluate projects, for instance Lifecycle Assessments (LCA). Such models can be used to project impacts for other similar projects and provide insight for decision making (Nogueira et al., 2017).
- Qualitative socio-economic studies: These studies are usually surveys made, especially to assess the impact of projects in retrospect. When done prior to the project, they usually address subjective issues like communities' readiness to embrace a biofuels venture. These studies are more relevant when done in retrospect, since they are based on historical facts and evidence, including statistical facts. Since statistics are used in a pure historical rather than modeling context in this case, they do not then qualify such literature as 'quantitative and systematic, according to this discussion.

As previously discussed, a number of authors have stressed on the importance of conducting socio-economic studies, especially prior to launching a biofuel project (Amundson, Sukumara, Seay, & Badurdeen, 2015; Batidzirai, Smeets, & Faaij, 2012; Friends of the earth, 2009; Gasparatos et al., 2015); although retrospective impact studies are also important in informing future policies for the same or other regions (Pradhan & Mbohwa, 2014).

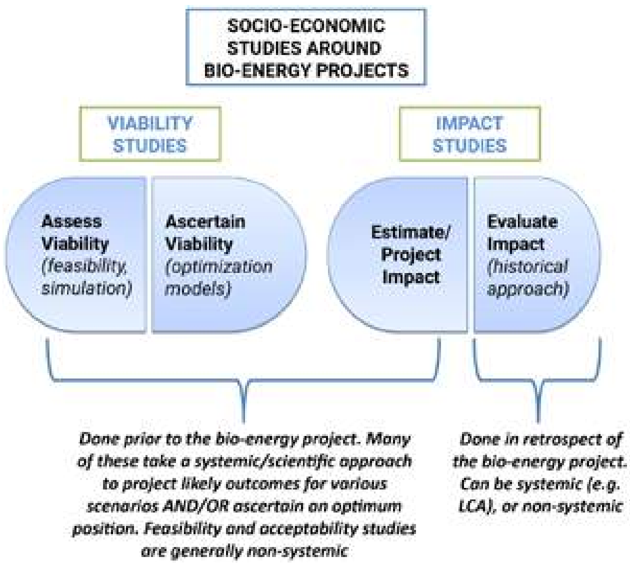

FIGURE 2: Classification of socio-economic studies around bio-energy projects.

3. SOCIO-ECONOMIC VIABILITY STUDIES

3.1 The distinction between 'techno-economic' and 'socio-economic' studies

A survey of literature shows a distinction between techno-economic and socio-economic studies around biofuels (Patel, Zhang, & Kumar, 2016; Timilsina & Shrestha, 2010). Techno-economic studies are mostly connected with the plant design/flow sheet (otherwise referred to as 'plant economics'), while socio-economic studies largely look at the flows of energy from the environment (societies) into or out of the plant and sustainability or impact of such flows (Badger, Badger, Puettmann, Steele, & Cooper, 2011; Dassanayake & Kumar, 2012; Patel et al., 2016). As discussed later, they are also integrated socio-economic studies that include detailed techno-economic schemes in the supply chain study, although these can be complex.

3.2 Socio-economic viability studies

Most reviewed literature that looks at the viability considerations for biofuel ventures has taken a quantitative and systematic approach. This is largely true for simulations and optimizations; while some feasibility assessments and acceptability surveys are mostly statistical. As depicted in Figure 2 socio economic viability studies have been classified in this study into those that assess viability or sustainability of a venture (viability assessments) AND those that aim at ascertaining viability through optimization.

3.2.1 Assessing viability

Noguiera et al. (2017) conduct a concise review of socio-economic studies made to assess and evaluate the sustainability of biofuel projects (Nogueira et al., 2017). The following models (Table 1), in their study, can be done prior to the launch of the project, as viability assessments.

Musango et al. (2011 & 2012) suggested the use of SD in assessing the sustainability of various conversion technologies in the African context if some renewable energy policies are enacted. They demonstrate an SD simulation approach through the Bioenergy Technology Sustainability Assessment (BIOTSA) model (Musango et al., 2012, 2011). Barisa et al. (2015) also used SD approach and looked into prospective biodiesel policy interventions and consumption patterns and their impact on ecosystem dynamics and services in Latvia (Barisa et al., 2015). Martine-Hernandez

TABLE 1: Types of viability assessments.

Study model	Description	Strengths and weaknesses	Literature
Simulation models e.g. System Dynamics (SD), Agent Based Modelling (ABM), Monte Carlo Simulation *Quantitative and systematic approach*	This is a well-established approach in many fields, that uses properly understood interrelationships between variables, metrics and indicators to determine how changes could produce overall system change over time. It is important to have an accurate conceptual model (e.g. a cause and effect diagram) from the onset; then an appropriate modelling software package is used to represent it. This cause-effect conceptualization is the basis of many other simulation packages. Usually used to simulate possible effects of policies.	**Strength**: it can give both a global and local view on socio-economic viability, especially when model is generalized. **Weakness**: it depends on the proper understanding of the cause-effect dynamics between variables.	(Barisa, Romagnoli, Blumberga, & Blumberga, 2015; Musango, Brent, Amigun, Pretorius, & Hans, 2012; Musango, Brent, Amigun, Pretorius, & Müller, 2011)
Feasibility assessments/ Enquiries *Quantitative and qualitative approaches*	These are preliminary surveys made to establish facts about the availability of adequate biomass, financial and people resources or skills to support a bioenergy venture. They could also look into the policy landscape, potential supply chain partners, markets and growth opportunities. Such a study is in a strict sense, classified as qualitative since it thrives on facts (statistical and non-statistical); unless models are employed.	**Strengths**: they give the first impression about the feasibility of having a bioenergy project. Such information is pertinent as it forms the basis for other detailed studies. **Weaknesses**: depending on approach, they may require intensive field surveys and accurate data acquisition methods, which can be difficult. Accessibility and availability of information also varies with country/region.	(Iakovou et al., 2010; Skoulou & Zabaniotou, 2007; Zhan, Chen, Noon, & Wu, 2005) Global organizational studies on biomass potential e.g. by FAO and EU, e.g. (Parikka, 2004), (Ericsson & Nilsson, 2006)
Acceptability surveys *Largely qualitative approach*	These surveys are usually done when there are ethical or acceptability issues in the region where the bioenergy project is targeted. In this case, a Public Consultation and Communication (PC&C) scheme can be carried out. These studies are usually integrated with other viability assessments.	**Strength**: they have a good social thrust and when recommendations are applied, they obtain support from surrounding communities. **Weaknesses**: alone, they are limited- they do not give a bigger picture around the bioenergy project (economic and environmental issues).	(Nogueira et al., 2017)

et al. (2015) used SD to simulate the potential effect of bioenergy production on ecosystem dynamics such as biomass production, carbon capture and nitrogen utilization in the soil (Martinez-Hernandez, Leach, & Yang, 2015). Cruz et al. (2009) applied the SD framework and developed a novel multi-time I-O based modeling framework that can be used to simulate bioenergy supply chain dynamics (Cruz, Tan, Culaba, & Ballacillo, 2009). Shastri et al. (2011) come up with an Agent Based Model using the theory of complex adaptive systems to simulate the system dynamics around agricultural biomass production, with farmers and the bio refinery as the 2 main independent agents (Y. Shastri, Rodríguez, Hansen, & Ting, 2011). The likely decisions and interactions of each agent are modeled/predicted using a set of socio-economic and personalized attributes deemed to govern the agent's behavior. They use the model to simulate the production of Miscanthus in Illinois; it can however, only be accurate to the extent at which the attributes and interrelationships between agents in the system are correctly modeled. For instance, the attributes and responses of farmers, while they can be market driven, can also be subject to their attitudes- a difficult attribute to model. Such a model is, however, useful in obtaining a near/approximate projection of likely outcomes a few years down the line (Y. Shastri et al., 2011).

Iakovou et al. (2010) claim that the majority of literature findings on the evaluation of biomass potential, selection of collection sites and capacity & location of conversion facilities take a feasibility study inclination (Iakovou et al., 2010). Shi et al. (2008) look into the feasibility/suitability of establishing new bio-power plants and optimizing their location using spatial information technologies like GIS and remote sensing; while Zhan et al. (2005) conduct a study to determine the economic feasibility of locating a switch grass-to-ethanol conversion plant in Alabama (Shi et al., 2008; Zhan et al., 2005). Both studies, though looking at feasibility, are very systematic since they use mathematical models and software solvers to assess the suitability/feasibility of potential biomass sites that with available and usable biomass. They also use GIS to find optimal plant location based on the spatial distribution of usable biomass.

Qualitative viability assessments come in the form of feasibility and acceptability studies. Most feasibility studies that only seek to locate and quantify available, usable and non-usable biomass fall in this category and they are usually the interest of global humanitarian organizations. For instance, the Food and Agricultural Organization and regional organizations like the European Commission have undertaken global, regional and country specific surveys to quantify biomass in defined geographical spaces(Ericsson & Nilsson, 2006; Parikka, 2004). Such studies already give the biomass potential of various regions, although some literature goes further to quantify biomass availability in smaller regions (Iakovou et al., 2010).

Integrated viability assessments

Viability assessments are often integrated with optimization techniques and to span a part or the whole supply chain. Shastri et al. (2013) incorporate informatics, modeling and analysis and a decision support for biomass feedstock production system in their integrated Concurrent Science, Engineering and Technology (ConSEnT) platform (Figure 3). This system then supports a regional bioenergy system, ensuring continuous operation of conversion facilities (Y. Shastri, Hansen, Rodriguez, & Ting, 2013). The model does not connect with the Midstream processing facilities, therefore could be more appropriately termed a semi-integrated study, since it does not cover the full SC.

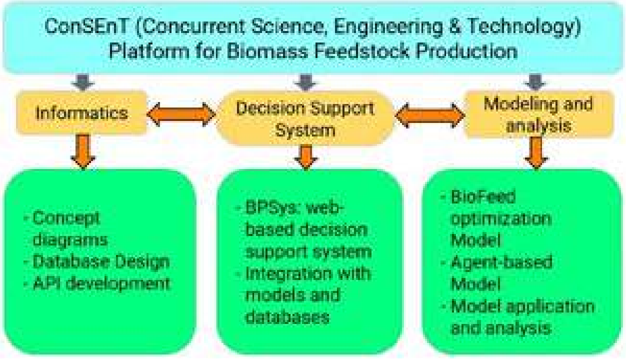

FIGURE 3: The ConSEnT integrated model for management of bio feed stocks (Y. Shastri, Hansen, Rodriguez, & Ting, 2013).

3.2.2 Ascertaining viability through optimization

The other set of viability studies try to ensure sustainability ahead of the bioenergy project through optimal use of resources. Whereas the viability assessments mostly inform policy makers or decision makers on viability issues at the macro socio-economic scale (spanning whole nations and regions), optimization techniques are usually project-centric and largely site specific. These optimization models are more at home in the viability rather than impact study category, with a goal to achieve the most sustainable/viable strategic, tactical or operational point. The optimal region in this case is always bound by the constraints introduced around the objective function- a mathematical model that represents pertinent social, economic and environmental goals. Inevitably, all optimization problems, by strict definition, follow the quantitative and systematic approach comprising objectives, constraints, a mathematical model and a software solver.

A number of authors concur that the two major constraints that hamper widespread uptake and dissemination of bioenergy projects: cost (a function of technical complexities, especially in the conversion technology) and the feedstock supply chain (SC) dynamics (Amundson et al., 2015; Batidzirai et al., 2012)(Ba et al., 2016). IRENA goes on to state that, for the advanced biofuel industry to be competitive compared to the fossil fuels, there is need for greater innovation in conversion technologies and supply chain models; market development and policy support (IRENA, 2016). Even where feed stocks are cheap, as in the case of forest residues (up to 50% cheaper than 1G feed stocks), the total cost for the feedstock supply significantly contributes towards high production costs; ranging from 40-70% (IRENA, 2016; Ji & Long, 2016). This is due to the low energy density of biomass compared to fossil fuels, making it imperative to optimize supply chain logistics and minimize costs (Amundson et al., 2015; Iakovou et al., 2010). Such a low energy density makes handling, storage and transportation of a unit of energy more expensive. Essentially, the complexities associated with the design and planning of biomass SCs emanate from the associated high costs of handling per unit energy, seasonal and uncertain nature of some feedstock supplies, variability of feedstock locations and other factors (Iakovou et al., 2010). These and other reasons make for a strong case in optimizing these SCs, with various objectives such as minimizing costs, maximizing conversion throughput, minimizing GHG emissions and maximizing social returns, e.g. employment.

Despite an equally compelling case for research around feedstock supply chain dynamics and costs, most research has focused on the conversion technologies (Paolucci, Bezzo, & Tugnoli, 2016). Recently, however, there has been an upward trend in research around biomass supply chains, though the initial bias was on assessment of potential biomass, allocation of collection sites and location of production facilities (Iakovou et al., 2010). However, SC optimization is increasingly spanning a broader scope thanks to recent advances in computational tools, subsequent improvements in mathematical models and the recent realization that SC logistics are a major bottleneck in most bioenergy projects (Ba et al., 2016; Hadidi & Omer, 2017; Pantaleo & Shah, 2013). Still, more research is required to ascertain bioenergy projects viability through SC optimization to contribute to a significant reduction in the cost of the integrated bioenergy system (Gold & Seuring, 2011; Hombach, Cambero, Sowlati, & Walther, 2016; Iakovou et al., 2010).

SC optimization literature generally concurs that supply chain complexities have to be addressed at 3 decision levels: strategic, tactical and operational (De Meyer, Cattrysse, Rasinmäki, & Van Orshoven, 2014; Iakovou et al., 2010), (Awudu & Zhang, 2012). These are defined in Table 2, along with the activities normally tagged along these levels.

The other important consideration in the SC optimization studies is the part of the supply chain they focus on, since in principle, the entire supply chain comprises the production, harvesting or collection of biomass; transportation; pretreatment; storage; subsequent conversion to bioenergy (heat, power or fuels) and supply to markets. Figure 4 shows a classic biomass supply chain divided into:

- the upstream process that delivers the biomass in the appropriate form to the conversion facility;
- the midstream conversion process;
- then, finally, the downstream SC which concerns the supply and distribution of the bio-product (heat, power or fuels) to the market.

A number of studies look into SC optimization at the different levels shown in Table 2. For the strategic level, most of the researches take a multicriteria decision analysis (MCDA) approach based on many hierarchical attributes or objectives, often conflicting, which are analyzed mathematically to obtain an optimal choice (De Meyer et al., 2014). Iakovou et al. (2010) analyze a synthesis of recent literature around the design and management of waste biomass supply chains (Iakovou et al., 2010). They do not focus only on optimization models, but on the design and management of, specifically, waste biomass supply chains (WBSCs). This article starts off at a strategic decision- sourcing of the biomass- with a number of researchers using geographical bibliographies like those already published by humanitarian organizations (Skoulou & Zabaniotou, 2007), and Geographical Information System (GIS) tools (Kinoshita, Inoue, Iwao, Kagemoto, & Yamagata, 2009; Voivontas, Assimacopoulos, & Koukios, 2001; Zhan et al., 2005). Of particular interest at the sourcing level are studies that try to minimize the costs of the supply chain by using a mix of biomass. Most of the studies are quantitative and systematic, using simulation and optimization models to compare options and combinations (Freppaz, Minciardi, Robba, & Rovatti, 2004; Frombo, Minciardi, Robba, Rosso, & Sacile, 2009).

A number of researchers look into the strategic capacity and location of conversion facilities; with some preferring to use GIS – based optimization. Panichelli and Gnansounou (2008) develop a methodology that integrates a GIS system with a biomass allocation algorithm to select suitable bioenergy facilities (Panichelli & Gnansounou, 2008). Papadopoulos and Katsigiannis (2002) develop a GIS tool to locate a conversion facility considering economic sustainability (Papadopoulos & Katsigiannis, 2002). Other facility location problems are solved use integer programming (IP); for instance Tembo et al. (2018) develop an mixed integer programming model to select the most economic biomass source and optimal bioethanol conversion facility location that maximizes net present profit (Tembo et al., 2018). Mixed Integer Linear Programming (MILP) is

TABLE 2: SC decision levels.

Description	Long term and usually investment intensive decisions that can be revised after several years	Address medium term decisions (usually between 6 months to 1 year) using guidelines provided by strategic decisions	Address short term decisions (weekly, daily and hourly)
Decision spheres and variables	Conversion facilities- size and technology to be used; biomass supply network design & configuration; facility location; sourcing and procurement (including supply contracts);	Inventory planning & control: How much to harvest/collect and store; selection, timing and place of treatment technology. Fleet management: transport mode, shipment size, routing & scheduling, outsourcing options.	Inventory planning & control: Daily inventory control and planning. Fleet management: vehicle planning and scheduling
Literature	(De Meyer et al., 2014; Iakovou et al., 2010), (Tembo et al., 2018)	(De Meyer et al., 2014; Iakovou et al., 2010), (Awudu & Zhang, 2012)	(De Meyer et al., 2014; Iakovou et al., 2010), (Awudu & Zhang, 2012)

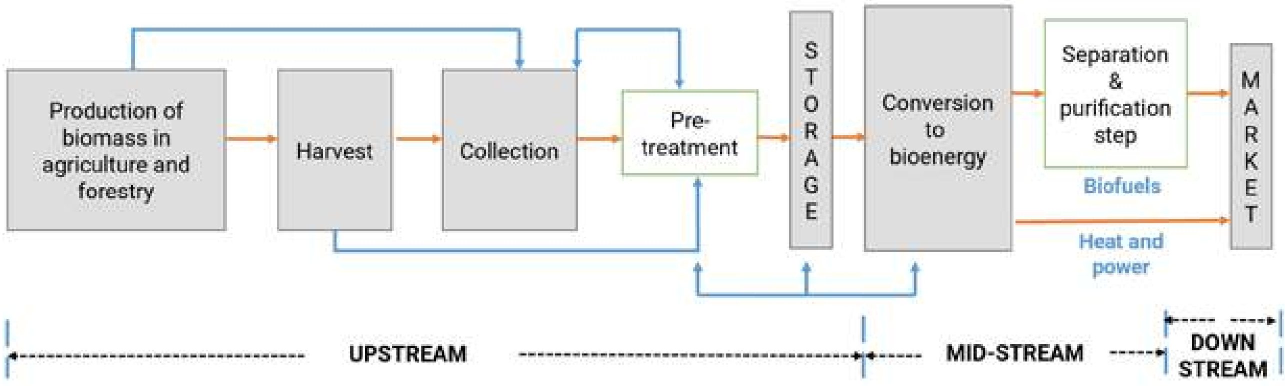

FIGURE 4: Biomass supply chain operations- Interrelationships and interdependencies. Arrows represent possible transport links (De Meyer et al., 2014).

used by many researchers at the strategic level spanning facility location and network design. Frequently, the MILP is embedded or combined with GIS, especially in facility location problems where a specific set of spatial criteria with respect to major highways, railroads or similar facilities are being considered. MILP tends to limit the researcher to one objective (usually the economic); whereas the common occurrence is that economic, ecological, energetic and social factors simultaneously affects supply chain decisions (De Meyer et al., 2014). Several authors therefore employ Pareto optimization to determine optimal Pareto trade off alternatives between various MILP objectives. Examples of researchers that have employed MILP only or along with GIS or Pareto for this purpose are summarized in Table 3.

Frombo et al. (2009) present an Environmental Decision Support System that optimizes the plant capacity and quantity of material harvested from a particular location, assuming a fixed plant location (Frombo et al., 2009).

A number of researchers combine strategic objectives with tactical; for instance network design or facility location (strategic) with fleet management or inventory planning (tactical) (Awudu & Zhang, 2012; Gold & Seuring, 2011; Iakovou et al., 2010). Paolucci et al. (2016) present a two tier approach for the optimal SC configuration by considering the environmental and economic aspects (Paolucci et al., 2016). Tier 1 uses simplified assumptions and average, limited geographical information; giving a streamlined multi-objective optimization of the studied system. It therefore sets up a firm basis for a more detailed optimization with Tier 2 using Multi-objective Mixed Integer Linear Program (MoMILP); involving both strategic (e.g. optical locations) and tactical objectives like transport flow optimization. A more informed basis for decision making results from Tier 2, generating a benchmark for further assessments (Paolucci et al., 2016). Iakovou et al. (2010) also point out that pre-treatment is also a critical tactical level decision to be made: i.e. to determine whether it is more effective to pre-treat before or after transporting, before subsequent storage (Iakovou et al., 2010). The biomass mix to be used also determines the intensity of the treatment schedule: fresh biomass will require more drying compared to biomass that has been left to dry for some time. On the other hand, biomass that was treated or grown in contaminated areas, with certain compounds that may later affect microbes or catalysis, will require a more rigorous treatment schedule. As such, the characterization of the different biomass types is good to ascertain their quality in terms of the presence of certain poisonous substances.

This review will not delve into operational level optimization publications. It is, however, worth noting that the day to day operations are better modeled closer to or during the running of the project, when the strategic and tactical objectives are clearly mapped.

Amongst computational, systemic methods that seek to ascertain viability are heuristic approaches. These look for satisfactory, but not always optimal solutions as in the case of optimization techniques, with the advantage of reduced runtimes. They usually find good application in complex problems characterized by high uncertainties requiring stochastic approaches or with many objectives or constraints. The most popularly used heuristics are population based, mainly genetic algorithms (GAs), Particle Swarm Optimization (PSO) and binary honey bee foraging (BHBF) (De Meyer et al., 2014). The modus operandi of these is evolving a population of solutions through a given number of iterations, then returning a solution subset of the populations evolved when the stop condition is fulfilled. It is observed that most literature that uses heuristics apply them to strategic level optimization problems. Authors like Celli et al. (2008), Rentzelas and Tatsiopoulos (2010) apply GA, a mimic of natural evolution, in facility location, type, sizing and biomass sourcing/allocation (Celli, Ghiani, Loddo, Pilo, & Pani, 2008; Rentzelas & Tatsiopoulos, 2010). According to De Meyer et al. (2014)its inherent advantages over other heuristics and optimization techniques is that it can handle multiple variables, both continuous and discrete (De Meyer et al., 2014). It therefore can optimize non-continuous, non-linear, and non- differential functions simultaneously; and it evaluates a large population, not a single point. PSO is also evolutionary, but based on the social flocking or swarming behavior of creatures like birds and fish; found effective in multidimensional optimization as espoused by Izquierdo, Minciardi, Montalvo, Robba, & Tavera (2008). BHBF is similar to PSO, however it is based on the swarm behavior of honey bees (De Meyer et al., 2014).

TABLE 3: MILP hybrid optimizations.

Study	Optimization model(s) used	Literature
Biodiesel supply chains from biomass produced by small scale Brazilian farmers	MILP	(De Campos Cesar Leão RR, Hamacher S, 2010)
Production of methanol from wood gasification, Austria	MILP	(Sylvain Leduc, Schwab, Dotzauer, Schmid, & Obersteiner, 2008)
Ethanol production from lignocellulosic biomass, Sweden	MILP	(S Leduc et al., 2010)
Optimal material flows and subsequent plant production costs for different demand scenarios and supply options. Also demonstrated differences between direct flow and flow via storage.	MILP and GIS	(Kanzian, Holzleitner, Stampfer, & Ashton, 2009)
Pareto optimization to determine optimal Pareto trade off alternatives between various MILP objectives	MILP and Pareto	(Mele, Kostin, Guillén-Gosálbez, & Jiménez, 2011; Zamboni, Shah, Bezzo, & others, 2009)
Optimal technology selection, bio refinery location and biomass flow according to a combination of objectives specified by user (e.g. maximize overall profit, minimize overall cost, minimize energy use etc.)	MILP model (Biocolo) combined with goal programming techniques	(Mol, Annevelink, & Dooren, 2010)

3.2.3 Integrated viability studies through optimization

Nogueira et al. (2017) propose that the evaluation of a bioenergy process in some locality should take an integrated approach that seeks to understand the interrelationships between environmental, economic, social and technological factors. They do acknowledge the complexity of such a model given the numerous direct and indirect factors involved, for each site, country or region (Nogueira et al., 2017). Such studies should span a large part or the whole supply chain: the upstream (Supply-to-conversion Chain), Midstream (conversion) and downstream (Market supply) as illustrated in Figure 4. It is evident however, that most SC optimization literature cover the upstream supply chain and rightly so, since the biomass supply can be a bottleneck if the dynamics at this stage are poorly managed. The Midstream has its own challenges, which mostly have to do with the choice of technology and optimizing technical parameters to enhance productivity/yield, reduce GHG emissions, increase efficiency and ultimately reduce the cost of producing a unit of biofuel. In many cases, the upstream SC has a significant bearing on the conversion process; while the choice of the conversion technology may in turn, also influence the nature of the upstream SC. This interdependence is explained by the points below:

1. For a given 2G conversion technology, there is a minimum supply threshold required for an economic biofuel production. The minimum required inputs of biomass for Lignocellulosic fermentation, Biomass to Liquid/Fischer Tropsch (BTL/FT) and Syngas fermentation conversion technologies are 2,280; 1,520 and 290 odt/day respectively (E4tech, 2009). This in turn affects the choice of biomass mix to be employed in order to meet the stipulated supply requirement. As the biomass feedstock increases, economies of scale may also lead to a reduction in cost of production, depending on the rate of increase in cost of obtaining the biomass.
2. If the conversion technology has already been picked, it will affect the choice of biomass and subsequent location of the conversion plant. For instance, lignocellulosic fermentation route does not have the luxury of accepting multiple feed stocks; therefore, to avoid technical complexities in cellulosic breakdown and fermentation, a uniform feedstock is ideal. In this case, the preferred feedstock is agricultural residues (IRENA, 2016), whose lignin fraction is smaller than woody biomass or forest residues, since this part cannot be broken down during the conversion process.
3. For a provided conversion technology, other considerations like the biomass feedstock type, quality and pretreatment requirements have to be taken seriously. For instance, if lignocellulosic fermentation is to be used, then the pretreatment formula should avoid the release of many inhibitory substances (Kennes, Abubackar, Diaz, Veiga, & Kennes, 2016; Walker, 2012). On the other hand, the catalysts in the BTL/FT process are sensitive to other contaminants that could be in the feedstock, like sulphur compounds, HCN, NOx and tar. This means that feedstock like treated poles with some traces of sulphur or, tar are not good for this process. That becomes a constraint on the biomass eligibility or otherwise implies higher treatment costs, which should be compared with the cost of alternative, distant feed stocks (E4tech, 2009). For syngas fermentation, vegetative matter brings in the danger of hydrogen cyanide contamination that is toxic to the acetogenic micro-organisms. This imposes a constraint on supplies of vegetative parts of the forest residues (E4tech, 2009).

Evidently, the type of conversion technology under consideration would impose more constraints around the biomass type, quantity, quality and pretreatment techniques to be used. As such, more integrated approaches that factor in, especially the type of conversion process to be used would bring in a broader perspective on SC optimization. Such approaches are scarce in literature because they usually require the collaborative input of different technical fields. The upstream SC part usually involves computational, industrial engineering and OR techniques, whereas conversion technology aspect will require core chemical engineering fundamentals. As Ba et al. (2016) suppose, integrated approaches will therefore require collaborative efforts between experts in these implicated fields, unless, the individual appreciates all these fields (Ba et al., 2016). Evidently, depending on the emphasis/thrust of the SC model (upstream/input or output parameters) the conversion module can be technically light or intensive.

The conversion module can also be technically intensive, comprising models that house rigorous mass balances and/or thermodynamic modules. For instance, Eason and Cremaschi (2014) describe a multi-objective, quantitative and systematic network flow system for an 'ideal' biofuel production process defined mainly around achieving low cost, a high energy recovery from feedstock and low carbon emissions (Eason & Cremaschi, 2014). The 3 available feed stocks are switch grass, corn and rapeseed, to be treated using alternative conversion technologies: gasification, anaerobic digestion, ethanol fermentation and transesterification. The result is a bio feedstock-to-biofuel super structure (BBSS) model with 17 production paths and requisite mass balance compositional data (Eason & Cremaschi, 2014). Aksoy et al. (2011) also conduct an integrated study that compares four biomass and sawmill waste utilization avenues defined by four bio refinery alternatives: BTL/FT through Circulating Fluidized Bed gasification, Simultaneous Saccharification and fermentation (SSF), Direct Spout Bed (DSB) of biomass with air and steam and direct combustion. They come up with a Decision Support system (DSS) that combines SC optimization with economic feasibility analysis, spanning the upstream, midstream and downstream sections of the SC. They also use the I-O models to evaluate the potential impact of these various avenues (Aksoy et al., 2011). Another case of integration is solved by Leduc et al. (2008) who look into optimal location for the polygeneration of ethanol, heat and power. However, they use a readily available steady state simulation model for a polygeneration plant for ethanol, heat and power, then use it to generate input data into the optimization model that covers the rest of the supply chain (Sylvain Leduc et al., 2008).

Evidently, such integrated modules that feature intensive conversion modules make the integrated module complex, with large computational times; however, they do give a holistic picture that factors in a lot of detail. Ultimately, there has to be a good trade-off between the research and economic value of the model's results and the time, effort and resources used to obtain them.

4. SOCIO-ECONOMIC IMPACT STUDIES

Impact studies using various indicators and approaches have been used at national, regional and international levels. The relevant studies to be considered in this scope are national and regional studies that are associated with specific biofuel projects. Most impact studies use isolated methods for a qualitative analysis of indicators that reflect on the socio-economic effects of such a system (Nogueira et al., 2017). There has also been a call for more integrated impact study approaches that can give a holistic overview of socio-economic and ecological factors (Amundson et al., 2015; Leimbach et al., 2011).

4.1 Isolated approaches

Macro-economic studies usually have a double pronged purpose: to evaluate the up-to-date socio-economic impact of the project in question, then to project its effects on a nation or region's economic growth. They can also use impact evaluations from other geographically or socio-economically similar nations or regions to project the likely effects on the particular country or region of study (Nogueira et al., 2017). Macro-economic studies have been carried out in Southern African countries like Tanzania and Mozambique and indicated that biofuel expansion could fuel their economic growth (Gasparatos et al., 2015). The study in Mozambique particularly took the form of a General Equilibrium Model (GEM) which concluded that biofuel production could contribute 0.37% of its Gross Domestic Product (GDP) and generate 271,000 rural jobs (Gasparatos et al., 2015). Bento et al. (2014) used an 'inter-regional, bottom-up, dynamic GEM' embedded with the 2005 Brazilian Input-Output (I-O) table to evaluate the effects of increased ethanol production and indirect land use change (ILUC) (Bento, Ferreira, & Horridge, 2014). In this approach, agriculture and land use were modeled separately for various regions and agricultural mixes.

I-O analyses are also widely used alone to assess macroeconomic impacts of bioenergy projects. They can be used to evaluate the impacts of new projects using I-O tables that show annual monetary flow of goods and services among various economic sectors. The interdependence between these flows is noted, especially with regards to the addition of a new major bioenergy project to the economy. A number of authors have used this approach: Martinez et al (2013). used an I-O model to demonstrate significant socio-economic impacts of expanding sugarcane ethanol bio-projects in North-east Brazil (Herreras Martínez et al., 2013). Kunimitsu et al. (2013) used an inter-regional I-O analysis to evaluate the economic ripple effects of bioethanol production on countries within the Association of Southeast Asian Nations (ASEAN) (Kunimitsu, Takahashi, Furubayashi, & Nakata, 2013). Evidently, the majority of hybrid approaches for impact studies have featured I-O analyses. For instance, You et al. looked into the optimum design of cellulosic biofuel SCs using multi-objective optimization (with socio-economic and ecological sustainability objectives) coupled with I-O analysis and LCA (You, Graziano, & Snyder, 2012). This approach has both a viability assessment and impact assessment dimension, although the latter is projected using known historical facts/experiences. Souza et al. (2016) integrated Social Life Cycle Assessment (s-LCA) with I–O tables to develop quantitative social and environmental metrics to evaluate various ethanol production technologies in Brazil using impact assessment (Souza, Watanabe, Cavalett, Ugaya, & Bonomi, 2016).

As with other impact studies, these macroeconomic studies depend on the accuracy of the facts tendered; for instance, the earlier macro-economic and life cycle assessment (LCA) studies made in Southern Africa on Jatropha could have misleading since they were based on inflated Jatropha yield statistics (Econergy, 2008; Gasparatos et al., 2015). Consequently, they reflected high developmental returns from such projects, which later proved inaccurate. Later reviews then suggested that the academics should have been gone on the ground to obtain comprehensive information rather than depend on reports (Gasparatos et al., 2015). This really brings an important aspect about impact studies, especially for such new projects; scholars should not naively accept all the information from project proprietors, who at times, are desperate to prove that their models are perfect. They should endeavor to do proper historical research and not be content with desktop studies.

LCA assessments are studies (using dedicated software) used to quantify and compare ecological and energy flows associated with agricultural and manufacturing or processing stages in a product value chain, in most occasions, including transportation (Sobrino, Monroy, & Pérez, 2011). Pradhan et al. (2014) comment on the fact that LCAs for biofuel projects are geographically specific. Consequently, a wide range of LCAs for somehow similar biofuel projects have yielded varying results due to differences in feedstock selection and types, conversion technology and system boundaries (Pradhan & Mbohwa, 2014). Pradhan et al. (2014) then assert that localized LCA studies could therefore help select the right feedstock and technologies best suited to the nation(s) and advise policy makers accordingly (Pradhan & Mbohwa, 2014). Generally, LCAs and their associated inventories (LCIs) are static models that do not consider socio-economic mechanisms like maximization of profit. However, the Consequential Life Cycle Assessment (C-LCA) can model socio-economic mechanisms through market factors of general and partial equilibrium, such that the relationships between activities and processes are not static connections but dynamic entities (Nogueira et al., 2017). Marvuglia et al. (2013) modeled such a C-LCA for biogas production in Luxembourg, with an emphasis on indirect land use change (ILUC) (Marvuglia, Benetto, Rege, & Jury, 2013).

4.2 Integrated impact study approaches

The majority of work done around evaluation of biofuel project impacts is based on isolated sections of the systems. Nogueira et al. (2017) bemoan the paucity of integrated, systematic methodologies for the comparison of the sustainability of various biofuel production systems (Nogueira et al., 2017). Indeed, such an integrated approach would give a holistic conclusion on the optimal parameters across the whole supply chain; however it is usually very complex, involving vast amounts of data and constraints, intricately constructed objectives and subsequently, complicated mathematical models and large computational times. These are the major drawbacks of an integrated approach either prior to the project or in retrospect; explaining why not so many scholars have used this route. However, it has been a growing area of interest in recent years. Nogueira et al. (2017) defines Integrated Assessment (IA) as a 'reflective and iterative participatory process that links knowledge (science) and action (policy) regarding complex global change issues such as bioenergy production and climate change' (Nogueira et al., 2017). Such an approach can be quantitative and systematic or qualitative. Leimbach et al. (2011) note that IA has grown popular as a tool for assessing strategies and policies around climate change; they assess the suitability of biofuel implementation strategies in the light of a complex ecological, and socio-economic matrix (Leimbach et al., 2011). However, literature reports that a few scholars use the IA approach in bioenergy production; one notable example being the systemic, 'SIByl-LACAf1 framework' proposed by Noguiera et al. (2017). It is presented as a sustainable, integrated approach where complementary evaluation methods are set in a logical and sequential array to assess the project's impact, along with a Strength, Weaknesses, Opportunities and Threats (SWOT) matrix (Nogueira et al., 2017). The mix of methods to be integrated can differ from project to project. Though largely a comprehensive approach due to the fact that it harnesses a pool of indicators derived from each method to give a holistic view, SIByl-LACAfl's robustness will always be subjective, depending on the individual suitability of the integrated methods selected, the logic used to arrange them sequentially and the method used to interpret the set of results obtained.

5. CONCLUSIONS

It is important to note that, though efforts have been made to distinguish between quantitative and systematic versus qualitative studies and viability assessments versus impact studies, there are occasional overlaps. For instance, some socio-economic impact case studies have been done and used as feedstock for viability studies in the same or similar regions (Pradhan & Mbohwa, 2014),(Bamière, 2013),(S Leduc et al., 2010). Similarly, some quantitative and systematic studies have featured qualitative methods like case studies as support for certain projects; while it is also not unusual to find qualitative studies featuring some small quantitative and systematic models, especially from a statistical angle. It is also not rare, as some studies have revealed, to have a mix of approaches for a more comprehensive, integrated outcome. All the same, classifying the socio-economic studies and characterizing them gives a better perspective into the broad subject. Consequently, any research entity that will desire to carry out a socio-economic study should be able to clearly define their objectives and strategy, guided by the taxonomy provided in this review.

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BIODEGRADABLE WASTE MANAGEMENT BY ANAEROBIC DIGESTION: A COMPARISON BETWEEN POLICY APPROACHES AND REGULATION IN ITALY AND ISRAEL

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ABSTRACT

Biodegradable waste is a significant component of municipal solid waste (MSW); anaerobic digestion allows the recycling of this waste. This paper presents a comparison between definitions, management, and usage of digestate/sludge and sewage from the anaerobic digestion (AD) of biodegradable waste in Italy and Israel in light of the legislation in both countries. Italian legislation is focused on three main components of the whole management/ recycling chain of bio-waste: the source of the waste, the characteristics of the digestate, the environmental matrix (i.e., soil and water) affected by the use of the digestate. Some relevant differences are currently present in the legislation concerning bio-waste and the sewage sludge management. In particular, both EU and Italian legislation lack specific "end-of-waste criteria" regarding the digestate from the AD of bio-waste. The legislation in Israel, on the other hand, is more focused on the application of the digestate and sewage from AD plants on soil rather than the source of the bio-waste. The focus on the end product (waste, water or sludge) is due to scarcity of water as well as soil sensitivity for agriculture use. The comparison indicates profound differences between the two countries, revealing inter alia advantages and disadvantages.

1. INTRODUCTION

In a dynamic, complex, and globalised world, an integrated and multidisciplinary approach is needed in order to analyse and solve complex problems. Such an approach is highly reflected in a joint international research (Brissaud, 2008). Scientific cooperation between Italy and Israel goes back to the early days of the Israeli state, inter alia as both are Mediterranean countries with similar agricultural crops and raw food materials (Tous & Ferguson, 1996). The scientific cooperation between the two countries is not limited to the natural sciences and Mediterranean studies, but extends to history, art, the classics, archaeology, and numerous other scholarly domains (Pagliaro, 2017). The current study compares policy approaches and regulation for biodegradable waste management by anaerobic digestion in Italy and Israel in light of the great challenges both countries are facing in the management of biodegradable waste and its by-products.

Thanks to a strong and reliable political, legal, and economic supporting scheme (EC, 2001), the EU has become a leader in production of renewable energy, with a total production of about 70 M ton oil equivalents. Anaerobic digestion (AD), with more than 17,500 facilities in the member states of the EU, and with total installed power of about 9,000 MW (EBA, 2016) contributes approximately 7.5% of the total renewable energy in Europe (EEA, 2016). Most diffused feedstocks for AD are represented by energy crops (ECR) (mainly maize), contributing to the production of more than 50% of the whole of the biogas generated (EC, 2017a), yet representing a cost increase from about 0.08 €/kWh to about 0.15 €/kWh (Schievano et al., 2015). This last aspect represents a serious threat to the viability of these facilities considering that many of them are now approaching the end of the period of economic subsidies.

A possible and widely studied solution (Pognani et al., 2009; Schievano et al., 2009) is the partial or total replacement of ECR with other substrates among which bio-waste is of particular interest. AD as treatment for bio-waste recycling is also considered a suitable technology for the implementation of a circular economy in this sector (EC, 2017b). Furthermore, since the bio-waste represents more than 30% of the whole of EU municipal waste, in order to achieve the overall recycling goals imposed by EU legislation (WFD, 2008) (i.e., 50% within 2020) recycling of bio-waste is crucial. Economic aspects limit the exploitation of AD in this

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sector to only about 10% of the EU28 bio-waste potential (ISPRA, 2017). For this reason, the replacement of ECR with bio-waste could provide an important opportunity for using the under capacity of existing ECR facilities the viability of existing ECR facilities and the further implementation of EU policy in the bio-waste sector at reduced investment costs. One should bear in mind that the management of the digestate is of particular concern in light of the absence of uniform EU end-of-waste (EoW) criteria. In fact, according to EU legislation (WFD, 2008), digestate from biomasses and ECRs are still considered biomasses, whereas digestate from waste is still considered waste. This legal distinction has affected successive management schemes. The most frequently adopted solution is preliminary solid/liquid separation with successive post-composting of the solid fraction to achieve the standard quality imposed by the organic fertilizer regulation. Since about 90% of AD used for ECR is of the wet type, more than 70% of the solid/liquid separation is still represented by the liquid fraction of the digestate. In some cases, its use on land can be authorized by legal entities in accordance with the R10 recovery operation "land treatment resulting in benefit to agriculture or ecological improvement" (Annex II, WFD, 2008) but in other cases its further processing in wastewater treatment plants (WWTP) could be requested in order to achieve standard water quality before discharge and/or reuse. In this case, even if the outlet water from WWTP is reused, the bio-waste cannot be considered recycled. WWTP is another important EU and Italian sector in which AD is widely exploited even if mainly for environmental considerations (i.e., the biological stabilized sludge before disposal/use) (Di Maria et al., 2016; Di Maria and Micale, 2017). Currently in the EU area there are some 36,000 WWTPs equipped with an AD section for sludge, representing another relevant source of digestate/sludge to be managed. Sludge can also be recovered by the R10 operation.

This approach arises from the Italian and EU legislation that imposes two main goals on the waste sector (CD, 1986; CD, 1991a,b: EC, 2015; WFD, 2008). The first goal is to manage waste without affecting the environment, including human health. The second goal is to make the best possible use of waste materials that can replace raw materials. In the specific case of digestate and sludge, the goal is to replace mineral fertilizers with the ones obtained from those processes. Due to great differences in climatic conditions and soil characteristics across Italy and EU, more specific details related to the quality of soils and specific features for use of these materials on land are usually outlined in local legislation.

Israel is characterised by an arid and semi-arid climate and its water resources are very limited. Water is one of the most significant environmental issues and a major concern in Israel, where the arable land area is approximately 4,200 km2 and the irrigated land area is about 1,866 km2 (Inbar, 2007). The water sector in Israel is subject to the Water Authority (WA), which has overall responsibility for it (Water Law, 1959) and legislation is created at the national level. The Water Authority also supervises the establishment of wastewater treatment facilities by the local authorities, mainly city associations or water corporations that are also required to maintain these systems (Sewage Law, 1962). The Water Law declares that all water resources are public property subject to the control of the state, thus there are no private water rights or resources in Israel and water may only be used by permit holders. As water consumption exceeds the natural rate of replenishment, while the intensity of freshwater use is extremely high by OECD standards (OECD, 2011).

Financial instruments for reducing consumption, such as a 40% increase in domestic water prices (introduced in January 2010) and financial penalties for pollution were also implemented in order to enhance overall water cycle management. Established in 1937, Mekorot, the National Water Company, supplies 70% of total water consumption. Water supplied to agriculture is mainly provided by Mekorot directly or by Agricultural Water Associations. Mekorot treats some 40% of the country's wastewater. The Ministry of Environmental Protection (MoEP) is responsible for protecting water quality and preventing water pollution. In the eastern Mediterranean region, irrigation with water of marginal quality has a long history, with Israel being the most prominent pioneer in advanced treated wastewater use policy and technology (Schacht et al., 2016).

This paper aims to compare and discuss the differences in legislation and practices related to biodegradable waste treatment and its liquid and solid digestate recycling between EU (and hence the Italian) and Israeli legislation.

1.1 List of Acronyms

AD	Anaerobic Digestion
CFU	Colony Forming Units
COLL	Collection
ECR	Energy Crops
EoW	End of Waste
EU	European Union
KWh	Kilo Watt hour
MBT	Mechanical and Biological Treatment
MCM	Million Cubic Meters
MoEP	Ministry of Environmental Protection
MoI	Ministry of Interior
MPN	Most Probable Number
MSW	Municipal Solid Waste
MSWM	Municipal Solid Waste Management
OECD	Organization for Economic Co-operation and Development
PFU	Plaque Forming Units
STD	Standards
TS	Total Solids
WA	Water Authority
WR	Water Regulations
WWTP	Wastewater Treatment Plants

2. METHODOLOGY

The scientific approach in this study is based on a joint international study for conducting a comparative analysis of the policy approaches and regulation in Italy and in Israel. Such a comparison is expected to point to advantages and disadvantages of the management systems in both countries and thus contribute to the enhancement of these systems.

The comparison implemented in this study required the collection, classification, and processing of various data, including documentation such as laws, regulations, government decisions, and qualitative data. The data was retrieved from literature and from official documents of legal entities charged with waste planning and monitoring. Furthermore, data from previous works of the authors were considered. The legislation in both countries was reviewed and processed into a visual scheme of the technical and legal recycling pathway of bio-waste via AD in both countries, providing an accessible way to understanding the various "decision junctions" along the pathway as a tool to support conclusions drawn from the comparative analysis.

The following definitions will be adopted in the study: liquid digestate, the fraction of digestate characterized by a Total Solids TS ≤ 10% w/w and sludge, the digestate characterized by a TS ≥ 15%. It is important to note that in the EU, as in Italy, the term "sludge" is usually used to refer to the sludge generated by the sludge treatment lines (primary and activated) of wastewater treatment plants; the above term "sludge" applies to Israel as well.

3. RESULTS

3.1 Italian legislation and scenario

3.1.1 Waste management legislation

The reference legislation for waste management in Italy arises from the adoption of the latest EU directive, the Waste Framework Directive 2008/98/EC (WFD, 2008). This directive imposes some relevant goals to be achieved by the member states at given times. In particular, by 2020 not less than 50% of waste, such as paper, plastics, cardboard, metals, and glass, is required to be prepared for reuse and/or recycled. The recycling of bio-waste, as defined by the EC Environment, by recovery operation R3 "Recycling/reclamation of organic substances which are not used as solvents (including composting and other biological transformation processes" (Annex II, WFD, 2008) is intended to contribute to the achievement of this goal.

Alternatively, bio-waste can be considered recycled after AD if the digestate is effectively used on land. In this case, due to the absence of EU EoW criteria, the authorization of this operation is subject to the standard qualities imposed by the Council Directive 86/728/EEC (CD, 1986) on the agronomic use of sludge from WWTP classified as the R10 recovery operation. This imposes limits regarding the concentration of heavy metals and other pollutants, including pathogens, for the sludge but also limits on the content of heavy metals for the soils on which the sludge is spread (Table 1). Another relevant legal aspect to be considered in use on land is Council Directive 91/767/EEC (CD, 1991a) concerning the protection of water against pollution caused by nitrates from agricultural sources. This Directive limits the amount of nitrogen in soils to 170 kgN/ha/year for vulnerable areas, and 340 kgN/ha/year for non-vulnerable areas. In any case, separated collection of bio-waste is a compulsory requirement for its recycling (main water and wastewater legislations are listed in Table 2).

TABLE 1: Chemical and physical features for use of sludge from WWTP on land (D.Lgs., 1999).

Parameter	Value
For sludge from WWTP	
Cd (mg/kg TS)	20
Hg (mg/kg TS)	10
Ni (mg/kg TS)	300
Pb (mg/kg TS)	750
Cu (mg/kg TS)	1,000
Zn (mg/kg TS)	2,500
TOC (%TS) (min)	20
Total P (%TS) (min)	0.4
Total N (%TS) (min)	1.5
Salmonella MPN/g TS (max.)	10^3
For soil	
N for vulnerable areas (kg/ha/year)	170
N for non-vulnerable areas (kg/ha/year)	340
Cd (mg/kg TS)	1.5
Hg (mg/kg TS)	1
Ni (mg/kg TS)	75
Pb (mg/kg TS)	100
Cu (mg/kg TS)	100
Zn (mg/kg TS)	300

3.1.2 Wastewater management legislation

In cases where the digestate from bio-waste cannot be used on land, it usually undergoes a liquid/solid separation. According to current legislation and standard quality, the solid fraction can be composted for the production of organic fertilizer, whereas the liquid fraction is moved to WWTPs with appropriate permits. In these facilities, the liquid digestate is usually co-treated with domestic wastewater, and the goal of the treatment is to reintroduce the water into the system in compliance with the water standard quality imposed by the current legislation. Specifically, there are two main water standard references (Table 3): one for discharge in surface water (e.g., lakes, rivers), the other for reuse.

In the latter case, the legislation refers to three possible reuses: agricultural, industrial, and domestic, with the exclusion of drinking and hygienic use. Currently, at the EU level, water reuse is strongly promoted (CD, 1991b), but no target has been defined yet. Italy currently reuses about 9% of its wastewater based on quality of water discharged by WWTPs, while the potential is estimated to be 60% (EC, 2015). Even if the purified water is reused, this cannot be considered recycling of bio-waste since the goal of WWTP is to remove N and P, which represent the real focus of recycling for the EU legislation (i.e., R3 and R10 operations).

3.1.3 Anaerobic digestion of bio-waste

In the EU28, the bio-waste production potential is of about 90Mtonnes. Currently, approximately 40 Mtonnes

TABLE 2: Main water and wastewater legislations in Italy.

Year	Legislation	Purpose
1896	Local regulations on hygiene of soil and house	To establish the main regulation for surface water cleaning, drinking water supply and delivery, wastewater disposal
1904	Legal regulation for hydraulic works	To establish the state as responsible for the protection of public water and related works
1933	Legal regulation for water and hydraulic power plants	To identify the users in terms of small and large public water withdrawal, define the regulations for the search for and extraction and use of ground water, roles for the transmission and distribution of electrical energy
1934	Sanitary legislation	To outline the hygienic conditions for water outflow and impose treatment for wastewater before discharge in water bodies
1963	Master plan for aqueduct	To plan the water supply and delivery system
1976	Legal regulation for protection of water from pollution	To represent the first legal framework regarding wastewater management, collection, and treatment
1898	Legal regulations for the reorganization and protection of the soil	To establish soil protection, water reclamation, management of water bodies
1994	Regulation on water resources	To rationalise the national water supply system
1999	Regulation on water protection from pollution	To define the general principles for prevention and reduction of the pollution, sustainable use and preservation of natural self-capacity of purification of water bodies
2003	Regulation on water and wastewater reuse	To impose possible reuse of the wastewater after purification process and the standard quality

TABLE 3: Main water quality standards for discharge in surface water and reuse in Italy (D.Lgs., 2006; D.M. 2003).

Parameter	Units	Surface Water (max)	Reuse (max)
pH	mg/l	5.5-9.5	6-9.5
SAR[a]	mmol/l	-	10
Solids	mg/l	None	None
BOD5	mg/l	25-40	20
COD	mg/l	125-160	100
Total P	mg/l	2-1	2
Total N	mg/l	15-10	15
N-Ammonia (as NH_4)	mg/l	15	2
Conductivity (mS)	mS/cm	-	3,000
Al	mg/l	1	1
As	mg/l	0.5	0.02
Ba	mg/l	20	10
Be	mg/l	-	0.1
Bo	mg/l	2.0	1.0
Cd	mg/l	0.02	0.005
Co	mg/l	-	0.05
Total Cr	mg/l	2.0	0.1
Cr+6	mg/l	0.20	0.005
Fe	mg/l	2	2
Mn	mg/l	2	0.2
Hg	mg/l	0.005	0.001
Ni	mg/l	2	0.2
Pb	mg/l	0.2	0.1
Cu	mg/l	0.1	1
Se	mg/l	0.03	0.01
Sn	mg/l	10	3
Tl	mg/l	-	0.001
V	mg/l	-	0.1
Zn	mg/l	0.5	0.5
Total CN	mg/l	-	0.05

Legend: a=Sodium Adsorption Ratio for soils

are recycled mainly by composting (3,500 facilities) and only 8 Mtones are processed or co-processed by AD (about 245 facilities). Italy has a bio-waste production potential of approximately 9 Mtonnes. Of this amount, as of 2016, 3.4 Mtonnes are recycled by composting in 274 plants, about 2 Mtonnes are recycled by integrated AD and post-composting facilities in 31 plants, and about 0.25 Mtonnes are processed by AD in 21 plants. A large part of the liquid digestate generated after solid/liquid separation is currently processed by WWTP. A minor amount is currently used on land in accordance with the R10 operation.

3.2 Israeli legislation and scenario

3.2.1 Waste management legislation

Until the early 1990s, 97% of the MSW produced in Israel was landfilled in hundreds of unregulated sites that were used and operated by local authorities. Following the closure of hundreds of unregulated dumps during the 1990s, the MoEP declared a “recycling revolution” that included a comprehensive program for transitioning from landfilling to turning MSW into a resource via recycling. The initial goal set by the MoPE in 1998 was to increase MSW recycling and recovery rates to 25% by 2007. Beginning in 2006, further steps were taken, including the imposition of a landfill levy and the establishment of a financial support program for local authorities to promote separation at source (Daskal et al., 2018). To date, separation at source of bio-waste is not mandatory and most bio-waste is landfilled without any treatment. In particular, AD is not mandatory, implementation of this treatment method is relatively low, and the definitions of this process are vague as there are no clear classifications regarding recycling vs. recovery. In light of the above, sludge management in Israel is mainly associated with WWTP.

3.2.2 Water and wastewater management legislation

Leapfrogging in the treatment and reuse of wastewater in Israel occurred when the state took the lead on this issue, set standards, and financed projects, making Israeli industry a world leader in wastewater treatment and disposal. There are numerous laws and regulations that relate to water and wastewater in Israel. Table 4 presents the most central of these.

3.2.3 Sludge management legislation

The Water Regulations (WR, 2004) are aimed to prevent the pollution of water resources and the creation of environmental nuisances as a result of uncontrolled disposal of sludge originating in municipal sewage. The regulations classify sludge according to various definitions based on the level of treatment and the characteristics of the material obtained. Table 5 presents classification of sludge and various materials according to the Water Regulations (2004).

TABLE 4: Main water and wastewater legislation in Israel.

Year	Legislation	Purpose
1957	The Drainage and Flood Prevention Law, 1957	The 11 drainage authorities are primarily responsible for drainage of agricultural runoff, including through channelisation of rivers.
1959	The Water Law, 1959	Establishes the framework for the control and protection of Israel's water sources.
1962	The Local Authorities Sewage Law, 1962	Prescribes the rights and duties of local authorities in the design, construction, and maintenance of sewage systems.
1971	The Water Law Amendment, 1971	Outlines prohibitions against direct or indirect water pollution, regardless of the state of the water beforehand.
1981	Discharge of Industrial Sewage into the Sewage System, Model Local Authorities Bylaw, 1981	Sets recommendations to local authorities on the treatment of industrial sewage and its disposal into the sewage system.
1988	Streams and Springs Authorities Order (Yarkon River Authority), 1988	Establishes the Yarkon River Authority, which includes: prevention and abatement of stream pollution, planning and implementation of rehabilitation schemes, and transformation of the area into a recreational site.
1991	Prevention of Water Pollution – Rinsing of Containers for Spraying, Regulations, 1991	Prohibits aerial spraying of biological and/or chemical substances for agricultural purposes near a water source, including Lake Kinneret, the open sections of the National Water Carrier, the Upper Jordan River and its tributaries, and other sources of drinking water.
1992	Prevention of Water Pollution – Cesspits and Septic Tanks, Regulations, 1992	Establishes prohibitions and restrictions regarding the construction of new cesspools and septic tanks and on existing ones, including timetables for the gradual elimination of cesspools under certain conditions.
1994	Prevention of Water Pollution – Reduction of Salt Use in the Regeneration Process, Regulations, 1994	Requires industries to undertake a number of technical steps to bring about salt reduction in the regeneration of ion exchange in order to reduce the quantity of salt used in the water-softening process and the consequent emission of brines into the municipal water system.
1994	Streams and Springs Authorities Order (Kishon River Authority), 1994	Establishes the Kishon River Authority, whose functions include: prevention and abatement of stream pollution, planning and implementation of rehabilitation schemes, and transformation of the area into a recreational site.
1997	Prevention of Water Pollution – Gasoline Stations, Regulations, 1997	Requires specific conditions for the establishment and operation of gas stations, including installation of fuel-water separators, use of impermeable construction materials, special measures and equipment to prevent leakage and oil pollution, measures for protection against corrosion, and monitoring equipment and procedures.
1997	Prevention of Water Pollution – Evaporation and Storage Ponds, Regulations, 1997	Aims to prevent water pollution from evaporation and collection (storage) ponds, on the one hand, and restricting their use, on the other.
1998	Prevention of Water Pollution – Prohibition on Discharge of Brines to Water Sources, Regulations, 1998	Prohibits the discharge of brines from ion-exchange renewal, from food, tanning and textile industries, and from hospitals to water sources and the municipal sewage system.
1998	Prevention of Water Pollution – Sewage Disposal from Vessels, Regulations, 1998	Prohibits the discharge of sewage from a vessel to a water source, requires commercial vessels to install adequate sewage collection facilities, and calls for the establishment of adequate reception facilities on shore.
2000	Prevention of Water Pollution – Metals and Other Pollutants, Regulations, 2000	Aims to protect water sources from heavy metals and other pollutants by limiting the volume of wastewater discharged from pollution sources and reducing the concentration of pollutants in it.
2001	The Water and Sewage Association Law, 2001	Increases efficiency of municipal water supply and sanitation services via public service entities called 'Water and Sewerage Corporations'.
2003	Prevention of Water Pollution - pH Values of Industrial Sewage, Regulations, 2003	Sets pH values of industrial sewage in order to protect the environment and prevent the pollution of water sources from the corrosive impacts of industrial sewage.
2003	Salt Concentrations in Industrial Sewage, Regulations, 2003	Sets threshold values for salt concentrations in industrial sewage.
2004	Prevention of Water Pollution – Usage of Sludge, Regulations, 2004	Aims to prevent water source pollution and environmental degradation as a result of improper disposal of sludge originating in municipal sewage treatment plants.
2006	Prevention of Water Pollution – Fuel Pipelines, Regulations, 2006	Reduces potential risks from fuel transport pipelines, thereby preventing environmental degradation and pollution of water sources.
2010	Effluent Quality Standards and Rules for Sewage Treatment, Regulations, 2010	Aims to protect public health, prevent pollution of water sources from sewage and effluents, facilitate the recovery of effluents as a water source, protect the environment, including ecological systems and biological diversity, soil, and agricultural crops.
2011	Prevention of Water Pollution – Wastewater Conveyance System, Regulations, 2011	Aims to prevent leaks from wastewater conveyance systems in order to protect water sources, ecosystems, biodiversity, and other natural resources and prevent environmental hazards, inter alia, by imposing charges and issuing directives in accordance with the provisions of these regulations.

TABLE 5: Classification of sludge and various materials according to the Water Regulations (2004).

Definition	Description
"Sludge"	A by-product of a sewage treatment process in a sewage treatment plant (except in a process in which crude filtering and separation of sand and oils is carried out)
"Stabilized Sludge"	Sludge that has undergone treatment according to a plan approved by the Ministry of Environmental Protection
"Class A Sludge"	Stabilized sludge that satisfies the following requirements: (1) The geometric mean of the density of faecal coliform type bacteria, determined from at least seven samples of the sludge, is less than 1000 MPN per one gram of dry material or the arithmetical mean of salmonella bacteria, determined from at least seven samples of the sludge, is less than 3 MPN per four grams of dry material (2) The arithmetical average of enteric viruses determined from at least seven samples of the sludge is less than one PFU per four grams of dry material (3) The arithmetical average of density of viable helminth ova determined from at least seven samples of the sludge is less than 1 to four grams of dry material, provided that the sampling was conducted in accordance with the method prescribed in Book 3 and explained in Book 4
"Class B Sludge"	Stabilized sludge in which the geometric average of the density of faecal coliform type bacteria determined according to at least seven samples is less than two million MPN or CFU per one gram of dry material
"Dry Material"	Material obtained after drying of sludge at a temperature of 105 degrees centigrade by the method prescribed in Book and explained in Book 4
"Volatile material"	Material found in sludge that evaporates after heating of the dry material at a temperature of 550 degrees centigrade, in the presence of oxygen, according to the method prescribed in Book 1 and explained in Book 4
"Total nitrogen"	The arithmetical amount of concentrations of Kjeldahl nitrogen, N- nitrite and N- nitrate according to the methods described in Book 1

In 2016, 118,019 tons of sludge were disposed of from 63 WWTPs. Thirty-three percent of this amount was discharged into the Mediterranean Sea and 67% was removed to land destinations as presented in Figure 1 (MoEP, 2017). In 2016, most of the sludge that was removed to land-based destinations (which did not flow into the sea) was used for agricultural purposes, after it passed additional sanitary processing and turned into fertilizer/soil enhancement for unlimited use ("Class A Sludge" in accordance with the regulations – see Table 5). The trend of sludge disposal from WWTPs between 2002 and 2016 is presented in Figure 2.

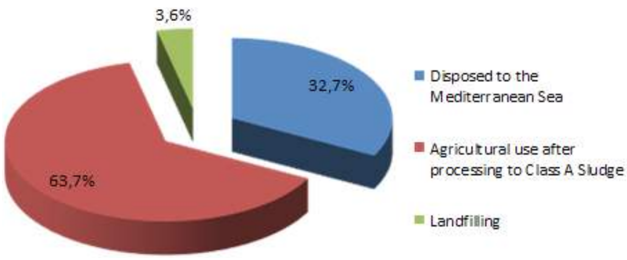

FIGURE 1: A diagram of sludge disposal from WWTPs for 2016 (MoEP, 2017).

3.2.4 Effluent management legislation

As water scarcity is a major concern, Israel has introduced ambitious water policies and pioneered cutting-edge water-efficient technologies, including drip irrigation, brackish and seawater desalination, and soil aquifer treatment for reuse of treated wastewater.

In Israel, the local authorities are responsible for the construction and operation of wastewater treatment plants. Israel's wastewater treatment plants use intensive (mechanical/biological) and extensive treatment processes. From a total of 500 million cubic meters (MCM) of sewage produced in Israel in 2008, about 70% of the effluents were reclaimed. Local authorities are responsible for the treatment of municipal sewage. In recent years new or upgraded intensive treatment plants have been set up in municipalities throughout the country. The ultimate objective is to treat 100% of Israel's wastewater in order to bring it to a level that enables unrestricted irrigation in accordance with soil sensitivity and without risk to soil and water sources (MoEP, 2014). The effluent quality and wastewater treatment regulations issued by the Ministry of Environmental Protection (MoEP) and the Ministry of Health in 2010 include 36 parameters that may not be exceeded in effluent whose use in irrigation will be unrestricted or that will be discharged to rivers. Sewage treatment effluent is the most readily available water source and provides a partial solution to the water scarcity problem. Table 6 presents

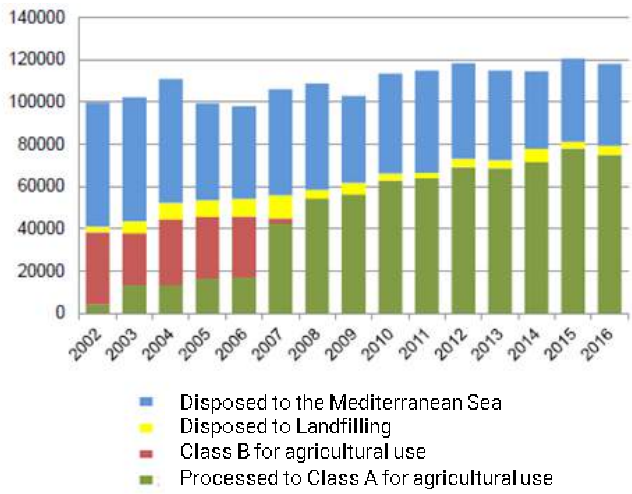

FIGURE 2: A diagram of sludge disposal from WWTPs between 2002 and 2016 (MoEP, 2017).

the restrictions on the use of effluents.

4. DISCUSSION

This comparative survey highlights some profound differences between Italy and Israel in the approaches and in legislation concerning the recycling of bio-waste and the management of sludge and liquid fractions generated from

TABLE 6: Israeli standards for effluent (average levels) (MoH, 2010).

Parameter	Units	Unrestricted Irrigation	Rivers
Electric conductivity	dS/m	1.4	n/a
BOD	mg/l	10	10
TSS	mg/l	10	10
COD	mg/l	100	70
N-NH4	mg/l	20	1.5
Total nitrogen	mg/l	25	10
Total phosphorus	mg/l	5	1.0
Chloride	mg/l	250	400
Fluoride	mg/l	2	n/a
Sodium	mg/l	150	200
Faecal coliforms	Unit per 100 ml	10	200
Dissolved oxygen	mg/l	>0.5	>3
pH	mg/l	6.5-8.5	7.0-8.5
Residual chlorine	mg/l	1	0.05
Anionic detergent	mg/l	2	0.5
Mineral oil	mg/l	n/a	1
SAR	(mmol/l)0.5	5	n/a
Boron	mg/l	0.4	n/a
Arsenic	mg/l	0.1	0.1
Mercury	mg/l	0.002	0.0005
Chromium	mg/l	0.1	0.05
Nickel	mg/l	0.2	0.05
Selenium	mg/l	0.02	n/a
Lead	mg/l	0.1	0.008
Cadmium	mg/l	0.01	0.005
Zinc	mg/l	2	0.2
Iron	mg/l	2	n/a
Copper	mg/l	0.2	0.02
Manganese	mg/l	0.2	n/a
Aluminium	mg/l	5	n/a
Molybdenum	mg/l	0.01	n/a
Vanadium	mg/l	0.1	n/a
Beryllium	mg/l	0.1	n/a
Cobalt	mg/l	0.05	n/a
Lithium	mg/l	2.5	n/a
Cyanide	mg/l	0.1	0.005

AD. Figures 3a and 3b schematically present the Italian and Israeli procedure for bio-waste recycling via AD, respectively.

This comparison suggests three main differences between the EU legislation (Italy) and the Israeli legislation:

1) Source of the bio-waste, which has to be collected separately (for Italy and EU);
2) Quality of the digestate in terms of physical, chemical, and biological features;
3) Quality of the soils receiving the digestate, mainly in terms of heavy metals content.

In Italy, if one of the last two steps are not verified, the liquid has to be processed in a WWTP, resulting in a failure of bio-waste recycling (Figure 3a). This approach arises from two main factors: the implementation of the waste management hierarchy and the absence of EoW criteria for the digestate. According to EU legislation, the goal of the hierarchy is to make the best possible use of the waste materials for replacing and/or avoiding the consumption of raw materials. Pursuing this goal in the bio-waste sector means effective use on land of its organic nutrients content, e.g. N, K, P, for replacing mineral ones avoiding the consumption of mineral resources. On the other hand, there are currently no defined criteria specifying when the status of the bio-waste changes from waste to product (i.e., EoW criteria). This means that the use on land of digestate is not really forbidden, but it is necessary to activate an alternative legal procedure for assessing whether or not this activity can be performed. The legal pathway for doing this is stated in article 6 of the WFD 2008/98/EC, which enumerates the general mandatory criteria for the end-of-waste status: (a) the substance or object is commonly used for specific purposes; (b) a market or demand exists for such a substance or object; (c) the substance or object fulfils the technical requirements for the specific purposes and meets the existing legislation and standards applicable to products; and (d) the use of the substance or object will not lead to overall adverse environmental or human health impacts. The absence of a uniform EU legal support leads local and member state legal authorities to adopt procedure from similar legislation that in the specific case are usually represented by the current one concerning the agronomic use of sludge from WWTPs (Figure 3a). Of course, the chemical, physical, and biological features of the WWTP sludge are significantly different from those of the digestate from bio-waste. One main reason for concern is the risk of pollutant compounds, such as heavy metals, that can be quite high in sludge (Table 1). The reason for these concerns arises from the impossibility of having stringent control over the quality of the wastewater collected by the WWTP. In fact, sewage grids are usually mixed systems that collect domestic, commercial, and industrial sewage that, depending on different context, can significantly affect the quality of the sludge. Furthermore, sewage grids also collect rainwater from public roads and parking areas, and thus bring to the WWTP large amounts of pollutant compounds.

If, on the one hand, the EU and Italian legislation is strongly oriented toward the implementation of the waste management hierarchy, on the other hand it shows some weaknesses regarding the implementation of an efficient management of water resources. In the current EU legislation, which imposes an efficient use of water resources, unlike the waste management sector, no reuse/recycling targets were defined. In addition, in this case the lack of legal and political framework limits the achievement of high performances in this sector.

Yet, as Figure 3b shows, in Israel the approach is different. The national goal is the reduction of landfilling via recycling and thus, a landfill levy has been imposed since 2007. AD in this case is considered a suitable technology for the recovery of bio-waste via the production of a soil amend-

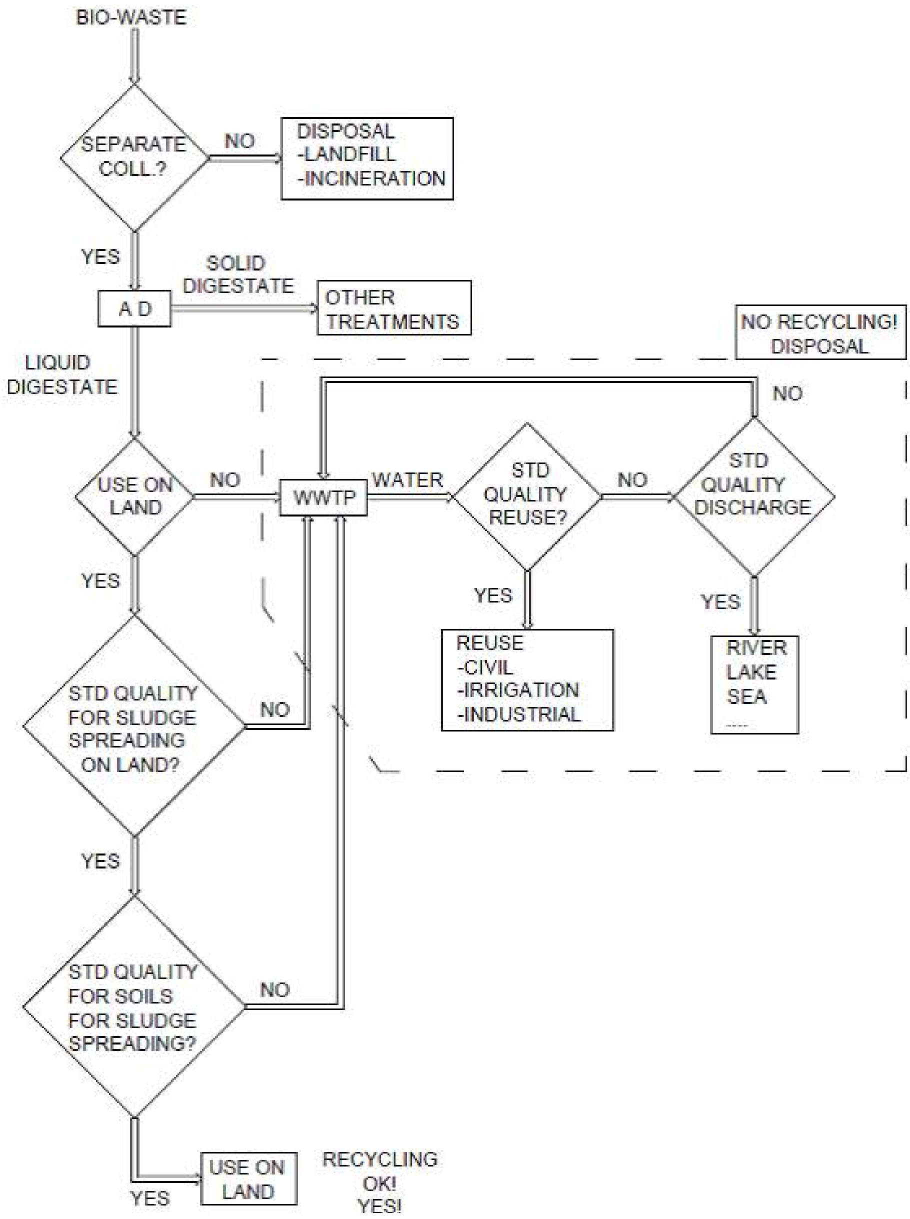

FIGURE 3a: Scheme of the technical and legal recycling pathway of bio-waste via AD in Italy.

ment and/or fertilizer. However, Israeli legislation is not focused on the source of the material, as demonstrated by the absence of mandatory rules on the source separation of bio-waste. Moreover, AD is not specifically indicated by national policy as a suggested technology and its application is left to the choice of local authorities and municipalities. AD of bio-waste is thus an optional recycling method, performed mainly to reduce landfilling and increase recycling, based on the availability of adequate treatment facilities. Additionally, in Israel, unlike in Italy, bio-waste can be landfilled without any pre-treatment and there are no EoW criteria. Concerning the other two aspects related to quality of digestate/sludge and the soils that receive them, some other differences and similarities exist. In fact, both in Italy and Israel there are policies for assessing the quality of the digestate/sludge, with some differences.

The use of solid digestate (sludge) and liquid digestate (effluent) in Israel are well regulated, since there is concern regarding the possible effects of the use of these products on soil and water quality. On the other hand, Israeli legislation concerning sludge (i.e., the solid part) is more oriented toward pathogen content (see Table 5) and less focused on other potential pollutant risks, such as those caused by nitrogen. Italian and EU legislation also interact with water protection legislation where limits on the concentration of different nitrogen and other compounds (e.g., P) are also carefully addressed. Finally different approaches to the quality of receiving soils can be also detected (Tables 1,5).

Concerning the standard quality of water generated by WWTP of liquid digestate effluents (Tables 3,6), it is possible to note that, in general, Israeli standard quality is more stringent for discharge in rivers than for reuse. In contrast, the Italian and EU approach imposes more stringent limits for reused water in particular with regard to the main

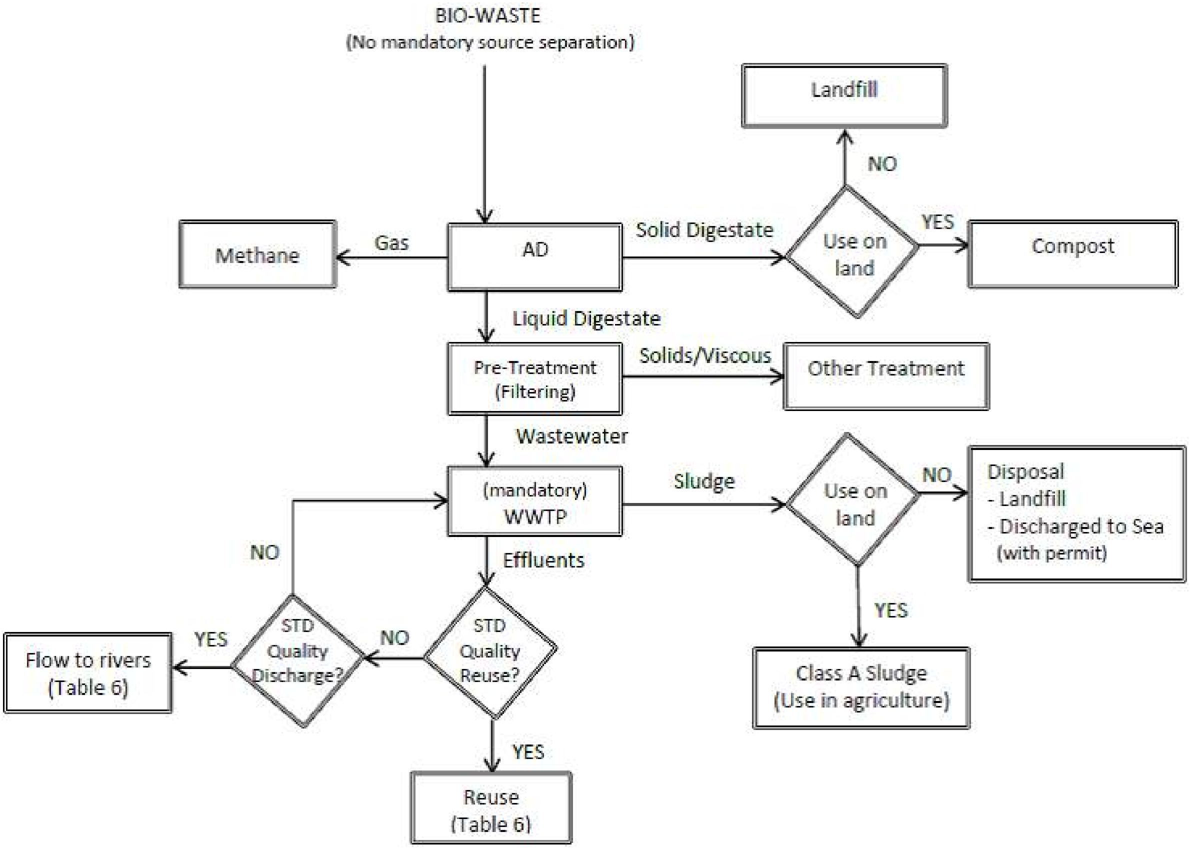

FIGURE 3b: Scheme of the technical and legal recycling pathway of bio-waste via AD in Israel.

parameters and COD, N and P. This difference highlights the priority Israeli legislation gives to the reuse of water, as opposed to Italy and EU where currently this priority is stated, but not fully addressed.

5. CONCLUSIONS

The comparison presented in this paper points out some profound differences between regulation in Italy and Israel and raises some substantive issues:

1. The main differences that emerge from this study are the following:
   I. Bio-waste definitions are different – the EU regulation addresses the source of the bio-waste, which dictates the product's destination, and thus in Italy the source of the bio-waste is the decisive factor that dictates its usage and destination. In Israel, on the other hand, the regulation relates to the receiving media of the products (soil/ agriculture use/rivers).
   II. In order to recycle bio-waste, separation at source is mandatory in Italy, as the EU is concerned about contamination and possible health hazards of the recycled materials (e.g., compost and reclaimed water) whilst in Israel separation may be implemented at the end point based on technologies such as MBT.
   III. The Italian legislation includes EoW criteria that apply inter alia to bio-waste only regarding its solid part, whereas in Israel such criteria are not yet anchored in legislation.
   IV. Even if largely promoted by EU legislation, full implementation of AD of bio-waste in Italy, as in the UE, suffers from the absence of uniform EoW for the digestate.
2. In order to "close the loop" of bio-waste via bio-waste recycling according to the EU legislation, separate collection must be of a very high quality. This requirement might be an obstacle in achieving the EU recycling goals, so further research should be implemented in order to determine whether the EU's strict legislation, which requires source separation, is indeed a must, or whether separation in an advanced sorting facility (MBT) is sufficient for further treatment in an AD facility.
3. The differences that arise from the comparison in this paper emphasize the crucial role of regulation and legislation. We conclude that adequate legal support is crucial for achieving sustainable systems.
4. Elaborating this comparison and further analysing the regulation and management systems of both countries may make it possible to enhance the wastewater cycle in a way that will contribute to the advancement of sustainable wastewater treatment systems in both countries, taking a step forward towards a circular economy.

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OPTIMIZATION OF THE PYROLYSIS OIL FRACTION: AN ATTAINABLE REGION APPROACH

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ABSTRACT
In this research, we focused on the application of the attainable region method as an optimization technique that has never before been used in the field of waste management, specifically in optimising the pyrolysis oil fraction from plastic waste. Experimental results obtained from this investigation showed that the optimum yield of the oil fraction from the Attainable Region (A.R) plot was found to be 95%, using a pyrolysis temperature of 450°C and after a residence time of 2 hrs. Within the conditions of experimental investigation, it was also determined that the optimum conversion attainable is 70% and this was achieved using a pyrolysis temperature of 450°C and a residence time of 2 hrs.

1. INTRODUCTION

Traditionally, the production of liquid fuels, organic chemicals and energy, has relied heavily on the gasification of coal, distillation of crude oil and hydro-power. As natural resources are becoming depleted, and continue to do so due to the exponential growth of our population, the need to look elsewhere for sources of alternative raw materials and processes is imperative.

Domestic and industrial plastic solid waste (PSW) collection and disposal systems in Botswana are not as efficient as those in other developing African countries like South Africa, Ghana and Ethiopia. Specifically, the mode of disposal of PSW in smaller cities and villages in Botswana is a major concern. Even in bigger cities, the current waste management systems mostly entail use of dumping sites and landfills, where the majority of the waste (Al-Salem et al. 2010) has been identified to be different classes of polymers namely: high density polyethylene (HDPE), low density poly ethylene (LDPE), poly vinyl chloride (PVC), poly propylene (PP), poly styrene (PS) and poly ethylene terephthalate (PET) (Singh et al. 2017)

Although the rate of reaction associated with the biodegradation process of plastic waste is said to be slow that it is kinetically approximated not to be taking place (Guerrero et al. 2013), it is however thermodynamically feasible. Some waste management researchers (Shah et al. 2008; Das & Tiwari, 2017 Hahladakis et al. 2018) have argued that pigments used in plastics as well as plastics can be broken down by acidic leachates under certain conditions of temperature (450°C) and pressure (2 kPa) to give products that are both deleterious and obnoxious to the environment and to the underground water sources which in some areas serve as drinking water for livestock, wildlife and even the rural folk.

Due to these highlighted PSW issues, the Botswana Government has responded to this societal challenge by implementing its environmental policies and inviting some NGOs and private companies on-board in assisting to combat this waste situation. But this approach has not yielded any positive results in villages like Palapye, Pilikwe, Serule and smaller towns like Serowe, Selibe Phikwe and Maun. In these parts of the country, PSW will continue to increase with negative implications on the local municipalities and environment. The major challenge in these locations is mainly the state of the roads, which make it practically impossible for refuse collection trucks to access the domestic and industrial waste. The finances usually allocated to the local municipalities by the central government in order to run their towns is also a constraint as it is never enough to address the water supply, sanitary, accommodation and other service delivery challenges, let alone cater for the

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waste collection and disposal services. Even at a government level, rarely is waste treatment a priority as there are more pressing national issues such as electricity supply, food security, agricultural inputs and mining challenges to take care of.

In this research, we focused on the application of the attainable region method as an optimization technique that has never before been used in the field of waste management, specifically in optimising the pyrolysis oil fraction from plastic waste.

1.1 Theoretical background

Botswana is a major market for most South African manufactured products. This is mainly due to the fact that the manufacturing industry in Botswana is hardly existing. This situation poses challenges on proposals of how to recycle PSW as there are no local plastic industries that will process the recycled material. In neighbouring countries such as Zimbabwe and South Africa, there are plastic processing companies like Megapak and Nampak, respectively. These companies have led to the development of downstream recycle companies that supply them with recycled material to be used as feed. These downstream companies have also created a lot of jobs for local people who are willing to collect the PSW.

We propose that PSW should be collected from the villages and processed into finished products that are of economic value to the same villagers. There are four generally acceptable and well developed techniques used to process PSW, namely gasification, hydrogenation, biodegradation and pyrolysis (Marshall & Farahbakhsh, 2013). For this research, we chose to use a batch pyrolysis method because of its simplicity in design, operation and relative ease in adjusting the experimental parameters.

1.2 The pyrolysis process

The pyrolysis process starts with higher molecular chain hydrocarbons (polymers) that are then broken down (cracking) using either heat, a catalyst or hydrogen gas (Al-Salem et al. 2017) into targeted smaller chain hydro-carbons such as pyrolysis oil, tar and char. Plastics, tyres and even coal can be used as feed material to a pyrolysis plant. In general, this process of waste treatment has added advantages over traditional approaches such as incineration in that products of socio-economic value are obtained. Charcoal, tar, oil, wax and combustible gases are some of the products of pyrolysis (Chen et al. 2014) and process parameters can be optimised for maximum production of either the solid, liquid or gaseous products.

Since pyrolysis is basically a thermal degradation process that occurs in the absence of oxygen, most researchers conducted their pyrolysis studies at atmospheric pressure and focused more on the temperature factor (Sharuddin et al. 2016). Other researchers (Martínez et al. 2013; Das & Tiwari, 2018) focussed on the rate aspect associated with pyrolysis, in which they proved that the product composition and yield are a function of pyrolysis time. The other factors that affect the efficiency of a pyrolysis process are pressure, type of reactor, temperature, residence time and cooling mechanisms (Sharuddin et al. 2016). Investigations on the development and effect of different types of reactors on pyrolysis were reported by Sannita et al., 2012 while Quek and Balasubramanian (2013) reported on the oil characteristics and upgrading techniques.

1.3 Batch reactor design

It is generally accepted that the pyrolysis process follows a first order irreversible reaction model, whose integral form (equation 1) is used here to determine the optimum residence time.

1.3.1 Conversion and reactor sizing

$$-r_A = N_{A_o}\frac{dX_A}{dt} \quad \left({}^{mols}/_{s}\right) \tag{1}$$

Where:

N_{A_o} is the initial number of moles of the reactant A
dX_A is the change in conversion of reactant A
dt is the change in time
$-r_A$ is the rate of reaction with respect to reactant A
V is the volume of the reactor

In the integral form:

$$dt = \frac{N_{A0}}{V}\frac{dX_A}{r_A} \tag{2}$$

Since the reactor was for a constant volume, the equations above reduces to:

$$t = \frac{N_{A0}}{V}\int_{X_{A0}}^{X_A}\frac{dX_A}{-r_A} \tag{3}$$

The pyrolysis of plastics is defined by an irreversible uni-molecular first order reaction of the form ($A \rightarrow products$):

$$-r_A = -\frac{dC_A}{dt} = kC_A \tag{4}$$

Where:
k is the rate constant

The equation simplifies to:

$$-\ln(1 - X_A) = kt \tag{5}$$

The kinetic parameters of pyrolysis under isothermal conditions as given by (Khaghanikavkani & Farid, 2011) are:

$\mathrm{Ea}=164.15\mathrm{kJ}/mol$
$K_0=4.89 \times 10^7/sec$

The rate constant is therefore calculated from the Arrhenius equation:

$$k = 4.89x\ 10^7 e^{-\left(\frac{164.15\ x\ 10^3}{(8.314)}\right)}$$
$$= 1.8\ x\ 10^{-4}/sec$$

The residence time was the calculated from the integrated rate law equation for a first order reaction with a conversion of 70%.

$$-\ln(1 - X_A) = kt \tag{6}$$
$$-\ln(1 - 0.7) = 1.8\ x\ 10^{-4}(t)$$
$$t = 1.85\ hours\ \cong 2\ hrs$$

2. MATERIALS AND METHODS

In preparing the feed sample that was used in this investigation, 80 kg of plastic waste was collected from the

Palapye landfill site, in Botswana. This waste was then grouped into the six different classes of plastics namely: HDPE, LPDE, PVC, PET, PP and PS using their resin identification number (RIN). Other different classification techniques that could have been used, if resources permitted include froth flotation, laser induced breakdown spectroscopy and the X-ray fluorescence spectrometer. The next step in the experimental procedure was to combine the HDPE and LDPE fractions together before taking the mixture to the laboratory scale plastic shredder for size reduction. The standard particle size analysis method was then carried out on a 100g cone and quartered shredded sample, using a stack of sieves arranged in the route mean square technique and having a top size of 2400 μm, bottom size of 300μm placed over a pan and electric shaker for 20 mins. After a particle size distribution (PSD) analysis, the experimental setup was arranged as shown in Figure 1. A measured 1000g sample of the size reduced and homogenised material was then introduced into the batch reactor, heated using an electric coil for periods ranging between 30-150 mins at temperatures ranging between 300-500°C. The system was pressurized to 2 bars by closing the outlet valve while heating the material. Using a specific pyrolysis temperature e.g. 300°C and at the end of each residence time, the mass of material remaining unconverted in the reactor was measured and a conversion value evaluated. Also at the end of each residence period the outlet valve was open to allow the flow of condensable vapour through to the condenser where it was condensed to liquid, collected, measured and the yield with respect to the liquid fraction evaluated. The condensates were collected, stored in glass jars and taken to the chemical engineering laboratory at the University of South Africa for chemical analysis using a capillary gas chromatography-mass spectrometer (GC-MS-QP2010 SE). A total of 25 samples obtained from the five pyrolysis temperatures and five residence times used, were sent for analysis. Three determinations were made for each sample and an average result reported (Table 1).

Figure 1, is a pictorial view of our batch pyrolysis process, showing the heating and condensation mechanisms. The process is simple and made from materials locally available at the University and in local communities. One of the main objectives behind setting up this pilot unit was to produce samples of the pyrolysis oil in order to be eligible for funding from the government, as this is a pre-requisite.

2.1 The Attainable Region Approach

In this article, we applied the Attainable Region (AR) optimization technique to optimize the objective function by way of manipulating the input variables in order to give result to maximum process outputs. The objective function was to maximise the conversion as well as the yield of the pyrolysis oil fraction. The AR method is a modern day geometric optimization technique that has been used successfully in the different disciplines of chemical engineering. This approach owes its origins to the field of chemical reaction engineering where Hildebrandt and Glasser (1990) tested it in choosing optimal reactor configurations. Over the years, different researchers (Katubilwa et al. 2011; Danha et al. 2015; Hlabangana et al. 2018) have applied this optimization method on their laboratory scale data with the aim of either minimizing an experimental manipulated variable or maximizing an associated process variable. Since one of the objectives of operating any process is to make profit, the AR technique assists in this regard by way of specifying optimal experimental parameters that will result in either a maximum or minimum condition of the objective function.

The greatest advantage of the AR method is its versatility. The versatility of the approach lies in that it is generic across the field of chemical engineering and a researcher can apply this technique on any process parameter of choice. Smith and Malone (1997) also applied the technique in organic industrial chemistry where they optimized the molecular weights, monomer conversions and residence time in isothermal polymerization systems. In 1998, McGregor et al. used the geometric ideas of the AR method in process synthesis in which they optimized a reactor-separator-recycle system. Godorr et al. (1999) extended the application of the technique in selecting optimal control and operating policies to situations where the rate vector depends on a control parameter. In the year

TABLE 1: Description of the sample (with respect to the year of the provider change).

Parameter	Specification
Batch reactor volume (litres)	100
Batch charge (g)	1000
Residence times (mins)	30; 60; 90; 120; 150
Pyrolysis temperatures (°C)	300; 350; 400; 450; 500
Reactor pressure (bars)	2

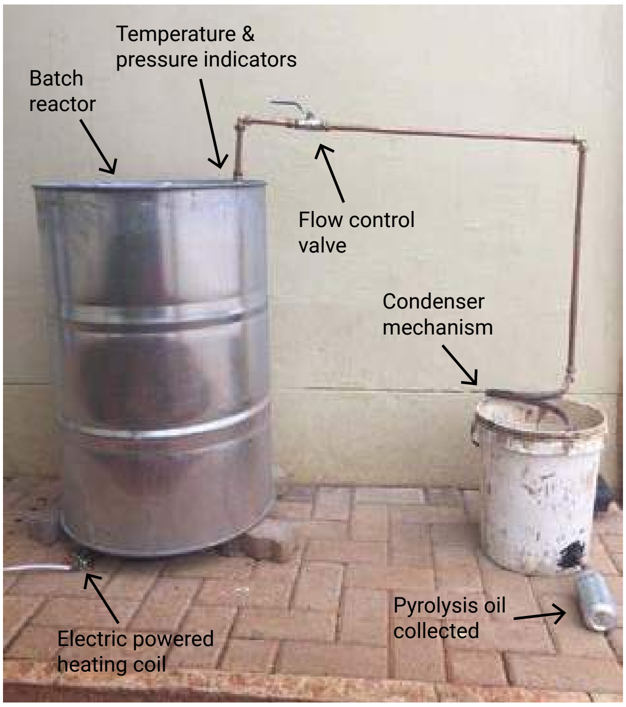

FIGURE 1: Experimental set-up.

2000, Book and Challagulla used the technique in order to obtain optimal design and operating conditions for the adiabatic oxidation of sulfur dioxide to sulfur trioxide. Nicol et al. (2001) applied the technique in order to find an optimum process design for an exothermic reversible reaction system where provision was made for an external heating and cooling source. Over the years, the technique has been further modified and employed in various fields of process engineering.

3. RESULTS AND DISCUSSION

Figure 2 shows a PSD plot of the feed material in the form of a cumulative mass percentage passing versus the sieve size. This plot reveals that the size of the material used as batch feeds were less than 2400μm in size. The size reduction process was performed for a number of reasons namely to: improve handling issues; enable homogeneous mixing of the feed material; enhance packing of material within the reactor; and increase the surface area for heat to act on the material.

Figure 3 shows an attainable region plot of conversion versus pyrolysis time for different pyrolysis temperatures. The region bounded by the curves and the x-axis is termed the wanted or attainable region and contains combinations of conversion and residence time. The boundary curve offers solutions to the optimisation problem relating to conversion. By applying this technique, Figure 3 reveals that within the conditions of experimental investigation, the optimum conversion attainable is 70% and this was achieved using a pyrolysis temperature of 450°C and a residence time of 2 hrs. All the other pyrolysis temperatures gave a maximum value that is less than that recorded for the 450°C case. Generally, these experimental results indicate that conversion increased with temperature, and there was an optimum temperature that gave result to maximum

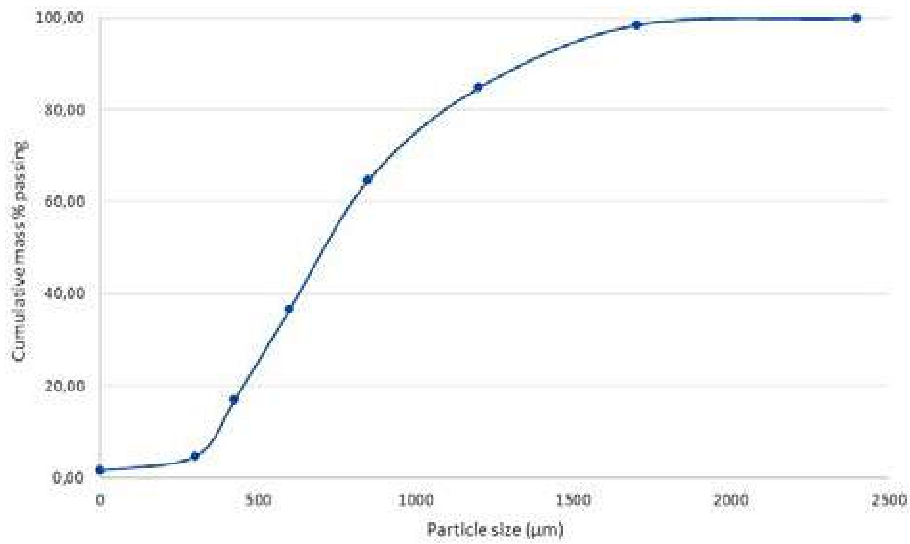

FIGURE 2: Feed particle size distribution.

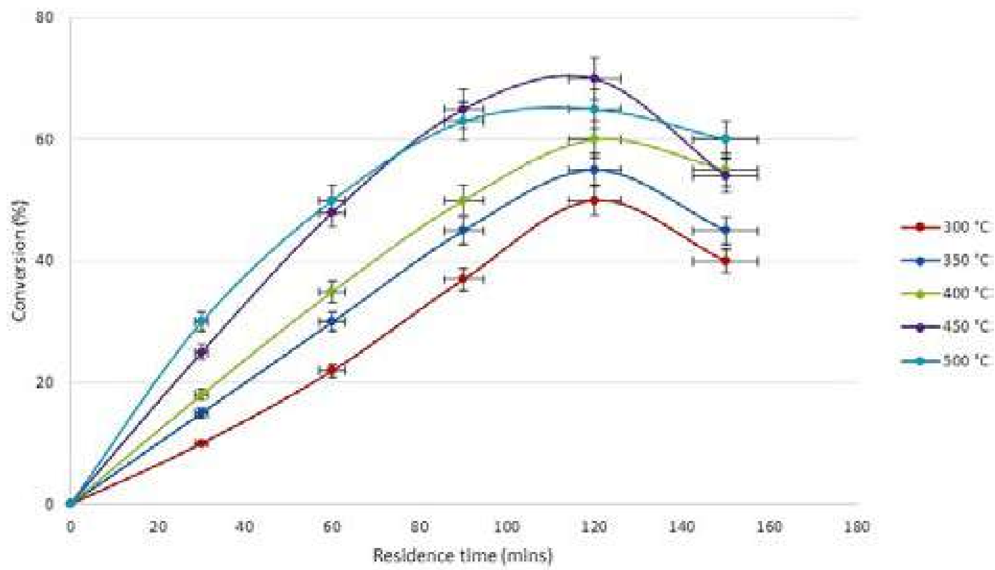

FIGURE 3: Attainable region plot of conversion versus residence time.

conversion. When temperature was increased beyond this optimum point, there was a decrease in conversion mainly due to a shift in the position of equilibrium towards the reactants.

Figure 4 shows the attainable region plot for yield of the pyrolysis oil fraction versus pyrolysis time for different pyrolysis temperatures. Again, the region bounded by the curves and the x-axis denote the wanted or attainable region. Within this space lies different combinations of yield and residence time for the different pyrolysis temperatures. Solution to the yield optimization question lies on the boundary of the curves, specifically at the turning point. Percentage yield of the oil fraction was calculated by taking a molar ratio of the amount of oil produced to the amount of feed material converted multiplied by a unit stoichiometric factor and 100%. Experimental results reveal that the optimum yield of the oil fraction from the A.R plot was found to be 95%, using a pyrolysis temperature of 450°C and a residence time of 2 hrs. Figure 4 also shows that the molecular vibrations are directly proportional to temperature therefore at higher temperatures the molecular vibrations are increased. The increase in molecular vibrations caused the bonds holding the longer molecules to break into shorter molecular chains (solid to liquid then vapour state). Hence increasing temperature up to 450°C increased yield of the oil fraction, but increasing beyond this optimum value resulted in a decrease in the yield of the oil fraction as a result of the increase in the production of the non-condensable gas fraction.

Where truncated sample results are shown in Table 2. Figure 5 and Table 2 show the qualitative and quantitative capillary GC-MS-QP2010 SE analysis results of the pyrolysis oil fraction (density = 0.7 g/cm^3, viscosity = 1.8 mm^2/s) obtained at the experimentally determined optimum conditions of pyrolysis time of 2 hrs, pyrolysis temperature of 450°C, and conversion of 70% resulting in a 95% yield with respect to the oil fraction.

The capillary GC-MS was an ideal technique to use in determining the volatile and semi-volatile mixture of components that made up the pyrolysis oil. The GC-MS is an instrument that combines two separate techniques namely, the gas chromatography (GC) and the mass spectrometry (MS). The GC part separates the volatiles but can-not always selectively detect them, while the MS part can selectively detect compounds but can-not always separate those (Sneddon et al. 2007).

Figure 5 is a plot of signal intensity, which is the relative abundance of the components versus the molecular weight. Each component in the pyrolysis oil has a unique identification parameter linked to its organic molecular mass. Using a software impeded in the instrument, the peaks from Figure 5 were then matched against a database of spectra for known compounds and each peak was then identified as shown in Table 2.

Figure 5 and Table 2 reveals that there are many different liquid components obtained by the co-pyrolysis of HDPE and LDPE classes of Plastics. This high number of organic components obtained could have resulted from the fact that different pigmented plastics were used as feed material.

A possible way of minimising the number of peaks would have been to pyrolysize either HDPE or LDPE samples separately as well as sorting feed material in terms of colour of their pigments.

4. CONCLUSIONS AND RECOMMENDATIONS

From our experimental results it can be concluded that the optimum yield of the oil fraction from the A.R plot was found to be 95%, using a pyrolysis temperature of 450°C and after a residence time of 2 hrs. Sharuddin et al. (2016) reported similar results in their review article in which they were summarising studies done on plastic pyrolysis. It was also determined that within the conditions of experimental investigation, the optimum conversion attainable is 70% and this is also achieved using a pyrolysis temperature of 450°C and after a residence time of the material in the

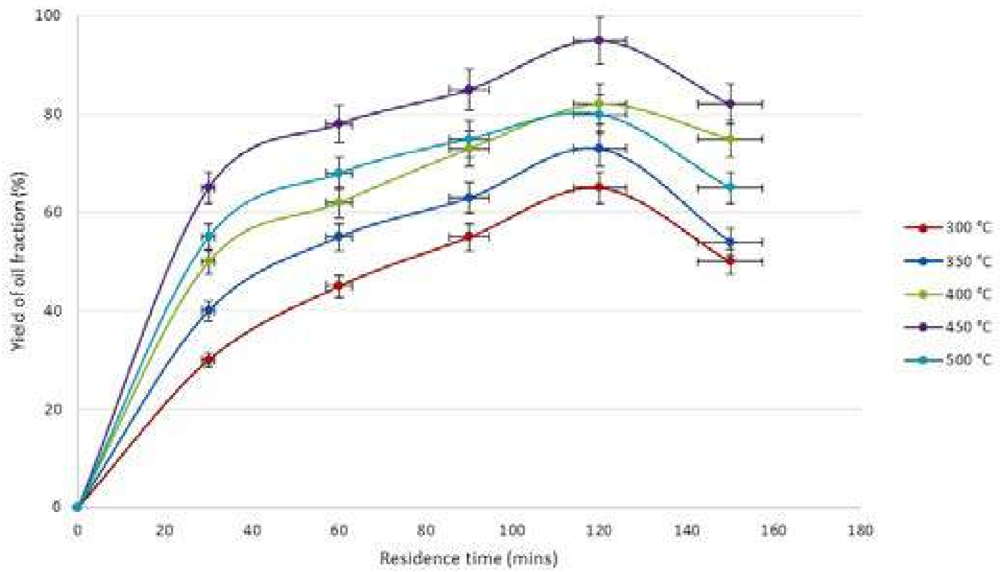

FIGURE 4: Attainable region plot for yield of oil fraction versus residence time.

TABLE 2: Truncated results of the GC-MS analysis.

Peak #	Name	R.T. (min:sec)	Weight	Similarity	Unique Mass
169	1,2-Di(prop-2-ynyl)cyclohexane	10:37.9	160	608	117
244	Cyclopentanecarboxylic acid, 2-amino-, cis-	19:42.8	129	610	56
76	Decane, 2,2,3-trimethyl-	04:01.3	184	610	72
203	2-Octene, 2-methyl-6-methylene-	13:31.9	138	620	109
274	Oxalic acid, hexyl octadecyl ester	26:25.8	426	624	153
32	Dodecylcyclohexane	02:08.9	252	625	89
113	2-Heptyne-4-one	06:44.4	110	629	68
27	1-Octyn-3-ol, 3-methyl-	02:00.9	140	634	88
37	Cyclobutene, 3,3-dimethyl-	02:15.9	82	635	65
213	7-Heptadecyne, 17-chloro-	14:54.8	270	638	117
38	Cyanic acid, 2-methylpropyl ester	02:17.3	99	640	60
171	Fumaric acid, 3-phenylpropyl tridec-2-yn-1-yl ester	10:49.5	412	640	117
30	4-t-Butylcyclohexylamine	02:05.6	155	646	105
153	4-Benzoyloxy-1-morpholinocyclohexene	09:13.0	287	651	110
47	2-Butyn-1-ol	02:29.8	70	652	95
70	Spiro[2.5]octane	03:38.8	110	654	110
114	2-Propenoic acid, 2-methyl-, oxiranylmethyl ester	06:45.3	142	658	107
231	Cyclohexane, 1-bromo-2-methyl-	17:51.9	176	659	97
178	Glycine, furfuryl ester	11:24.1	155	662	97
5	2-Propenoic acid, oxiranylmethyl ester	01:31.2	128	664	45
191	2,7-Octadiene-1,6-diol, 2,6-dimethyl-, (E)-	12:31.6	170	665	71
192	2-Pentene, 4-bromo-	12:43.1	148	667	253
44	2-Butanone, 1-(2-furanyl)-3-methyl-	02:24.8	152	670	86
50	(7S,8R)-7-Hydroxy-8-amino-trans-anti-trans-tricyclo[7.3.0.0(2,6)] dodecane	02:34.0	195	672	96

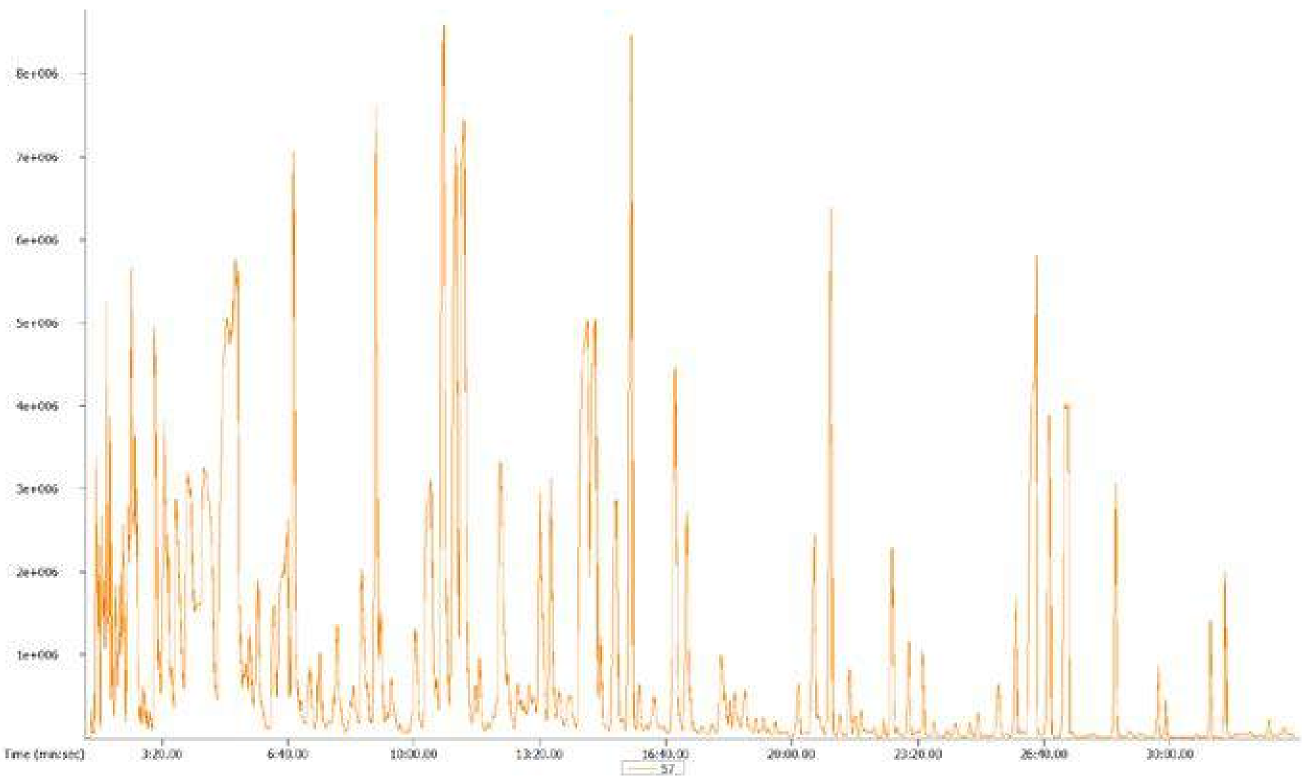

FIGURE 5: GC-MS results of the pyrolysis oil fraction.

reactor of 2 hrs. Hence the objective of optimizing the oil fraction was achieved as we managed to employ the AR technique in specifying the optimal pyrolysis temperature and conversion. Even though the investigation was successful there are possible areas of improvement that we recommend in order to improve the efficiency of the process. A catalyst should be used to improve the degradation process and lower the degradation temperature. As a general rule of thumb in the field of analytical chemistry, no single analytical technique should be used to give comprehensive results in any investigation, we therefore recommend that future studies should include complementary analysis techniques such as the nuclear magnetic resonance spectroscopy (NMR) for completeness.

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KINETIC PARAMETERS OF TORREFACTION PROCESS OF ALTERNATIVE FUEL PRODUCED FROM MUNICIPAL SOLID WASTE AND CHARACTERISTIC OF CARBONIZED REFUSE DERIVED FUEL

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Carbonized refuse derived fuel

ABSTRACT

Torrefaction is next to drying, pelletizing and briquetting one of the methods for pre-treatment of fuels for later use for energy purposes. Torrefaction is a thermo-chemical process, carried out in the temperature range from 200 to 300°C, under atmospheric pressure and inert gas environment. The study involved a refuse derived fuel (RDF) produced from municipal solid waste in a mechanical-biological plant. The aim of this work was to determine the kinetic parameters of the torrefaction process of RDF and to examine the effect of temperature and the residence time on fuel properties of biochar. Torrefaction process was carried out in the temperature range from 200 to 300°C with the temperature interval of 20°C. The residence was respectively 20, 40 and 60 minutes for each temperature. RDF and the resulting carbonized refuse derived fuel (CRDF) have been subjected to the following analysis: moisture content, organic matter, combustible and volatile content, ash content, and higher heating value. The determined activation energy of RDF torrefaction was 3.71 $kJ \cdot mol^{-1}$. The thermogravimetric analysis indicated that during torrefaction, mostly lingo-cellulosic, and hemi-cellulosic biomass present in RDF decomposes during torrefaction. Studies have shown the influence of residence time and temperature on fuel properties of the obtained CRDF. The highest heating value of the CRDF was obtained for the temperature of 260°C, and residence time 20 minutes.

1. INTRODUCTION

Waste conversion for energy purposes offers an effective way of recycling. This approach is very important in times of increased energy demand and the requirements of waste utilization and recycling. One of the ways to maximize the production of electricity and heat from waste is to produce Refuse Derived Fuel (RDF) and direct it for later thermal recovery and/or recycling.

RDF is a converted waste material that is generated by mechanical treatment (grinding, sorting) of combustible municipal and industrial wastes, including primarily plastics, paper and wood. RDF is most often used for the production of electricity and heat (Dalai et al., 2009; Preston and Kollberg 2016).

In Poland, municipal wastes are sent first to the Regional Municipal Waste Treatment Plant or to a waste incineration plant. At the Regional Municipal Waste Treatment Plant waste is subjected to mechanical biological treatment. The waste stream in the plant is divided into two fractions using a sieve with a mesh size of 80 mm. From the over-sieve fraction, materials that can be recycled are selected, and then unselected waste is converted into alternative fuel (RDF) for a waste incineration plant or cement plant. The under-sieve fraction undergoes a biostabilization process. After this process, the obtained stabilized waste is sieved on a 20 mm sieve. The obtained over-sieve fraction is most often disposed at the landfill, and the fine fraction is used as a material for landfill covering (Pietryszyn and Primus 2015, SPC report 2015, Kinitz 2014).

At present, there are 127 Regional Municipal Waste Treatment Facilities in Poland, and their total capacity is 8164 thousand Mg per year. RDF incineration plants can be divided into two types: cement plants and waste incineration plants. 9 cement plants in Poland have the total capacity of 1200 thousand Mg of RDF per year. In Poland, waste incinerators are currently in the initial stages of com-

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missioning or in the final phase of construction. Ultimately, 7 incineration plants will operate in the area of Poland, with a total capacity of 1 014 thousand Mg per year (Pietryszun and Primus, 2015).

The total mass of the produced RDF amounts 2 million Mg per year. The RDF produced in Poland is characterized by the following properties: moisture content fluctuates from 3.20% to 24.80%, ash content from 11.00 to 24.30% d.m., sulphur content from 0.12% to 0.76% d.m.. Higher Heating Value (HHV) ranges from 15.80 to 23.08 MJ·kg^{-1}, is Low Heating Value (LHV) from 14.53 to 21.59 MJ·kg^{-1}. The RDF is also characterized by high heterogeneity, which adversely affects the possibility of maintaining constant fuel properties (Ahn et al., 2013; Nowak and Szul, 2016).

RDFs are used for energy and heat production in cement plants and waste incineration plants with a total capacity of 1.996 thousand Mg per year (Krawczyk and Szczygieł 2013; Kinitz N., 2014). It should be noted that the total capacity of RDF utilization installations is two times smaller than the estimated annual volume of RDF produced. This situation results in a decrease in costs for wastes utilization in a thermal processing plant and increase in the requirements for the quality of RDF produced by the MBPs. At present, the essential parameters to be met by RDF are: calorific value >20 MJ·kg^{-1}, moisture content <15%, heavy metal content <2500 mg/kg, chlorine content <1%, sulphur content <1.5%, and ash <15%. In addition, the RDF should have a granularity of less than 40 mm and be a homogeneous mixture (Hryb and Biegańska, 2013).

Torrefaction also called roasting, high-temperature drying, low-temperature pyrolysis could be a helpful solution for overcoming problems with RDF qualitative requirements. Torrefaction is a thermo-chemical process, with following characteristics: temperature 200-300°C, heating rate <50°C·min^{-1}, residence time <60 minutes, no oxygen, atmospheric pressure (Tumuluru et al., 2011). Five process phases can be distinguished: pre-heat, pre-drying, drying and transient heating, torrefaction, cooling of the product (Bergman et al., 2005). As a result of the process, two products are obtained: biochar and torrefaction gas one with a mass balance of 70-80% and 23-30% respectively. The solid product is called biocarbon when agricultural or forestry biomass is used as a substrate. For other substrates, it is called carbonate or biochar (Malińska, 2015). The gas product is referred to as a tor-gas (Bergman et al., 2005).

The solid product resulting from the processing of agricultural or forestry biomass is characterized by:

- High energy density. Processed biomass contains 70-80% of the initial mass and 80-90% of initial energy (Tumuluru et al., 2010);
- Decrease in moisture content. After the torrefaction process, the moisture content of the obtained product is approximately 1-2% mass (Tumuluru et al., 2010);
- Hydrophobic properties. Processed biomass manifests high hydrophobicity. Maximum water uptake is 1-6% (Tumuluru et al., 2010) (e.g. water content in unprocessed wood biomass ranges from 12 to 22%, bark from 45 to 55% (Kordylewski et al., 2008));
- Increased carbon content. The concentration of carbon in the structure of the compound results in increased biocarbon reduction properties (Bergman, 2005);
- Reduction of oxygen and hydrogen. The O/C and H/C ratios are reduced, resulting in an increase in the attractiveness of biocarbon as a substrate for the gasification process (Prins, 2005);
- Better milling properties. Due to the depolymerisation of cellulose fibres, lignin and hemicellulose biochar grinding requires less energy since the structure and form of the particulate matter is similar to carbon (Bergman et al., 2004).

Due to the above, torrefaction process may be a good way to increase the fuel properties of RDF. However, this process has not been characterized or understood deeply. This paper presents the characteristics of the thermal decomposition of RDF using thermogravimetric analysis (TGA). Using the qualitative interpretation method of the TGA curve, changes in the mass decrease pattern of the sample under linear temperature increase were observed, whereby a comparison of the mass drop within temperature range with the values given in literature of the individual RDF components was conducted. The quantitative interpretation of the TGA allowed for the determination of kinetic parameters such as the reaction rate constant and activation energy. These parameters are indispensable in the torrefaction modelling process.

RDF and produced carbonates are characterized in terms of fuel properties. The conducted analysis allowed to determine the suitability of the torrefaction as an alternative fuels valorisation process.

2. MATERIAL AND METHOD

2.1 RDF used in the study

The RDF used in the study was taken from a mechanical-biological waste treatment facility with the status of a regional waste treatment plant. The facility is located in the village of Gać, Poland (in the region of Lower Silesia). The process of production RDF from municipal solid waste is presented in Figure 1.

A general 250 kg sample was taken from RDF's production line and then a laboratory sample of 5 kg was separated from the general sample by quartering (PN-Z-15006:1993). In order to homogenize the material (RDF and RDFs), it was ground to particle size ≤0.425 mm with the use of the LMN 100 knife mill. The material to be tested was prepared in the way presented below.

2.2 Carbonized Refuse Derived Fuel production method

The biochar, previously referred to as Carbonized Refuse Derived Fuel (CRDF) (Białowiec et al., 2017) was obtained by means of the SNOL 8.1/1100 muffle furnace (Figure 2). CRDF samples were generated under the following conditions:

- Temperature range from 200 to 300°C (temperature interval of 20°C);
- Retention time: 20, 40, 60 minutes for each temperature;

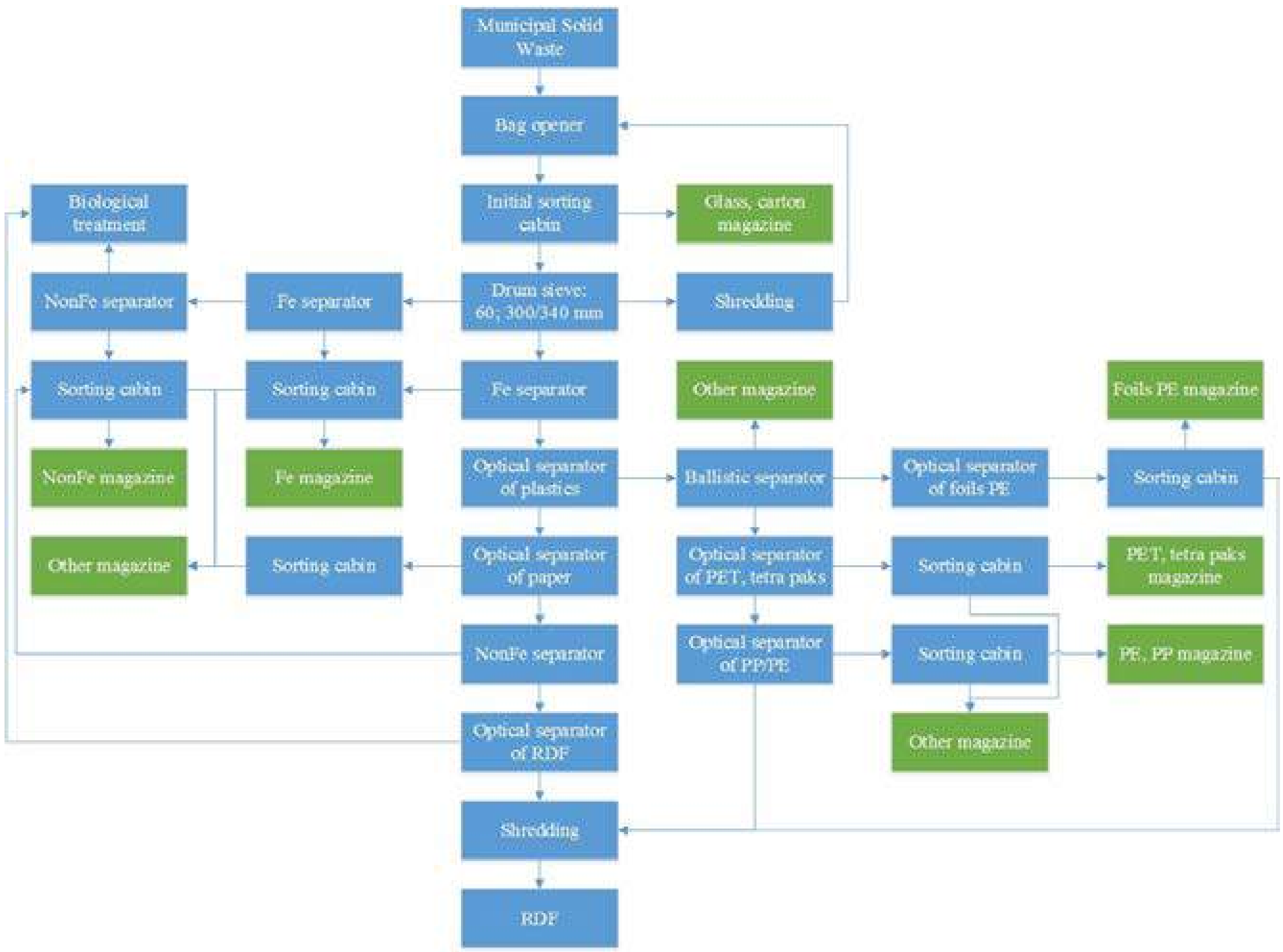

FIGURE 1: The configuration of RDF production lines at MBT plant in Gać.

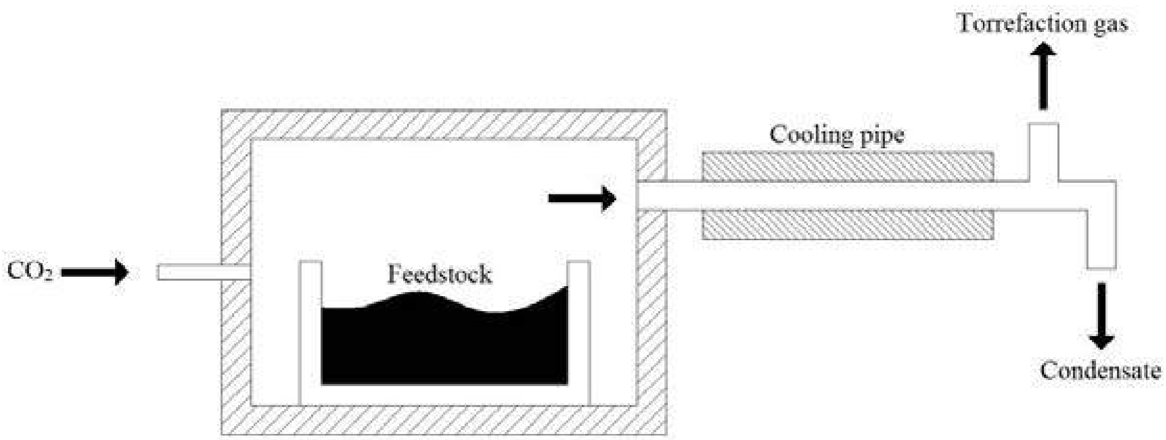

FIGURE 2: Schematic figure of the experimental set-up of CRDF generation.

- Temperature rise: 50°C·min^{-1} (maximum heating rate);
- Used gas: carbon dioxide;
- Gas flow: 10 dm^3·h^{-1}.

The heating of the reactor was commenced 5 minutes after gas introduction into the device began. Carbon dioxide was cut off when the temperature inside the reactor during the cooling phase reached 100°C (Madanayake et al., 2016).

2.3 Physical and chemical analysis of RDF and CRDF

The RDF and generated CRDF from torrefaction were tested for:

- Morphological composition (only RDF) in accordance with Malinowski and Wolny-Koładka (2012);
- Moisture content by means of the KBC65W laboratory dryer in accordance with the PN-EN 14346:2011 stan-

dard;

- Content of organic matter by means of the SNOL 8.1/1100 muffle furnace in accordance with the PN-EN 15169:2011 standard;
- Combustible and non-combustible content by means of the SNOL 8.1/1100 muffle furnace in accordance with the PN-Z-15008-04:1993 standard;
- Volatile content by means of the SNOL 8.1/1100 muffle furnace in accordance with the PN-G-04516:1998 standard;
- Higher heating value by means of the IKA C2000 Basic calorimeter in accordance with the PN-G-04513:1981 standard.

Each of the designations was repeated 3 times.

2.4 Thermogravimetric analysis (TGA) of RDF

The thermogravimetric analysis was carried out by means of the Czylok RST 40·200/110P stand-mounted tubular furnace (Figure 3).

The study used the method of qualitative and quantitative interpolation of the TGA curve. The qualitative method allows for determining the mass deviations of the sample to be tested at a set temperature. By this analysis, the distribution of particular chemical compounds that build up the sample may be observed. A quantitative method allows for determining the kinetic parameters of the process. The measurement is based on accurate determination of the mass change and its rate at particular temperatures.

The first method was carried out under the following conditions:

- Temperature from 10 to 850°C;
- Temperature rise: 10°C·min^{-1} (maximum heating rate);
- Used gas: carbon dioxide;
- Gas flow: 10 dm^3·h^{-1}.

The second method was carried out under the following conditions:

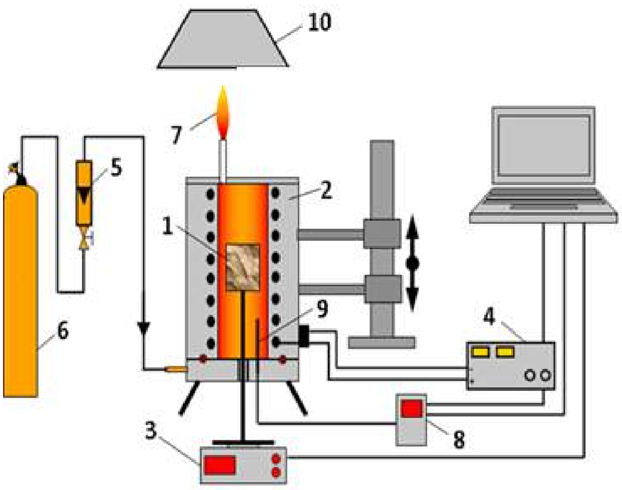

FIGURE 3: Reactor set-up: 1 - a vessel filled with solid fuel sample, 2 - electrically heated reactor, 3 - electronic balance, 4 - electric power feeder (regulator), 5 - rotameter, 6 - bottle with carbon dioxide, 7 - gaseous products of pyrolysis/torrefaction process, 8 - temperature indicator, 9 - thermocouple, 10 - exhaust chimney.

- Temperature range from 200 to 300°C (temperature interval of 20°C);
- Retention time: 60 minutes for each temperature;
- Used gas: carbon dioxide;
- Gas flow: 10 dm^3·h^{-1}.

Based on TGA results, the reaction rate and activation energy within the torrefaction temperature range was calculated by Statistica 13.1 software. The reaction constant rate of the thermal transformation of the material was calculated on the basis of a first-order reaction (Eq. 1, Eq. 2) (Bates et al., 2012):

$$M_S = M_S^0 \cdot e^{-k \cdot t} \quad (1)$$

$$\ln \frac{M_S^0}{M_S} = k \cdot t \quad (2)$$

where: M_S^0 is initial mass, g, M_S is mass per unit, g, k is reaction rate constant, 1·s^{-1}, t is time, s.

The Arrhenius equation (Eq. 3) (Bates et al., 2012) represents the dependence of the reaction constant rate k on the temperature T:

$$k(T) = A \cdot \exp - \frac{E_a}{R \cdot T} \quad (3)$$

The logarithmic form of the equation (Eq. 3) is shown below:

$$\ln k(T) = \ln A - \frac{E_a}{R \cdot T} \quad (4)$$

where: R is universal gas constant, 8.314 J·(mol·K)$^{-1}$, T is temperature, K, A is a pre-exponential factor, 1·s^{-1}, E_a is activation energy, J·mol-1, k is reaction constant rate, 1·s^{-1}.

Using the Arrhenius equation, activation energy can be calculated by means of the reaction constant rate. Ln(k) is a linear function of 1·T-1 (Eq. 5) (Soria-Verdugo, 2015).

$$y = a \cdot x + b \quad (5)$$

where:

$y = \ln(k)$,

$b = \ln A$,

$a = \frac{E_a}{R}$,

$x = 1 \cdot T$.

3. RESULTS AND DISCUSSION

3.1 Results of the physical and chemical analysis of the substrate

RDF morphological composition is shown in Table 1.

The percentage share of highly calorific waste (plastics, paper, wood, textiles) was 54.3%. This value is low, but lies within the lower range of values given in literature, where the proportion of highly calorific waste was from 53.2% to 100% (Seo i in. 2010; Miskolczi i in., 2011; Kara, 2012; Kruger i in., 2014; Akdag i in., 2016; Çepolioğullar Ö i in., 2016).

The average results of the physical and chemical analyses of RDF are presented in Table 2.

Moisture content in the RDF was 17.31%. This value is within the upper limit of the moisture content in RDF. The

TABLE 1: The average morphological composition (N=3) of the analysed RDF.

Waste group	Share of a waste group (%)
Plastics	30.3
Paper	11.3
Wood	10.5
Composite waste	8.3
Rubber	5.6
Textiles	2.2
Metal	0.1
Glass	0.1
Kitchen and garden waste	0.1
Other unidentified waste and mineral waste	31.6

TABLE 2: The average (±SD - standard deviation) values of physical and chemical properties of the analysed alternative fuel.

Sample	Alternative fuel
Moisture (%)	17.31 ± 4.48
Organic matter content (%)	85.80 ± 15.32
Volatile content (%)	85.13 ± 1.04
Combustible content (%)	86.75 ± 1.82
Ash (%)	13.25 ± 1.82
High calorific value (MJ·kg^{-1})	25.41 ± 1.58

moisture content reported in the literature ranges from 1.6% to 17.4% (Akdag et al., 2016; Edo et al., 2016; Manya et al., 2015; Seo et al., 2010; Singh et al., 2011).

Analysing the results of the content of organic matter and the content of combustible components, it must be noted that RDF is mostly built of organic materials that break down to 550°C. This is reflected in the morphological composition of the alternative fuel, which consists mainly of materials (plastics and wood) that undergo decomposition at temperatures of 550°C (Robinson et al., 2016). The residual matter is ash, whose average value in the analysed material was 13.25%. The ash content in RDF ranges from 8.64% to 26.29% (Ahn et al., 2013; Akdag et al., 2015; Çepolioğullar et al., 2016; Miskolczi et al., 2011; Seo et al., 2010; Singh et al., 2012).

The determined high calorific value was higher than usually given in the literature - from 17 to 22 MJ·kg^{-1} (Akdag et al., 2016; Çepolioğullar Ö et al., 2016; Whyte et al., 2015).

3.2 Results of a thermogravimetric analysis

During the thermogravimetric analysis, two weight drops were observed. The mass decreases occurred between 210°C and 380°C, and between 380°C and 730°C. Material transformation followed one by one (Figure 4).

The first thermal decomposition may be linked of hemicellulose and cellulose breakdown. The temperature ranges of the thermal decomposition of these two compounds are as follows: hemi-cellulose 220-315°C, cellulose 315-370°C (Akdeg et al. 2016; Carrier et al., 2011; Lu et al., 2012). The second peak may be linked to the decomposition of plastics. Degradation of these materials begins above 400°C (Robinson et al., 2016; Sanchez-Silva et al., 2012; Stępień et al., 2017).

An equation (1) was used to calculate reaction constant rates and activation energies for temperatures within the range of 200-300°C (Table 3).

The estimated activation energy was 3.71 kJ·mol^{-1}. Literature review showed that activation energy for RDF depends on temperature ranges: between 240-380°C and between 250-370°C. The values obtained by Singh et al. (2012) and Grammelis et al. (2007) were much higher and were 97.8 and 121 kJ·mol^{-1}, respectively.

3.3 Results of the physical and chemical analysis of the CRDF

The average results of the physical and chemical analyses of carbonized refuse derived fuel are presented on 3D

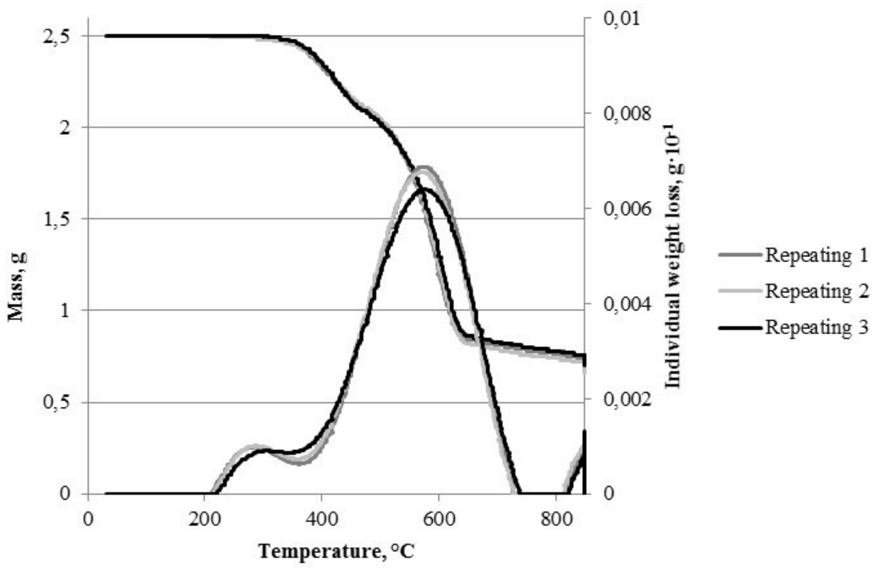

FIGURE 4: DTG and TGA characteristics of RDF.

charts with spline interpolation (Figures 5-10).

Moisture content for untreated RDF was 17.31%. The residual water content in carbonates was the smallest at temperature 200°C and retention time 20 minutes. In that case, the moisture decreased to 0.13% (Figure 5). The moisture content increased to 1.27% along with the increase of the temperature and retention time of the torrefaction process. The studies conducted by Nobre et al., (2016), found that the torrefaction process had a positive effect on the moisture content reduction. They indicated that the content of water 6.02% in the raw material dropped to 2.23% during torrefaction at 300°C for 30 minutes.

The contents of organic matter (Figure 6), combustible (Figure 7) and volatile components (Figure 8) were similar. This was correlated with characteristics of thermal decomposition of materials present in RDF, including plastic and lingo-cellulosic compounds. These materials undergo thermal degradation of up to 550°C. The higher content of combustible components in relation to the organic matter is associated with a high temperature distribution of volatiles (Lu et al., 2012, Robinson et al., 2016). The content of organic matter, combustible and volatile components in relation to unprocessed biomass was reduced by 6%, 7%, and 7%, respectively. These values are low compared to the results obtained by Nobre et al. (2016), where the decrease in the volatiles amounted to 22% at 300°C torrefaction with 30 minutes residence time. Such a large difference may be due to the different morphological composition of the tested samples. The highest residual organic matter has been found for temperature range from 240 to 260°C, and residence time 40 minutes (Figure 6). The tendency of combustibles content decrease with the increase in torrefaction temperature, for all tested residence times, was observed (Figure 7). A similar relationship was found for volatiles content, with the highest value for a variant with temperature 200°C, and 60 minute of residence time (Figure 8).

The ash content in CRDF is indirectly related to the increase in temperature and residence time that affects the sample gasification. It should be noted that ash content in carbonate increased to over 23% (Figure 9). The raw material was characterized by the ash content of 13%. The maximum ash content in torrefied materials was lower compared to RDF. The maximum value of the quoted parameter can be as high as 26%.

The decrease in HHV is related to the ash content increase and the gasification of the volatile component. According to the principles of torrefaction, the substrate should have a low content of inert parts, because after the process, when partial degassing of the volatiles occurs, the mass ratio of the ash to the entire mass of the particle increases (Tumuluru et al., 2011). In the case of RDF torrefaction, the optimum value of the process was 260°C with 20 minutes of residence time. For this torrefaction parameters, the average value of HHV was 26.22 MJ·kg^{-1}. Comparing this value with the average HHV of unprocessed material, the obtained value was higher by 0.81 MJ·kg^{-1}.

The observed tendency of HHV decrease along with the increase in temperature and retention time was also reflected in the research carried out by Nobre et al. (2016), in which the HHV heat decreased from 17.68 to 15.70 MJ·kg^{-1}.

Due to the low content of lingo-cellulosic compounds in the waste, they could have reacted completely at a temperature lower than 300°C. The main component of RDF were plastics, whose distribution starts at 400°C (Robinson et al., 2016). The complete conversion of the lignocellulose parts caused a rise in the ash content, which had an impact on the ash presence in the sample. The final effect was that the higher ash content resulted in the decrease in HHV.

Table 4 shows the comparison of raw material with the obtained CRDF's.

4. CONCLUSIONS

The TGA analysis has shown that one of the materials groups contained in RDF is decomposing within the temperature range of the torrefaction process. The observed transformation was attributed to the decomposition of cellulose and hemicellulose which build wood and paper in RDF. The calculated activation energy in the temperature range of 200 to 300°C was 3.71 kJ·mol^{-1}.

The torrefaction process has a positive effect on reducing the moisture content. Moisture decreased from 17% to 1%.

As the temperature and retention of the torrefaction process increased, the material degassed significantly, resulting in an increase in ash content in the product. This parameter unfavourably influences the HHV.

The highest HHV of CRDF was achieved for temperature 260°C, and residence time 20 minutes.

Scientific research on the torrefaction of RDF is still at an early stage and needs to be further developed in order to accurately characterize the process and the obtained products.

TABLE 3: Reaction constant rate and activation energy.

T, K	k, 1·s^{-1}	R^2	1·T^{-1}	ln(k), 1·s^{-1}	E, J·mol^{-1}	R^2
473	1.41E-05	0.89	2.11E-03	-1.12E+01	3.71E+03	0.55
493	1.44E-05	0.77	2.03E-03	-1.11E+01		
513	1.47E-05	0.78	1.95E-03	-1.11E+01		
533	1.37E-05	0.78	1.88E-03	-1.12E+01		
553	1.66E-05	0.80	1.81E-03	-1.10E+01		
573	1.66E-05	0.67	1.75E-03	-1.10E+01		

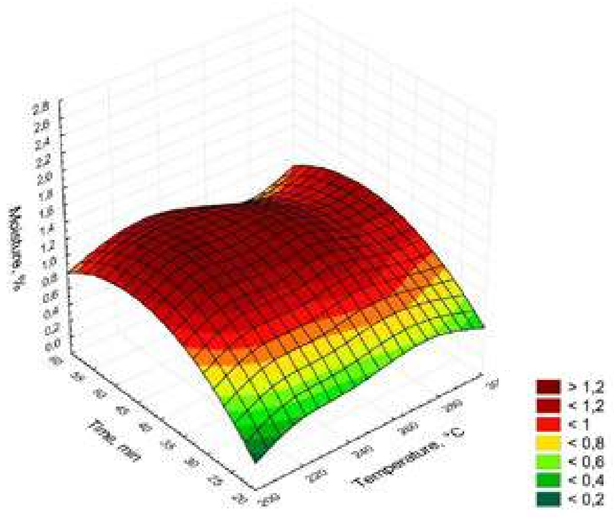

FIGURE 5: Effect of torrefaction temperature and retention time on the moisture content in CRDF.

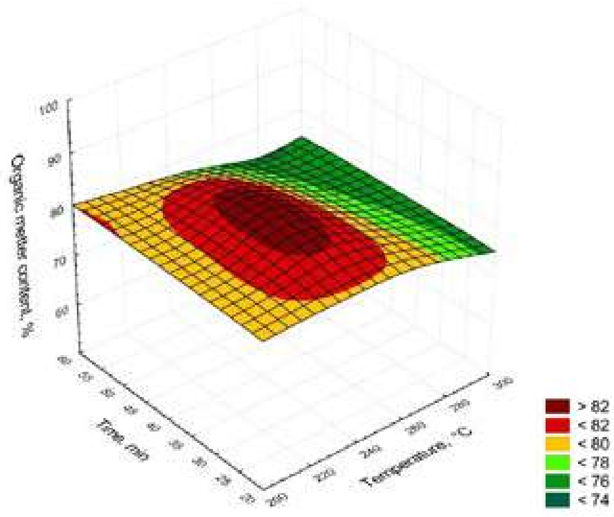

FIGURE 6: Effect of torrefaction temperature and retention time on the organic matter content in CRDF.

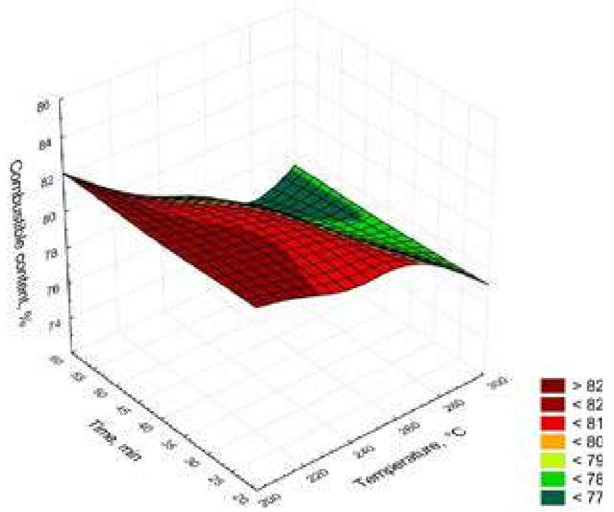

FIGURE 7: Effect of torrefaction temperature and retention time on the combustible content in CRDF.

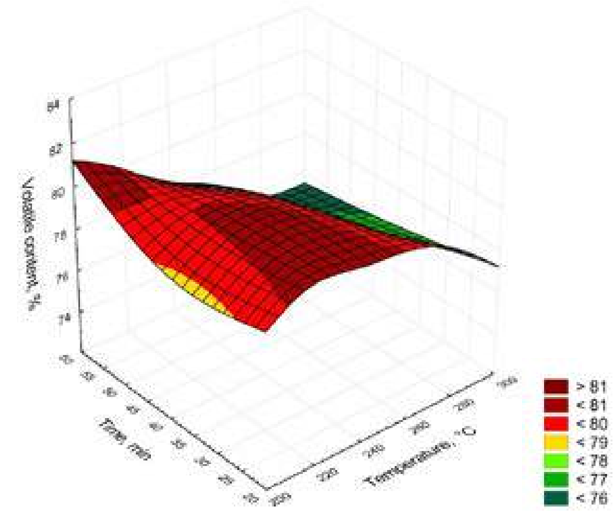

FIGURE 8: Effect of temperature and retention time on the volatile content in CRDF.

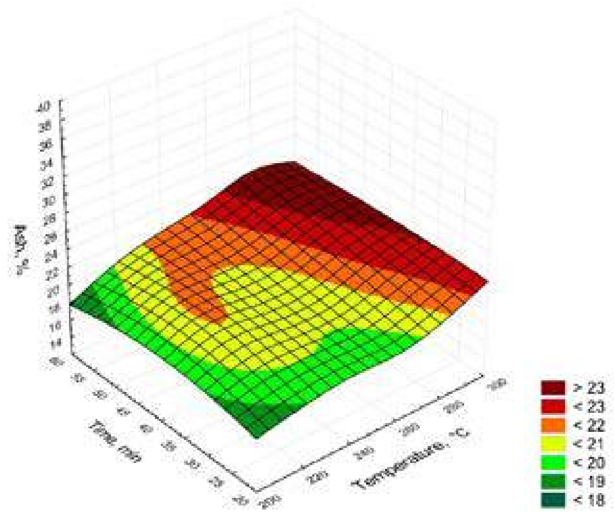

FIGURE 9: Effect of torrefaction temperature and retention time on the ash content of CRDF.

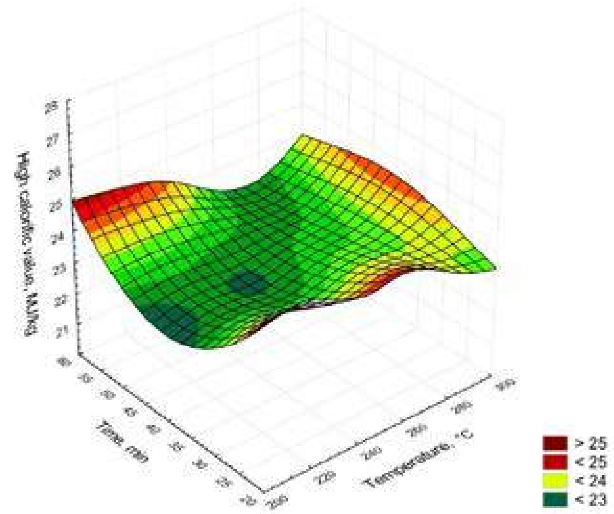

FIGURE 10: Effect of torrefaction temperature and retention time on the HHV of CRDF.

TABLE 4: The average (±SD - standard deviation) values of physical and chemical properties of the analysed alternative fuel and CRDF's.

Sample		Moisture (%)	Organic matter content (%)	Volatile content (%)	Combustible content (%)	Ash (%)	High calorific value (MJ·kg^{-1})
Alternative fuel		17.31 ± 4.48	85.80 ± 15.32	85.13 ± 1.04	86.75 ± 1.82	13.25 ± 1.82	25.41 ± 1.58
200	20	0.08 ± 0.07	79.33 ± 0.30	79.29 ± 0.38	81.95 ± 0.30	18.04 ± 0.30	24.94 ± 0.31
	40	0.94 ±0.07	80.05 ± 0.89	77.50 ± 0.57	81.34 ± 1.00	18.65 ± 1.00	22.18 ± 0.49
	60	0.76 ± 0.10	80.83 ± 0.61	82.01 ± 0.85	82.65 ± 0.47	17.34 ± 0.47	25.15 ± 0.03
220	20	0.54 ± 0.05	79.93 ± 0.49	81.65 ± 1.10	81.70 ± 0.47	18.29 ± 0.7	26.53 ± 0.38
	40	1.63 ± 0.77	73.85 ± 0.47	80.60 ± 0.57	80.94 ± 3.08	24.25 ± 11.49	23.50 ± 0.19
	60	0.97 ± 0.03	78.42 ± 0.71	79.90 ± 0.31	79.69 ± 0.70	20.30 ± 0.70	24.64 ± 0.47
240	20	0.26 ± 0.02	77.41 ± 1.13	79.05 ± 0.76	78.85 ± 1.12	21.14 ± 1.12	23.10 ± 0.50
	40	0.80 ± 0.45	92.41 ± 0.85	80.84 ± 0.70	81.49 ± 0.39	19.01 ± 0.51	21.98 ± 0.32
	60	1.27 ± 0.08	76.10 ± 0.35	76.87 ± 0.67	77.82 ± 0.27	22.17 ± 0.27	25.34 ± 0.29
260	20	0.55 ± 0.05	80.45 ± 0.44	80.27 ± 0.44	81.79 ± 0.46	18.20 ± 0.46	26.22 ± 0.84
	40	1.77 ± 0.14	78.86 ± 0.60	80.11 ± 0.27	80.71 ± 0.60	19.28 ± 0.60	24.11 ± 0.31
	60	0.73 ± 0.02	76.55 ± 1.41	78.06 ± 0.88	78.49 ± 1.32	21.50 ± 1.32	22.65 ± 0.38
280	20	0.79 ± 0.01	78.31 ± 0.27	79.98 ± 1.07	80.11 ± 0.28	19.88 ± 0.28	23.74 ± 0.31
	40	0.73 ± 0.04	73.19 ± 0.11	75.13 ± 0.26	75.77 ± 0.14	24.22 ± 0.14	22.91 ± 0.44
	60	0.37 ± 0.06	70.62 ± 0.31	74.09 ± 0.70	73.30 ± 0.28	26.69 ± 0.28	22.26 ± 1.21
300	20	0.25 ± 0.023	74.40 ± 0.19	76.60 ± 0.23	76.38 ± 0.17	23.61 ± 0.17	22.91 ± 0.13
	40	1.23 ± 0.10	75.33 ± 1.06	76.73 ± 1.00	77.29 ± 0.94	22.70 ± 0.94	25.47 ± 0.82
	60	1.08 ± 0.07	76.27 ± 0.59	76.34 ± 1.04	78.52 ± 0.61	21.47 ± 0.61	24.12 ± 0.76

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ECONOMIC EVALUATION OF A HYDROTHERMAL LIQUEFACTION PROCESS

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ABSTRACT

Wet waste streams include a wide variety of products such as food processing residues, sewage sludge, but also the organic fraction of municipal solid waste. Hydrothermal liquefaction is a thermochemical conversion in hot compressed water that produces a hydrophobic product. This paper gives presents how hydrothermal liquefaction can produce a biocrude or a heavy fuel oil from blackcurrant pomace, grape marc and sewage sludge. The paper presents experimental results as well as a technical and economic evaluation of the process. The results from hydrothermal liquefaction depend on the resource. Typical biocrude yield is 50% of the dry resource while bio-oil yield can be up to 25%. High ash resources are however less interesting for this technology. The production costs are high compared to their fossil counterparts but gate fees in the order of 50 to 130 € tonne^{-1} could ensure economic competitiveness compared to fossil fuels.

1. INTRODUCTION

Waste streams are an extremely variable and diffuse resource. Examples include sewage sludge, food processing residues and the organic part of municipal solid waste. Humidity typically varies from 50 to 90%. Basic incineration but also more advanced techniques such as gasification and pyrolysis, are interesting for dry feedstocks but lose much of their interest when the humidity of the resource is higher than 50%. Dewatering and drying is possible for most feedstocks but at a significant cost. These wet waste streams are often used or abandoned in low value applications such as composting, incineration or landfill. Many environmental problems are associated to those waste streams such as bad odours but also due to the production of secondary pollutants such as dioxins during incineration.

Current disposal routes include composting, anaerobic digestion but also landfill and incineration (often after drying). Hydrothermal Liquefaction (HTL) is an alternative waste treatment that makes it possible to produce liquid fuels potentially replacing fossil fuels. Hydrothermal liquefaction produces a biocrude that can be further upgraded to biofuels. This paper shows how the operation of HTL plants can be made economically feasible.

Hydrothermal liquefaction converts biomass in hot compressed water into a biocrude. This biocrude is an oily material containing bio-oil and char. Hydrothermal liquefaction has been known for some time. The developments started simultaneously in Europe (Goudriaan & Peferoen, 1990) and in the United States (Elliott & Schiefelbein, 1989). The conversion takes place at temperatures between 300 and 400°C and at pressures above the saturation pressure to ensure that water remains in the liquid phase, typically above 100 bar. Under these conditions the ionisation of water increases while its polarity decreases (Kruse & Dahmen, 2015), favouring depolymerisation and dehydration of biomass polymers to produce hydrophobic compounds.

Figure 1 shows a typical resource, black currant pomace, an autoclave batch reactor and the biocrude obtained. The Heating Value of the biocrude is typically 30-35 MJ kg^{-1} whereas the original biomass has heating values in the 15-21 MJ kg^{-1} range. This biocrude can either be used directly as a combustible liquid, fed into a refinery as crude oil (Buisonjé et al., 2010), or it can be upgraded to a diesel type biofuel (Zhu et al., 2014). The initial development of the technology in the 1970s has been hampered by low oil prices in the 1990s but also by technical difficulties and the increasing cost of biomass. Increasing oil prices in the early years 2000 lead to a regain in interest. The application of hydrothermal liquefaction to wet waste streams can procure a new momentum for this technology. Traditional HTL laboratories such as PNNL are actively working on this subject as well as many newcomers.

The chemical composition of the resource plays a major role in the product yield and quality as has been shown by

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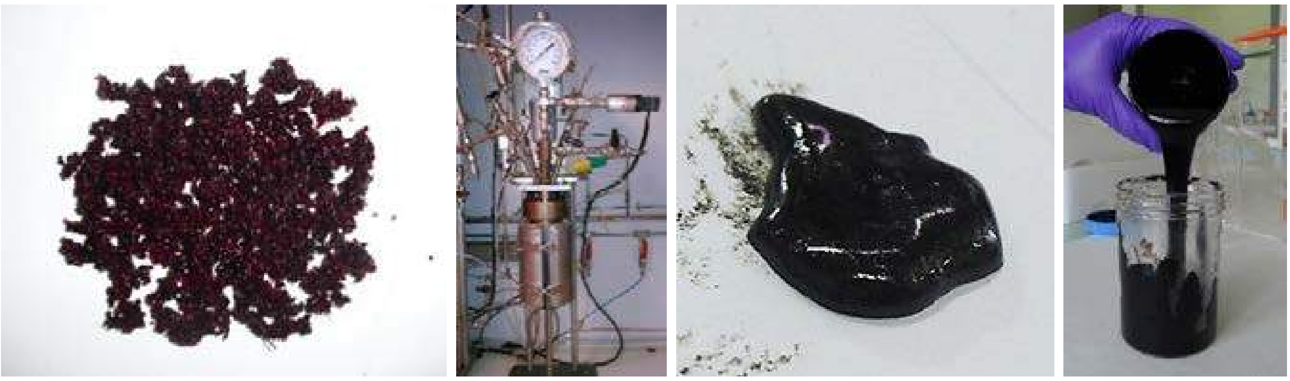

FIGURE 1: Example of the resource blackcurrant pomace, the HTL reactor and the obtained biocrude.

(Déniel at al., 2017). Important parameters include ash content, fibre composition and content, protein and lipids. This study presents results with biomasses rich in lipids and proteins but also with ligno-cellulosic biomasses. The differences in the results as well as the impact on the economic evaluation will be presented. It has been shown that certain additives (Deniel et al., 2017) and operating conditions (Déniel et al., 2016) also greatly influence biocrude and bio-oil yields but also their quality.

There are many technical-economic evaluations of biomass to fuel processes. There are however few evaluations of HTL processes, most on the conversions of biomass into biofuel. The majority of the evaluations of the HTL process are done on either algae (Hognon et al. 2015; Ou, Thilakaratne, Brown, & Wright, 2015) or wood (Goudriaan & Peferoen, 1990; Zhu et al., 2014). Other studies exist on swine manure as a resource (Buisonjé et al., 2010; Minarick et al., 2011). Typical production costs for diesel type fuels from cultivated algae are in the 2-3 € L^{-1} range (Hognon et al., 2015) considering a fully integrated production site. Prices of defatted (waste) algae are much lower as the extracted lipids are sold at a premium price. HTL fuels from defatted algae may be much cheaper, less than 1 € L^{-1} (Ou et al., 2015) for very large plants (2000 tonne day^{-1}). Wood conversion plants at a large scale are also expected to be (nearly) profitable at a large scale with production prices in the 0.6 to 1.2 € L^{-1} range (Goudriaan & Peferoen, 1990; Zhu et al., 2014). More complicated feedstocks such as sewage sludge and swine manure received less attention for technical-economic evaluations of the HTL process. Buisonjé (Buisonjé et al., 2010) estimated that an integrated swine manure conversion plant should be economically viable with a gate fee of at least 15 € $tonne^{-1}$ applied to the wet swine manure to produce a biocrude that can be sold to a refinery for further upgrading.

Sewage sludge conversion in HTL plants has an additional challenge in that the resource is very distributed, available throughout the territory in small quantities. Transport of wet sludge over significant distances is not recommendable. Local processing should be favoured. The waste water treatment plant of a typical metropolitan area as Grenoble (France) produces around 7000 tonne of dry matter per year, around 1 tonne dry matter per hour, or around 10 m^3 per hour of biomass slurry. This capacity is 10 to 100 times smaller than projected wood processing facilities and extremely small compared to fossil fuel refineries. There remain important uncertainties on the chemistry and technological issues on HTL plants. The optimal residence time in the reactor will probably range from 10 to 20 minutes depending on the resource and the temperature. This means that the reactor volume should be around 2.5 m^3 which is already quite large for a pressurised reactor. Alternatively multiple smaller reactors in parallel could be considered. Being limited to low scales make economic viability even more difficult. Gate fees are common place in the waste treatment industry and typically 100 to 200 € $tonne^{-1}$ is charged for waste treatment in France (Awiplan, 2015). The use of sewage sludge as an agricultural resource is more and more constrained and is also costly (Ferry & Wiart., 2002), with prices in the same range.

The focus of this paper is on wet solids wastes such as food processing residues and municipal sewage sludge. Many other resources are suitable for hydrothermal liquefaction, such as micro and macro algae or even dry resources such as wood. The actual resources presented in this study include grape marc and blackcurrant pomace representing food processing residues. Three types of sewage sludge were also tested, mixed, activated, and anaerobically digested sewage sludges. These resources are characterised by a humidity varying from 50 to 90wt.% and an extremely variable chemical composition. The analysis of the resources is performed by following regular food analysis norms for fibres, lipids and proteins.

Hydrothermal liquefaction produces a biocrude with an interesting energy content. The biocrude can be further separated into bio-oil and char by means of solvent extraction. The produced oil can be compared to heavy fuel oil (Anouti, Haarlemmer, Déniel, & Roubaud, 2016). This bio-oil can be further refined into a biofuel by catalytic upgrading, typically to produce a biodiesel (Zhu et al., 2014). The higher the degree of refinement considered, the more uncertain the technical and economic feasibility is.

The objective of this study is to show how these low value resources can be valorised and upgraded to biofuels. The paper presents experimental results of how different resources behave under hydrothermal liquefaction conditions. However, the emphasis of this paper is not on the experimental work. The product yields of different resources, converted at the same conditions, are used to estimate the cost of the hydrothermal conversion. Gate fees are estimated to ensure economic viability of the plants.

2. MATERIALS AND METHODS

2.1 Materials and experimental procedure

Food processing residues presented in this study are grape marc and blackcurrant pomace. These are procured via local producers (UNGDA and Les Vergers de Boiron). Additionally, three types of sewage sludge were tested, mixed, activated, and anaerobically digested sewage sludge from municipal waste water treatment plants (WWTP) in the Grenoble region in France (Aquantis in Voreppe and Aquapole in Le Fontanil).

The resources have been analysed by well-known techniques to establish the chemical composition of the resource. The results are presented in Table 1. Simple sugars cannot be quantified by standard methods and are typically calculated by difference (everything that is not ash, protein, lipid or fibre).

Experiments were performed in a 0.6 L stainless steel (SS316) stirred batch reactor (Parr Instruments Company). In a typical experiment, the reactor was filled with 240±5 g of biomass slurry, with a constant 14 wt.% dry matter to water ratio in the case of blackcurrant pomace and grape marc. Sludge 1 was diluted to 10% dry matter to ensure good rheological properties. Sewage sludges 2 and 3, were used as received. The autoclave was leak tested, purged and pressurised to 1 MPa with nitrogen gas, to ensure sufficient pressure for gas analysis after the transformation. The pressure inside the reactor is a function of the reaction temperature, the amount of water and the amount of produced gas during the process. The reactor was stirred at 600 rpm and was heated to the reaction temperature by an electrical heater. Once the reactor reaches the reaction temperature, it was held during a specified time (holding time) within ± 1°C of the specified operating temperature. For these experiments a 15 min holding time was applied. All resources have been treated at 300°C, this temperature was reached in about 35 minutes. After the holding time, the reactor was rapidly cooled to room temperature in 20 min by an air quench.

After venting the reactor for gas analysis, the content of the reactor was first filtered on a Buchner filter to separate the aqueous phase from the raw organic residue. The raw organic residue (biocrude) was generally sticky, and removed from the reactor. The reactor was then weighed and the weight difference with the empty reactor is counted as raw organic residue. The produced biocrude, was dried at room temperature under air circulation until a stable mass was obtained (variation less than 0.1 mg). The experimental procedure is further detailed in the Figure 2.

The biocrude was separated into char and bio-oil using a solvent, ethyl-acetate in our case. Bio-oil was recovered after evaporation of the solvent at room temperature under air circulation, until a stable weight is obtained. GC-MS analysis confirmed that no residual solvent is left in the bio-oil. The char was also dried at room temperature under air circulation, until a stable weight was obtained. Weight loss of the char after extraction and drying was used to determine the proportion of solvent-soluble organics in the raw residue, and therefore the bio-oil yield. The bio-oil can alternatively be estimated by extraction from wet biocrude followed by solvent evaporation or by weighing the bio-oil after extraction. Determination of the water content by Karl-Fisher and comparison with the water content found by oven drying can provide an estimate to the amount of volatiles in the bio-oil that cannot be quantified otherwise. To limit the loss of volatile compounds the products are dried at room temperature. All yields reported in this study are expressed in weight percentage of the dry biomass (wt.% dry matter).

TABLE 1: Characterisation of blackcurrant pomace, grape marc and sewage sludge used in this work.

	Blackcurrant pomace	Grape marc (dried)	Sludge 1 Mixed	Sludge 2 Activated	Sludge 3 Digested
Moisture content (wt.%) [1]	59.6	7.4	83	94	97
HHV resource dry basis (MJ kg^{-1})	18.5	23.3	20.1	19.6	14.6
Fibre content (wt.% of dry matter) [2]	62	70	40	38	50
NDF (Neutral Detergent Fibres)	62	70	40	38	50
ADF (Acid Detergent Fibres)	53	63	28	30	26
ADL (Acid Detergent Lignin)	35	49	21	7	18
Cellulose (ADF-ADL)	18	15	7	23	8
Hemicelluloses (NDF-ADF)	9	6	12	8	25
Lignin (ADL)	35	48	21	7	18
Proteins (wt.% of dry matter) [3]	17	9.7	11	5	3
Lipids (wt.% of dry matter) [4]	15	8.1	10	15	13
Ash content at 550°C (wt.% of dry matter) [5]	4.3	4.8	14	14	38

[1] *EN 14774-1 (AFNOR, 2010a)*
[2] *NF V18-122 (AFNOR, 2013)*
[3] *Kjeldahl method*
[4] *Hydrochloric acid digestion + Petroleum ether extraction*
[5] *NF EN 14775 (AFNOR, 2010b)*

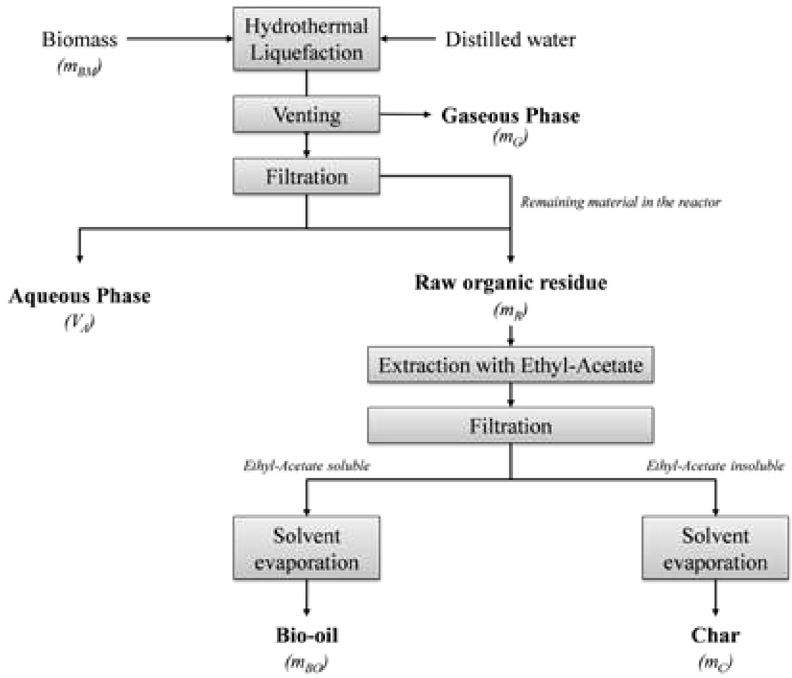

FIGURE 2: Products recovery procedure after hydrothermal liquefaction.

2.2 Economic Evaluation

The technical-economic evaluation is based on a process simulation with the ProSimPlus software (ProSim, 2012). The simulation was used to design the equipment in terms of heat exchange surfaces and electrical power. The evaluation of the equipment cost and economic evaluation is based on the methods described by Turton (Turton et al., 2003) and Chauvel (Chauvel et al., 2001). The main economic parameters as they enter in the production costs are presented in Table 2. The total installed equipment cost (Inside Battery Limits, ISBL) served as a basis to estimate the overall investment (CAPEX), including buildings, utilities, and engineering.

The approach was that the plant is located on an existing industrial site, either a food processing plant or a waste water treatment plant. The assumption was that the hydrothermal plant will be operated by an existing team of operators that is reinforced by one person for each shift. The capacity of the proposed plant was chosen to match an urban sewage treatment works of a city like Grenoble treating the water of 500 000 inhabitants. This a common capacity, even though much large treatment works. The majority of treatment works are much smaller.

Discounted cash flow methods take into account the erosion of the value of the invested money and the value of the cash flow by discounting operating costs and revenues in time. A euro earned in 2017 has more value to a company as a euro in 2027 as profits earned earlier it can be reinvested early to earn more money. The cash flow (CF) in any operating year n is discounted to a "present value".

$$CF_n = \frac{Revenues_n - Costs_n}{(1+DR)^n} \quad (1)$$

The operation is evaluated as a project with a start, operation and a clearly defined ending. The sum of the discounted investment, all yearly cash flows and the salvage value (value of the plant after service) of the plant is the Net Present Value (NPV) of the project after N years. This means that the project has generated a return on investment equal to the discount rate.

$$NPV = DeprCapCost + \sum_{n=1}^{N} CF_n + \frac{SalvageValue}{(1+DR)^N} \quad (2)$$

The minimum selling price is found by imposing the NPV to zero with a selected depreciation time.

Fixed costs consist of financial, personnel, maintenance and general overheads. The financial costs are essentially the costs of the bank loan. The cost of a Full Time Employee (FTE) is based on a French salary. With five

TABLE 2: Financial parameters for the economic evaluation.

Parameter	Value
Discount rate	8% (typical value)
Interest rate bank loan	5% (fixed)
Part bank loan in investment	50%
Stream factor	7000 h $year^{-1}$
Capital depreciation	10 years
Loan duration	10 years
Technical lifetime	20 years
Tax rate	30%
Personnel	5 Full Time Employees (FTE)
Personnel costs FTE	70 k€ $year^{-1}$ FTE^{-1}
Electricity cost	150 € MWh^{-1}
Treatment cost waste water	0.5 € m^{-3}
Salvage value plant	10% du CAPEX

shifts, one operator specific to the HTL plant is added to each shift. Maintenance and overheads are proportional to the size of the plant (and therefore its cost) and are typically estimated from a percentage of the CAPEX, 4% in our case. Variable costs include electricity usage and the cost of water treatment. Even recycled locally in the treatment works, the process water will generate some additional costs.

In this study we assumed that the products have a negative value and that the producer is prepared to pay for their disposal. In our case, the WWTP will internally shift funds from the disposal to the HTL unit. Taking into account this additional revenue allows the sale of the products on the general market at the price of fossil fuels. For all cases a gate fee was calculated to lower the production costs to match fossil fuel market prices.

3. EXPERIMENTAL RESULTS

Hydrothermal experiments always produced a mixture of solids (char), extractable (bio-oil) and an aqueous phase rich in ash and organic molecules. The products were separated according to the procedure described earlier. The results of the experiments are presented in Table 3.

As mentioned in section 2, the bio-oil yield can be evaluated by different methods. Drying of the biocrude or evaporating an extraction solvent always entrain the loss of light volatile compounds. Comparing the water content in the wet biocrude after filtration obtained by Karl-Fisher titration and that obtained by oven drying effectively showed that volatiles are lost in the drying and evaporation process. In practice for the blackcurrant pomace, 4% of initial dry ash free biomass was converted in bio-oil without being detected as such. We presented earlier (Anouti et al., 2016) a very detailed analysis of the bio-oil obtained from blackcurrant pomace.

We observed significant variations between the results from the different resources. Resources rich in lipids and proteins such as sewage sludge but also blackcurrant pomace produce significant amounts of oil. The lipids initially present in the resource clearly help increasing the bio-oil yield. The lignin rich grape marc produced less oil than the other resources under these conditions. Digested sewage sludge was very rich in ash and as a consequence contains less organic material. In addition, the organic material remaining after anaerobic digestion contains few proteins and lipids. It has lost much of its proteins and lipid content, making it less interesting for HTL.

Some of the sulphur was found in the gas phase as hydrogen sulphide. The produced gas was rich in CO_2, but it did contain some hydrocarbons and badly smelling molecules. The gas needs to be oxidised in a fired heater before it can be vented to atmosphere. The aqueous phase contains a significant amount of organics and cannot be disposed without further treatment. The process water must be treated before disposal.

4. TECHNO-ECONOMIC EVALUATION

The technical-economic analysis is presented on these five resources. Two different cases are presented. A simple conversion plant that produces biocrude that is sold to a refinery as a crude oil replacement. In the second case, the same conversion plant is equipped with a solvent extraction unit to produce a bio-oil. Bio-oils are corrosive due to their high acidity (Anouti et al., 2016; Haarlemmer et al., 2016). This means that stainless steel should be used as construction material.

The water content positively affects the results of the liquefaction, it has been shown that increasing dry matter content decreases the oil yields of the process (Déniel et al., 2016; Yang et al., 2016). However, increasing the water content also increases the volume of the installations. The water content in the feed is 90 wt.% (10 wt.% dry matter) for this evaluation to obtain sufficiently good rheological properties for sludge 1. This ensures good pumpability and optimal yields. The sewage sludge is very wet when produced and can easily be dewatered to the desired water content. Grape marc and blackcurrant pomace is much dryer when produced and will have to be diluted with process water. This actually has a beneficial effect on the yield and the biocrude quality (Déniel et al., 2016).

The biocrude plant is described in Figure 3. The plant consists of a two major subunits, these are the biocrude production unit and a separation unit that separates biocrude into bio-oil and char. Most (80%) of the required heat is recovered from the product stream in heat exchangers HX1 and HX2. The products are sticky when cold. Full heat recovery is therefore problematic as heat exchangers tend to foul when the biocrude contacts cold surfaces. Some additional heating (20% of the total heat requirement) is therefore necessary on feed of the HTL Reactor (two reactors are required to model the reactions). This is done by burning char or some of the produced biocrude in the

TABLE 3: Results of batch liquefaction experiments at 300°C.

Yields	Blackcurrant pomace	Grape marc	Sludge 1 Mixed	Sludge 2 Activated	Sludge 3 Digested
Biocrude (%)	52	35	51	61	54
HHV / LHV Biocrude (MJ kg^{-1})	32 / 30	30 / 28	26 / 24	24/ 23	13 / 12
Char (wt.%)	27	22	27	35	37
Bio-oil (wt.%)	25	13	24	26	17
HHV / LHV Bio-oil (MJ kg^{-1})	33.4 / 31.3	34 / 32	33 / 31	Not Available	Not Available
Gas (wt.%)	12	8.0	5.5	8.5	6.6
Aqueous phase (by difference) (wt.%)	24	57	44	31	40

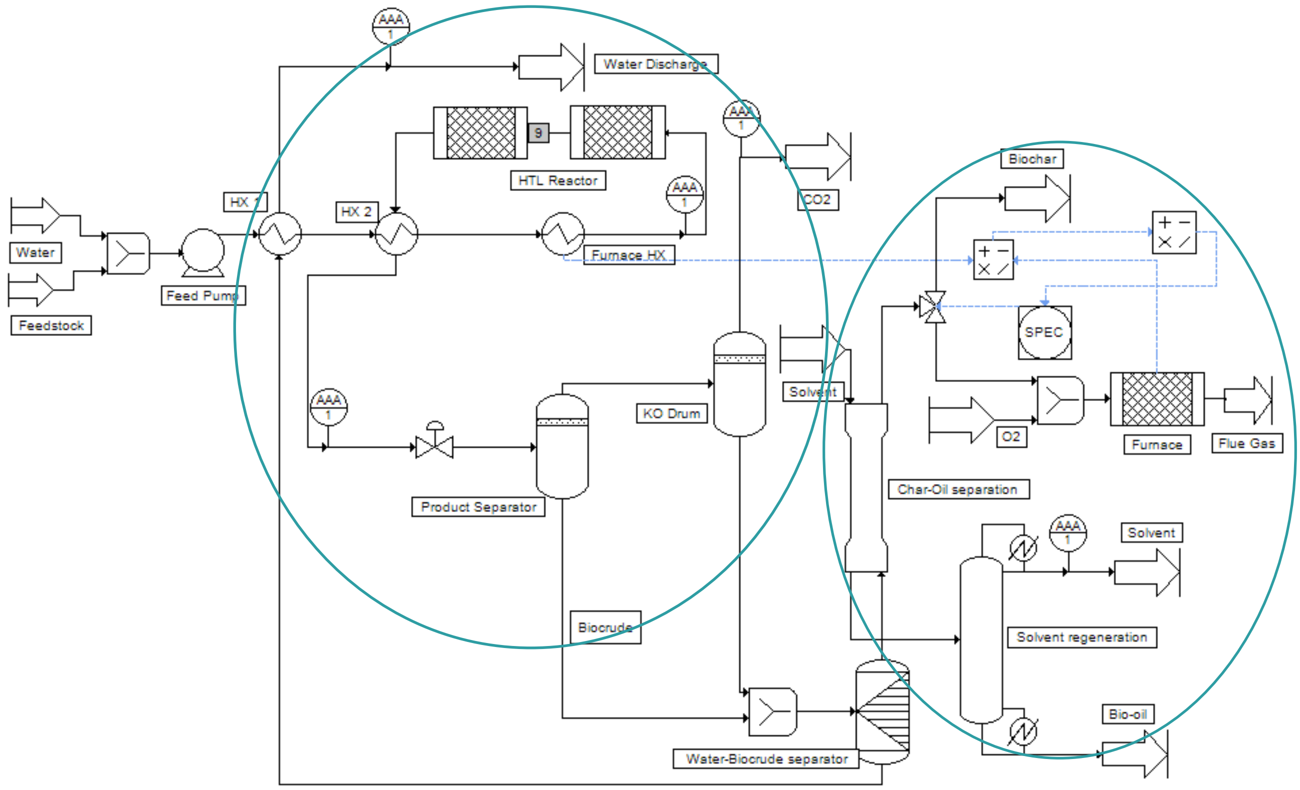

FIGURE 3: Process scheme biocrude and bio-oil plant.

Furnace. The consumption of these products is taken into account in the economic evaluation.

The HTL reactor converts the feed into a biocrude. The residence time is assumed to be 15 minutes. The products are cooled by heating the feed and the aqueous phase, the biocrude and the gas are separated in the Product Separator and the Water-Biocrude Separator. The biocrude should be maintained above 60°C to prevent plugging. Water is recycled into the process as much as possible or purged. The purged water is recycled or discharged to the waste water treatment plant, not included in the process diagram. There are some costs associated with this. Globally, the amount of water is not very large, proportional to the water entering the system. In the case of sewage sludge, the process water is locally reprocessed. The waste water treatment works in Grenoble process 240,000 m^3 per day, while the corresponding HTL plant would produce 240 m^3 per day. In the case of the blackcurrant pomace and grape marc process water is sent to external water treatment plant leading to additional costs in these cases.

4.1 Biocrude plant

The different resources were evaluated and presented below in Table 4. Biocrude is produced by processing a biomass slurry. In this study we assumed the same heat of reaction for all biomasses. The amount of biomass pumped, heated and products cooled having the same volume, the cost of the plant is insensitive to the actual biomass type. As the volumes of treated slurry are the same, the investment costs are the same. Fixed and operating costs are also the same between the different cases as they are estimated from a fixed percentage of the capital costs. The differences are in the yields and the energy content of the products. A gate fee is calculated in the cases when the biocrude production costs are higher than the reference crude oil price. The gate fee is the negative value of the feed to make sure the products can be sold without further losses. When the gate fee is lower than alternative disposal ways, the operation is beneficial. The heating values used for the energy equivalence are reported in Table 3.

The precision of the reported data does not correspond to the actual precision of the estimations. CAPEX estimations are notoriously difficult and uncertain early in the development of a technology. Typical uncertainties are in the 50% range or even higher at this stage (Dysert, 2003). Economic evaluations largely depend on business plans and on the economic structure of the exploiting organisation. The methods and results presented in this paper are fairly standard but variations exist. For these reasons it is difficult to quantify the precision of the presented results.

4.2 Bio-oil plant

The second case concerns the same plant extended with a solvent extraction unit to separate the biocrude into bio-oil and char as shown in Figure 3. A solvent is mixed into the biocrude stream to dissolve the bio-oil. Char is separated from the mixture as an insoluble part in the Char-Oil Separator. The solvent is separated from the bio-oil by distillation. The biocrude contains insoluble char, heavy oil but also light compounds (Anouti et al., 2016; Haarlemmer et al., 2016). In practice, the initial solvent will be rapidly replaced by the light compounds included in the biocrude that are separable by distillation.

The different resources were evaluated and are presented in Table 5. The investment costs are now dependant on the amount of oil produced. The production costs only concern the oil produced, the char is used as fuel for the process and is not further valorised. The heating values used for the energy equivalence are reported in table 3. For sludges 2 and 3 the heating value of sludge 1 were used.

The results show that the production costs of hydrothermal oil and vegetable oils are very similar. They are both significantly more expensive than fossil fuel oil. There remains a major issue with the quality differences between the products. Fuel oils are refined products ready for use. Vegetable oils need some upgrading, but this process is well understood. Hydrothermal oils are probably slightly too viscous and acidic to directly replace heavy fuel oil. The actual market value of these oils are unknown.

4.3 Comparison with existing practices

Table 6 presents an overview of the gate fees required for economic viability for both solutions. These calculated values are compared to typical values found for waste incineration and agricultural use of sewage sludge. Hydrothermal liquefaction, combined with the sales of biocrude or bio-oil, can indeed be economically viable. The work shows that gate fees are comparable to current waste incineration

TABLE 4: Results of the economic evaluation of an HTL biocrude plant.

	Blackcurrant pomace	Grape Marc	Sludge 1	Sludge 2	Sludge 3
Investment (CAPEX) - M€			5.55		
Heat Exchangers			0.86		
Pumps			0.70		
Reactor			1.30		
Storage			0.35		
Utilities and terrain			1.79		
Electricity consumption - MW	0.3	0.3	0.3	0.3	0.3
Fixed costs - M€ $year^{-1}$	0.61	0.61	0.61	0.61	0.61
Variable costs - M€ $year^{-1}$	0.35	0.35	0.35	0.35	0.35
Minimum selling price - € $tonne^{-1}$	462	787	494	383	598
Minimum selling price - € GJ^{-1}	14.4	26.3	19.0	12	46.0
Gate fee - € $tonne^{-1}$ dry matter	108	154	131	89	173
Crude oil (Brent 2015) - € GJ^{-1}			6.7		
Fossil coal (2015) - € GJ^{-1}			1.5		

TABLE 5: Results of the economic evaluation of an HTL bio-oil plant.

	Blackcurrant pomace	Grape Marc	Sludge 1	Sludge 2	Sludge 3
Investment (CAPEX) - M€	6.53	6.36	6.50	6.56	6.30
Heat Exchangers	0.86	0.86	0.86	0.86	0.86
Pumps	0.70	0.70	0.70	0.70	0.70
Reactor	1.30	1.30	1.30	1.30	1.30
Storage	0.35	0.35	0.35	0.35	0.35
Utilities and terrain	1.90	1.90	1.90	1.90	1.90
Bio-oil extraction	0.42	0.25	0.39	0.45	0.19
Electricity consumption - MW	0.35	0.35	0.35	0.35	0.35
Fixed costs - M€ $year^{-1}$	0.81	0.79	0.81	0.81	0.72
Variable costs - M€ $year^{-1}$	0.56	0.56	0.56	0.56	0.56
Minimum selling price - € $tonne^{-1}$	1040	1990	1083	1000	1390
Minimum selling price - € GJ^{-1}	30	57	31	29	40
Gate fee - € $tonne^{-1}$ dry matter	202	228	204	200	196
Heavy Fuel Oil - € GJ^{-1}			9.4		
Domestic Fuel Oil - € GJ^{-1}			15		
Crude Palm Oil - € GJ^{-1}			17 (September 2017)		
Soy Bean Oil - € GJ^{-1}			30 (September 2017)		

TABLE 6: Comparison between established gate fees and the projected gate fees in this study.

	Blackcurrant pomace	Grape Marc	Sludge 1	Sludge 2	Sludge 3
Biocrude					
Gate fee - € $tonne^{-1}$ dry matter	108	154	131	89	173
Gate fee - € $tonne^{-1}$ wet	44	77	22	5	5
Bio-Oil					
Gate fee - € $tonne^{-1}$ dry matter	202	228	204	200	196
Gate fee - € $tonne^{-1}$ wet (hum)	82 (60%)	114 (50%)	35 (83%)	12 (94%)	6 (97%)
Typical agro sludge disposal (Ferry & Wiart., 2002)	200 - 400 € $tonne^{-1}$ dry matter or 25 – 100 € $tonne^{-1}$ wet at 75% humidity				
Waste incineration (ADEME, 2015)	200 - 300 € $tonne^{-1}$ dry matter or 100 - 150 € $tonne^{-1}$ wet at 50% humidity				

plants, in the range of 100 to 150 € per tonne of wet waste. The organic matter has a typical humidity of 50 wt.% leading to a cost of 200 to 300 € per tonne of dry matter. Agricultural sludge disposal costs typically between 200 and 400 € per tonne of dry matter. Hydrothermal liquefaction costs are at the low end of agricultural disposal costs.

As is was mentioned before it seems rather unpractical to have large plants, beyond 1 to 10 tonne dry matter per hour. Small scale production facilities, close to the resources such as food processing factories and population centres appear to be an obvious application for hydrothermal liquefaction. Economic viability will necessarily come via gate fees to compensate for these relatively small capacities. Figure 4 presents the required gate fee for the treatment of blackcurrant pomace as a function of the production capacity for the two case studies.

Figure 4 shows that the required gate fee increases (due to increasing production costs) rapidly with decreasing production capacity. The reference point are placed well below the typical incineration or agro disposal fees so even with escalating construction costs the plant may still be viable.

5. CONCLUSIONS

Food processing wastes and sewage sludge are interesting carbonated resources. Rather than looking for low value valorisation, more value can be added to these waste steams by hydrothermal liquefaction. The technology is not able to compete economically with the fossil energy industry. Most organic waste producers are used to pay to dispose of these waste. The cost varies greatly with the nature of the waste and with the local legislation. With gate fees in the 50 to 130 € $tonne^{-1}$ dry matter range hydrothermal liquefaction can produce liquid fuels that can compete with fossil fuels. Significant uncertainties subsist however about the quality of the fuels and their compatibility with existing applications.

Not all resources are however equally suited for this technology. The results are however variable and optimal conditions need to be found for each resource. Lignin rich resources such as grape marc yield much lower oil yields at low temperatures. These resources should be processed

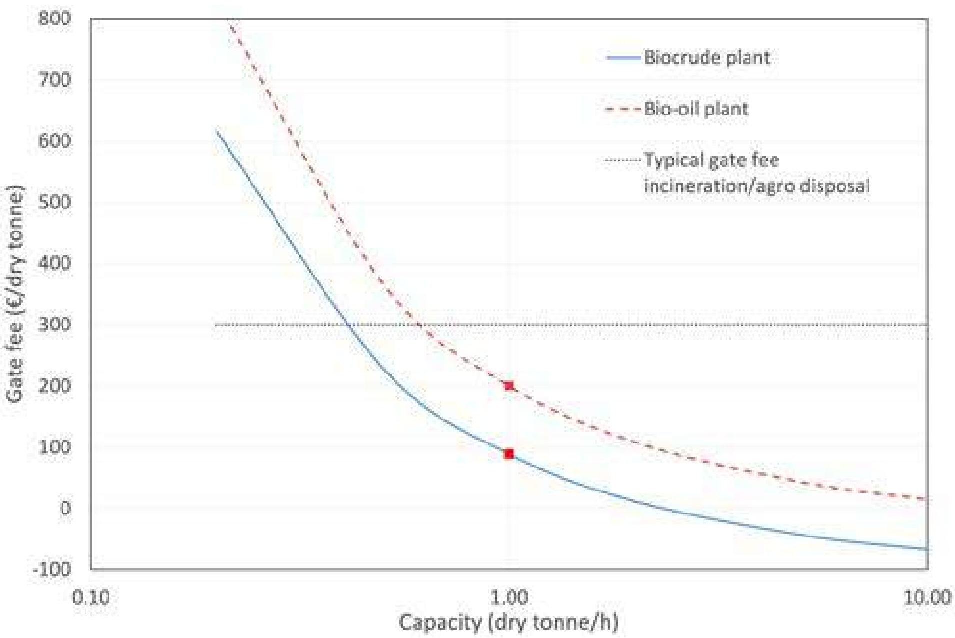

FIGURE 4: Sensitivity of the gate fees to the production capacity for the blackcurrant pomace resource (reference cases indicated with ■)

at higher temperatures (Pedersen et al., 2016). High ash resources, low in organic material such as digested sewage sludge are less interesting. The oil yields are low and the biocrude is of low quality as it is very rich in inorganic material.

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RECENT DEVELOPMENTS ON THE USE OF DRAINAGE GEOCOMPOSITES IN CAPPING SYSTEMS

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ABSTRACT

This paper reviews the different requirements imposed from the European harmonised standards for the use of different types of drainage geocomposites in the capping systems of landfills. The results of specific experimental investigations on the behaviour of drainage geocomposites in such applications are presented.

1. INTRODUCTION

The need to follow the regulations foreseen by the European Union – Council Directive 1999/31/EC (hereforth referred to as EUCD) related to the design and maintenance of landfill, and most of all by the local national codes that have been adopted country by country starting from the Directive itself, can sometimes represent a technical and economical challenge for designers and for landfill owners.

The EUCD defines a landfill as a site for the deposit of the waste onto or into land (i.e. underground), including internal waste disposal sites (i.e. landfill where a producer of waste is carrying out its own waste disposal at the place of production), and a permanent site (i.e. more than one year) which is used for temporary storage of waste. In particular, the Landfill Directive defines three different categories of wastes: hazardous waste, non-hazardous waste and inert waste.

As already known, the function of the capping system is to: insulate waste from the external environment; control rain water from entering into the landfill body; prevent surface water from entering into the landfilled wasteand avoid the risk of subsidence and sliding.

According to the EUCD, the materials to collect and drain biogas, the creation of the barrier system and the removal of rainfall water, should consist of thick layers of natural materials, respectively gravel and compacted clay. In practise, this is sometimes very difficult to achieve, as natural materials may not be available close to the site.

Depending upon the type of waste, different systems are proposed to allow gas drainage (required for non-hazardous wastes only), an artificial sealing liner (required for hazardous wastes only), an impermeable mineral layer (required for every type of waste), a drainage layer > 0.50 m and a top soil cover > 1.00 m, both required for every type of waste.

It's important to note the following: although the Directive prescribes a gas drainage layer for non-hazardous waste in general, in reality it is necessary for all landfills receiving biodegradable waste.

Wherever the national regulations allow it, the materials listed above are widely substituted, with geosynthetics (see for example the case history described by Cazzuffi et al., 2009).

The scenarios foreseen by the Italian code derived from the European directive and the corresponding sections using geosynthetics are shown in Figure 1.

There are several reasons to substitute natural material with geosynthetics. The most relevant are technical reasons: the stratigraphy foreseen by the European directive and by the Italian regulation are sometimes not compatible with the geometry of the landfill bodies. There are some cases where the landfill geometry has been designed and finalised well before the Directive was active, and therefore the surfaces are too steep and long to guarantee the sta-

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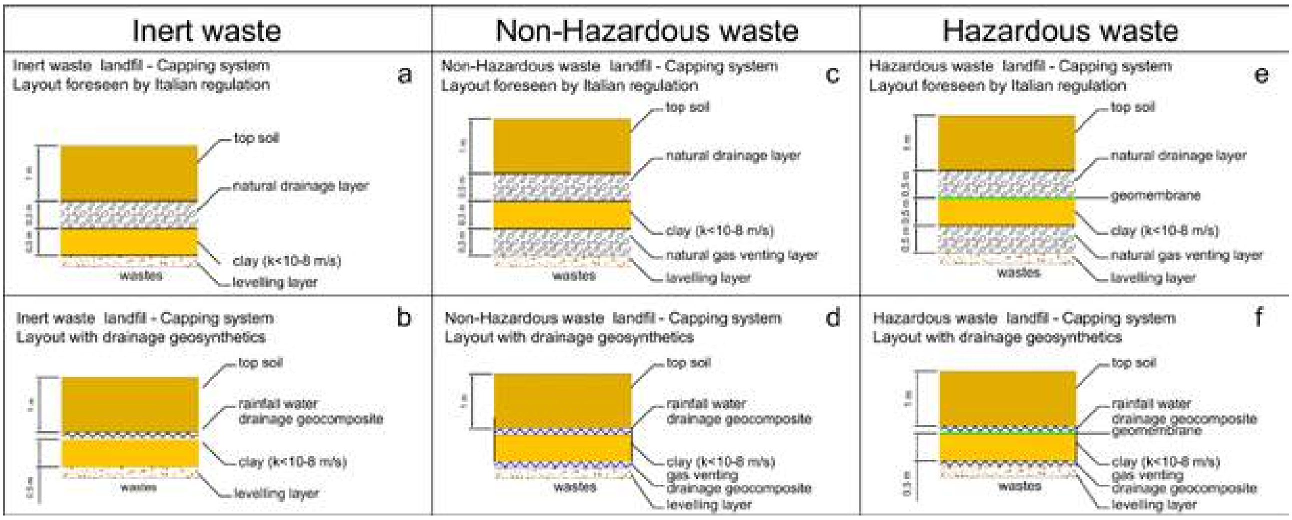

FIGURE 1: Capping system: layout foreseen from Italian code for inert waste landfill (a), non-hazardous waste landfill (c) and hazardous waste landfill (e); possible alternatives with geosynthetics for inert waste landfills (b), non-hazardous waste landfills (d) and hazardous waste landfills (f).

bility of the natural layers. This type of problem is amplified whenever the site is in a seismic areawhen the Eurocodes 7 and 8 have to be followed.

Another reason is economical: granular materials used to guarantee the proper drainage have to be clean coarse sands or gravel. The need toimport large quantities of an expensive material may present difficulties to collect all the required material from the same quarry or source. This could lead to difficulties to guarantee a proper quality control on site, making this solution extremely expensive (Riot and Cazzuffi, 2013).

For the drainage systems, in particular, the continuous evolution of the manufacturing process, together with a wider range of laboratory tests, allows us to obtain increasingly higher performance with evident advantages; not only in economic terms but also from the environmental point of view as less natural materials from quarries and subsequent excavation works are required.

The general rules of the Directive have been adopted in different ways in European countries, using more or less restrictive approaches. In Italy for example, the drainage layers for gas and water have to be granular layers having a minimum thickness of 0.50 m, and as for the water drainage layer, it is recommended that no water head should develop within the granular layer, but no indication is given in terms of hydraulic conductivity. The same lack of information about the hydraulic conductivity is also present for the gas venting layer (Recalcati and Salis, 2012). Another point that can be made is the lack of any clear reference to the possibility to adopt geosynthetics; a method widely used even well before the Directive was written. Because of this situation there are local authorities that are not allowing the use of alternative solutions to the natural layers.

The paper describes in detail some examples of technical developments related to testing and installation of drainage geocomposites in the drainage systems for landfills capping.

2. HARMONIZED STANDARDS FOR DRAINAGE GEOCOMPOSITES

European standards on geotextiles and geotextile-related products are developed by CEN/TC 189 Geosynthetics. International standards are developed by ISO/TC 221 Geosynthetics.

Over the past 20 years both committees have issued more than 100 standards and amendments to standards. In particular, CEN TC 189 is the Technical Committee taking care of harmonized European product standards (hENs) related to geosynthetics.

The scope of this group is the standardization related to geosynthetics; terminology, sampling before testing, identification and marking rules, test methodsand requirements related to their intended use.

The TC is currently divided into 6 Working Groups (WG) each relating to specific items related to geosynthetics definition and properties:

- CEN/TC 189/WG 1 - Geotextiles and geotextile-related products - General and specific requirements of harmonized technical specifications;
- CEN/TC 189/WG 2 - Terminology, identification, sampling and classification;
- CEN/TC 189/WG 3 - Mechanical testing;
- CEN/TC 189/WG 4 - Hydraulic testing;
- CEN/TC 189/WG 5 – Durability;
- CEN/TC 189/WG 6 - Geosynthetic barriers - General and specific requirements.

In particular, WG1 is the working group taking care of the development of Harmonized Standards for geotextiles and geotextile-related products to a specific field of application.

During recent years, two specific European harmonized standards have been developed for application of geosynthetics in waste disposals (EN 13257: 2016 - Geotextiles and geotextile-related documents — Characteristics required foruse in solid waste disposals and EN 13265: 2016 - Geotextiles and geotextile-related products — Characteristics required for use in liquid waste containment projects).

Moreover, in the specific case of drainage systems, the harmonized standard EN 13252: 2016 (Geotextiles and geotextile-related products — Characteristics required for use in drainage systems) should be used.

2.1 EN 13252: 2016 - Geotextiles and geotextile-related products - Characteristics required for use in drainage systems

This standard specifies characteristic properties, test method limits and their significance level for drainage systems.

The main functions of geotextiles and geotextile-related products used in drainage systems are filtration, separation and drainage. The specification defines which functions and conditions of use are relevant (see Table 1 for drainage function).

The manufacturer of the product shall provide the necessary data based on the requirements and test methods described in this European Standard.

In particular, the characteristics of a product are divided in:

- A: essential characteristic relevant to all conditions of use;
- S: relevant to specific conditions of use.

3. DRAINAGE GEOCOMPOSITES IN CAPPING SYSTEMS

3.1 Design aspects

Geosynthetic products that can effectively substitute granular drainage layers are the so-called drainage geocomposites. They are characterised by a draining medium, capable to allow a planar flow within its surface (drain core), and one or two filter layers (geotextiles) bonded to the surfaces of the geonet, whose function is to protect the passage of the fluid prevent the solid particles from entering the drainage medium and clog it.

The use of geosynthetics first of all gives to the owner and to the designer proven and certified information about the water flow capacity of the product. To have the same type of information with granular materials it would be necessary to run a larger number of tests and there can be uncertainties due to the possible variation of the properties of the gravel used.

Furthermore, the capping system has a reduced weight, a lower thickness (thus allowing an increase of the landfill volume) and the overall stability of the capping system can be improved.

The design of a drainage system using geosynthetics is based on the evaluation of the effective discharge capacity of the geocomposite and of the required water flow capacity under the design conditions. The drainage capacity of geosynthetics is evaluated though specific laboratory tests, while the required design flow rate has to be evaluated on the base on hydrologic studies.

As previously stated, the granular layer used (either natural or synthetic) should guarantee that no hydraulic head is developed on top of it in case of rainfallIt is then necessary to choose a product capable to guarantee the discharge of the whole water amount that it is reasonable to assume will reach the drainage layer after migrating from the top soil layer.

If the drainage capacity is not adequate, the excess of water can cause the development of an uplift due to the water itself, and as a consequence a dramatic reduction in the functional properties at the interface between the topsoil and the drainage layer. Then, the stability of the topsoil can be compromised.

Precipitation intensity should be determined with a return time that is considered relevant and sufficient from the designer, and it should be carefully chosen. It is the task of the designer to evaluate if a persistent, long-lasting but not heavy rainfall is more dangerous than a heavy storm. In the second case the amount of water most probably will never reach the drainage layer but will runoff over the surface.

The critical rainfall intensity has to be evaluated to also consider the geometry of the surface and of the nature of the soil, or better of its hydraulic conductivity.

Once the critical rainfall intensity has been evaluated, it is then necessary to give an estimation of the percentage of water that will reach the drainage layer, taking into account the water that will be lost due to evapotraspiration, runoff and absorption in the soil.

The hydraulic flow rate within a granular layer can be determined in a complex way on the base of the geometry of the piezometric surface (Figure 2). However, if the length of the granular layer is much larger than its thickness, the real shape of the piezometric surface is almost linear.

A simplified (yet conservative) approach, based on the assumption that the hydraulic flow within the granular layer is linear and uniform, is suggestible and strongly recommendable.

The maximum flow rate that can be transmitted by a drainage layer characterised by a hydraulic conductivity ‘k’ under a gradient ‘’ can be calculated using the flow equation proposed by Darcy:

$$Q = k \cdot A \cdot i \tag{1}$$

The flow rate corresponding to 1 meter width of draining layer is whose thickness is

$$t\,Q = k \cdot t \cdot 1,00 \cdot i \tag{2}$$

The transmissivity is equal to the specific flow rate divided by the gradient

$$\Theta = k \cdot t \cdot 1,00 \tag{3}$$

The flow rate of a drainage geocomposite is measured according to EN ISO 12958. The flow rate per unit width is determined by measuring the quantity of water that passes through a test specimen in a specific time interval under different normal stress and different hydraulic gradient (typically i=1.0, i=0.50 and i=0.10).

The hydraulic gradient is defined as the ratio between the difference in the piezometric water head between the

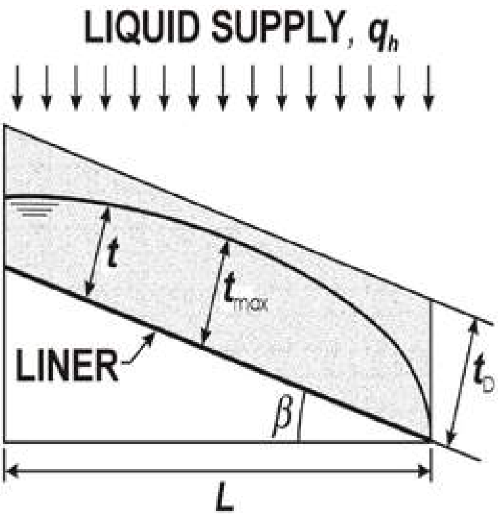

FIGURE 2: Piezometric surface within the granular medium of a capping system (Giroud e al., 2000).

upstream and downstream edges of the slope and the length of the slope itself; in the hypothesis of a steady flow, it is equal the ratio between the difference in level (DH) and the length of drainage (L), that corresponds the sinus of the slope angle.

$$i = \sin\beta \tag{4}$$

The results of the tests are represented in diagrams, having the normal pressure on the x axis and the flow rate measured in the lab on the y axis; usually the y axis is represented in logarithmic scale.

If the design gradient "' is represented on the flow rate diagram, then it is enough to choose the geocomposite that under the same gradient and the same normal pressure can guarantee a flow rate equal or greater to the design one Q.

If the design gradient "' is different from the values of the flow rate diagrams, it is possible to calculate the equivalent flow at the specific gradient by knowing the actual values at a different gradient using empirical formulas (Cancelli and Rimoldi, 1989).

Replacement of a natural granular layer with a geosynthetic can be justified only if it is possible to prove that the latter can guarantee at least the same performance not only in the short term, but most of all in the long term. It is important to remember that EN ISO 12958 gives information about the short-term behaviour of the geocomposite, being an index text, but by itself it is not sufficient to asses any long-term performance.

To take into account the real long-term performance of a draining geosynthetic under a constant normal load, it is necessary to apply factors of safety to the required flow rate in order to define an allowable flow rate.

In case of long steep slopes, the state of stress to which the drainage layer will be subjected has also a tangential component that needs to be taken into account (Muller et al, 1998; Yeo and Hsuan, 2007).

A fundamental parameter, sometimes neglected in design, is the long-term compressive creep resistance of the geocomposite (see also Cazzuffi and Recalcati, 2016). The geosynthetic shall be capable to resist to high pressure for short duration (operation machineries passing over the material during installation) and lower but long-lasting pressures during the whole design life (both normal or inclined).

3.2 Experimental investigations on the behaviour of drainage geocomposites in capping systems

Geocomposites have been successfully designed as surface water removal layer in landfill final covers or as gas venting layer for decades. The most critical engineering property of a geocomposite is its in-plane flow capacity under design loads and site-specific boundary conditions. The design parameter used to quantify the in-plane flow capacity is either the flow rate per unit width of the geosynthetic or hydraulic transmissivity (flow rate per unit width of geosynthetic and per unit of hydraulic gradient "i"). Transmissivity is applicable to laminar flow conditions (EN ISO 12958) and it is defined as:

$$\Theta = k_p \cdot t = \frac{q}{i} \tag{5}$$

where:
Θ = Hydraulic transmissivity ($m^3/s/m$)
q = Flow rate per unit width ($m^3/s/m$)
k_p = In-plane hydraulic conductivity (permeability) (m/s)
i = Hydraulic gradient (-)
t = Geocomposite thickness (m)

EN ISO 12958 covers the procedure for determining the flow rate per unit width within the manufactured plane of geosynthetics under varying normal compressive stresses and a constant head. This test method is limited to geosynthetics that allow continuous in-plane flow paths to occur parallel to the intended direction of flow.

The flow rate per unit width is determined by measuring the quantity of water that passes through a test specimen in a specific time interval under a specific normal stress and a specific hydraulic gradient. The hydraulic properties are measured with a testing equipment derived from that originally used by Darcy to study the water permeability of soil (Figure 3).

In order to measure the drainage capacity of the geonets and of the geocomposites the test apparatus is capable of applying differing values for the hydraulic gradient "i", as well as for the applied normal pressure, so as to simulate different possible operating conditions (varying overburden pressures).

According to EN ISO 12958, the geocomposite specimen may be tested in accordance to specific project boundary conditions that consist of:

- Geocomposite between two stiff HDPE liner (abbr. H/H);
- Geocomposite between one stiff and one layer of soil (abbr. S/H);
- Geocomposite between two soil boundaries (abbr. S/S).

Whichever is the type of test performed, the result obtained is an index value, representing the behaviour in the short term; however, a designer needs to have a value

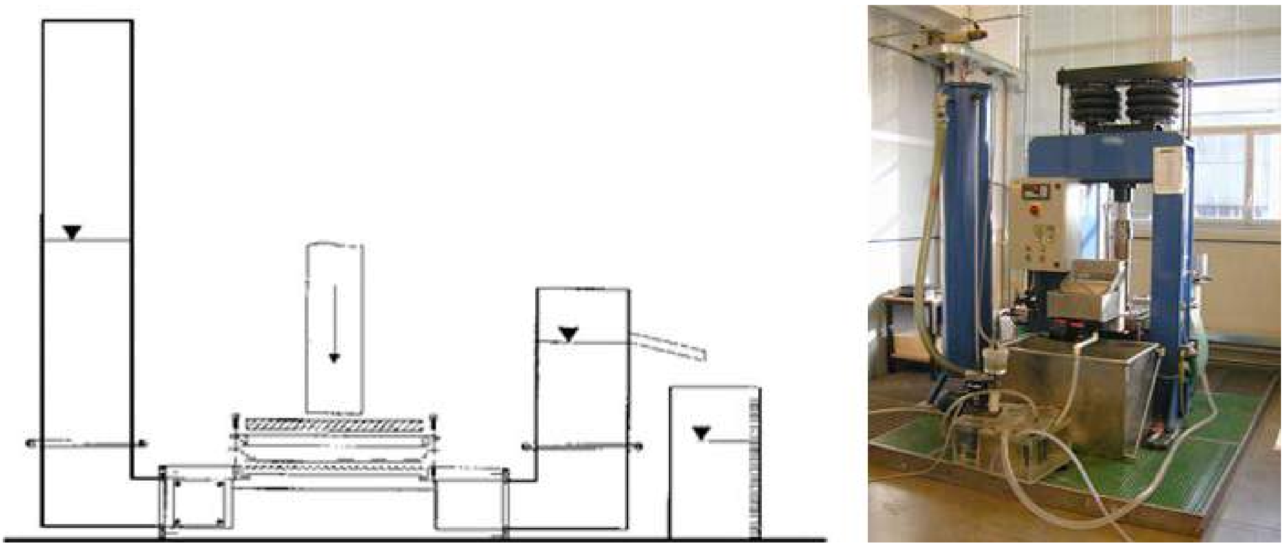

FIGURE 3: A constant head (in-plane) flow rate testing device for the evaluation of the drainage capacity of drainage geocomposites under normal pressure and for different hydraulic gradients.

that is representative of the performance of the product for the whole design life, or in other words a long-term allowable flow rate.

GRI Standard GC8 (2001) "Determination of the Allowable Flow Rate of a Drainage Geocomposite" presents a possible methodology for determining the allowable flow rate of a drainage geocomposite; from which the resulting value can be used directly in a hydraulic-related design. The method is based upon the concept of identifying the reduction factor for creep, chemical and biological clogging that may affect the long-term performance of the geocomposite.

The definitions and symbols specified in the GRI Standard GC8 apply. The fundamental equation for the evaluation of the Allowable Flow Rate Q_{allow} is:

$$Q_{allow} = q_{100} \cdot \left[\frac{1}{RF_{CR} \cdot RF_{CC} \cdot RF_{BC}} \right] \tag{6}$$

where:
Q_{allow} = allowable Flow rate
q_{100} = initial flow rate determined under simulated condition for 100 hours duration
RF_{CR} = Reduction Factor for creep to account for long-term behaviour
RF_{CC} = Reduction Factor for chemical clogging
RF_{BC} = Reduction Factor for biological clogging

It is very well known that, in the design by function approach, a drainage geocomposite must meet the following equation:

$$FS = \frac{Q_{allow}}{Q_{reqd}} \tag{7}$$

where:
Q_{reqd} is a required (or design) flow rate; the required flow rate can be determined from a water balance model such as the HELP model or other well-documented methods;
FS is the overall safety factor; generally, for landfill drainage, it is recommended a value of Safety Factor between 2 to 3 be used.

As seen before, the first aspect is measuring the flow rate under at different gradients but under a specific load condition (contacts and surcharge) with a test lasting at least 100 hours (Figure 4). The extended duration of the tests allows us to evaluate the long-term compressibility of the product; this is particularly important for geocomposites showing a brittle behaviour after a period longer than the normal duration of the transmissivity test.

To determine the Creep Reduction Factor, according to the standard EN ISO 25619-1:2008 "Geosynthetics -- Determination of compression behaviour -- Part 1: Compressive creep properties" the drainage core is placed under compressive stress and its reduction in thickness (deformation) is monitored over time.

Creep reduction factor RFcr is determined from 10,000 hours compressive creep data. In the absence of 10,000 hours creep data, designers must assess the applicability of the geocomposite with respect to structural stability under sustained loads.

The reduction factor for creep of the core is interpreted according to the following formulas, after Giroud et al. (2000) and they are summarized in the equation below:

$$RF_{CR} = \left[\frac{\left(t_{CO} / t_{original}\right) - \left(1 - n_{original}\right)}{\left(t_{CR} / t_{original}\right) - \left(1 - n_{original}\right)} \right]^3 \tag{8}$$

where:
RF_{CR} = reduction factor for creep
$t_{original}$ = original thickness (m)
t_{CO} = thickness at 100 hours (m)
t_{CR} = thickness at >>100 hours, e.g., at 10.000 hours (m)
$n_{original}$ = original porosity

$$n_{original} = 1 - \frac{\mu}{\rho \cdot t_{original}} \tag{9}$$

where:
m = mass per unit area (kg/m²)
r = density of the formulation (kg/m³)

A creep curve where percentage thickness retained is plotted against a specific normal stress over time; if there is a linear-Log relationship between percentage thickness retained and time, this linear-Log relationship can be extrapolated to design life of a project to obtain thickness and reduction factor at, e.g., 30, 50 or 100 years (Figure 5).

There are specific conditions that can occur either during the installation of a product or during the lifetime of the product that cannot be simulated with the conventional creep test, but can be derived by a simple modification of it and can give important information to the designer or the job director in the choice of a product or at least in the procedure to be followed during installation.

In normal conditions, the surcharge applied to a drainage geocomposite for a landfill capping does not exceed 20-50 kPa. However, it is possible that, for many reasons, during installation a higher state of stress is applied for a short time to the product.

This can happen for example during the installation of the geocomposite if any site equipment remains on top of the geocomposite or on top of a thin layer of soil. The equipment can apply a surcharge at least equal to 100-200 kPa; after the removal of this surcharge the geocomposite should come back to a thickness as close as possible to the one it would have without the increased surcharge.

Different products can bring quite surprising results; in the following chart the results of a test performed on a PA monofilament geocomposite having an initial thickness of

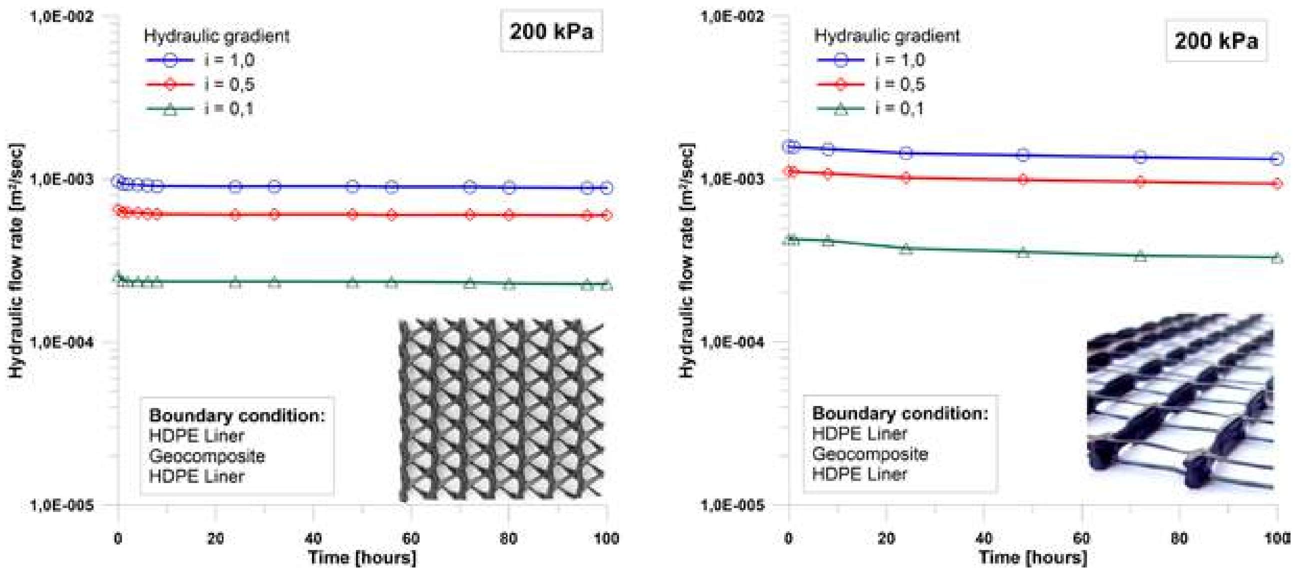

FIGURE 4: Long-term hydraulic flow rate on different geocomposites at 200 kPa.

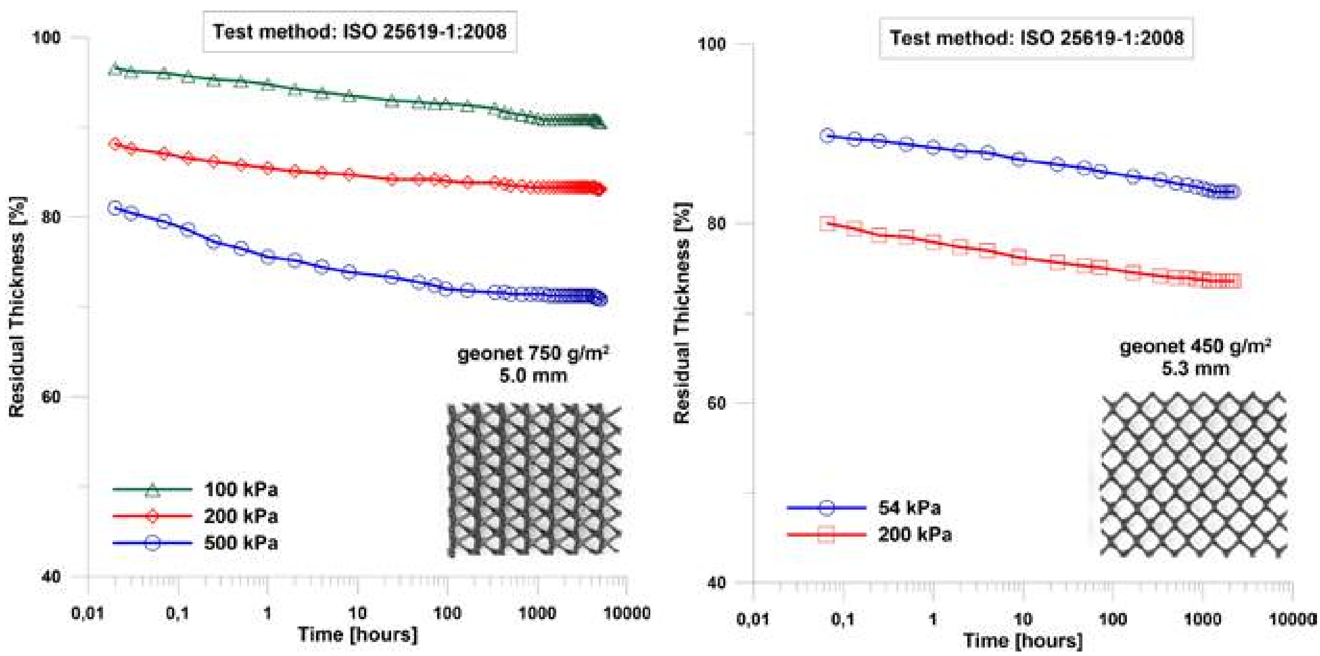

FIGURE 5: Creep curves obtained from compressive creep tests on different types of geonet.

about 20 mm is shown. Five different samples were tested; four were subjet to a surcharge of 50 kPa for over 100 hours; in one case a surcharge of 200 kPa was applied for 1 hour and then removed rapidly reaching a constant surcharge of 50 kPa (Figure 6).

By comparing the results, the presence of a residual plastic deformation in the last case that is not recovered even after 1000 hours is evident.

4. CONCLUSIONS

The use of drainage geocomposites in landfill capping systems represents a solution technically valid and sustainable; this is confirmed by thousands of examples all over the world.

European and International standards on geosynthetics are considered to be in continual progress. It is of fundamental importance that national technical regulations are continuously updated in order to take into account the progressive evolution typical of the development in manufacturing technologies of those materials and also in the related preparation of European and International standards on geosynthetics.

The influence of the the long-term water flow capacity as well as the compressive behavior under normal or inclined state of stress has been shown: these aspects should be evaluated in future revisions of the harmonized standards on drainage geocomposites.

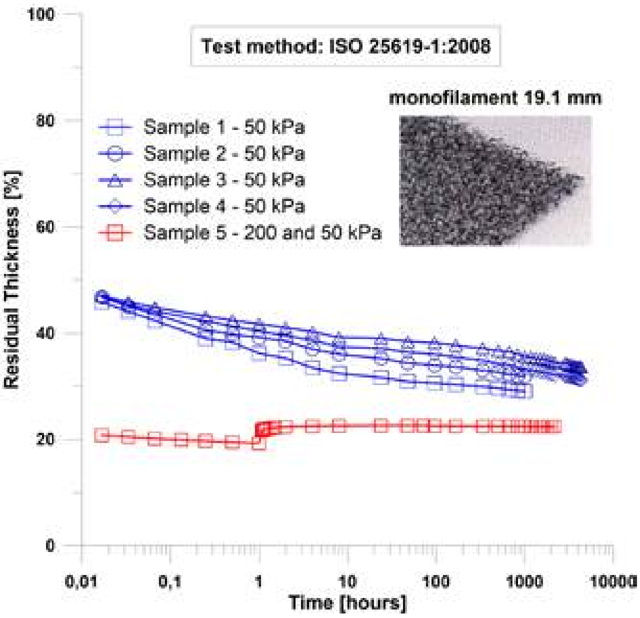

FIGURE 6: Creep curves obtained from compressive creep tests on monofilament geocomposite for different test procedures.

In order to avoid a wrong choice of drainage geocomposites because of an excessively restrictive interpretation of the rules and of the European Directive, it is necessary to test specific conditions that can occur during the installation of a product or during the lifetime of the product. This is important particularly when the real situation cannot be simulated with conventional creep tests, but it can be derived by a simple modification of it. This can give important information to the designer or the project manager for a correct choice of a product or at least in the procedure to be followed during installation.

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BIOREACTOR LANDFILLS: COMPARISON AND KINETICS OF THE DIFFERENT SYSTEMS

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ABSTRACT

The need for more sustainable landfilling has increased interest in bioreactor landfills as a suitable tool for optimising degradation processes. Bioreactors can be categorised as follows: anaerobic, aerobic, semi aerobic and hybrid. The choice of a specific bioreactor can be strongly influenced by the desired treatment objectives (i.e., energy recovery, increased rate of waste stabilisation, washing) as well as by the specific site conditions (e.g., waste characteristics, climate and social/economic situation, regulations). However, the increased rate of waste stabilisation should be the primary driving principle in the bioreactor landfill design (Cossu, 2010). Full-scale bioreactor landfills are still uncommon and one of the reasons is the perception that the effectiveness of this technology is not well demonstrated. This paper aims to contribute to filling this knowledge gap by analysing and comparing the lab scale applications of different types of bioreactors available in the literature and providing a survey of the different methods by considering their respective advantages and disadvantages. Qualitative analysis of the main types of bioreactor landfills is provided according to a few selected characteristics (i.e. energy recovery, biochemical kinetics, technological complexity, costs). Considering landfill sustainability, the discussion is primarily focused on the quantification of the stabilisation capability of the different bioreactors which is calculated in terms of COD and ammonia removal kinetics. The results demonstrate that the optimisation of COD removal kinetics is the highest in aerated bioreactors, while ammonia removal kinetics is maximum in hybrid bioreactors (i.e., 6 and 10 times higher, respectively, compared to the anaerobic bioreactors).

1. INTRODUCTION

Although recent legislation tends to limit landfilling as much as possible, it will continue to play a key role in future modern solid waste management systems (Cossu, 2012). Even with circular economy thinking, the zero-waste concept cannot currently be realistically achieved and a final disposal step is needed for residues that cannot be technically or economically exploited. Landfilling assumes the role of providing a final sink to close the loop in the material cycle in order to isolate, from the environment, concentrated residual waste that are no longer usable. In particular sustainable landfilling has been introduced as a system that should be operated in such a way to minimise the emissions potential by achieving waste stabilisation as quickly as possible in order to preserve the next generations from potential environmental risks and remediation costs.

From an environmental and health point of view, the most problematic issue dealt with in a landfill system is the putrescible fraction of waste. This fraction is responsible for the main long-term impacts, including methane and carbon dioxide emissions (contributing to the greenhouse effects and ozone depletion) and leachate emissions resulting in surface and groundwater pollution as well as soil pollution. In order to achieve the sustainability requirements, several strategies can be adopted to control the effects caused by the landfilling of biodegradable waste. These control strategies can be implemented before landfilling by means of the diversion of the putrescible fraction from the waste stream going to the landfill (separate collection), thermal or mechanical/biological pre-treatment and washing of the waste, and during the operational and/or aftercare phases by using in-situ treatments approaches.

Among the other solutions, the need for the implementation of innovative landfill management techniques has increased the interest in bioreactor landfills as a viable in-situ treatment tool (Cossu, 2012; Reinhart et al., 2002).

A bioreactor landfill is typically defined as a system purposely planned and operated for the in-situ treatment

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of degradable waste with the aim of enhancing conversion processes. The possible in-situ measures include injection of air and/or water, leachate recirculation, and other combinations of in-situ treatments. These treatments create a more suitable environment for degradation processes by controlling biochemical kinetics, nitrification, moisture content, pH, redox conditions, and gas emissions.

Moisture control particularly supports the metabolic processes, nutrients transport, microorganisms movement, and dilutes high concentration of inhibitors, while air injection speeds up the biodegradation processes and allows for the removal of nitrogen compounds (Cossu et al., 2003; Ritzkowski and Stegmann, 2013).

Bioreactor landfills can have several advantages over conventional landfills, from both an economic and environmental point of view:

- Reduce environmental impacts, by improving leachate quality and controlling landfill gas (LFG) emissions;
- The aftercare time is generally shorter due to the increased stabilisation rates therefore reducing aftercare costs and returning the site for different uses in a shorter timeframe;
- The leachate treatment is cheaper, since the in-situ treatment enhances leachate quality;
- The landfill gas (LFG) generation in an anaerobic bioreactor is enhanced;
- Refuse settlement and density are increased while less post-closure care operations are necessary (Berge et al., 2005; Omar and Rohani, 2015; Price et al., 2003; Warith, 2002).

On the other hand, a bioreactor landfill can have some disadvantages such as increased odours, physical instability of the waste mass due to the increase in moisture. Moreover, the need for aeration and/or leachate recirculation may increase capital and management costs.

According to the process, landfill bioreactors can be divided into four main types: anaerobic, aerobic, semi aerobic and hybrid. The hybrid bioreactor is a sequence of aerobic and anaerobic conditions (EPA, 2018a; Omar and Rohani, 2015).

Landfill bioreactors were mostly operated under anaerobic conditions (Price et al., 2003; Valencia et al., 2011; Vigneron et al., 2007) improving the methane generation rate, leachate quality, and reducing the period needed for long term maintenance and monitoring through recirculation, compared to traditional anaerobic landfills (Christensen, 2011). However, ammonia accumulation in leachate and the landfill body still remains one of the main challenges in anaerobic bioreactors. Furthermore, the anaerobic degradation process is still very slow.

According to the sustainable landfilling concept, the aerobic process is considered to be a better alternative to the traditional anaerobic landfills (Nikolaou et al., 2009; Read et al., 2001). Nevertheless, aerobic landfills are not always technically and economically feasible due to the need for forced ventilation systems, complex operation and management, and large energy consumption which translates to high operating and capital costs (Slezak et al., 2015). In order to overcome the cost disadvantage of forced aerated systems, the semi-aerobic landfill could be considered as an alternative solution to the aerated system (forced aeration). The semi-aerobic landfill aims to achieve aerobisation of the waste mass with a proper engineering design in which the ambient air naturally flows into the waste mass through leachate collection pipes, moved by the temperature gradient between the inside and outside of the landfill (Hanashima et al., 1981; Theng et al., 2005). Although developed at the Fukuoka University more than 20 years ago, this method is not widely spread around the world but field tested in Japan and in different on-going pilot projects in Italy, Pakistan, Iran, Nepal, Thailand, Malaysia, China, Vietnam, Samoa, and Mexico (Ministry of the Environment (Japan), 2018; JICA, 2004).

A limiting factor of aerobic bioreactors is the potential for complete inhibition of methane generation leading to the absence of any energy recovery. More recent developments have been shown in hybrid bioreactors, which are operated under various combinations of aerobic and anaerobic conditions (He et al., 2011; Long et al., 2009b; Sun et al., 2014; Xu et al., 2014). In a hybrid system, aerobic and anaerobic conditions can be purposely alternated to enhance the methane production for energy recovery and to achieve relatively faster waste stabilisation, facilitate conditions for nitrification and denitrification, improve leachate quality, reduce treatment costs (Berge et al., 2009), and potentially fulfil sustainability requirements. Bioreactor landfills are in some cases more economically advantageous than a traditional landfill (Berge et al., 2009; Hater et al., 2001; Theng et al., 2005), when accounting for landfill space recovery and a reduction in the post-closure care period (Anex et al., 1996).

A bioreactor landfill can also be operated as a flushing bioreactor. In a flushing bioreactor a large volume of water is applied in order to wash-out soluble waste constituents and accelerate waste stabilisation processes (Christensen et al., 2011). The magnitude of the flushing process is defined by the liquid to solid (L/S) ratio and according to Walker et al. (1997) the passage of approximately 4.6 times the bed volume of fluid is required to reduce leachate concentrations by two orders of magnitude, corresponding to a L/S ratio of ~3 m^3/t (Hupe et al., 2003; Christensen, 2011). However, the flushing process is strongly influenced by the solubility of various compounds in leachate (ammonia (NH_4), chemical oxygen demand (COD), Na, and Cl) (Christensen et al., 2011). Overall costs for this type of bioreactor may be two to four times higher than a conventional landfill (Karnik and Perry, 1997; Reinhart et al., 2002). Moreover the hydrodynamics of a landfill limits in time the potentialities of the flushing process. The high-water quantity addition increases the density of the waste, the hydraulic conductivity decreases and the short-circuiting phenomena tends to dominate with a limited portion of bulky waste subjected to water flow (Karnik and Parry, 1997; Walker et al., 1997).

The choice of the bioreactor landfill type is driven by the specific treatment objective to be achieved (e.g., energy recovery from landfill gas and/or leachate quality improvement) as well as by specific site conditions, such as waste characteristics, climate, and the social/economic situation. However, the sustainable landfill concept should be the driving principle in the bioreactor landfill design in order to assure the capability of achieving faster waste stabilisation (Cossu, 2010).

Several bioreactor landfill types have been successfully applied with promising results at lab or pilot scale, but full scale bioreactor landfills are still uncommon. The reasons for the lack of full scale systems are on one hand the regulatory constrains and on the other the technical complexity and cost investment associated with poorly demonstrated processes (Reinhart et al., 2002). This paper aims to review the state of the art bioreactor landfill research and elaborating on data to quantify the different kinetics with the goal of increasing the knowledge of bioreactors performances and potentialities.

Several literature lab-scale applications of different bioreactors have been analysed, compared, and an overview of different types is provided. The paper proposes a possible classification of the bioreactors, grouping them according to the main bioreactor types in literature, in order to simplify the bioreactors discussion. Advantages and disadvantages are discussed for each bioreactor category, although specific bioreactor performance should be considered individually. A qualitative analysis is then provided that takes into account some selected characteristics that are useful for the deciding on a specific bioreactor type such as methane production and energy recovery, biochemical kinetics velocity, nitrogen removal, technological complexity, and maintenance and leachate treatment costs. The ability for a bioreactor to achieve waste stabilization was quantified by the authors by mean of first-order kinetics which was determined by the approximation of the overall removal process of the selected relevant contaminants.

2. DATA COLLECTION AND ELABORATION METHODOLOGY

To provide an overall qualitative analysis of the different bioreactors types lab-, pilot- and full-scale applications of landfill bioreactors were considered. In order to quantify the stabilization performance and sustainability of the different systems, further and much more specific elaboration has performed based on lab-scale applications. Results from these studies have been published since 2005.

Variation kinetics of organic and nitrogen concentrations in leachate have been selected as criteria for the evaluation of the bioreactor stabilization performance (Ritzkowski et al., 2006) through the approximation of the combination of all the different processes involved in the stabilization of the bioreactor (e.g., biodegradation, flushing, volatilisation, etc.) in order to determine the overall first-order kinetics. These first-order kinetics were used for representing the removal process of the considered contaminants.

First-order kinetics (Heimovaara et al., 2014) for COD and ammonia conversion processes was performed by extrapolating the concentration values from graphs through the use of dedicated Matlab code and calibrating the following first-order kinetic equation:

$$C_t = C_{peak} * e^{-kt}$$

where:

C_t = concentration of considered contaminant at time t [mg/L];

C_{peak} = peak concentration [mg/L];

k = kinetic constant [d^{-1}];

t = time of process [d]

This equation is a strong approximation for a complete landfill simulation test (Fellner et al., 2009; Morello et al., 2017), but is acceptable for a qualitative discussion of the results of the investigated lab-scale tests. The concentration are clearly influenced by the water addition, but information about L/S ratio or water input were not clearly expressed in most of the cited papers. Starting from the data collection of the gas composition of the different bioreactors types, data elaboration has been performed in order to provide a graphical representation of the typical quality of the gases generated under different process conditions.

3. DISCUSSION

The results obtained from the leachate and gas literature data elaboration are presented in Table 1 and Figure 1. Peak and final concentrations (C_{peak} and C_{end}) of the considered contaminants in leachate are summarized in Table 1 for each analysed case study including the Putrescible Organic Fraction (POF) content of the studied waste. The C_{peak} has been considered as the beginning of the contaminant removal process, while the fraction of time required to reach the C_{peak} has been defined as Lag phase and has been indicated as the fraction of the whole experiment. COD and ammonia first-order kinetics have been calibrated to represent the contaminants removal process. In case where ammonia removal processes were not present, the related kinetics were not calculated.

Figure 1 summarises the typical composition of landfill gases under anaerobic, semi-aerobic, and aerobic conditions.

According to the U.S. EPA (2018), the contribution of landfills to the total non-CO_2 GHGs emissions will count for approximately 7% of the total GHGs emissions worldwide by 2030. The quality improvement of landfill gas represents a current challenge to limit the impact of landfills on climate change. The GHGs from landfill consist of primarily CO_2 and CH_4, along with several other trace gaseous components, such as non-methane volatile organic compounds (NMVOCs), nitrous oxide (N_2O), nitrogen oxides (NOx), and carbon monoxide (CO). But only CH_4 is counted towards a landfill's contribution to the GHG emissions (IPPC, 2006), being the most significant among the other emissions. In this study, the improvement of landfill gas quality performed by the landfill bioreactors has been considered only in terms of CH_4 reductions. Nitrous oxide emissions can become an issue when bioreactor landfills are implemented, since both leachate recirculation (Price et al., 2003; Vigneron et al., 2007; Watzinger et al., 2005)

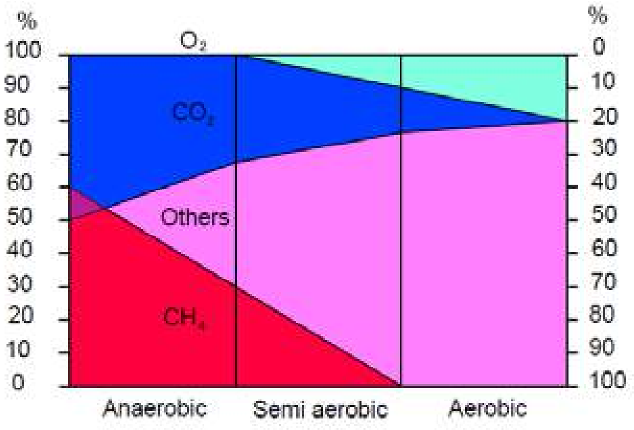

FIGURE 1: Composition of landfill gases in anaerobic, semi-aerobic and aerobic lab-scale bioreactors (Graph adapted using data from Ahmadifar et al. (2016), Borglin et al. (2004), Cossu et al. (2016, 2003), de Abreu et al. (2005); Erses et al. (2008), HUANG et al. (2008), Huo et al. (2008), Kim (2005), Nikolaou et al. (2008), Shao et al. (2008), Slezak et al. (2015), Sutthasil et al. (2014), Yang et al. (2012)).

and aeration (Berge et al., 2006; Powell et al., 2006; Tsujimoto et al., 1994) may induce N_2O production. N_2O production can result both from partial nitrification and partial denitrification (Mummey et al., 1994; Venterea and Rolston, 2000). Particularly, depending on the concentration of oxygen, the presence of oxygen during denitrification or oxygen below optimal levels during nitrification may result in the production of N2O (Berge et al., 2006; Khalil et al., 2004). A detailed study on the effects of the combination of the leachate recirculation and landfill aeration has been carry out by He et al. (2011). This study demonstrated the occurrence of N_2O under different leachate recirculation and aeration conditions. However, results showed that the conversion of the total nitrogen added to columns into N_2O occurred at a maximum of 0.18% and the significant reduction in nitrogen mass was mainly due to the production of N_2. Moreover, although some N_2O has been detected in several lab scale tests, the complete reduction of N_2O to N_2 can be expected within a full-scale landfill, due to the longer retention time of the gas (Price et al., 2003). Landfill N2O is considered globally negligible, although these emissions may need to be considered locally in case of aerobic/semi-aerobic bioreactor landfill.

3.1 Anaerobic bioreactor landfills

The anaerobic landfill bioreactor is the most common application of bioreactor systems where the biological degradation is enhanced by means of leachate recirculation and has been applied since the 80s at several landfills in USA (Reinhart et al., 2002). The literature review of several lab-scale tests identified the peculiarities which are typical in all anaerobic bioreactors, regardless of the differences in the putrescible waste content. In particular the maximization of carbon removal occurs when methanogenesis starts. Once methane gas production increases, the concentrations of COD, five-day biochemical oxygen demand (BOD_5), and volatile fatty acids (VFAs) decrease and a subsequent rise in pH to the ranges of 6.8-8 is observed. The BOD_5/COD ratio decreases from 0.8-0.4 to 0.4-0.1. The typical gas composition during the methanogenic phase shows between 30-60% CH_4 and 30-50% CO_2 (v/v) (Figure 1). These values are consistent with interstitial gas concentration in full-scale landfill bioreactors during the stable methanogenic phase (Raga and Cossu, 2014; Ritzkowski and Stegmann, 2007).

The main benefits associated with anaerobic bioreactors are both the increase in methane generation and the improvement of leachate quality compared to traditional landfills (Filipkowska, 2008; Read et al., 2001; Sanphoti et al., 2006). Sanphoti et al. (2006) compared the cumulative methane generation in anaerobic bioreactor with a traditional landfill. Anaerobic bioreactors with and without water addition generated 17 L_{CH4}/kg_{TS} and 54.9 L_{CH4}/kg_{TS}, respectively, while only 9 L_{CH4}/kg_{TS} was produced in a traditional landfill simulation.

Despite the proven advantages associated with the anaerobic bioreactor compared to the traditional landfill, anaerobic bioreactors represent the least preferable option compared to the other bioreactor types when considering the concept of sustainability. The slow anaerobic degradation is confirmed by the lower COD and ammonia removal kinetics compared to other bioreactors (Table 1) which leads to contaminant emissions lasting for several decades in case of landfill gas and even for centuries in case of leachate (Rich et al., 2008; Ritzkowski et al., 2006). In particular the treatment of nitrogen in leachate remains to be the major challenge in aftercare, which is limitedly removed by flushing processes. Moreover leachate recirculation can even enhance ammonification, resulting in an increased ammonia concentration compared to traditional landfills (Berge et al., 2006; Long et al., 2009a; Price et al., 2003). This increase often causes the partial or complete inhibition of methane production, increases the costs for leachate treatment, and may create a significant long-term impact (Cossu et al., 2016).

Slow degradation rates and ammonia persistence puts the anaerobic bioreactor far from meeting sustainability requirements, threatens the public health and the environment over the long term and increases the costs associated with aftercare (Berge et al., 2006; Giannis et al., 2008; Read et al., 2001). Moreover, considering that a robust gas collection system is required in order to achieve a high collection efficiency, this infrastructure is not always technically and economically feasible in particular in developing countries (Sutthasil et al., 2014).

3.2 Aerated bioreactor landfills

Bioreactor landfills can be treated aerobically by injecting air in order to create an aerobic environment within the waste mass and to promote the growth of aerobic microorganisms. According to Ritzkowski and Stegmann (2012), different technologies and strategies have been developed for in-situ aeration, such as high pressure aeration, low pressure aeration, and active aeration with or without off-gas extraction.

One of the first experiments on aerobic stabilization of municipal solid waste (MSW) was carried out by Stessel and Murphy (1992) to define the optimum air injection and

TABLE 1: Description of the sample (with respect to the year of the provider change).

Parameters	POF % wet weight	ANAEROBIC					AEROBIC					SEMI AEROBIC					HYBRID					REFERENCE
		C_{peak} (g/L)	C_{end} (g/L)	Lag phase %	Test duration (d)	$k \cdot 10^{-2}$ (d^{-1})	C_{peak} (g/L)	C_{end} (g/L)	Lag phase %	Test duration (d)	$k \cdot 10^{-2}$ (d^{-1})	C_{peak} (g/L)	C_{end} (g/L)	Lag phase %	Test duration (d)	$k \cdot 10^{-2}$ (d^{-1})	C_{peak} (g/L)	C_{end} (g/L)	Lag phase %	Test duration (d)	$k \cdot 10^{-2}$ (d^{-1})	
COD	80	64900	5300	33	160	1.46	44800	3840	3	160	4.5	56000	5000	6	110	1.57						Ahmadifar et al. (2015)
	44	94000	1600	13	500	0.98	68500	5000	0	250	1.59											Sinan Bilgili et al.. (2007)
	14	77000	3000	24	305	1.15											74364	1844	12	307	2.09[1]	Cossu et al. (2015)
					305												98000	2458	3	307	1.19[1]	Cossu et al. (2015)
	14	1942	500	15	310	1.79											2423	50	8	310	3.2[2]	de Abreu et al. (2005)
	25	68000	2550	15	271	0.85						54000	1000	9	271	1.45						He et al. (2012)
	70	86068	15600	15	370	0.46						82600	2270	15	370	1.9						Huo et al. (2008)
	15	79000	42050	4	690	0.1	70000	5880	15	335	0.85											Kim et al. (2005)
	60	60000	4000	8	260	1.17	42700	1160	0	260	6.51											Nikolaou et al. (2009)
	70	90200	79200	51	132	0.36											57300	3800	18	132	2.92[2]	Shou-liang et al. (2008)
	28	30000	2020	13	195	1.35	14920	460	5	195	2.34											Slezak et al. (2015)
	45	67520	900	10	630	0.5	19237	664	4	378	4.82											Erses et al. (2008)
	55	84300	75000	50	300	0.16											62000	7100	5	300	1.63[2]	Xu et al. (2014)
	30*	11100	700	5	900	0.29						9700	260	4	900	0.64						Yang et al. (2011)
MEAN						0.82					5.1					1.39					2.21	
SD						0.55					4.13					0.54					0.85	
$N\text{-}NH_4$	44	2100	1050	39	500	0.27	1700	120	26	250	3.06											Bilgili et al. (2007)
	14	155	80	8	310	0.39											304	24	8	310	3.01[2]	de Abreu et al. (2005)
	49	1340	1040	67	100	0.79											1040	120	31	100	6.22[2]	He et al. (2007)
	25	2800	1750	42	271	0.36						2100	700	42	271	0.95						He et al. (2012)
	70	2950	2490	25	370	0.06						3198	22	16	370	3.81						Huo et al. (2008)
	60	625	55	12	260	0.89	410	2	24	100	3.2											Nikolaou et al. (2009)
	60	1900	1840	18	200		1440	330	18	200	1.5	1300	22	30	200	1.27						Shao et al. (2008)
	70	3025	2550	60	132	0.3											1590	120	23	132	2.5[2]	Shou-liang et al. (2008)
	45	1064	638	42	630	0.14	407	5	10	378	1.47											Erses et al. (2008)
	30*	1580	510	9	900	0.19						1270	50	2	900	0.38						Yang et al. (2011)
MEAN						0.38					2.31					1.6					3.91	
SD						0.28					0.95					1.52					2.02	

** Estimated value. [1]Hybrid aerobic-anaerobic. [2]Hybrid facultative.*

leachate recirculation rate for degradation. Faster stabilization and improved settlement were demonstrated (Stessel and Murphy, 1992).

The positive effects of aeration on waste stabilization have been confirmed by several studies by comparing anaerobic and aerobic conditions (Table 1). Despite variation in the POF, lab-tests revealed similar results in terms of stabilization performance with higher carbon and nitrogen removal kinetics and/or shorter lag phase compared to anaerobic conditions. Aeration also lowered the leachate carbon and nitrogen values and achieved a final BOD_5/COD ratio between 0.02-0.003 (Table 1). Volatile organic acids production decreased by limiting the anaerobic fermentation processes and resulted in pH ranges between 6-8 after the initial acidic phase.

Although the aerobisation (establishment of aerobic conditions) of the waste mass prevents methane generation and thus energy recovery, there are several advantages compared to the anaerobic bioreactor landfill, which can be summarised as follows:

- Acceleration of the degradation processes in the landfill due to the higher biochemical aerobic degradation kinetics, reducing the long-term emission potential as well as the post closure management costs. In addition to faster settlement of the landfill, the site can be used for other uses in shorter time period (Yuen et al., 1999, Read, 2001);
- Higher waste settlement that generates additional landfill capacity
- Reduction of leachate volumes and enhanced remediation of recalcitrant carbon molecules and nitrogen compounds, improving the leachate quality resulting in the subsequent financial savings for secondary treatment;
- Reduction of CH_4 generation and increased carbon gasification dominated by CO_2;
- Reduction of odours generally produced from anaerobic degradation, such as hydrogen sulphide and volatile acids (Jacobs et al., 2003).

Among others, nitrogen removal is one of the most significant benefit of an aerobic system. In anaerobic landfills, nitrogen removal from leachate, in form of ammonia ion, is generally performed ex situ using costly and complex treatment plants. In order to avoid these costs, in-situ techniques have become an attractive solution and to date the most used alternative is the aeration of the waste mass to facilitate nitrification-denitrification processes (Berge et al., 2006; Shao et al. 2008). Although air injection will theoretically inhibit the denitrification process, the complete aerobisation of the waste mass is never achieved in the field. Therefore anaerobic and anoxic areas still exist inside the landfill and both processes can take place simultaneously even under low biodegradable matter conditions (Berge et al., 2006; Giannis et al., 2008; Ritzkowski, 2011; Ritzkowski and Stegmann, 2005, 2003; Shao et al., 2008). Air stripping and volatilisation can also occur since these processes are favoured by higher pH levels and temperatures reached in an aerobic system and can also be facilitated through the gas flow associated with air injection (Berge et al., 2005).

The forced air flow and the temperature rising up to more than 60°C results in a high evaporation of water and in a low quantity of leachate (Berge et al., 2005; Read et al., 2001).

Recirculation still represents an additional in situ leachate treatment tool to improve stabilization performance (Sinan Bilgili et al., 2007). In particular, the increased frequency of leachate recirculation accelerates the stabilization rate of waste, even if too much recirculation leads to saturation, ponding, and acidic conditions (Šan and Onay, 2001). Slezak et al. (2015) observed that the higher recirculation rate, increased the reduction of carbon and nitrogen parameters in leachate over a shorter time period but O_2 diffusion was limited leading to lower waste stabilization.

Aeration rates and modes influence the degradation performance differently. Slezak et al. (2010) compared stabilization performance of four aerobic lysimeters with different aeration rates obtaining similar changes in leachate parameters and demonstrated that above the minimum aeration requirements the increased rates do not provide any additional benefits. Intermittent aeration has been demonstrated to be much more effective than continuous aeration (Cossu et al., 2016; Morello et al., 2017); however optimum aeration rate is strongly influenced by oxygen consumption, which varies according to waste composition, age, and operating parameters.

Fate of metals in aerobic and anaerobic landfill bioreactors was investigated by Kim et al. (2011). Apart from the initial acidic phase, heavy metals mobility was reduced under aerobic conditions due to the high pH and positive redox conditions, affecting solubility and sorption properties. Metals were retained in the waste by sorption, carbonate precipitation, and hydroxide precipitation (Borglin et al., 2004; Giannis et al., 2008).

Typical composition of off gases reported in lab scale tests consists of 10-20% O_2 and 0-20% CO_2 (Figure 1). Methane generation is almost completely inhibited under aerobic conditions and mostly CO_2 is produced (Mertoglu et al., 2006; Slezak et al., 2015). On one hand aerobic conditions impede energy recovery while on the other environmental impacts are limited when biogas collection and control is not technically or economically feasible and uncontrolled emissions are expected. Ritzkowski and Stegmann (2007) demonstrated that in situ aeration could avoid more than 72% of the total GHG emissions occurring under anaerobic conditions.

Since the faster waste stabilization under aerobic conditions, carbon gasification is enhanced. Slezak et al. (2015) compared CO_2 and CH_4 gasification from anaerobic and aerobic lysimeters. The results showed that carbon gas released from aerobic lysimeters was about 5 times higher than that the one from anaerobic ones.

Potential disadvantages, which limit the use of this technology are the risks associated with the drying of the waste mass due to the high temperatures which may limit the highly sensitive nitrogen removal biological processes and may create an elevated temperature or fire potential. However, limited methane production, proper moisture content, and waste pre-treatment can overcome these problems (Berge et al., 2005). The high costs due to the energy requirements for compressed air injection may be limited by the appropriate selection of operating parameters, including aeration and recirculation rates, providing optimum conditions for waste decomposition, and minimizing energy consumption (Rich et al., 2008). According to the hypothetical cost model developed by Read et al. (2001), aerobic landfills could be a cost-effective solution when considering the potential recovery of valuable materials from the site, even if the operational costs and the regulatory requirements of closed landfills represents an obstacle for the full-scale development of aerobic landfills (Read et al. 2001).

Forced aeration is nowadays mostly used for remediating old anaerobic landfill, instead of being only a designed option for active landfill management. This is because aeration of old landfills represents a feasible solution to biologically stabilize waste, reduce nitrogen concentrations, and significantly control liquid emissions (Hrad et al., 2013; Ritzkowski and Stegmann, 2005, 2003). Moreover, the aeration of the landfill mass is a fundamental pre-treatment for landfill mining procedures (Raga and Cossu, 2014; Ritzkowski and Stegmann, 2012). In remediation, this technology is generally preferred over flushing: although on one hand flushing has been demonstrated to be the most effective approach (Bolyard and Reinhart, 2015), on the other hand it requires large volumes of water, off-site leachate treatment costs, and is not always technically or economically feasible (Ritzkowski et al., 2006).

Combination of both flushing and aeration processes however, have been suggested as alternative landfill management approaches by Cossu et al. (2003). PAF model was proposed as a combination of mechanical-biological Pre-treatment with Aeration and Flushing to exploit the advantages of the individual options. PAF and flushing reactors were compared to the traditional anaerobic, semi aerobic, and aerated landfills. Among the others, flushing bioreactors revealed faster kinetics and lower concentration values for carbon and nitrogen control parameters, even if the aerobic reactor presented lower residual carbon in the final solids and greater gasification. Gas generation is limited in flushing reactors since the washing of waste tends to remove the soluble biodegradable substance available to gasification (Cossu et al., 2003; Purcell et al., 1997).

3.3 Semi aerobic bioreactor landfills

The semi-aerobic system has been developed in Japan by Hanashima (1961). This system could be considered as a lower cost alternative solution to the aerobic landfill system, by providing the same benefits but lowering the operational costs by avoiding the direct air injection. Aerobic bacteria activity is improved by the natural flow of the external air into the waste mass through the leachate collection pipes, moved by the temperature gradient between the inside and outside of the landfill (Theng et al., 2005). The movement of air is particularly enhanced in winter and during the night when the temperature differences are higher. Hirata et al. (2012) observed that aerobic bacteria count in semi-aerobic systems were higher compared to anaerobic bacteria, demonstrating the effectiveness of the semi-aerobic system in the aerobisation of the waste mass.

Reproducing the aerobic process, the semi-aerobic system achieves the same benefits described for the aerated bioreactor landfill which has been proved by several lab-scale studies as well as by large-scale applications.

According to the data elaboration presented in Table 1, results show that regardless of the differences in the POF fraction of waste, the semi-aerobic system is able to achieve a much higher organic matter stabilization than the anaerobic system. The COD and ammonia concentrations in the leachate are always lower under semi-aerobic conditions, achieving higher removal kinetics. In particular, ammonia oxidation was achieved by creating aerobic conditions, while the simultaneous presence of anaerobic, anoxic, and aerobic zones within the waste mass creates conditions for denitrification of the nitrate. Shao et al. (2008) obtained higher efficiency under semi-aerobic conditions rather than in fully aerobic bioreactor since denitrification was limited due to the persistent presence of oxygen.

Despite the capability of the semi-aerobic system to partially simulate aerobic conditions, aerated bioreactors remain the best performing systems in terms of COD concentrations, degradation rates, and removal efficiencies (Table 1) (Ahmadifar et al., 2015).

A benefit of the aerobisation of the waste mass is the higher gasification occurring under semi-aerobic conditions dominated by CO_2 (Figure 1). According to Matsufuji et al. (1996) the proportion of gas to leaching emissions was 3:2 from the semi-aerobic lysimeter and 1:4 from anaerobic lysimeters. Similar results were obtained by Shimaoka et al. (2000) with a ratio of 4:1 and 2:3 under semi-aerobic and anaerobic conditions, respectively. Lavagnolo et al. (2018) achieved up to a 60% initial carbon gasification under semi-aerobic conditions compared to only 20% in anaerobic reactors.

Intergovernmental Panel on Climate Change (IPCC) guidelines (IPCC, 2006) estimates that the degradation process within a semi aerobic waste mass is supposed to occur simultaneously under anaerobic and aerobic conditions in line with the heterogeneity of the waste mass. According to this, the biogas composition in a semi-aerobic landfill is described by a CH_4/CO_2 ratio of 0.48 (Jeong et al., 2015). This value seems to align well with the majority of the values reported in the literature. The average methane concentration in the semi-aerobic process mostly ranges between 0-30% (v/v) with CO_2 and O_2 at 10-30% (v/v) and 0-20% (v/v), respectively (Figure 1).

3.4 Hybrid bioreactors

Hybrid bioreactors are conceptually based on the principle of combining a sequence of aerobic and anaerobic conditions with the purpose of achieving the benefits from both conditions in order to maximise the potential of bioreactors in terms of sustainability and/or methane generation. In particular methane production and energy recovery are maximized during the anaerobic phase while during the aerobic phase the nitrification-denitrification processes are enhanced for complete removal of nitrogen from landfill. Overall waste stabilization is achieved in a shorter period of time by improving the degradation of recalcitrant compounds such as lignin and aromatic substances

(Berge et al., 2006, 2005; He et al., 2011; Long et al., 2009b; Ritzkowski and Stegmann, 2013; Sun et al., 2013). A challenge with a Hybrid Bioreactor is the economic cost since continuous injection-extraction plants are expensive or alternatively require a biological leachate treatment plant. Consequently, this technology is applied for limited periods of time when traditional degradation processes cannot decrease the pollution any further (Berge et al., 2006).However, the high maintenance costs associated with air injection and leachate recirculation are generally covered by the increasing methane generation and/or by leachate treatments savings due to recirculation and aeration (Berge et al., 2009). Several different hybrid conditions have been tested at lab scale with promising results through combining various sequences of aerated and non-aerated phases, aeration modes (continuous or intermittent), and application (leachate aeration or in situ waste aeration).

3.4.1 Anaerobic-Aerobic sequencing

Long et al. (2009) proposed a hybrid bioreactor landfill sequencing the anaerobic and aerobic phases. At the end of the second phase, the system was able to achieve more than a 97% removal efficiency of COD and ammonia, nitrifying and denitrifying more than 70% of the initial content of nitrogen in the waste sample, produced methane for energy recovery, and dropped the main pollutants concentration to low levels (COD < 400 mg/L and ammonia < 20 mg/L). Aerobic conditions through air injection significantly improved the stabilization of the refuse, the readily biodegradable organic matter was mineralized during the initial anaerobic phase, and the hardly biodegradable organic matter was stabilized mainly during the aerobic phase. Ammonia was converted to NO_3^- and NO_2^- in ex-situ nitrification, while nitrate was reduced into nitrite and then to N_2 gas in in-situ denitrification. A simple example of the application of the hybrid bioreactor is the aeration of old landfills, in which the long lasting anaerobic process occurred over the lifetime of the landfill is followed by forced aeration. Forced aeration is an efficient technology applied worldwide for the remediation of persistent pollution (Ritzkowski and Stegmann, 2013). The same has been applied in some more recent landfills which were built as anaerobic bioreactors in order to achieve methane production leaving the possibility of applying in-situ aeration as a subsequent phase. This type of operation would convert this landfill to a Hybrid Bioreactor.

3.4.2 Aerobic-Anaerobic

When aerobic-anaerobic sequencing is applied completely in situ, aeration could be addressed to maximize the methane production by accelerating the initial acidogenic phase and anticipating optimum pH and VFA conditions for methanogenesis (Xu et al., 2014; Morello et al., 2017). Mali Sandip et al. (2012) showed that pre-aeration in combination with leachate recirculation and/or inoculum injection could increase the methane production by 25%. Similar results were obtained by Xu et al. (2014) using a lab scale hybrid bioreactor with intermitted air injection before a second anaerobic phase which achieved a higher methane production (about 32 L_{CH4}/kg_{TS}) and a higher consumption of organic compounds compared with a full anaerobic one in which methane production never started due to excessive acidity. Aeration frequencies, depth and rates strongly influence the methane production, the decomposition of organic carbon, and nitrification. Xu et al. (2015) operated two hybrid bioreactors with two different initial aeration frequencies (twice and 4 times per day) with same unit rate of 0.1 L/min/kg_{TS} until pH>7, obtaining similar trends in COD and ammonia values but higher methane generation in the case of low frequency aeration (85 L_{CH4}/kg_{TS} compared to 72 L_{CH4}/kg_{TS}). Cossu et al. (2015) tested aerobic-anaerobic hybrid bioreactors with continuous and intermittent aeration until optimum pH and VFA concentrations for methanogenesis were achieved. Both aeration modes were beneficial in accelerating waste stabilization and the acidogenic phase, however intermittent aeration until optimum pH values was more efficient in enhancing stabilization kinetics and methane generation (Table 1). According to Wu et al. (2014), aeration at the bottom layer achieved enhanced decomposition of organic carbon, while high air injection rates lead to effective simultaneous nitrification-denitrification. This combination accelerated waste decomposition but may limit methane generation. Despite the cited benefits of pre-aeration, it does not solve the problem of persistent nitrogen pollution in leachate and in all previous studies strong ammonification occurs during the first aerobic phase with positive trend in ammonia concentration which accumulated during the second anaerobic phase (Cossu et al., 2016; He et al., 2011; Morello et al., 2017; Xu et al., 2015, 2014). For this reason, S.An.A landfill model has been suggested, including a third final phase of post-aeration to drop down nitrogen indexes in leachate (Cossu et al., 2016; Morello et al., 2017). The Semiaerobic-Anaerobic-Aerobic (S.An.A) Landfill model is a hybrid system with an initial semi-aerobic phase to enhance the methane production occurring in the anaerobic step which is then followed by forced aeration for the abatement of the residual emissions. According to Morello et al. (2017) with this approach it was possible to achieve a methane potential 50% higher than that of a traditional anaerobic bioreactor which equates to an estimated reduction of aftercare by 25-35%.

A Mechanical Biological Pre-treatment (MBP) of waste before anaerobic landfilling could be regarded as a form of a hybrid bioreactor, with off-site forced aeration followed by in situ anaerobic reactions. MBP aims to achieve a quick stabilization of the waste and during landfilling the production of landfill gas might not be significant for energetic exploitation.

3.4.3 Facultative landfill

In order to overcome the challenge of ammonia accumulation under anaerobic conditions, an alternative solution consists of an external aerobic pre-treatment of leachate prior to recirculation in an anaerobic bioreactor, to allow for simultaneous nitrification and denitrification to occur in order to remove nitrogen compounds (Berge et al., 2005; de Abreu et al., 2005; Price et al., 2003; Zhong et al., 2009). This system aims at ensuring that the energy recovery due to methane production is maintained throughout the whole landfill by facilitating anaerobic conditions. In order to remediate nitrogen pollution in the leachate, the leachate is aerobically treated to nitrify the ammonia and then it is re-injected into the landfill to denitrify the produced nitrates. This system is also patented in the United States (US639895, 2002) by the name of a facultative landfill and has been tested at the lab scale by Price et al. (2003) in order to verify that the bioreactor is capable of denitrifying the nitrates produced during aerobic leachate treatment. The options available for ex situ leachate treatment are chemical-physical (ion-exchange, air stripping, chemical precipitation, reverse osmosis) and biological. Among the others, biological treatment is the most common since costs are limited compared to other processes (He et al., 2007). Several lab scale ex situ biological leachate nitrification options have been studied including the aerobic biofilter (Jokela et al., 2002), sequential anaerobic and air-lift loop sludge blanket reactors (He et al., 2007), continuous stirred tank reactor (Zhong et al., 2009), activated sludge reactor (Huo et al., 2008), fluidized bed reactors (de Abreu et al., 2005), and aerobic landfill reactor (Sun et al., 2017). All these studies demonstrate the capability of the facultative bioreactors to remove nitrogen through ex-situ nitrification of NH_4 to NO_2 and NO_3 and in-situ denitrification to convert nitrates to N_2 gas.

De Abreu et al. (2005) compared the performance of an anaerobic bioreactor with that of a facultative bioreactor with external aerobic biological leachate treatment consisting of an electrocoagulation/settling unit for metals removal and two fluidised bed reactors. According to Table 1 there are clear benefits in both COD and ammonia removal observed in the facultative bioreactor with higher removal kinetics (1.8-fold and 7.7-fold for the anaerobic column for COD and ammonia, respectively), achieving a final COD and NH_4 concentration much lower compared to the anaerobic bioreactors. Shou-liang et al. (2008) compared the performances of an anaerobic bioreactor with those of a facultative bioreactor. The latter consisted of a fresh waste landfill reactor for denitrification, a well decomposed waste landfill reactor for methanogenesis, and an aerobic-activated sludge reactor for nitrification. The obtained results showed the capability of the system to improve the methane generation and promote ammonia removal since nitrification and subsequent denitrification occurred with removal kinetics 8-folds higher than anaerobic conditions. The acidogenic phase was accelerated in the hybrid reactor with a higher methane concentration during the experimental period, while inhibiting methanogenesis in the anaerobic reactor due to the VFA accumulation and low pH level. He et al. (2007) studied the performance of a facultative reactor with an external leachate treatment consisting of a sequential up flow anaerobic sludge blanket reactor for organic matter removal and an air-lift loop sludge blanket reactor for nitrification. Even if the COD removal was quite similar to the control reactor, the ammonia removal was strongly enhanced with final NO_3 values of about 4 mg/L, suggesting the occurrence of denitrification. This kind of Hybrid Bioreactor is promising because it allows for the reduction in ammonia in the landfill without any aeration systems while ensuring methane recovery at the same. The downside of this process is the continued need for a biological leachate treatment plant.

The high concentration of nitrate produced in ex-situ nitrification may inhibit methanogenesis in a facultative bioreactor. For this reason, Sun et al. (2017) studied the use of ex situ simultaneous nitrification-denitrification in an aged refuse bioreactor for nitrification prior to in-situ denitrification, in order to enhance the methane production. Hirata et al. (2012) proposed the SeRA system (recirculatory semi-aerobic landfill) with ex situ leachate aeration in order to improve the semi-aerobic landfill performance by reducing the in situ oxygen demand, expanding the aerobic zone in the waste mass, and improving the nitrification denitrification process. SeRA achieved a similar TOC degradation performance compared to the aerobic lysimeter and an even better total nitrogen degradation performance confirmed by the higher gasification rates.

4. BIOREACTORS COMPARISON IN TERMS OF SUSTAINABILITY

A comparative qualitative analysis of bioreactor types are summarised in Table 2 based on selected characteristics, such as persistent emissions, technological complexity, maintenance costs, and leachate treatment costs.

Considering the prior need of achieving landfill sustainability, ammonia is generally recognized as the main long-term pollutant in leachate. Therefore almost all the bioreactor types involved some form of a nitrification-denitrification process with different methodologies. Even if the carbon and nitrogen emissions can be reduced efficiently, leachate can also be polluted by saline compounds and heavy metals, which are difficult to be removed biologically.

The performance of each type of bioreactor may highly depend on the-situ conditions, such as waste characteristics and climate, which should be taken into consideration beyond the objectives to be pursued (i.e. energy recovery, faster waste stabilization, washing of soluble compounds). For example, according to the recent European Regulations (EU, 2015), the reduction of the POF in landfilled waste and waste pre-treatment limit the practicability of bioreactors that are intended for energy recovery, while these bioreactors will surely have a central role in waste management outside of Europe (Reinhart et al., 2002). Moreover, the capability of bearing the costs and the technological complexity will strongly depend from country to country. Nevertheless, knowing the general behaviour in stabilization performance of each bioreactor type at the lab scale may help to identify the best bioreactor solution at field scale. The best performance would be based on the aim to fulfil the sustainability concepts according to the specific site objectives and in-situ conditions. For this reason, the quantification of the stabilization performance and thus the sustainability of the different systems has been carried out.

According to Berge et al. (2009), the main parameters that influence bioreactor economics are air space recovery, gas recovery for the subsequent energetic use, and savings resulting from reduced leachate treatment requirements. Therefore, faster biological stabilization provides a metric for measuring the successfulness of any landfill

TABLE 2: Qualitative analysis of different landfill bioreactor types compared to the traditional landfill.

Bioreactor Landfill Type	Objective: Methane production & energy recovery	Objective: Nitrogen removal	Biochemical Kinetics	Other Persistent Emissions	Technological complexity	Maintenance Costs*	Leachate treatment costs *
Traditional Landfill	Traditional Recovery	by leaching	slow	NH_4^+, Salinity, Heavy metals	Gas collection	Low	High
Anaerobic	Enhanced recovery	by leaching	Medium-slow	NH_4^+, Salinity, Heavy metals	Leachate recirculation, Gas collection	Leachate recirculation	Savings from leachate recirculation
Aerobic	No	Nitro-Denitro	fast	Salinity, Heavy metals	Leachate recirculation, Air Injection	Air injection, Leachate recirculation	Savings from leachate recirculation and aeration
Semi-aerobic	No	Partial Nitro-de-nitro	medium	Salinity, Heavy metals	Build to enhance natural convection	Sometimes Leachate recirculation	Savings from aeration
Hybrid	Enhanced recovery	Nitro-Denitro	fast (limited for NH_4 in aerated-anaerobic)	Salinity, Heavy metals (NH_4^+ in aerated-anaerobic)	Two stage aerobic-anaerobic or vice versa; Gas collection; ex situ treatment before reinjection	Air injection, Leachate recirculation	Savings from leachate recirculation and aeration, ex-situ treatment cost if present

** The costs are referred to the operational phase.*

bioreactor type, both by reducing leachate treatment costs and by assuring sustainability requirements are achieved. Stabilization criteria of landfills is still a debated topic in the scientific literature (Barlaz et al., 2002; Laner et al., 2012; Stegmann et al. 2003; Valencia et al., 2009) since the criteria are not absolute and site specific conditions significant influence the values. In order to evaluate the sustainability achievement and aftercare completion, several approaches have been proposed such as the compliance with Final Storage Quality (FSQ) which defines the target emission values that must be achieved, impact risk assessment approaches, and performance based systems (Laner et al,. 2012). All of these approaches require a site-specific assessment in order to take into consideration the potential of natural attenuation or vulnerabilities (Barlaz et al., 2002; Laner et al., 2012; Rich et al., 2008).

In this study first order removal kinetics of organic and nitrogen concentration in leachate have been selected as criteria for the evaluation of the bioreactor stabilization performance (Ritzkowski et al., 2006) of the investigated lab-scale tests (Table 1).

A general overview of the stabilization capability associated with the different bioreactor types were calculated by the mean values of the COD and ammonia removal kinetics and standard deviations. The latter ones are represented as bar errors in Figure 2 in order to describe the distribution of values. Although there are variations in the operational management in the different investigated case studies, including the recirculation rate, waste composition, L/S ratio, air injection and experimental period (Table 1), the obtained mean COD kinetics can represent the general behaviour of each bioreactor type, as demonstrated by the standard deviations. The benefits of aerobic conditions are evident in the maximization of the COD removal with an average COD removal kinetic of 0.051d^{-1}. Hybrid and semi-aerobic bioreactor performances are between the

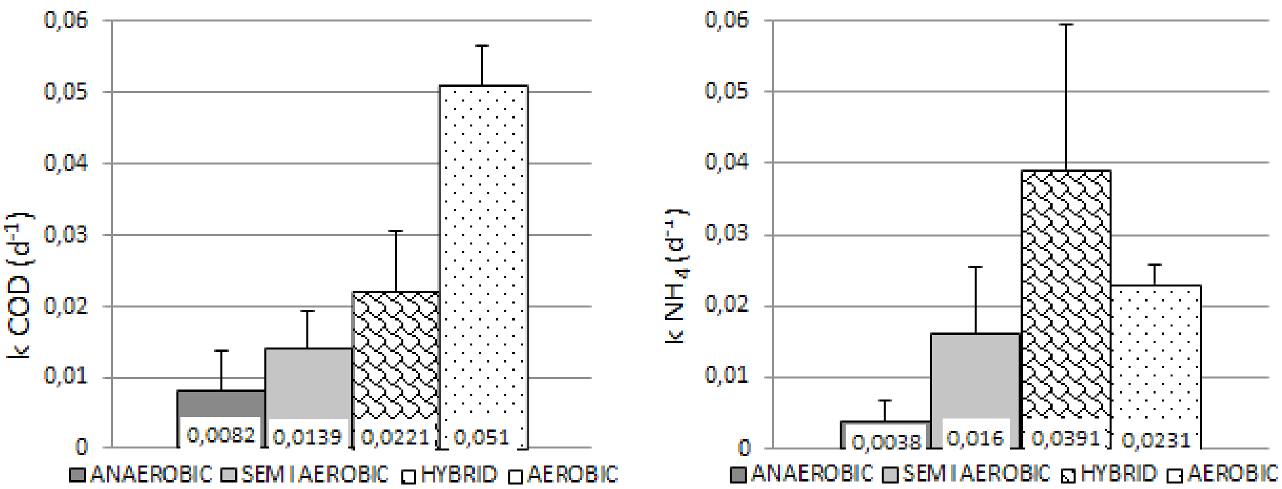

FIGURE 2: Mean values (numerically represented) and associated standard deviations (bar errors) of the COD and ammonia removal kinetics associate with each landfill bioreactor type starting from data collection and elaboration.(2012)).

results for anaerobic and aerobic conditions presenting a 0.0221 d^{-1} and 0.0139 d^{-1} average removal kinetics, respectively. Different results were obtained for ammonia removal kinetics in which hybrid bioreactors demonstrated better average value compared to the other bioreactors (0.0391 d^{-1}). The higher variability of the values around the mean makes these results carefully reliable since they are strongly influenced by the specific hybrid bioreactor application.

By the use of the mean COD and ammonia removal kinetics, it is possible to foresee and compare the stabilisation time for each bioreactor type. Considering the reference time (T) required under aerobic conditions to achieve a 95% contaminant removal (Figure 3), the time to achieve the same COD removal performance under hybrid, semi-aerobic, and anaerobic conditions increased by 2.3, 3.7, and 6.2-fold, respectively. In the case of ammonia removal, time is reduced by 0.7-fold under hybrid conditions, while time increased by 1.7 and 3.7-fold under semi-aerobic and anaerobic conditions, respectively. According to these results, the faster the stabilization, the shorter the aftercare time and the lower the post closure care costs.

5. CONCLUSIONS

Anaerobic bioreactors improve, by leachate recirculation, the methane generation rate and the leachate quality compared to the traditional anaerobic landfills. However, ammonia accumulation and slow degradation kinetics remain the main challenges in anaerobic bioreactors compared to the others, putting anaerobic bioreactors far from sustainability requirements. Aerobic reactors increased the ammonia and COD average removal kinetics up to 6 times more than under strictly anaerobic conditions and reduced the time required to achieve a 95% removal of COD and ammonia by 6.2- and 3.7-fold, respectively. Aeration appears to be an effective alternative to the traditional anaerobic processes, although the need for forced ventilation systems, the complex operation and management, and the large energy consumption, with high operational and capital costs, make the aerated landfill not always technically and economically feasible. A semi-aerobic landfill achieves a performance between the anaerobic and aerobic bioreactors but lowering the typical operational costs of aerated landfills by removing the need for direct air injection. For this reason, the semi-aerobic system is recognized as a cost–effective, low technology landfill system. This system can also be feasibly implemented in developing countries, where financial constraints and limited technical knowledge are generally the main reasons for inadequate disposal. A limiting factor of aerobic bioreactors is the complete inhibition of the methane generation, making any energy recovery impossible. Hybrid bioreactors, which are operated under various combinations of aerobic and anaerobic conditions, achieve both energy recovery and/or faster waste stabilization. In particular aerated-anaerobic hybrid reactors aim to enhance the biogas generation but this system will experience ammonia accumulation challenges, while facultative bioreactors combine both objectives which provides the best performance in terms of ammonia removal kinetics. In general, the best ammonia removal performance is achieved under hybrid conditions.

Due to the careful operation and construction requirements of bioreactor landfills, capital and operating costs would be greater compared to traditional landfills. However these costs will be recouped through future economy benefits from bioreactor landfills. In particular, the obtained results demonstrate the possibility of achieving shorter aftercare, reduced leachate treatment costs, reduced long term environmental risks, and an earlier reuse of the land. Detailed analysis of costs related to full-scale bioreactors is still a crucial aspect to be further investigated.

Moreover, the transfer from a lab-scale to full-scale bioreactor still remains a significant issue to be explored since much higher benefits are achieved under lab-scale investigation rather than at full-scale application due to the challenges with reproducing optimum and homogeneous conditions. However, knowing the general behavior of each bioreactor type at lab scale allows the identification of the best bioreactor solution at a larger scale according to the site specific objectives and in-situ conditions.

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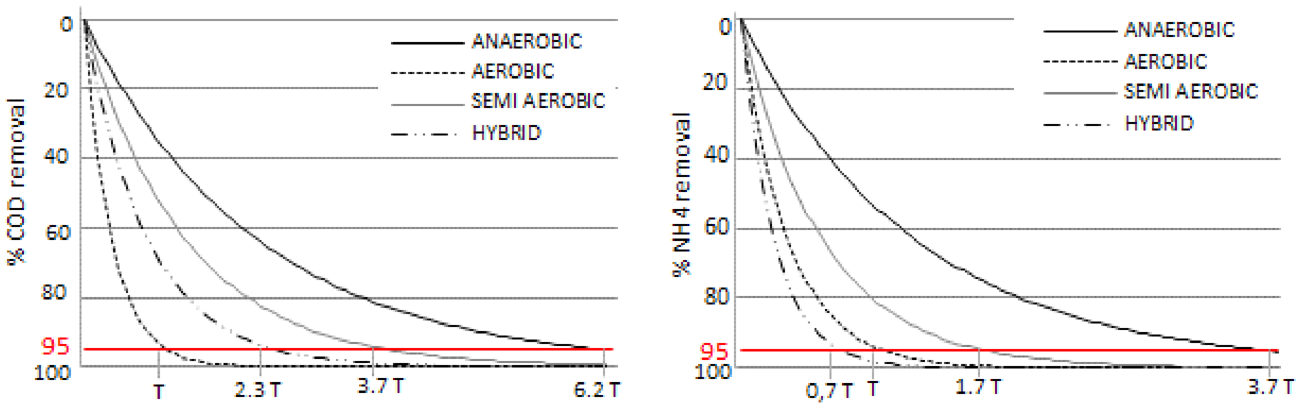

FIGURE 3: Variation of the percentage removal of COD and ammonia over time, according to the mean obtained removal kinetics. Reference time (T) is the time required to achieve a 95% removal under aerobic conditions.

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RECENT STATE-OF-THE-ART LEACHATE TREATMENT PLANTS IN EASTERN ENGLAND

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ABSTRACT
The paper presents detailed design and performance data for two full-scale leachate treatment plants that have been designed and operated in Eastern England during recent years, in which reliable performance has been achieved for an extended period. The first plant is a modified Sequencing Batch Reactor system, treating relatively diluted leachate (COD about 500 mg/l, ammoniacal-N about 180 mg/l) from a closed landfill site, to provide complete nitrification of ammoniacal-N and degradation of all degradable COD, in a manner requiring minimal site attendance. This is made possible by means of reliable and robust operational software, which can run the plant in a completely automated manner, but nevertheless alerts the operator to any issues. The second state-of-the-art leachate treatment plant was designed and built at the Masons Landfill Site in Ipswich. It was designed to treat 160 m^3/day of strong methanogenic leachate, often containing more than 2000 mg/l of ammoniacal-N. Discharge of treated leachate is to sewer, under a consent in which the main parameters that are limited are ammoniacal nitrogen, and COD. Treatment comprises full biological nitrification, with ultra-filtration membranes providing additional removal of COD, to achieve challenging consent limits. Taken together, the two case studies provide valuable, robust and real, full-scale data, for the degree of treatment which can realistically be delivered, by well-designed and operated, aerobic biological leachate treatment plants, where each plant has succeeded in treating leachates to well below the consented quality limits for discharge.

1. INTRODUCTION

Treatment of leachates is now an established technology, in which fitness for purpose, and process reliability are, without doubt, the most critical aspects. Nevertheless, it remains a fact that many leachate treatment plants continue to be designed inadequately, by over-confident but inexperienced contractors, so they fail to achieve required standards of effluent quality.

Many academic research papers are published each year, which present very detailed laboratory results describing small-scale and pilot-scale studies of leachate treatment, the great majority of which, although providing interesting and challenging topics for MSc and PhD students, never result in any substantial advances in treatment processes being provided on full-scale landfill sites.

What are needed, and prove to be far more useful to the landfill industry, are well-reported case studies of the application of state-of-the-art science, process designs, engineering, and automated control systems, which contain real and reliable data, that can be applied more widely to other applications. There is presently a large gap between academic research, and the reality of leachate treatment plant design and operation, to achieve required standards of effluent quality, and maintain compliant discharges of treated leachate into public sewers, and sensitive surface watercourses.

The authors have previously published many case studies of the design, operation, and performance of full-scale leachate treatment plants (e.g. Robinson, H et al., 2005; 2008; 2009; 2013a; Strachan et al., 2007), and in 2007 drafted current UK guidance on the treatment of landfill leachates (UK Environment Agency, 2007). We believe that availability of real performance data from well-designed and operated full-scale leachate treatment plants is of far greater value to landfill operators than are academic papers, in helping to ensure that plants do not continue to be constructed which are not capable of achieving required effluent standards.

This paper therefore presents very detailed design

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and performance data for two leachate treatment plants that have been designed and operated in Eastern England, during recent years, for which reliable performance has been achieved for extended periods. The first plant at Hatfield, comprises a relatively straightforward Sequencing Batch Reactor system, treating leachate from a closed landfill site, to provide complete nitrification of ammoniacal-N and degradation of all degradable COD, in a manner which requires minimal site attendance. This plant was commissioned during Summer 2016. The second plant, at Masons Landfill, treats much stronger leachate from an operational landfill, and faced more serious challenges in terms of reliable compliance with tight limits for COD in treated leachate. On this basis, the extended aeration process was complemented by incorporation of an ultrafiltration system for solids separation, following detailed pilot-scale studies and investigations.

Each plant has operated reliably and robustly, to achieve complete compliance with discharge limits, and very detailed operational data are presented.

2. HATFIELD LEACHATE TREATMENT PLANT, HERTFORDSHIRE, UK

2.1 Hatfield Landfill Site

2.1.1 Background Information

CEMEX UK Operations Limited manages Hatfield Closed Landfill Site, which is located near to St Albans in Hertfordshire, UK, in the commuter belt about 30 km north of Central London. The site is a working sand and gravel extraction site, but infilling of extracted areas with primarily commercial and industrial wastes took place into initially unlined, and later clay-lined cells from the 1960s to 1990s. Cells were a maximum of about 15 m deep. For several years before 2010, untreated leachates from the site were pumped safely into the local public sewer, but when concentrations of ammoniacal-N began to approach consented limits, pumping ceased, and leachate levels and composition within the site were monitored carefully for several years. During 2014, a decision was made to proceed with the design and construction of a small on-site leachate treatment plant, in order that leachate abstraction could be resumed to comply with Environmental Permit leachate depth limits. This would enable discharges of treated leachate to be made compliantly into the sewer again. Following detailed pilot-scale treatability trials, a plant was designed, and constructed during late 2015/early 2016.

Design of the plant had to be revisited, at short notice, following publication of new guidance by the Construction Industry Research and Information Association (2014), which dealt with secondary containment requirements for commercial and industrial premises, which although not formally adopted by the UK Environment Agency, was nevertheless first applied in 2015, as guidance as to what was acceptable for construction of process tanks in leachate treatment plants. Accordingly, the Hatfield plant became the first UK leachate treatment plant to be completely compliant with this guidance. Modifications included provision of a concrete bund which surrounds the entire plant, as well as completely independent secondary containment systems, complete with leak detection systems, beneath individual process tanks. These were constructed onto piled foundations into chalk bedrock, beneath the overlying silty ground.

2.1.2 Design and Construction of the Hatfield Plant

The Hatfield treatment plant is designed to treat relatively weak methanogenic leachates from the closed landfill, at rates of up to 60 m^3/d, before controlled discharge into the sewer via a pipeline. The plant is shown in Plate 1 and 2 includes; a roofed Sequencing Batch Reactor (SBR) tank, with twin 7.5 kW venturi aerators, bellmouth with actuated stopper, and an array of probes and sensors, and an operational range from 310 to 360 m^3. A roofed Raw Leachate Balance Tank, and a unroofed Treated Leachate Balance Tank, each with a capacity of just less than 100 m^3. The plant is designed and operated as an unmanned operation, with a SCADA system incorporating automated alarms to designated operatives, and fail-safe protection.

PLATE 1: View of Hatfield Leachate Treatment Plant, showing fully bunded area, chemical dosing compound in right foreground, and control building at the rear left.

PLATE 2: Hatfield Leachate Plant: Detail of small roofed Raw Leachate Storage Tank, roofed SBR tank with twin venturi aerators on the right, and unroofed Treated Leachate Balance Tank.

2.2 Results from Leachate Treatment at Hatfield

The Hatfield plant was designed and constructed by Phoenix Engineering during late 2015/early 2016, and commissioned during mid-2016. The plant rapidly (within days) achieved the design treatment rate of 50 m^3/d, and since then, the plant has treated a total of 13,900 m^3 of leachate, often at up to design rates, shown in Figure 1 below.

One interesting issue at Hatfield was that, although extended and routine monthly monitoring of leachate quality within landfill boreholes/extraction points had been carried out for more than 5 or 6 years, which indicated relatively weak leachates (ammoniacal-N about 100 mg/l), when pumping began during April and May, much stronger leachate was initially extracted, before leachate strength again reduced, see Figure 2.

Subsequently, concentrations of ammoniacal-N in blended leachate being treated stabilized at between 100 and 200mg/l, with COD values between 350 and 500 mg/l. What also occurred was that within about 4 months, after extraction and treatment of about 5300 m^3 of leachate during summer months, leachate extraction wells in the permitted landfill dried up, producing little further leachate. Additional leachate was obtained, as planned, by extending the pumping to existing abstraction wells in older engineered landfill cells, for which the permit had been surrendered. From January 2017, despite unusually dry weather conditions over an extended period, leachate has continued to be extracted throughout the summer. Overall mean concentration of ammoniacal-N in raw leachate was 181mg/l (maximum 400 mg/l), reduced to less than the detection limit of 0.40 mg/l in more than 60 per cent of treated leachate samples. Mean COD values in leachate were 476mg/l. During the 3 months following commissioning, as leachate pumping became established, each value

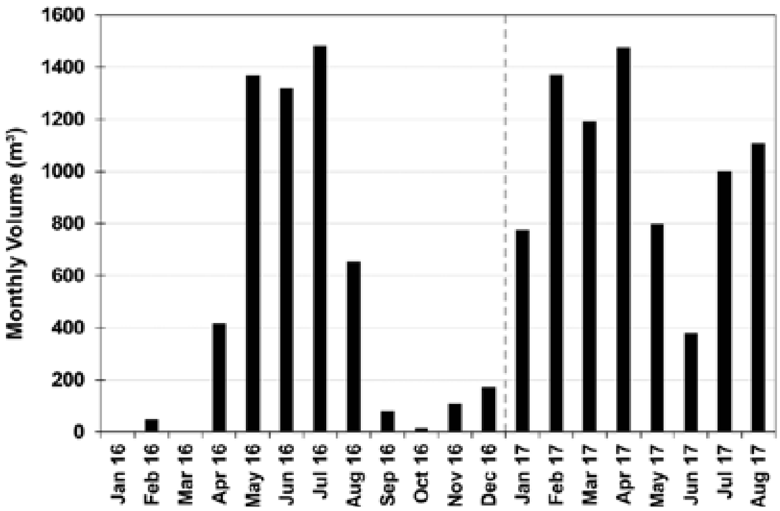

FIGURE 1: Rates of treatment achieved at Hatfield, 2016-2017 (m^3/month).

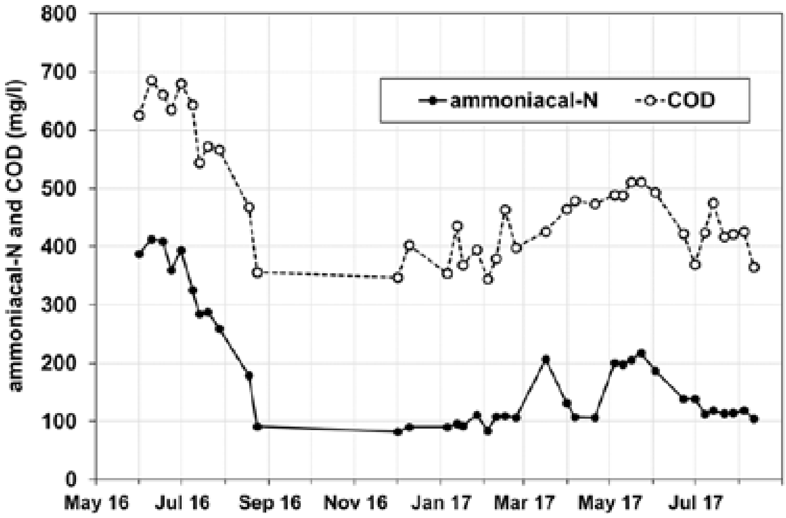

FIGURE 2: COD values and concentrations of ammoniacal-N in raw leachate blend at Hatfield.

was more than 50% greater overall. Overall mean values in treated leachate were 1.12 mg/l for ammoniacal-N, and 173 mg/l for COD, and each was always well below consented limits of 125 and 1000 mg/l respectively.

2.3 Summary of Results from Leachate Treatment in the Hatfield Plant

The treatment plant at Hatfield has demonstrated that a well-designed, but relatively simple leachate treatment plant can operate successfully and reliably on a closed landfill site, with instrumentation and SCADA controls in-place to alert a remote operator to any problems, and able to shut the treatment process down automatically, in the event of any problems. Similar treatment plants on closed and remote landfill sites, where sewer access is not available, can readily be fitted with simple polishing processes such as reed beds, to enable high quality treated leachates to be discharged safely, directly into surface watercourses. At Hatfield, the plant is reliably achieving required treatment of leachates, with very little operator input, in a similar fashion to a previously constructed treatment plant at Small Dole (Robinson, T, 2017).

3. MASONS LANDFILL, IPSWICH, EAST ANGLIA

3.1 Masons Landfill Site

3.1.1 Background Information

Masons Landfill Site is operated by Viridor Waste Management and is located near to the village of Great Blakenham, and about 6km NW of Ipswich, in Suffolk, UK. The site is a former chalk and clay quarry, with an area of 74ha, containing about 5 million tonnes of household and commercial wastes, tipped to depths of 30 m since it opened in 1992. Prior to the year 2010, leachates generated by decomposing wastes were discharged directly into the pubic sewer, receiving only simple aeration to reduce concentrations of dissolved methane to safe levels.

However, during 2010, as negotiations progressed between Viridor and Anglian Water plc, for continued discharge of leachate into their public sewer, it became clear that far tighter restrictions would be imposed going forward. This would require a significantly greater degree of treatment than hitherto, involving the design of a full biological treatment process at the Masons site. It was also intended that the Masons leachate treatment facility would also receive and treat leachates from a number of other landfills in the region, which would be imported by road tanker, providing an environmentally sound and reliable discharge route for these. Viridor was informed that a key discharge requirement would demand that COD values in treated leachate did not exceed 1500 mg/l, and experience at many sites indicated that when treating concentrations of ammoniacal-N in excess of 2000 mg/l, a simple SBR process could probably not be relied upon to achieve this 100 per cent of the time. Design work therefore needed to address this issue, to allow a suitable and completely reliable treatment process to be provided.

3.1.2 Treatment Process Design

In extensive experience of treating landfill leachates successfully, using aerobic biological processes optimised within Sequencing Biological Reactor systems, at both pilot-scale and full-scale, it has been demonstrated consistently that levels of residual and intractable "hard" COD in treated effluents are not related to levels of COD in raw leachates being treated, but rather are much more closely related to concentrations of ammoniacal-N in the leachates. This may well be due to both being the product of the same anaerobic processes of degradation, taking place within landfilled wastes, or possibly also because some hard COD is generated during the processes of nitrification of ammoniacal-N itself.

Figure 3 provides correlations between concentrations of ammoniacal-N in raw leachates being treated, and COD values in final effluents, for a large number of full-scale SBR plants and pilot-scale trials (after Robinson et al., 2005).

For treatment of blended leachates containing between 1500 and 2000 mg/l of ammoniacal-N at Masons, the graph demonstrates that a normal modified SBR process cannot be relied upon to achieve less than 1500 mg/l of

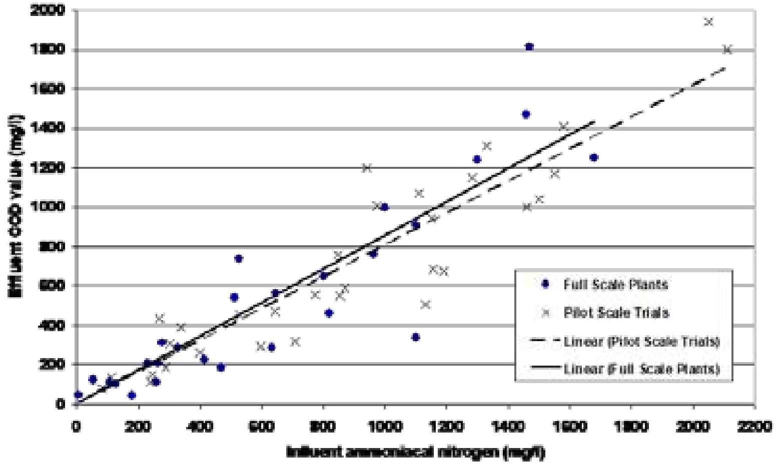

FIGURE 3: Correlation between concentrations of ammoniacal-N in leachates, and residual "hard" COD in settled treated effluents, for full-scale treatment plants and detailed pilot-scale studies (all results in mg/l). (After Robinson et al, 2005).

COD in treated leachate, all of the time. This was confirmed by specific pilot-scale leachate treatment trials that were undertaken on a representative blended leachate sample from the Masons site.

On this basis, further detailed studies were carried out by Phoenix staff, to examine the possibility of incorporating ultrafiltration (UF) membranes into the on-site treatment process, in order to significantly and reliably reduce COD values in treated leachates being discharged. A decision was made not to consider a standard Membrane Bioreactor (MBR) process design, as our belief and experience was that the extended aeration process provided within the SBR process would combine well with the UF process. This would provide the benefits of stable, robust, and cost-effective biological treatment and nitrification, coupled with the advantages of an effluent filtration process. In addition, it was anticipated that passage of mixed liquor from an extended aeration process, through membranes, would minimise the need for heavy chemical treatment of the membranes, increasing their long-term efficiency, and indeed working life.

Those pilot-scale studies of UF treatment have been described in detail previously, (Robinson et al., 2013), and are summarised here. Temporary incorporation of a pilot-scale UF membrane plant into the extended aeration process, at twelve leachate treatment plants across the UK, did indeed enhance removal of COD from treated leachate, as shown in Figure 4. Despite variability between different sites, overall mean rates of additional COD removal achieved by incorporation of the UF membranes were about 60 per cent.

All of these studies confirmed that a modified SBR process, with simple discharge of clarified effluent, would be unlikely to achieve required COD values of less than 1500 mg/l as required for discharge into the local public sewer. Therefore, incorporation of UF membranes for solid/liquid separation would be essential, and likely to achieve additional COD removal of about 60 per cent. This would provide assurance for reliable and complete compliance with the discharge consent.

In fact, during the construction of the full-scale Masons plant, after discussions, the proposed consent limit of 1500 mg/l of COD in treated leachate was relaxed to 2000 mg/l by Anglian Water, which provided even greater confidence for plant design, but did not change it.

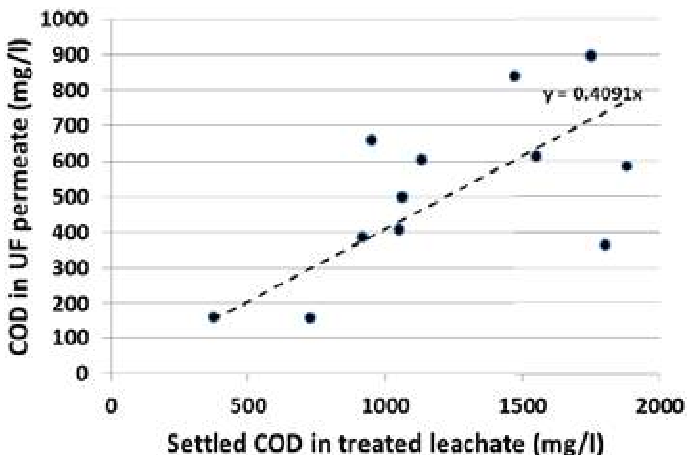

FIGURE 4: Relationship determined between Settled COD in SBR effluent, and COD in UF permeate, at each of the 12 SBR treatment plants examined (after Robinson et al., 2013).

3.1.3 Design and Construction of the Masons Plant

The Masons Leachate Treatment Plant (Plates 3 and 4) was therefore designed to treat leachate from the Masons site, as well as similar quality strong leachates transported by tanker from other nearby landfills. Overall, blended leachate to be treated was taken to typically contain about 4000-5000 mg/l of COD, and about 1500 to 2000 mg/l of ammoniacal-N, which has proved to be the case in practice. The plant is designed to treat leachate at rates of up to 160 m^3/d and comprises a large (operational volume up to 1900 m^3) roofed and part-buried reinforced concrete extended aeration tank. This tank is aerated continuously, 24 hours per day, using venturi aerators. Raw leachate is introduced gradually and evenly into this tank, from which mixed liquor is drawn and passed through a UF membrane plant, which produces effluent for discharge to sewer, via a Treated Leachate Balance Tank.

Because of the sensitivity of the receiving public sewer, some 1500 m from the treatment plant, after detailed investigations and hydraulic modelling of the sewerage network, it proved necessary to install flow measurement equipment into the receiving manhole, complete with a communications link, such that in times of high flows of wastewater within that sewer, discharges of treated leachate into it can be discontinued until wastewater flows reduce. To cater for this, a large Treated Leachate Balance Tank, providing at least four days' effluent storage capacity was provided. Similarly, a relatively large Raw Leachate Balance Tank (500 m^3) was provided to maximise blending of leachates from the various sources, before treatment.

PLATE 3: Masons Leachate Treatment Plant, Ipswich, UK.

PLATE 4: UF Membrane Tubules at Masons Leachate Treatment Plant.

3.2 Results from Leachate Treatment at Masons

The Masons plant was designed and constructed by Phoenix Engineering during 2012, and commissioned during early 2013. Since then the plant has treated a total of 204,000m^3 of leachate, at rates of up to 182 m^3/d, shown in Figure 5. Typical rates have been between 3500 and 5000 m^3/month (about 120 to 165 m^3/d, comparing well with the design capacity of 160 m^3/d).

Figure 6 presents detailed operational results for the removal of COD during treatment, demonstrating effluent quality results that are in compliance with the consent limit of 2000mg/l at all times. Figure 7 presents equivalent data for removal of ammoniacal-N.

Table 1 below compares results from the original treatability trials (without UF membranes, with those from operation of the plant, including the UF membrane system.

Results demonstrate consistent and complete compliance with required limits, not just for COD and ammoniacal-N, but for all other contaminants. The distributions of actual values that have been achieved, for COD values and for concentrations of ammoniacal-N in final effluent being discharged from the plant, are summarised in Table 2, as cumulative distributions showing the percentage of sample analytical results below specific stated values. These demonstrate very comfortable and robust compliance, although the skill of the plant operating team must certainly be recognised, in achieving such reliable performance.

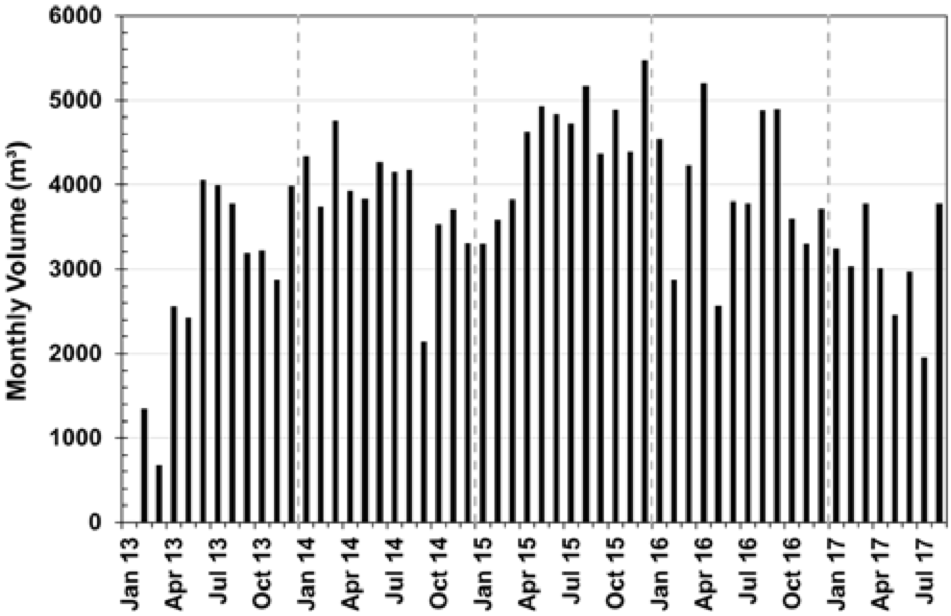

FIGURE 5: Monthly volumes of leachate treated at Masons, January 2013 to August 2017.

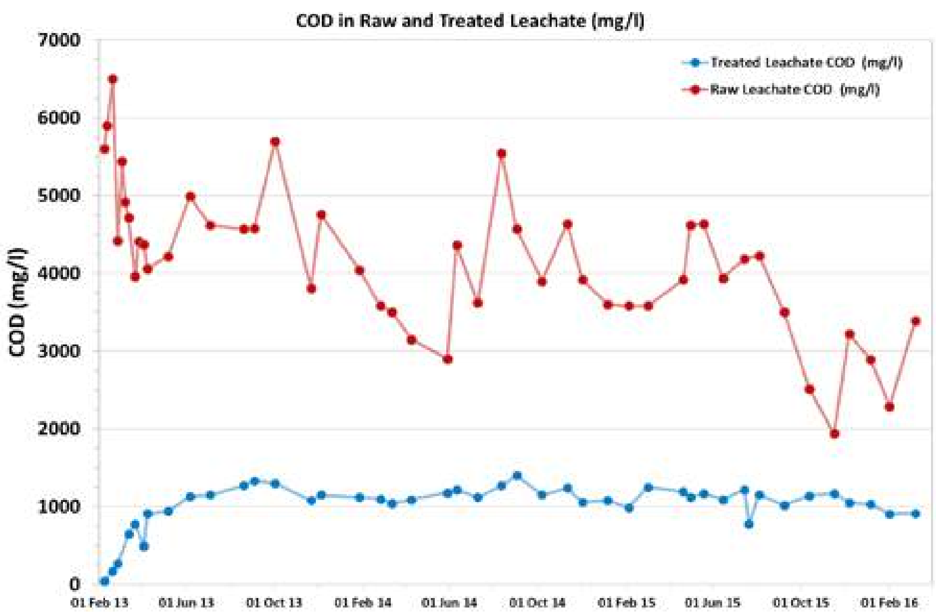

FIGURE 6: Masons Landfill: COD removal efficiency, February 2013 to March 2016.

TABLE 1: Masons Landfill: comparison of data from initial SBR trials with data from the full-scale plant during 2014. (after Robinson, T, 2014).

	Treatability Trials (2010)		Full-scale treatment plant (2014)	
Determinand	**Leachate**	**Effluent**	**Leachate**	**Effluent**
COD	3456	1460	3830	500
BOD_5	185	<10	992	2.1
TOC	1100	555	1490	177
ammoniacal-N	1818	0.59	1590	1.19
nitrate-N	1.13	1717	<1.3	667
nitrite-N	<0.3	<0.3	2.2	2.1
alkalinity (as $CaCO_3$)	9140	209	7960	1660
pH-value	8.09	7.52	7.79	7.70
chloride	2422	2443	2080	2330
sulphate (as SO_4)	515	585	-	348
phosphate (as PO_4)	11.5	10.3	-	7.45
conductivity (as µS/cm)	20,100	16,100	-	10,500
sodium	1878	3710	-	3180
magnesium	83	86	-	44
potassium	1310	1375	-	966
calcium	73	102	-	93
chromium	360	310	242	85
manganese	385	30	-	38
iron	709	141	-	240
nickel	255	260	-	88
copper	<40	56	-	<40
zinc	52	143	-	132
cadmium	<5	14	-	<5
lead	16	12	-	<5
arsenic	415	340	408	379
mercury	<0.02	0.04	-	<0.02

Notes: all results in mg/l, except heavy metals in µg/l, conductivity and pH as shown. - = no data.

Table 3 summarises all operational data from the Masons plant, also for the 3-year period from February 2013 to March 2016.

3.3 Summary of Results from Leachate Treatment in the Masons Plant

The successful and reliable treatment of leachate at Masons Landfill, demonstrates the significant benefits not only of experience at many other similar plants, but also of an initial stage of detailed design work, incorporating pilot scale studies as required, in order to ensure that the full-scale plant will operate exactly as required. All new treatment plants bring with them a degree of learning. At Masons, lessons learned included the fact that by providing a robust, extended aeration biological process, then this enables the UF membrane system to operate very reliably indeed, with chemical cleaning of the membranes rarely required, and excellent membrane performance being maintained simply by routine and automated cold water washes, with occasional hot water flushing.

In addition, although the plant was anticipated to operate at concentrations of Mixed Liquor Suspended Solids of only up to about 8000 mg/l, experience has demonstrated that successful operation at solids concentrations as high as 15,000 mg/l (still lower than routinely used in MBR systems), very much minimises net generation of sludge solids requiring disposal. A heat exchanger system was also fitted retrospectively, which during warmer months readily maintains plant operational temperatures below 37°C, to prevent harm to nitrifying bacteria.

4. CONCLUSIONS

Real performance data from full-scale, well-designed examples of leachate treatment technologies are of enormous value when making decisions about which process is most suitable for a given application on a landfill site. Real full-scale results are essential to enable operators to select treatment systems that will be able to achieve specific effluent discharge consent limits, reliably, robustly, and with minimal operator input. It is a fact that far too many on-site leachate treatment systems have been procured and con-

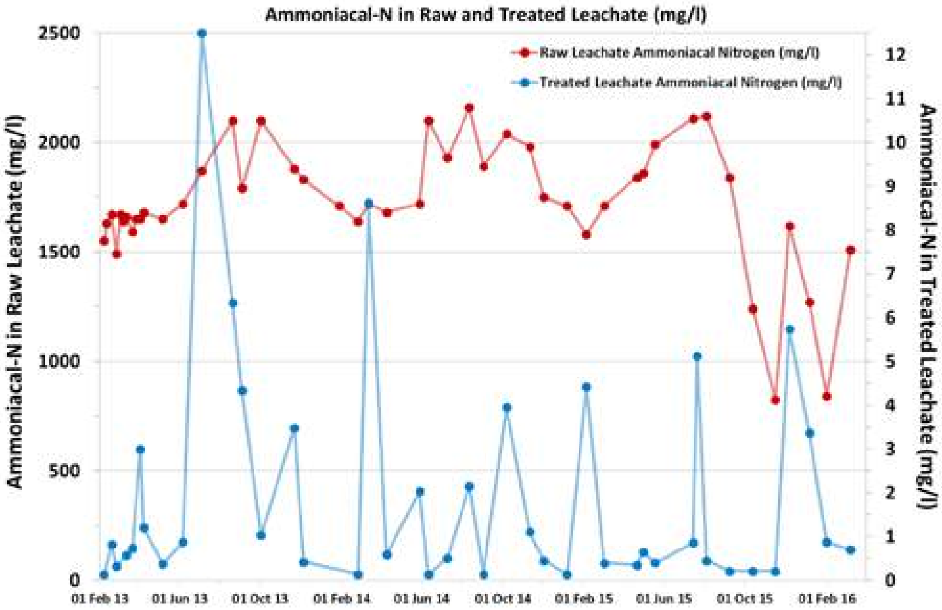

FIGURE 7: Masons Landfill: ammoniacal-N removal efficiency, February 2013 to March 2016.

structed, on landfill sites throughout the world, which have failed to perform as required.

This paper presents such data, from two recent, but very different, leachate treatment plants on UK landfill sites. The first, at Hatfield Landfill, is a state-of-the-art simple modified SBR system, treating relatively weak methanogenic leachate (ammoniacal-N from 100 to 400 mg/l) to sewer discharge standards, and doing so automatically but reliably, with intuitive SCADA software, capable of providing confidence in that performance.

The second leachate treatment plant constructed at Masons Landfill during 2012, on a large, operational landfill site, has similar automation and SCADA protection, but treats leachates almost an order of magnitude stronger (ammoniacal-N typically from 1500 to 2200 mg/l), where a modified SBR system alone could not have been guaranteed to meet challenging discharge standards for residual COD. The Masons plant is innovative in the UK, in bringing together the robustness of extended aeration biological treatment, and the advantages of UF filtration in achieving significantly enhanced COD removal, and essentially complete retention of solids in a relatively simple manner. Detailed operational data, and effluent quality results, from each plant, will be of great value to landfill operators considering their options for on-site treatment of leachates.

The treatment systems described have treated leachates typical of both old and restored landfills, and from large modern operational waste disposal sites where very strong leachates are being generated. In each case, the plants have readily and robustly achieved limit values for all contaminants, allowing safe discharge of the treated leachates. At both sites, complete nitrification of all ammoniacal-N (>99.5%) has been achieved reliably. However, each leachate type contains a significant level of residual, non-biodegradable "hard" COD materials. Although of very low toxicity, presence of this COD in treated leachates may constrain their discharge into both surface watercourses and the public sewer.

Operational results have demonstrated that incorporation of UF membranes for solids separation, can readily

TABLE 2: Masons Landfill: removal of COD and ammoniacal-N, February 2013 to March 2016.

COD (consent limit 2000mg/l)		ammoniacal-N (consent limit 50mg/l)	
COD value (mg/l)	**% samples below value**	**ammoniacal-N (mg/l)**	**% samples below value**
1400	100.0	13.0	100.0
1300	95.3	10.0	97.7
1200	79.0	5.0	88.4
1100	48.8	2.0	69.8
1000	27.9	1.0	60.5
800	16.3	0.75	51.2
		0.5	37.2
		0.2	11.6

Notes: Results represent the per cent of samples below the stated contaminant concentration, between February 2013 and March 2016.

TABLE 3: Masons Landfill: summary of all operational data, February 2013 to March 2016.

Determinand	Leachate Feed	Final Effluent	Consent Limit
COD	4124	1043	2000
BOD_5	1730	1.62	-
TOC	1010	428	-
Suspended Solids	58	14	500
ammoniacal-N	1726	1.95	50
nitrate-N	0.55	1176	-
nitrite-N	0.03	0.71	-
alkalinity (as $CaCO_3$)	7835	6320	-
pH-value	8.25	7.39	-
chloride	2230	2213	3500
phosphate (as PO_4)	11.0	7.8	-
conductivity (as µS/cm)	18250	15492	-
sodium	-	1670	-
magnesium	-	124	-
potassium	-	1630	-
calcium	-	81	-
chromium	223	73	-
manganese	31	25	-
iron	770	610	-
nickel	196	20.5	-
copper	13.0	4.86	-
zinc	134	57	-
cadmium	1.51	0.45	10.0
lead	28	5.7	-
arsenic	465	0.58	-
mercury	0.11	0.03	-

Notes: all results in mg/l, except trace metals in µg/l, conductivity and pH value as shown. - = no data. Results represent mean values from well over 40 samples for main determinands, and from more than 25 samples for trace metals.

provide further COD reductions of about 60 per cent, which can be important in some circumstances. Rather than simply adopting Membrane Bioreactor (MBR) processes, combination of the extended aeration biological treatment process with UF membranes provides significant additional benefits, which include far greater process stability, and extended membrane life.

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THE USE OF REED BEDS FOR TREATMENT OF LANDFILL LEACHATES

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ABSTRACT
Safe treatment and disposal of leachates is an important issue at many older landfill sites, where the ingress of rainfall or groundwater is a significant issue needing consideration. Such leachate may typically be relatively weak, but flows are often characterised by large seasonal variations in both volume and strength, in response to winter rainfall. This paper presents long-term data from several long-term, full-scale studies, where reed beds have been used successfully, to provide pre-treatment, polishing treatment, and full treatment of landfill leachates, and to achieve required standards for discharge into public sewers, or into surface watercourses.

1. INTRODUCTION

The on-site treatment of leachates has become an essential part of operations at many landfills in the UK, and at many sites reliable and cost-effective biological treatment systems have been designed and installed. Almost all of these operate as Sequencing Batch Reactors (SBRs), the first such UK system having been designed as long ago as 1982. Several papers have presented detailed operational results from such plants in recent years (Robinson et al., 2003, 2005; Robinson, 2015b; Novella et al., 2004; Carville et al., 2003; Robinson, 2003;) many of these describing plants which have made safe discharges into sensitive watercourses over many years, by use of reed beds for effective effluent polishing to high standards.

A continuing problem, however, remains the uncontrolled discharge of leachates from old landfill sites, many of which were originally engineered to standards far lower than is now acceptable. Although leachates may be relatively diluted, often because of groundwater ingress into unlined landfills, their impact on local watercourses can still be significant.

Reed bed treatment systems have found wide application as robust polishing processes after SBR treatment of raw leachates, prior to the final discharge of very high-quality effluents into watercourses (e.g. see Robinson, 1996; Robinson and Knox, 2001; 2003). However, reed beds are unable to provide good treatment of concentrations of ammoniacal-N much greater than about 20 or 30 mg/l (Cooper, 1999; Cooper and Green, 1995; Cooper et al., 1997), especially during colder winter months. Nevertheless, at older closed landfills, where much weaker leachates may be generated and released, and where low maintenance solutions are essential, reed beds can have a role to play. This chapter provides design information for both Vertical and Horizontal Flow reed bed systems, and performance data from detailed case studies at four closed landfill sites, for which several decades of data are available.

2. REED BED DESIGN AND OPERATIONS

2.1 Reed bed Design

Reed beds are designed to pass flows of wastewater either horizontally (Figure 1), or vertically (Figure 2). For each design type, most successful applications involve subsurface flow within gravel or sand media into which reeds have been planted – avoiding surface free-water flow, which would bypass the main treatment surfaces. Horizontal Flow Reed Beds (HFRBs) receive an inflow from an overflowing halfpipe structure at the inlet end of the bed, before water flows across and through the flooded bed, at a depth which can be adjusted by means of an adjustable overflowing outlet. Single-size gravel media (typically 10mm pea gravel) is generally flooded to just below the gravel surface, avoiding surface flows bypassing treatment, and allowing water to flow horizontally, at a steady rate.

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In a Vertical Flow Reed Bed (VFRB), the packing media can be a range of sizes, and water levels in the bed vary during treatment cycles. Incoming leachate, or pre-treated leachate, enters as occasional 'slug' doses (ideal for use in combination with SBR pre-treatment, where biological effluent is discharged in batches), and floods the bed surface. The liquid gradually passes down through the bed, contacting oxygen in the spaces between the media particles. The bed becomes fully flooded, and effluent continuously drains from the bottom of the bed at a controlled rate. As the liquid drains out, fresh air, containing oxygen, is drawn down into the media of the bed. Eventually the bed drains completely, ready for another dose of feed. Vertical flow beds therefore have greater oxygen inputs, so can provide more treatment (e.g. nitrification of ammoniacal-N) but are usually not so good at solids removal (Morris and Herbert, 1997).

2.2 Contaminant removal mechanisms within reed beds

The types of reed beds in case studies described below are four lined, gravel-filled, horizontal flow beds, and one combined system with both vertical and horizontal beds used to polish leachates that have been pre-treated in a modified SBR process. The four UK reed beds discussed in this paper are as follow; Monument Hill Landfill (Devizes), Shirley Landfill (West Midlands), Efford Leachate Treatment Plant (Hampshire), and Small Dole Leachate Treatment Plant (West Sussex).

Reeds, *Phragmites Australis*, have been planted into the gravel at each site. Effluent enters at the inlet of the beds, travelling slowly through the bed following a horizontal flow-path, before overflowing via a level control device. Although vertical flow reed beds have been reported to provide higher rates of removal of ammoniacal-N, their reduced performance in achieving removal of solids, and the intrinsic simplicity of the horizontal bed, were key to horizontal beds being selected at each site below.

Iron and suspended solids are readily removed in a reed bed system, principally by oxidation and physical filtration processes. The rhizome system of the reeds within the gravel bed may contribute to improved performance, by enhancing the supply of oxygen available by passive diffusion, which is required to convert soluble iron to insoluble iron hydroxide.

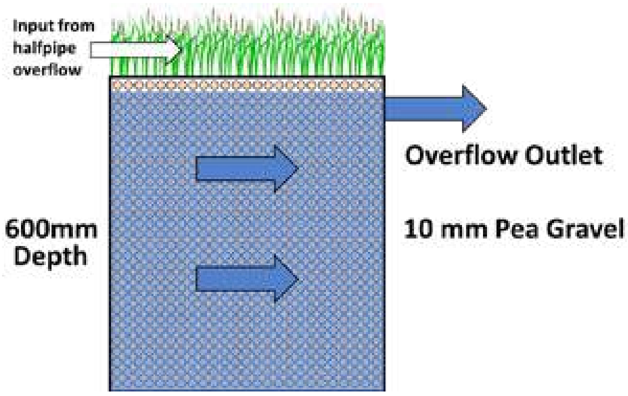

FIGURE 1: Cross-section of a horizontal flow reed bed.

Reed beds are particularly good at removing methane from effluents by means of aerobic biological degradation. Methane is readily oxidised biologically by bacteria, in the presence of oxygen. Therefore, because oxygen enters the reed beds by passive diffusion, assisted to some extent by oxygen transfer via the reed plants, methane can be removed successfully. This removal has been demonstrated at a reed bed at Shirley Landfill Site in the UK, where dissolved methane levels must satisfy a 0.14 mg/l discharge consent (see Robinson, 2017a).

Although reed beds have a poor record for removal of ammoniacal nitrogen from effluents containing high levels of COD and BOD (for example, widely noted for direct treatment of domestic wastewaters), they are generally more successful in situations where concentrations of organic contaminants are much lower, (as in the Monument Hill leachate, or for biologically pre-treated leachates), and more oxygen is therefore available to nitrifying organisms, principally *Nitrosomonas* and *Nitrobacter*, which convert ammoniacal nitrogen to nitrite, and then to nitrate. The full-scale case studies provide detailed design and operational information.

3. MONUMENT HILL LANDFILL

3.1 Background

Monument Hill Landfill Site is an infilled valley, 2 km east of the town of Devizes, Wiltshire, in Southern England, and was filled with household wastes during the 1970s, and is unlined, with a culverted stream beneath the landfill in the valley bottom. The 10-15 m overburden of wastes previously caused failure of the culvert, resulting in contamination of the stream over many years.

In 1985, to improve this situation significantly, a new culvert was prepared to divert the stream around the landfill, but the old culvert remained in place and caused continuing, albeit substantially reduced, minor downstream pollution of the Stert watercourse downstream of the site (see Figure 3). In 1992, after a detailed monitoring exercise, a reed bed leachate treatment scheme was installed on top of the old landfill, capable of treating up to 300 m^3 of leachate per day, and compatible with the nature reserve in which the restored site is located (Robinson et al., 2007).

As the site is remote, closed and unmanned, a low maintenance, low cost, vandal-resistant system was required for treatment of pumped leachate flows, which were typically in the range 200-300 m^3/d. Based on physical constraints posed by the site, and wildlife sensitivity, the only area available for construction was over-infilled parts of the site, and based on required effluent standards indicated by the Environment Agency, an engineered reed bed scheme was developed.

3.2 Leachate quality

An intensive programme of monitoring of the site began during Autumn 1993, to complement the long period during which samples of leachate had been taken by Wiltshire County Council prior to this. Monitoring included continuous measurement and recording of flows of leachate,

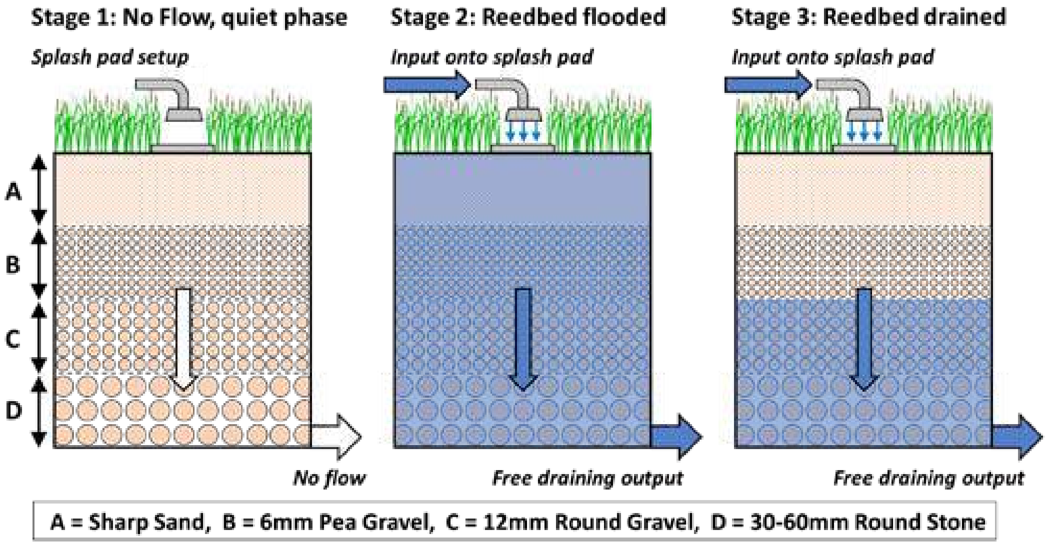

FIGURE 2: Cross-section of a vertical flow reed bed.

and of the flow within the Stert Watercourse.

Water samples were routinely obtained and tested, and results are summarised in Table 1.

Contaminants present in the leachate discharge, considered to have continuing potential for significant adverse impact on the Watercourse, were iron, suspended solids, and ammoniacal-N. Iron was unlikely to be a health concern; its main impact being the orange staining that was evident for a distance of 10 m below the discharge point. Levels of suspended solids in leachate, associated to some extent with particulate iron, were typically about 60 mg/l, and needed to be reduced. Ammoniacal-nitrogen was of concern due to its potential toxic effect on aquatic organisms, salmonid fish, such as trout, being particularly sensitive.

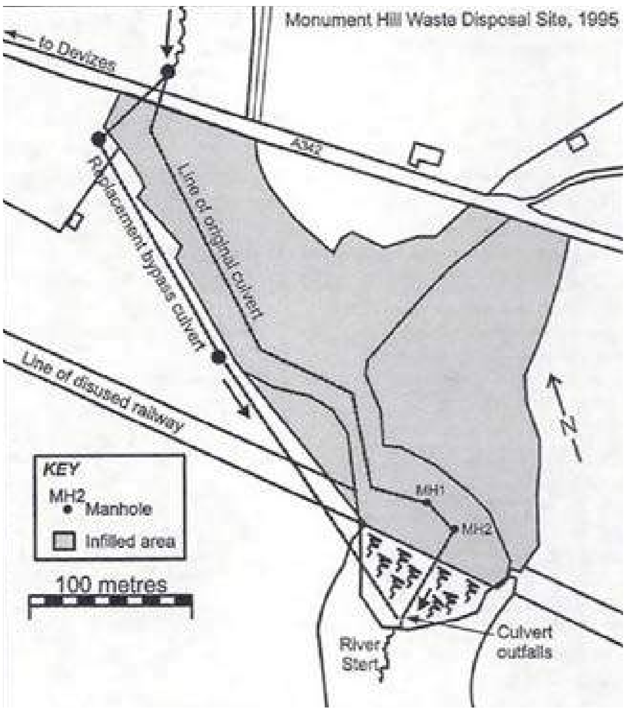

FIGURE 3: Monument Hill waste disposal site in 1995, prior to implementation of remedial works (after Robinson et al., 2007).

Leachate analyses determined that concentrations of up to 19 µg/l Mecoprop (MCPP) (a phenoxy alkanoic herbicide) were also present in the leachate flows (values as high as 0.6 µg/l were also measured in the upstream Stert Watercourse, presumably of agricultural origin). Even though mecoprop is of low toxicity to mammals, fish, and insects, and is readily and completely degraded in aerobic situations such as soil (Heron and Christensen,1992), UK guidance states that it should not be applied near to watercourses. In the light of the above, it was considered likely that treatment would significantly reduce the concentrations of MCPP entering the stream.

Toxic trace metals are often stated to be of concern by regulators in dealing with discharges of raw or treated landfill leachates, either for treatment in sewage works, or directly into surface watercourses. Previous research has demonstrated that the speciation of metals within landfill leachates is the main contributing factor as to the toxicity of several trace metals within leachates (Jensen et al., 1999; Baun and Christensen, 2004).

Jensen and Christensen (1999) stated that in leachates, concentrations of some heavy metals can be very low, whilst further research work has demonstrated that heavy metals are rarely found at significant levels in any methanogenic leachates, unless the landfills have received specific direct inputs of such metals within incoming waste streams (e.g. Robinson, 1996; Robinson and Knox, 2001; 2003). No significant concentrations were detected in samples of leachate at Monument Hill (Table 1).

Presentation and discussion of monitoring results with the Environment Agency, including specific discussion of ammoniacal nitrogen removal, led to the Agency defining the discharge consent conditions as follows:

- BOD (10 mg/l);
- ammoniacal nitrogen (23 mg/l);

TABLE 1: Summary of design data for leachate quality entering the HFRB at Monument Hill Landfill site, December 1993 to October 1994.

Determinand	Units	Samples	Mean	Min	Max
pH-value	pH	14	7.1	6.8	7.8
COD	mg/l	22	43.6	25	64
BOD_5	mg/l	21	<5	1.4	5.0
Ammoniacal-N	mg/l	21	25.5	16.7	31
Chloride	mg/l	20	94.7	83	108
Suspended solids	mg/l	14	57.5	50	70
Conductivity (µS/cm)	µS/cm	7	1,330	1,210	1,472
Sulphate (as SO_4)	mg/l	6	48.3	26	86
Phosphate (as P)	mg/l	2	0.3	-	0.3
Sodium	mg/l	7	59.3	54	67
Magnesium	mg/l	7	16.8	15	20
Potassium	mg/l	12	31.8	26	36.4
Calcium	mg/l	12	215	196	235
Chromium	mg/l	7	<0.1	<0.01	<0.1
Manganese	mg/l	13	0.81	0.5	0.99
Iron	mg/l	20	21.2	12	28
Nickel	mg/l	7	<0.05	<0.01	<0.05
Copper	mg/l	7	<0.05	<0.01	0.03
Zinc	mg/l	18	0.08	0.05	0.11
Cadmium	mg/l	8	<0.02	<0.002	<0.01
Lead	mg/l	7	<0.05	<0.01	0.02
Arsenic	mg/l	1	0.005	-	0.005
Mecoprop	µg/l	15	5.34	1.06	18.91

- iron (6.5 mg/l);
- suspended solids (25 mg/l).

Remedial works comprised a new sump to intercept leachate flows, a settlement chamber to remove precipitated iron, and an 1,800 m^2 area of lined, 600mm deep, gravel-filled Horizontal Flow Reed Bed, for degradation of low levels of BOD and mecoprop. Some reduction in concentrations of ammoniacal nitrogen was also anticipated, especially during warmer summer months, when the watercourse, which receives the final effluent, is most sensitive, but was not generally required by the consent, which took account of dilution available within the receiving watercourse.

3.3 Leachate flows

Flows within the diverted Stert Watercourse (which would receive treated leachate from the site), and of leachate draining from the landfill via the old culvert, were continuously monitored during an initial twelve-month investigation period. Figure 4 is a plot of the relationship between measured daily flows in the Stert Watercourse (range 1,000 to 4,000 m^3/d), and daily flows of leachate from the old culvert (range 60 to 300 m^3/d). Results demonstrated that during 1994 the minimum dilution available at any time was at least 5:1. Dilution exceeded 6:1 more than 99% of the time; and exceeded 10:1 for 70% of the time. This fact was considered in the design of remedial works.

Rainfall records clearly demonstrated that flows of leachate from the old landfill were not rainfall dependent. It was calculated that mean infiltration rates through the old landfill surface were likely to lie in a range between 25 and 33 m^3 per day, compared with flows of leachate, which were typically between 180 and 220 m^3/d. It was therefore concluded that most of the leachate being discharged via the old culvert almost certainly represented groundwater inflows into wastes, and the drainage system in the landfill base. Efforts were therefore concentrated on treatment of leachates, rather than in trying to reduce volumes being generated.

3.4 Design and construction of the reed bed

The Horizontal Flow Reed Bed was sized using experience gained from an experimental reed bed designed that had successfully polished effluent from a leachate treatment plant at Compton Bassett, Wiltshire (Robinson, 1993). Being pre-treated, that effluent had a low BOD, similar to that of raw leachate at Monument Hill. The Compton Bassett bed was therefore extrapolated to give a required bed size of 1,800 m^2 at Monument Hill. 10 mm single-sized pea gravel, placed to a depth of 600 mm, and with a porosity of about 40%, provided the required 2-3 days hydraulic retention time. This size of bed resulted in an iron loading rate of 4g/m^2/d, which was considered adequate, with additional spare capacity to account for the bed possibly becoming clogged with iron deposits over time.

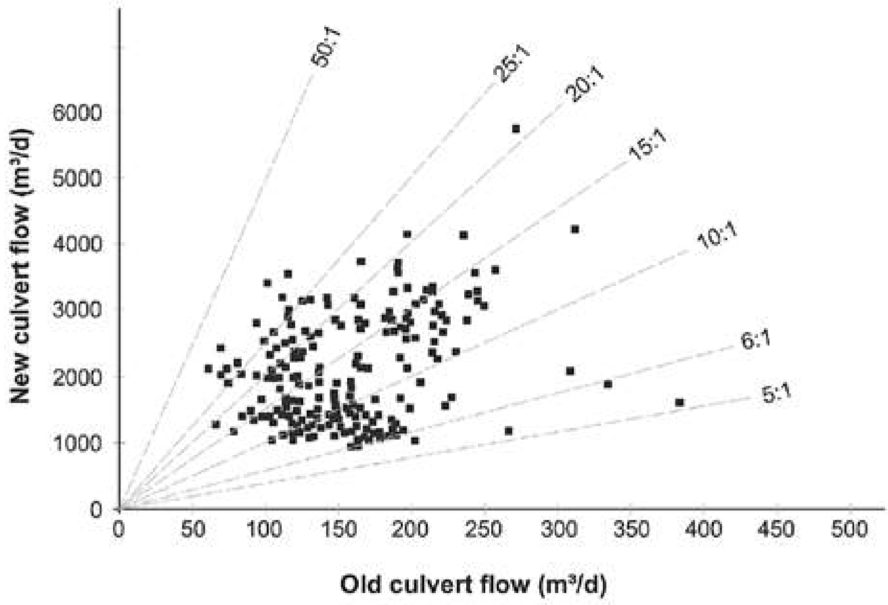

FIGURE 4: Comparison of total measured daily flows in the old and new culvert outfalls at Monument Hill landfill site during 1994 (results in m^3/d) (as presented in Robinson et al., 2007). Lines represent degree of dilution available within the watercourse.

In recognition of the fact that the reed bed would gradually accumulate precipitated iron, a preliminary settlement tank was included at the front end of the bed, to be cleaned out occasionally, by vacuum tanker.

Table 2 summarises the raw leachate quality used for design purposes. The reed bed, was completed during July 1996, lined with 2.5 mm HDPE with a geofabric protection layer, and filled with 600 mm of gravel (Figure 5). It was planted with 20,000 9 cm pot-grown plants of *Phragmites australis*. Water levels were initially maintained at the surface of the gravel, to avoid short-circuiting of flows, and to discourage weed growth, but then reduced by a few centimetres for final operation.

3.5 Performance of the Monument Hill Reed bed

Table 3 presents results obtained from analysis of samples taken 8 weeks after commissioning. The removal of iron could be traced through the system, with 28% being removed in the settling tank, and the remainder being removed within the reed bed, resulting in the iron concentrations in final effluent discharge being reduced to below detection limits. The header tank had no effect on the concentration of ammoniacal-N, and was not expected to. The removal rate for ammoniacal-N within the reed bed was 40%, with subsequent dilution within the Watercourse occurring at the agreed effluent discharge point.

Chloride values demonstrated that the removal of iron, ammoniacal nitrogen and mecoprop in the reed bed were not due to dilution. The removal of mecoprop by the reed bed, from 10.5 µg/l in the influent, to 2.68 µg/l in the effluent was extremely encouraging at such an early stage in the commissioning of the scheme.

Figure 6 presents initial results for the concentrations of suspended solids in treated leachate being discharged to the Stert Watercourse, in samples taken from April 1994 to December 1999. A dramatic and immediate improvement in levels of suspended solids entering the stream from the landfill was evident as soon as operation of the reed bed

TABLE 2: Results from analysis of samples taken from the old culvert at Monument Hill landfill site, on 8 January 1996 (results in mg/l).

Determinand	Total leachate	Filtered on-site	Filtered @ 24 hours
COD	47	47	47
BOD_5	3	3	<2
Ammoniacal-N	19.2	18.9	19.4
Iron	16.6	14.3	<0.6

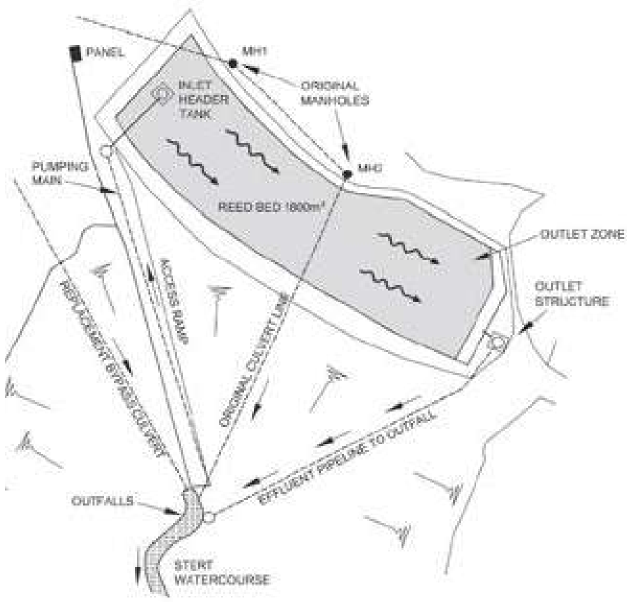

FIGURE 5: Layout of the reed bed at Monument Hill Landfill (Robinson et al., 2007).

TABLE 3: Initial results from analysis of sample from various locations at Monument Hill, in September 1996, soon after introduction of the reed bed scheme.

Determinand	Units	Raw Leachate	After Settling Tank	Reed Bed Effluent	Upstream Watercourse	Agreed Downstream Sampling Point
pH value	pH	6.8	6.9	7.4	8	7.6
BOD_5	mg/l	<2	<2	<2	<2	<2
Ammoniacal-N	mg/l	19.4	19.6	11.8	<0.3	1.8
Iron	mg/l	16.9	12.2	<0.6	<0.6	0.7
Suspended solids	mg/l	42	42	3	19	16
Chloride	mg/l	78	77	76	23	32
Mecoprop	µg/l	9.4	10.5	2.68	<0.1	0.44

began in July 1996, with concentrations falling from about 50 or 100 mg/l in leachate, to levels rarely above 10 mg/l in effluent. This improvement has continued for more than 10 years (see Figure 7). Data demonstrate that although high levels of solids remain present in leachate, and sometimes in the Stert Watercourse, levels in treated leachate continue to rarely exceed 10 mg/l.

Introduction of the reed bed immediately effected reliable and almost complete removal of iron, generally to background concentrations. Removal of iron during the initial 10-year period of reed bed operation is summarised in Figure 8, where essentially complete removal of iron has continued to be achieved by the combination of the preliminary settlement tank, (desludged as required, approximately once per year) and the bed itself. After 10 years there was no evidence that accumulation of iron within the bed had caused any reduction in treatment performance whatsoever. In addition, levels of iron in raw leachate have gradually fallen, to values typically between 5-18 mg/l.

Figure 9 contains equivalent early data for ammonia-

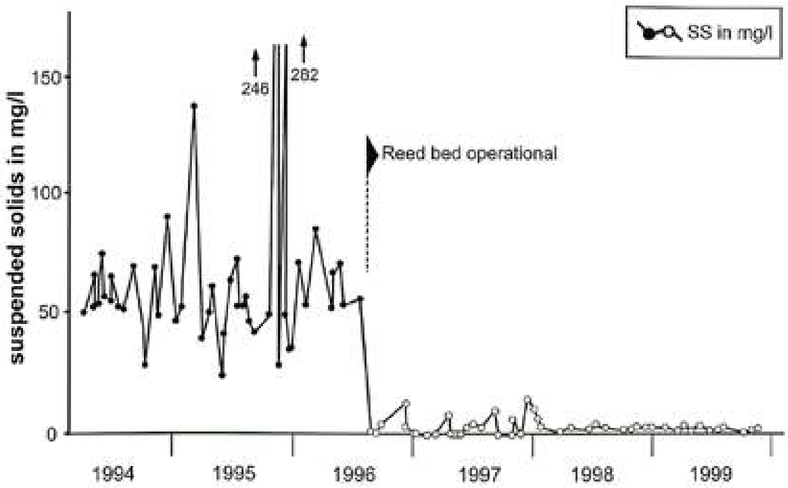

FIGURE 6: Concentrations of suspended solids in the discharge to the Stert Watercourse, 1994-1999 (Robinson et al., 2007).

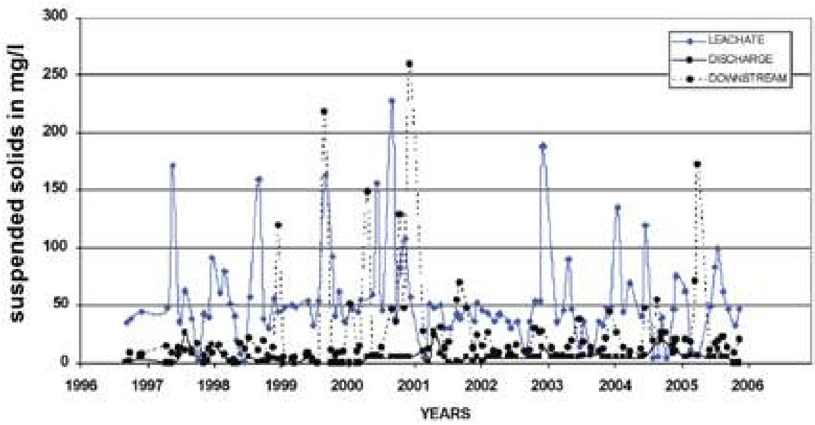

FIGURE 7: Concentrations of suspended solids in leachate, treated effluent, and downstream of the Monument Hill reed bed, within the Stert Watercourse, 1996 to 2006 (Robinson et al., 2007).

cal-N. Results for ammoniacal-N in leachate had historically shown slightly elevated concentrations (typically 25-30 mg/l) during summer months, compared with values near to 20 mg/l during winter months. Introduction of the reed bed in July 1996 resulted in a significant and consistent reduction of about 50% in concentrations of ammoniacal-N, typically to between 10 and 15 mg/l.

Results demonstrate that, during an extended period from 1996 to 2006, not only did concentrations of ammoniacal-N in leachate fall to some extent (presently 15-20 mg/l), but removal rates during treatment in the bed have also improved. Typically, between 50 to 70% of incoming ammoniacal-N is removed, leaving between 3 and 10mg/l of ammoniacal-N in effluent discharged from the reed bed into the watercourse. Dilution available within the Watercourse, as anticipated, has meant that concentrations of ammoniacal-N in the stream below the landfill rarely exceed 1 or 2 mg/l.

Although there is evidence of increased concentrations of nitrate after treatment in the bed, this does not account for all of the removal of ammoniacal-N being observed. Other processes such as uptake into the reeds, or some denitrification, must therefore be taking place. Although no consistent records of volumes of leachate being treated in the reed bed are now being kept, evidence from occasional flow monitoring and pumping records, indicate that flows of 100-200 m^3/d remain typical. At these flow rates during the last 5 or 6 years, reductions in concentration of 10-12 mg/l of ammoniacal-N are common (higher removal during summer months), allowing a range of removal rates in terms of grams of ammoniacal-N removed per m^2 of bed area to be estimated broadly as follows:

- Summer: 0.65-1.35 $gN/m^2/day$
- Winter: 0.55-1.10 $gN/m^2/day$

Concentrations of mecoprop in leachate have remained at generally similar levels throughout the period 1994-2006, typically 4-8 µg/l. Treatment in the reed bed has always kept concentrations in effluent below 2 µg/l. Results for chloride in leachate give a general indication of changes in raw leachate strength at Monument Hill, and show that although this remained fairly stable from 1994 to late 1999, since that time values have reduced by about 25%. Chloride levels remain unaffected by passage through the reed bed, as would be expected. COD removal in the bed has typically been about 15-20% because of low levels of degradable or-

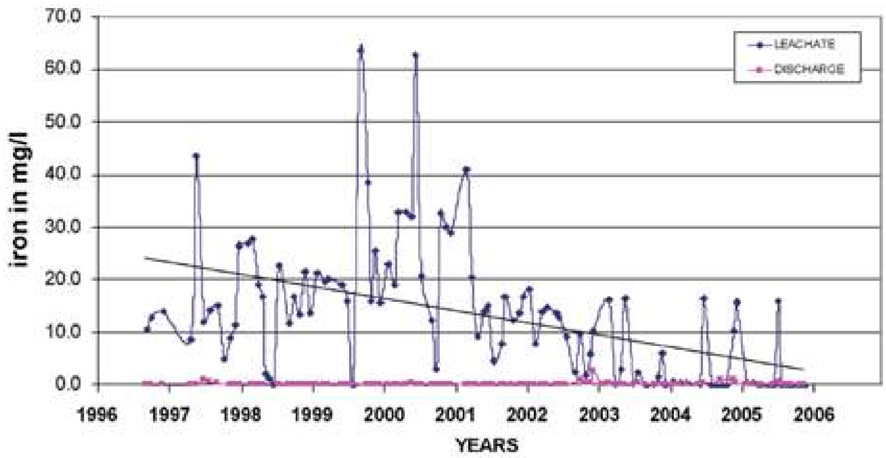

FIGURE 8: Removal of iron after passage through the Monument Hill reed bed, 1996-2006 (as presented in Robinson et al., 2007).

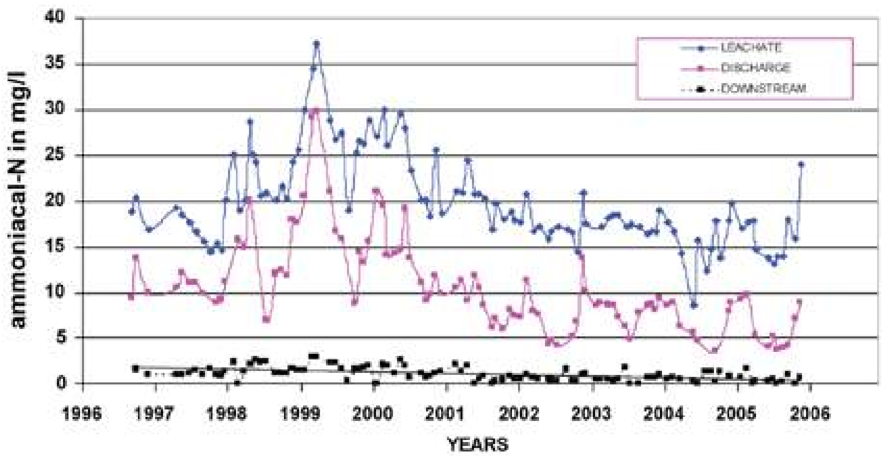

FIGURE 9: Concentrations of ammoniacal-N in raw leachate, in treated leachate, and in the downstream watercourse from the Monument Hill reed bed, 1996-2006 (as presented in Robinson et al., 2007).

ganic matter present in the methanogenic leachate.

3.6 Performance

Analytical results demonstrate that after twenty years, the leachate treatment scheme continues to provide reliable, efficient, and cost-effective protection of the watercourse. Accumulation of iron within the bed, meant that refurbishment was required during 2010, during which gravel was extracted, washed, replaced and replanted, but treatment has since continued to achieve required consents adequately (see Table 4).

4. SHIRLEY LANDFILL

4.1 Background

Shirley Landfill Site is located South West of the city of Birmingham, in the UK Midlands, and is the responsibility of Worcestershire County Council. The site was originally quarried for sand and gravel during the 1970s, and was restored between 1981 and 1988 by filling with 1.2M m^3 of household wastes, over an area of 15 hectares. The average depth of the waste is about 8 m with a maximum of 12 m and a minimum of 3 m.

A reed bed at Shirley was designed and constructed during 2013, primarily to reduce concentrations of methane in leachate draining by gravity from the landfill, where it was recognised that uncontrolled inflow of groundwater was a significant contributor to leachate generation rates. Concentrations of dissolved methane were routinely exceeding a recently-imposed limit of 0.14 mg/l, and removal would take place by means of aerobic biological degradation, since methane is readily oxidised biologically by bacteria, in the presence of oxygen. Oxygen would enter the reed bed by passive diffusion, assisted to some extent by oxygen transfer via the reed plants. Four years' data are available to demonstrate not only successful removal of methane (which is discussed in detail elsewhere, see Robinson, 2017a), but also provide valuable information on the limited and seasonal removal of ammoniacal-N being achieved by the bed.

4.2 Design and construction of the reed bed

The design was based on flow information provided by the Council; that mean flow rate would be about 50 m^3/d, and within a range from 24 m^3/d to a maximum flow of 78 m^3/d. Leachate draining from the site is captured by a series of French drains and a pipeline that runs to a chamber within the site, before being discharged into the public sewer. On a number of occasions, the limit set by the discharge consent for dissolved methane was being exceeded, which had the potential to be hazardous.

As at Monument Hill, uncontrolled inflow of groundwater was a significant contributor to leachate generation rates. A reed bed was a far more sustainable and practical option for an unmanned, relatively remote, closed landfill site, than a mechanical methane stripping arrangement, and it was recognised that the development was necessary to avoid pollution, and that the only alternative would have been to take leachate off-site in tankers, generating traffic and causing amenity impacts.

There was no means of buffering gravity leachate flows from the landfill, and the reed bed design did not seek to provide any flow buffering. Results indicated that although flow rates showed seasonal variation, they did not respond rapidly to rainfall events; as might be expected from a landfill where significant groundwater inflows were involved (Robinson et al., 2015a).

Leachate transfer arrangements required modification, with construction of a new deep chamber into which leachate would now drain from the site by gravity, and from where it would be pumped by duty/standby pumps into a surface-mounted precast concrete Header Tank, having a volume of 5 m^3. This header tank was designed to encourage the quiescent settlement and retention of any silt or precipitated iron solids, with supernatant leachate overflowing to the reed bed inlet.

The reed bed has a length of 50 m, a width of 7 m, a gravel depth of 0.6m, and an estimated hydraulic volume of about 85 m^3, giving a mean hydraulic retention time (HRT) of between 1 and 2 days at anticipated flow rates. Effluent from the bed drains into a discharge chamber at its remote end, over a variable level control mechanism, which maintains water level within the bed just below the gravel surface. Plate 1 gives an overview of the reed bed treatment system.

4.3 Performance

The reed bed performs well, removing all methane from leachate entering it on most occasions, including when flows were more than double design rates during early 2014. However, of main interest, are data for removal of ammoniacal-N. Since the bed was commissioned in July 2013, routine sampling of raw and treated leachates has been carried out regularly, and all flow meters and record-

TABLE 4: Performance of the Monument Hill Reedbed during 2016 (10 samples).

Determinand	Reedbed outflow	Downstream Watercourse
BOD	1.5	2.75
iron	0.56	0.11
ammoniacal-N	15.1	1.33

Notes: results expressed represent mean value of 10 samples taken during 2016.

PLATE 1: General view of Shirley Reedbed from the inlet end, showing the Leachate Header Tank in the foreground, September 2014. (Robinson, 2017a).

ing instruments have performed accurately and reliably.

Having observed loading rate data for removal of ammoniacal-N, it is evident that some seasonal removal of ammoniacal-N is taking place. However, this was not part of the original design purpose of the bed.

The most significant impact on operation of the bed, since it was commissioned, has been the flows of leachate passing through it, which have exceeded the original design specification. Extreme and record-breaking levels of rainfall during the early months of 2014, with more than double average rainfall amounts during January and February, led to the bed receiving and treating leachate flows as high as 160 m^3/d, with highest values recorded during late February/early March 2014 (see Figure 10).

During the full year from 1 October 2013 to 30 September 2014, mean leachate flow rate was just over 65 m^3/d; 30 per cent greater than predicted values, and maximum flow rate of 163 m^3/d was more than double the anticipated maximum flow rate of 78 m^3/d.

During the first 3 months of 2014, more than 10,000 m^3 of leachate passed through the bed (10,348 m^3), at a mean flow rate of 115 m^3/d, with a maximum monthly flow of 3,766 m^3 during February 2014 (mean rate 134.5 m^3/d); 45 per cent greater than predicted maximum instantaneous flow rates, throughout the month.

Table 5 presents the criteria for the discharge consent, as set by Severn Trent Water Limited, for discharges of effluent from the Shirley Reed Bed. The maximum volume of effluent to be discharged to sewer, was set at 137 m^3 during any single 24-hour period.

Results comparing concentrations of various contaminants in incoming leachate flows are compared with values determined in treated leachate discharged to sewer, in Figures 11 to 13. Results for chloride in raw and treated leachate are presented in Figure 11.

These results confirm that no significant dilution or concentration of contaminants took place during passage of leachate through the reed bed, which means that changes in concentrations of other contaminants can entirely be attributed to treatment being provided by biological and chemical changes taking place within the bed.

Of interest is the fact that although flow rates of leachate from Shirley Landfill, increased substantially during early 2014, this was not associated with equivalent dilution of the leachate being received for treatment. This is characteristic of landfills where high proportions of leachate being produced are derived from groundwater inflows.

4.4 Other contaminants

Reed bed performance in terms of removal of other contaminants is discussed below. Figure 12 examines changes in COD values through the bed, which were minimal. Figure 13 presents results for ammoniacal-N in raw and treated leachates, which show an interesting picture.

Although concentrations of ammoniacal-N were lower during the period October 2013 to May 2014, typically between 8 and 11 mg/l, removal rates were minimal (<10 per cent), no doubt due at least in part to the very high flow rates during this period. However, during warmer months of each year, when flow rates were also reduced, although ammoniacal-N was typically present at between 12 and 14 mg/l, removal rates of up to 50 per cent were achieved during the period July to September 2013, and again during the summer periods of 2015 and 2016. At slightly greater flow rates during summer 2014, Ammoniacal-N removal

TABLE 5: Discharge conditions set by Severn Trent Water Limited on 14th August 2014, for wastewaters being discharged into the Upper Cole Valley Sewer.

Condition / Determinand	Units	Discharge consent set by the EA
Maximum Discharge Rate	l/sec	2
Dissolved Methane	mg/l	<0.14
pH value	pH-Value	6 to 10
COD	mg/l	300
Ammoniacal-N	mg/l	50
Phosphorus	mg/l	25
Suspended solids	mg/l	200

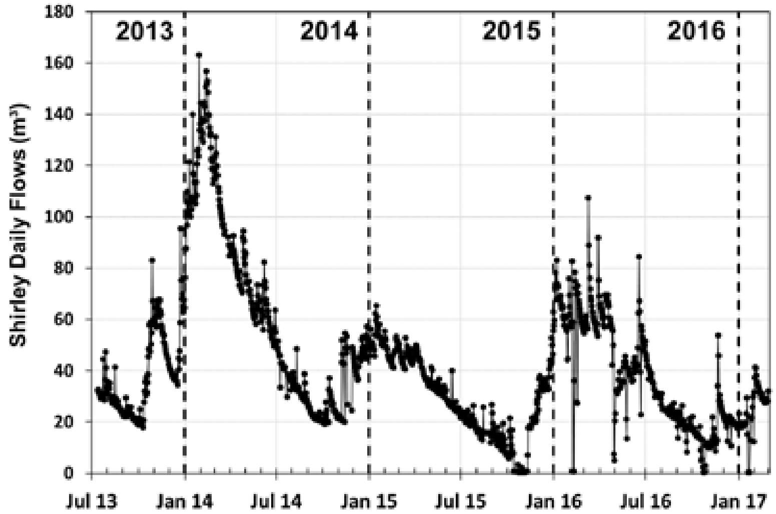

FIGURE 10: Daily volumes treated at Shirley, July 2013 to February 2017 (in m^3/d).

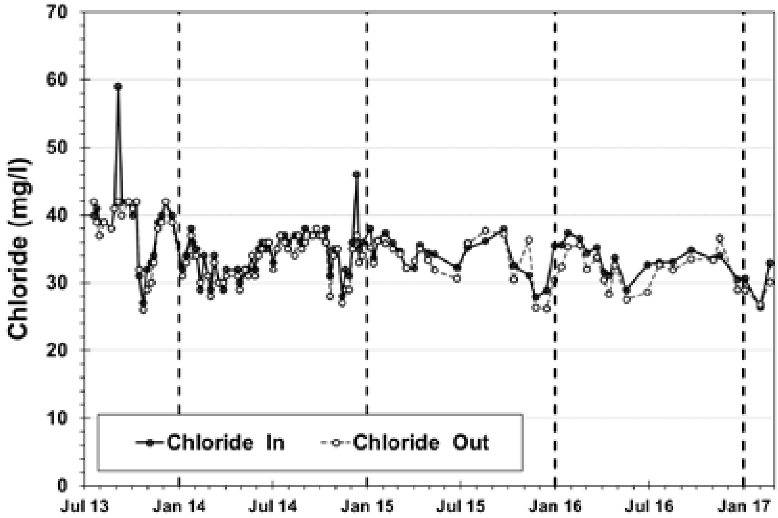

FIGURE 11: Variation in concentrations of chloride during passage through the Shirley reed bed, July 2013 to February 2017 (all results in mg/l as chloride).

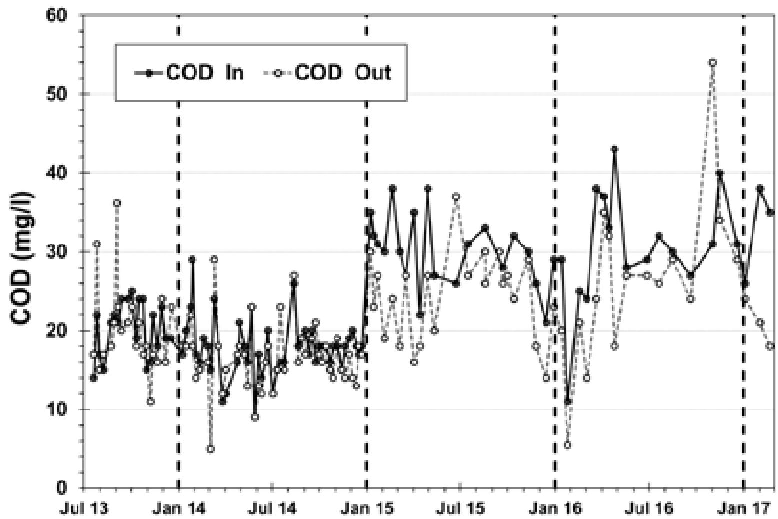

FIGURE 12: Variation in COD values during passage through the Shirley reed bed, July 2013 to February 2017 (all results in mg/l).

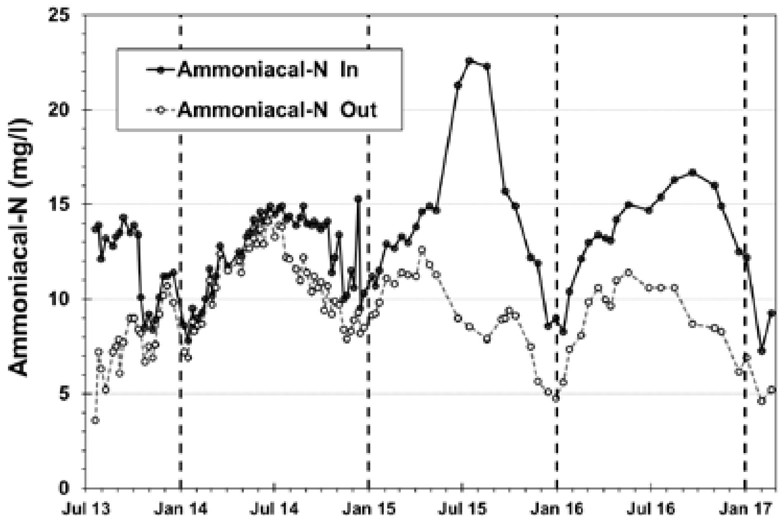

FIGURE 13: Variation in concentrations of ammoniacal-N during passage through the Shirley reed bed, July 2013 to February 2017 (all results in mg/l as N).

rates of up to 25 or 30 per cent were still achieved.

Removal of ammoniacal-N was not any part of the specific design of the reed bed at Shirley, but is clearly being achieved to a significant extent during warmer summer months:

- Summer: 0.9 to 1.0 gN/m^2.day
- Winter: 0.4 to 0.5 gN/m^2.day

5. EFFORD LEACHATE TREATMENT PLANT

5.1 Background

Efford is a closed landfill site on the south coast of England, where during early 2003, a leachate treatment system was constructed, which incorporates a fully-automated SBR treatment process, with an engineered reed bed polishing system to achieve very high effluent quality standards. The plant treats up to 150 m^3/d of strong methanogenic leachate, and more than thirteen years of operational data, collected since 2004, demonstrate the ability of the plant to meet stringent effluent discharge standards. Of particular interest are results which demonstrate the effectiveness of the reed bed polishing system, in providing removal of residual ammoniacal-N, suspended solids and BOD, allowing safe discharge of treated leachate into a small rural sewage treatment works, which itself discharges effluent into the very sensitive River Avon in Hampshire (Robinson and Olufsen, 2007).

TABLE 6: Discharge conditions set by the Environment Agency for treated leachate at Efford Landfill Site.

Condition / Determinand	Units	Discharge consent set by the EA
Maximum Discharge Rate	l/sec	4
Dissolved Methane	mg/l	<0.14
pH value	pH-Value	6 to 10
COD	mg/l	2,500
Ammoniacal-N	mg/l	80
Chloride	mg/l	2,000
Suspended solids	mg/l	400
Tin	mg/l	0.15
Chromium	mg/l	0.25
Copper, lead, nickel	mg/l	0.50
Zinc	mg/l	1.5

Leachate being produced at the Efford landfill site is strong, with a mean ammoniacal-N concentration of nearly 600 mg/l, mean COD of just under 1,000 mg/l, chloride of 1,400 mg/l, and alkalinity of 4,000 mg/l. Because of the small size of the receiving sewage treatment works, and the fact that it makes discharges of effluent directly into the Avon, the following effluent discharge conditions in Table 6 were set.

The leachate treatment system was designed to be capable of treating up to 150 m^3/d of strong leachate and is typical of many similar systems routinely being installed at similar sites globally (e.g. Novella et al., 2004). The plant is shown in Plate 2.

The performance of the Efford plant has exceeded design values, at all times, and all significant determinands in effluent have consistently been almost an order of magnitude below consented limits. Table 7 demonstrates the effectiveness of the SBR treatment, as well as of additional reed bed polishing.

Figure 14 provides details of total monthly volumes of leachate that have been treated by the plant. Since it was commissioned in January 2003, a total volume of 320,000 m^3 of leachate has been treated and discharged off-site. In recent years, daily flows have averaged about 62 m^3/d.

5.2 Polishing of biologically pre-treated leachates to high standards

Following extensive regular sampling and analysis, the performance of the Efford reed bed in treating key determinands can be observed.

Data for ammoniacal-N in Figure 15, show that con-

PLATE 2: General arrangement of the Leachate Treatment Plant and reed bed at Efford. (Robinson, 2018).

TABLE 7: Overall performance of Efford Leachate Treatment Plant, in terms of removal of key determinands during the treatment process, January 2003 to February 2017.

Years	2003-17	2004-17	2004-17	2004-17
Determinand	**COD**	**BOD_5**	**NH_4-N**	**chloride**
Raw Leachate Median	866	63.85	556	1,380
SBR Effluent Median	234	7.77	0.8	1,350
Reed Bed Effluent Median	205	2.25	0.22	1,260
Overall removal %	**76.3**	**96.5**	**99.9**	**8.70**
Reed bed removal %	**12.4**	**71.0**	**72.5**	**6.67**

Notes: all results in mg/l, over 600 samples tested for each result.

sistently high levels of ammoniacal-N within raw leachate are reliably treated down to concentrations below 10 mg/l, by the SBR system; the reed bed then providing polishing treatment to values below 1 mg/l.

Figure 16 displays a similar pattern for BOD_5, where values as high as 300 mg/l are consistently treated down to below 50 mg/l by biological SBR treatment, and then to much lower values by passage through the reed bed.

From January 2003 for 18 months, the plant was only treating leachates from older parts of the landfill, typically

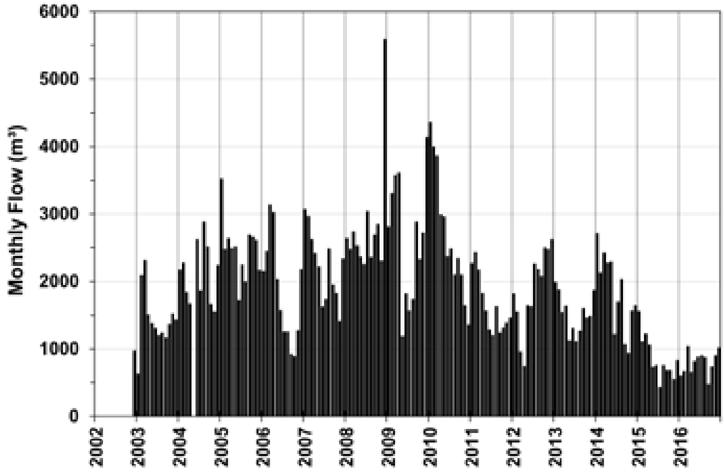

FIGURE 14: Monthly volumes of leachate treated at Efford Leachate Treatment Plant.

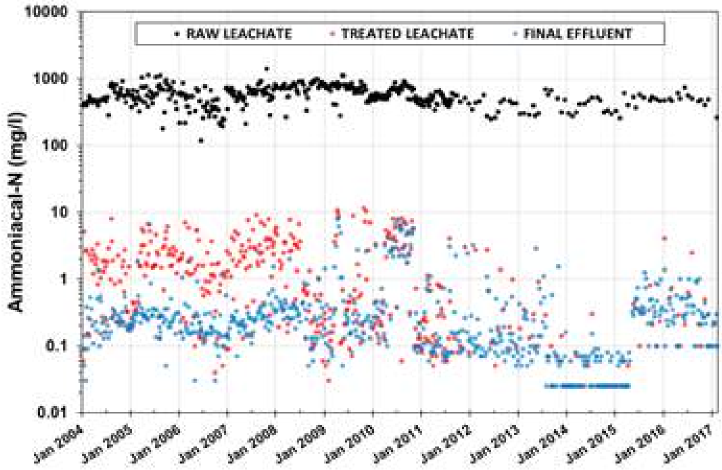

FIGURE 15: Treatment of ammoniacal-N at Efford, January 2004 to February 2017.

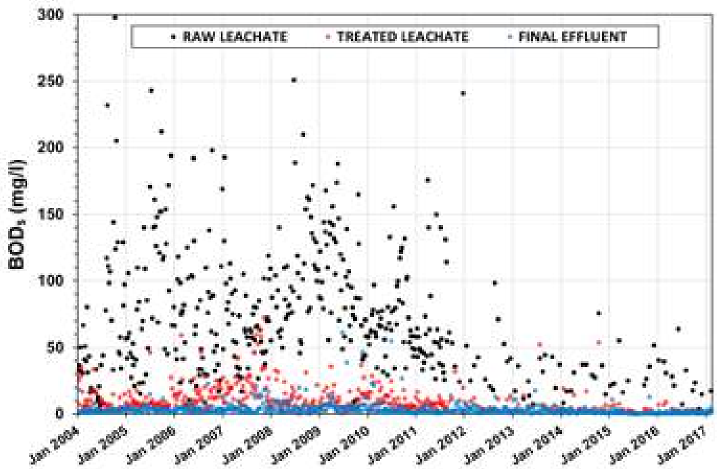

FIGURE 16: Treatment of BOD_5 at Efford, January 2004 to February 2017.

containing COD values from 500 to 1200 mg/l, and concentrations of ammoniacal-N from 400 to 700 mg/l. Raw leachate quality was very stable. After July 2004, variable amounts of stronger leachate, from more recent phases of the landfill, began to be introduced and treated. These leachates were much stronger in both COD and concentrations of ammoniacal-N (to well above 1,000 mg/l). This blending resulted in raw leachate feed that was both stronger (COD values to 2,500 mg/l, ammoniacal-N to 1,200 mg/l), and also far more variable in strength.

The plant nevertheless continued to maintain excellent final effluent quality. The value of the reed bed was clear, in dealing with occasional "spikes" in SBR effluent quality, which arose from the more variable quality of daily contaminant loads. This is particularly evident in data for ammoniacal-N in Figure 15, where levels in final effluent very rarely exceeded 1 mg/l, despite occasional spikes in values in SBR effluent of up to 10 mg/l.

5.3 Performance

Treatment of ammoniacal-N is by means of reliable and complete nitrification to nitrate, typically with about 75 to 90 percent appearing as nitrate-N in final effluent. The reed bed removes very little nitrate nitrogen, in spite of its excellent performance in taking out residual levels of ammoniacal-N, probably because at flow rates in the order of 100 m^3/d, small reductions in concentration of nitrate-N in effluent still represent significant supplies of nutrients to the reeds.

6. SMALL DOLE LEACHATE TREATMENT PLANT

6.1 Background

The final case study will describe use of both vertical and horizontal flow reed beds at the older closed Small Dole Landfill Site in West Sussex, where leachate quality is strongly methanogenic, but year-round contains typically between about 60 and 150 mg/l of ammoniacal-N. Leachate flow rates have varied between 80 and 700 m^3/d since 2010, when a full-scale leachate treatment system was designed and constructed, by substantial refurbishment and reconstruction of an existing treatment plant (Robinson, 2017b).

Treatment involves twin Aeration Tanks, which operate within a modified Sequencing Batch Reactor (SBR) system, by means of an external and separate batch Settlement Tank, shown in Plate 3. Because treated leachate must achieve very strict effluent discharge standards, in order to be disposed of into a small, slightly tidal watercourse, which flows around the perimeter of the landfill site, SBR effluent is passed first through Vertical Flow Reed Beds (VFRB), and then Horizontal Flow Reed Beds (HFRB), to provide polishing to high standards (Robinson, 2017b).

The SBR arrangement at Small Dole enables small volumes of leachate, containing 80 to 150 mg/l of ammoniacal-N, to be diluted within the continuously aerated treatment tanks, so that bacteria are not inhibited. In each 24-hour period, mixed liquor is transferred alternately from each of the 2 aeration tanks every 6 hours, to the settlement tank, before clarified effluent is decanted, and remaining mixed liquor returned to the aerated SBRs.

During discharge of treated leachate from the Settlement Tank, this effluent is fed through vertical and horizontal flow reed beds in series, as a successful effluent polishing process. Reed beds were installed during refurbishment, to provide additional final treatment of the effluent. Effluent then drains into a treated leachate balance tank, designed to enable balancing of discharge flows into the tidal River Adur.

Plate 4 shows the vertical flow reed bed (VFRB) to the right, and the two horizontal flow reed beds (HFRB) to the left, with the river visible in the distance.

Since 2010, flows of leachate have varied significantly; from 80 m^3/day during summer months, to maximum recorded volumes of up to 700 m^3/day during early 2014. Typical mean daily leachate flows during summer periods are below 100 m^3/day, while in winter mean daily flows are typically 400 m^3/day. Figure 17 presents detailed daily flow

PLATE 3: Aerial view of the updated Small Dole Leachate Treatment Plant, following modifications made in 2010. (Robinson, 2017b).

PLATE 4: Aerial view of the Small Dole vertical flow reed bed, and the two parallel horizontal flow reed beds, following construction by Phoenix Engineering in 2010. (Robinson T., 2017).

data for leachate being collected within the Raw Leachate Balance Tank (RLBT).

Records of the flows of leachate treated between 2011 and 2017 have enabled mean seasonal values for leachate generation to be calculated:

- Spring / Summer: (May to October) = 125 m³/day
- Autumn / Winter: (November to April) = 280 m³/day

Because of increased dilution during winter months, leachates generated during summer months are shown to contain more than double the levels of COD and BOD when compared to winter. Similarly, leachates produced during the summer contain 50% more ammoniacal-N than those generated during the winter periods.

Table 8 demonstrates that leachates are consistently treated with COD, BOD_5, and ammoniacal-N all treated to very low levels. during both summer and winter periods.

Although strengths of leachate are much lower during winter months, the overall loading of contaminants are significantly higher during winter periods. Despite lower concentrations of contaminants within the leachate being generated, the sheer volume of leachate containing these contaminants, means a higher load is put through the LTP during winter months.

Figure 18 presents data for ammoniacal-N concentrations and loading results. Although concentrations of up to 150mg/l are reached during summer months, mean daily loads are much higher during winter periods, exceeding 20kg/day of ammoniacal-N during every winter period; and reaching 40kg/day during the winter of 2013/14.

Figure 19 compares results for the concentrations of ammoniacal-N within the leachate at Small Dole, with concentrations of nitrate-N in final effluent. Because values for ammoniacal-N in leachate, and nitrate-N in effluent match so well, this shows that all ammoniacal nitrogen is being effectively fully nitrified. Combined with trace levels of ammoniacal-N in final effluent (presented in Table 8 earlier), this demonstrates the success of the system in achieving complete nitrification, as required by the discharge consent.

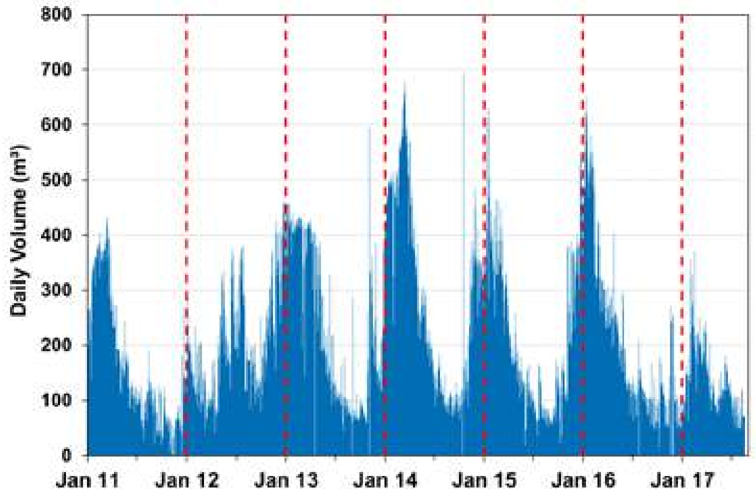

FIGURE 17: Daily Raw Leachate Flows at Small Dole from January 2011 to August 2017 (m3). (Robinson, 2017b).

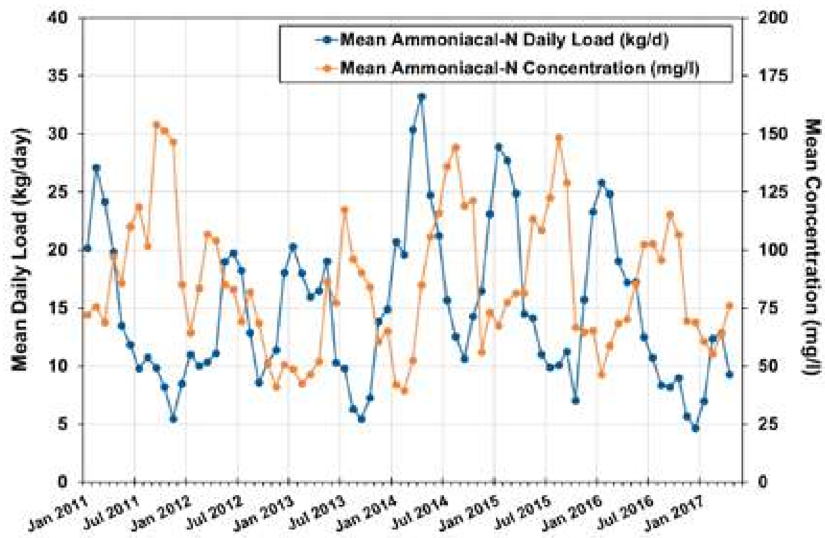

FIGURE 18: Ammoniacal-N mean concentration (mg/l) and mean daily load (kg/day) at the Small Dole reed bed. (Robinson, 2017b).

6.2 Performance

The refurbished plant has performed extremely well, always achieving discharges that are compliant with the site's Environmental Permit, and the combined reed bed polishing system readily provides a back-up to the main biological treatment plant. Research work is continuing (see Robinson, 2017b; Wilson et al., 2017) to examine in detail the contribution to treatment of the individual system components.

Results obtained at Small Dole demonstrate how effectively SBR and reed bed treatment options can be combined, to treat large volumes of leachates and achieve stringent discharge consents; allowing final effluents to be discharged to sensitive watercourses. It remains a hope that ultimately, the reed beds alone may provide a passive system, capable of managing all leachates from the site.

7. OVERALL SUMMARY

All four reed beds, at Monument Hill, Shirley, Efford and Small Dole, continue to perform successfully during 2017, ensuring that effluents from each site readily satisfy the discharge consents set by regulatory authorities.

The Monument Hill reed bed provides removal of suspended solids and iron to very high standards, with significant levels of reduction in concentrations of ammoniacal-N; whilst the degradation of residual levels of BOD_5, COD and mecoprop is also evident. This removal is most effective and important during warmer summer months, when a stream receiving final effluent is most sensitive.

The Shirley reed bed has removed all methane from leachate entering it, even when flows were more than double design rates during early 2014 (see Figure 20). Seasonal removal of ammoniacal-N has taken place (up to 50 percent during 2013, 2015 and 2016), but this was not part of the design purpose of the bed, and as more data are obtained, it has been possible to obtain valuable loading rate data for this removal.

A reed bed receiving treated leachate discharged from an SBR system operating at Efford Landfill Site continues to provide very successful removal of any residual levels of ammoniacal-N and BOD_5.

Table 9 summarizes the removal that each of the reed beds provide for key determinands. All beds demonstrate similar levels of removal for suspended solids, and high corresponding removal of iron (over 90% removal at each site).

Each of the reed beds demonstrate significant removal of ammoniacal-N, with Monument Hill and Efford both removing nearly 80% of NH_4-N, while Shirley removes over a quarter (26%), on a seasonal basis.

Shirley reed bed is very successful at removing high initial levels of dissolved methane (95% removal), as per the intended requirements; ensuring that methane remains well below the 0.14mg/l discharge consent.

Following biological treatment of stronger leachate at Efford, the reed bed there provides additional effluent polishing, by removing close to 70% of residual BOD_5.

TABLE 8: Variations in strength of Leachate produced at Small Dole. (Robinson, 2017b).

Season	Summer Period		Winter Period	
Months	May - October		November - April	
Samples (no.)	160		168	
Sample	Leachate	Effluent	Leachate	Effluent
COD	1,377	99.0	548	77.9
BOD	50.4	1.30	20.9	0.84
Ammoniacal-N	104	0.22	69.0	0.24
Nitrate-N	1.17	101	0.50	71.9
Chloride	606	655	460	391

TABLE 9: Comparison between the performance of three reedbed systems.

Determinand (mg/l)	Monument Hill Reedbed			Shirley Reedbed			Efford Leachate Treatment Plant			
	Raw	Final Eff	%	Raw	Final Eff	%	Raw	SBR	Final Eff	%
COD	54	30	**44.4**	22.9	20.5	**10.5**	963	239	207	**13.4**
BOD_5	3	3	**0.00**	1.23	1.15	**6.50**	74.0	11.43	3.58	**68.7**
NH_4-N	17.8	3.7	**79.2**	12.8	9.49	**25.9**	579	5.95	1.08	**81.9**
Alkalinity	640	505	**21.1**	393	381	**3.18**	3,692	811	774	**4.56**
Suspended Solids	68	6	**91.2**	-	11.71	**NA**	95.1	111	31.1	**71.9**
Sodium	52	51	**1.92**	25.36	24.8	**2.17**	867.9	1,364	1,271	**6.78**
Chloride	92	76	**17.4**	34.9	33.8	**3.15**	1,444	1,427	1,319	**7.57**
Methane	-	-	**-**	1.2	0.06	**95.0**	0.172	0.005	0.004	**25.8**
Iron	10.1	<0.05	**>99.5**	6.29	0.56	**91.1**	13.06	9.11	0.469	**94.9**

Notes: all results in mg/l; % = Percentage removal; Final Eff = Mean concentration in final effluent.

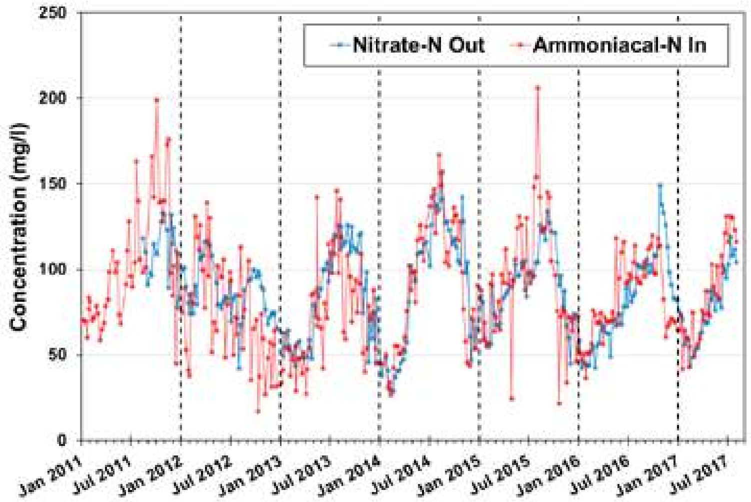

FIGURE 19: Concentrations of ammoniacal-N within raw leachate and Nitrate-N within final effluent at Small Dole, January 2011 to August 2017 (mg/l). (Robinson, 2017b).

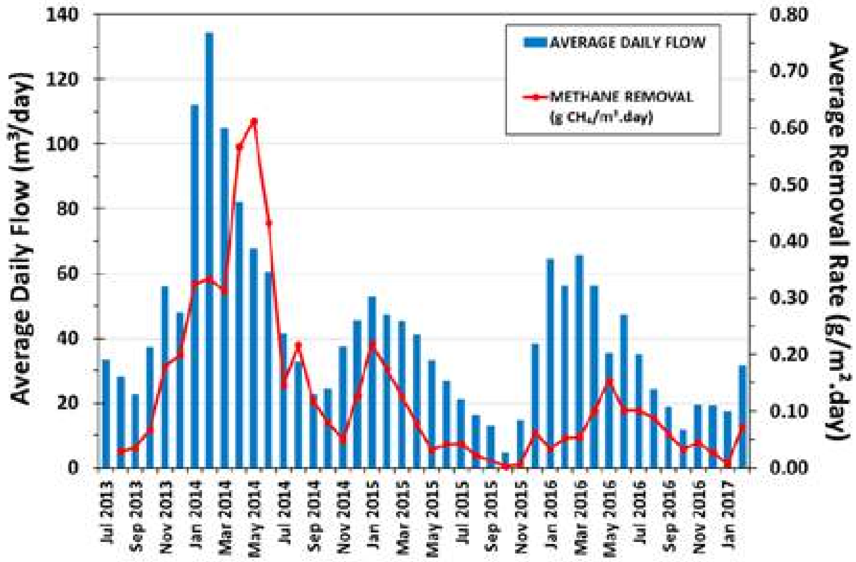

FIGURE 20: Removal of dissolved methane by the Shirley Reed Bed in terms of grams of methane per m2 of reed bed area, July 2013 to July 2016 (Robinson, 2017a).

8. CONCLUSIONS

Reed bed treatment systems are becoming increasingly common on UK landfill sites, although relatively few detailed data have been published from the operation of such systems.

The case studies presented in this report demonstrate that well-designed reed bed systems are able to operate consistently, reliably, and cost-effectively, to meet stringent effluent discharge standards for specific contaminants at all times. Detailed operating data that this paper provides provide great confidence to both treatment plant operators, and to landfill regulators.

In future, similar schemes will have widespread application at many closed landfill sites, where low levels of BOD_5, COD, ammoniacal-N and methane, in weaker leachates, will need reducing to below consented levels. Additionally, metals such as iron, associated with suspended solids, can be readily removed in a similar horizontal flow reed bed system; principally by oxidation and filtration. For unmanned closed sites, a reed bed is a reliable, low-cost leachate treatment solution, requiring little maintenance, supervision and operator input. Nevertheless, it is important to recognise that for leachates containing more than about 10mg/l of ammoniacal-N, complete removal to low levels cannot be guaranteed during winter months.

Reed bed polishing systems such as those operated at Efford and Small Dole for many years, will continue to be incorporated at many future leachate treatment plants, to achieve additional removal of residual low concentrations of ammoniacal-N (less than 5mg/l or maybe 10mg/l), and of BOD values following biological treatment.

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ADAPTATION OF IPCC DEFAULT VALUES ON NATIONAL LANDFILL CONDITIONS

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ABSTRACT

Over the next 10 to 30 years methane emissions from landfills in Germany will continue to represent a major source of greenhouse-gas emissions in the waste sector. Methane emissions have been estimated for the National Inventory Report on the German Greenhouse Gas Inventory (NIR) using the First Order Decay (FOD) method and the IPCC Guidelines for National Greenhouse Gas Inventories. Both national data and default factors (standard values) provided by the IPCC were used in calculating the estimations. A review of the methodological basis and input data used to determine methane formation in landfills indicates that the previous approaches adopted in the NIR reports led to methane formation rates that clearly exceeded actual emissions from German landfills. Both the state of knowledge and the results of investigations focussed on landfill gas production and emissions from numerous German municipal solid waste landfills confirm the latter. To obtain a more accurate description of the methane generation potential and methane generation in landfills, DOC parameters, fraction of degradable organic carbon (DOC_f), half-life and methane correction factor (MCF) can be adapted for the individual organic fractions of waste. When the adapted values for use in the estimation of methane emissions of German landfills are applied, the results yielded are in a range of approx. 50% compared to the estimated values reported in the German NIR to date.

1. INTRODUCTION

Since the early 1990s a series of legal provisions have been issued pertaining to the waste-management sector in Germany, and a number of relevant organisational measures have been implemented. These moves have had a strong impact on trends in emissions from waste-landfilling. Relevant developments have included an intensified collection of biodegradable waste from households and the commercial sector, an intensified collection of other recyclable materials, such as glass, paper/cardboard, metals and plastics; separate collection of packaging; and recycling of packaging. In addition, incineration of settlement waste has been expanded, and mechanical biological treatment (MBT) of residual waste has been introduced. As a result of these measures, the amounts of landfilled settlement waste decreased very sharply from 1990 to 2006, and have been stabilising at a low level since 2006 (Figure 1). As the figure shows, more than half the settlement waste produced in Germany today is collected separately and gleaned for recyclable materials (separate collection of recyclable materials and biodegradable waste) (UBA, 2017).

In 2004, approx. 330 settlement waste landfills were in operation in Germany and strict legal regulations were implemented whereby this type of landfill was required to be equipped for the collection and treatment of landfill gas. These regulations were fundamental in extensively reducing methane emissions from such facilities. In June 2005, in keeping with new, stricter requirements, more than half of all existing landfills were closed. As a result, as few as 150 settlement waste landfills remain in operation today. Pursuant to regulations in force since June 2005, the landfilling of biodegradable waste is no longer permitted, thus wastes with a potential for significant methane formation cannot be landfilled. To comply with the relevant requirements, settlement and other biodegradable wastes should undergo pretreatment by means of thermal or mechanical biological processes. From the year 2006, only a few waste components present in landfilled waste, with a minimal methane-formation potential (such as residues from treatment in MBT facilities) have contributed to the formation of landfill gas. As the formation of landfill gas in older landfills decreases, methane emissions from landfills will likewise decrease extensively, in the long term stabilising at very low levels (UBA, 2017).

By reducing landfill methane emissions from 1.4 million Mg CH_4 in 1990 to 0.4 million Mg in 2014, the waste-man-

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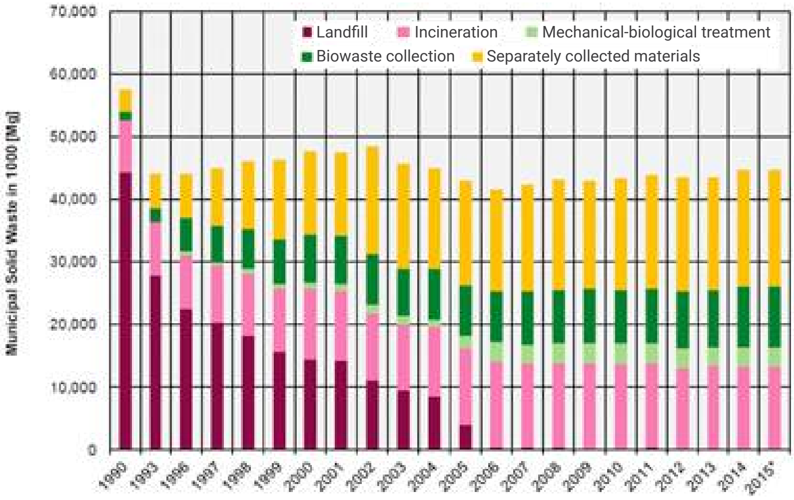

FIGURE 1: Changes in pathways for management of settlement waste from 1990 to 2015, with intermediate years (UBA, 2017).

agement sector in Germany has made an important contribution to climate protection (Figure 2). The lower methane emissions amount to a decrease of 24 million tonnes of CO_2 equivalents per year and, thus, to a 2.2% reduction of Germany's entire greenhouse-gas emissions. Experience gained by Germany's waste-management sector shows that a reduction in the quantities of biodegradable waste landfilled will provide a significantly higher contribution to climate protection than the collection and treatment of landfill gas.

Although the landfilling of biodegradable waste in Germany has been prohibited since June 2005, methane emissions from landfills will remain a major source of greenhouse-gas emissions in the waste sector for the foreseeable future.

Methane emissions from landfills were estimated for the National Inventory Report on the German Greenhouse Gas Inventory (NIR) using the First Order Decay method (FOD method) according to the IPCC Guidelines for National Greenhouse Gas Inventories. Calculations were based on data and factors assigned to the waste at the time of deposition, as well as by modelling biological degradation over the deposition period. Both national data and default factors (standard values) of the IPCC were used for the calculation.

To quantify methane emissions, the amount of methane extracted should be subtracted from the amount of methane generated. Based on the data collected by the German Federal Statistical Office on the amount of methane captured for the entire landfill status in 2014, the average collection rate corresponds to approx. 24%, very low in comparison to international rates (Figure 2) (Oonk, 2012, Krause et al., 2016). Consequently, questions were raised on the validation of the FOD method to produce more realistic estimates of the formed methane and its quantities. Therefore, the objective of a research project for the German Federal Environmental Agency was to investigate the estimation of landfill gas generation and emission (IFAS & RUK, 2017) and validate the same.

The-project comprises the following tasks:

- Verification of the estimation of landfill gas formation. For this purpose, the results of an initial expert report (RUK, 2014) are to be reviewed and updated;
- Verification of former MSW landfill sites to investigate gas quality and quantity, with sampling of solid waste and determination of the residual gas formation potential;
- Investigations on the gas formation potential of fresh waste at laboratory scale;
- According to the analytical results, potential adjustments and modifications of the estimated actual gas production rates using the FOD method will be determined and subsequently applied.

The current state of the report and the investigations are presented herein. The project has not yet been completed. The main aim of the research project and this paper is to develop a revised and improved set of values to be applied in the anaerobic degradation of relevant organic fractions such as food waste, garden waste, paper, wood, textiles or sludge, with particular regard to the following parameters:

- Degradable organic carbon in the landfilled waste (DOC);
- Fraction of degradable organic carbon which is anaerobically decomposed under landfill conditions (DOC_f) half-life ($t_{1/2}$)
- Methane correction factor (MCF)

2. MATERIAL AND METHODS - GAS PROGNOSIS, APPLICATION TO GERMAN LANDFILLS

2.1 Gas prognosis according to FOD method of the IPCC

According to the IPCC Guidelines for National Greenhouse Gas Inventories (IPCC, 2006), the following formula applies to the methane formation potential Lo of a landfilled waste:

$$Lo = W * DOC * DOC_f * MCF * F * 16/12 \quad (1)$$

Lo = Methane formation potential [Gg CH_4]
W = Mass of landfilled waste [Gg waste]

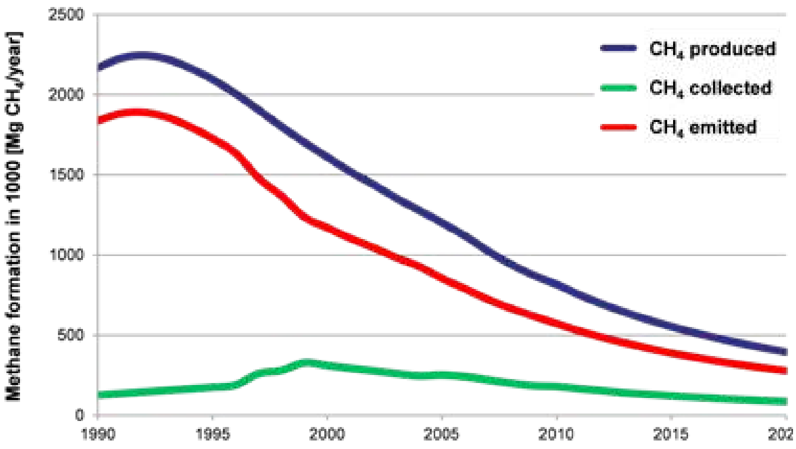

FIGURE 2: Impacts of waste management measures on the formation and emission of methane from landfills (Butz, 2014).

DOC = Fraction of degradable organic carbon in the landfilled waste in the year of landfilling [GgC/Gg waste]
DOC_f = Fraction of degradable organic carbon which is anaerobically decomposed under the conditions prevailing in the landfill [-]
MCF = Methane correction factor [-]
1-MCF = Fraction of degradable organic carbon which is aerobically degraded
in the year of landfilling [-]
F = Methane concentration in the formed landfill gas [-] (0.5)
16/12 = Molecular weight ratio CH_4/C [-]

The anaerobically degraded carbon quantity per year is estimated as follows:

$$DDOCm\ decomp_T = DDOCma_{T-1} * (1 - e^{-k}) \quad (2)$$

T = Year for which the calculation is performed
$DDOCma_{T-1}$ = Available DDOCm in the landfill body by the end of year T-1 [Gg]
DDOCm decompT = Anaerobically degraded DDOCm in year T [Gg]
k = Degradation constant [1/a]
= ln(2) / $t_{1/2}$
$t_{1/2}$ = Half-life [a]

The methane produced from the degraded organic carbon (DDOCm $decomp_T$), can be calculated as follows:

$$CH_4\ generated_T = DDOCm\ decomp_T * F * 16/12 \quad (3)$$

CH_4 $generated_T$ = Quantity of methane formed in year T [Gg methane]

By applying the FOD-model, the quantity of methane emitted during the considered year is calculated as follows:

$$CH_4\ emitted_T = (CH_4\ generated_T - RT) * (1 - OX_T) \quad (4)$$

CH_4 $emitted_T$ = Amount of methane emitted in year T [Gg methane]
R_T = Amount of collected and combusted methane [Gg methane]
OX_T = Fraction of methane oxidized in the landfill covering layer [-]

2.2 German NIR standard values and first preceding modification

In a previous expert report, standard values from the German NIR 2014 were used and compared with the operating results of landfill gas collected from 5 fully encapsulated landfills. On this basis, values differing from the standard NIR values were set, to ensure that the predicted time course of landfill gas production accurately reflected the time course of the landfill gas actually collected (RUK, 2014).

Table 1 shows the different values for the half-life and DOC_f. As a comparison, the standard values of the German NIR used to date are listed.

Reasons for the previous adaptation:

- DOC_f-values: The approach used in the IPCC model, according to which 50% of degradable organic waste is consistently degraded under landfill conditions, represents an unrealistic case. The readily degradable waste fraction is indeed substantially larger than that of scarcely degradable waste. In addition, readily degradable waste is often deposited with a high water content, thus rendering dehydration unlikely. Wood in bulk waste deposits is often deposited in a very dry condition, thus implying the risk that the wood may not come into contact with water at all. Therefore, biodegradable carbon DOC_f, and the fraction of paper and cardboard, were adjusted.
- Half-life: Experience with German landfills has shown how in phases with a high degradation potential, half-lives ranging from 4 (at the beginning of the phase) to 7.5 years can be assumed (Rettenberger, 2004). By contrast, when the IPCC model is applied, all five landfills evaluated yielded a half-life of around 7 years or longer within the first year.

3. EVALUATION OF GERMAN NIR VALUES AND RESULTS OF INVESTIGATIONS

3.1 Evaluation of DOC and DOC_f parameters for anaerobic degradation

The DOC and DOC_f values obtained are evaluated below.

TABLE 1: Comparison of the present German NIR standard values (UBA, 2017) and modified values from a previous preliminary expert report (RUK, 2014).

Waste fraction	German NIR Approach (UBA, 2017)		Previous Expert Report Approach (RUK, 2014)	
	half-life (years)	DOC_f	half-life (years)	DOC_f
Organic waste	4	0,5	3	0,8
Garden and Park waste	7	0,5	4	0,4
Paper and cardboard	12	0,5	7	0,4
Wood and straw	23	0,5	50	0,1
Textiles	12	0,5	7	0,4
Disposable nappies	12	0,5	7	0,4
Sewage sludge	4	0,5	3	0,8
Composites	12	0,5	7	0,4
MBT * - Waste	12	0,5	12	0,5

** Waste from mechanical biological treatment plants.*

The IPCC standard default values and the slightly modified current German NIR approaches were compared with reference values from the literature and results of scientific investigations, mainly by means of lab-tests undertaken to determine gas formation potential.

The overview in Table 2 indicates the following:

- Compared to a large body of data from the literature, values used in the NIR report are relatively high;
- In the fraction "Food waste", DOC contents and their bioavailability (DOC_f) are comparable;
- Garden waste yielded comparable results, although playing only a subordinate role in landfilling (self-composting, separate collection of green waste);
- Leaves (and woody shrub cuttings) as part of garden waste were characterised by a low bioavailability or anaerobic degradability (DOC_f), in the range of 25% of NIR values;
- Literature data refer to a DOC range between 0.28 - 0.34 for paper and cardboard fraction, compared to the NIR value of 0.4, also taking into account water content. Due to the high quantity of paper and cardboard deposited in German landfills, these parameters produce a significant influence on the estimation of methane emissions;
- Regarding the textile fraction, DOC contents are somewhat higher in the literature, although based on water contents of 15-25%. Data relating to anaerobic degradability are not available for this single fraction. It is, however, to be assumed that this would be significantly below 50% (DOC_f = 0.5), since a considerable carbon fraction is made up of plastic fibres;
- Digested sewage sludge, frequently disposed in the past, also displays a very low anaerobic degradability, in the range of 10-15% related to the NIR value of 0.5;
- The average DOC contents of three MSW sorting analyses in Bavaria and Austria from the period 1998-2003 are in a relatively narrow range of 128 to 132 kgC / t DM; thus, still below the average carbon content of the German nationwide domestic waste value from an analysis performed in 1985 (Barghoorn et al., 1986). Between the 1960s and up until termination of landfilling of unpretreated municipal waste in Austria and Germany, a gradual decrease of the carbon content of residual waste, derived from the various listed organic fractions but without plastic, has been determined;
- Application of the standard value of 0.5 (50%) for all fractions is therefore unrealistic. The fraction of readily degradable waste is substantially higher than the fraction of scarcely degradable waste.

3.2 Evaluation of the gas formation potential under anaerobic conditions

In order to further assess the plausibility of NIR values, the gas formation potential, which can be derived from these NIR values for individual organic fractions as well as for the average value of the total organics, should be considered in detail.

Taking into account DOC values in the NIR and a DOC_f of 0.5 (Table 2), an average gas formation potential of 231 m^3/ Mg (wet matter) is obtained, as an example, for the deposited organic fractions in the year 1993, when a high quantity of organic waste was deposited in German landfills (quantity see Figure 1, composition see UBA, 2017). This average gas formation potential may then be compared with results of investigations focussing on gas formation potential, largely from the 1990s (Table 3).

This comparison with literature data indicates that the landfill gas formation potential resulting from NIR default or input values tends to be too high, and the actual conditions of landfill gas production on German landfills are not adequately quantified:

- Thus, the majority of data relating to organic waste fractions (food waste) and garden and park waste are below the gas formation potentials derived from the NIR values. A few significantly higher values were determined by lab-tests. However, these higher gas formation potentials e.g. for grass clippings (Ramke, 2010) exert only a minor effect on the total gas formation in landfills, due to the paucity of landfilling of grass clippings in German landfills;
- The difference observed for paper and cardboard fraction, for which the gas formation potential is only

TABLE 2: Default values in the German NIR (UBA, 2017), reference results from literature and investigations (weight refers to wet mass).

Waste type	Approach German NIR	Approach German NIR	Reference results literature, investigations etc.		
	DOC (Mg C/Mg)	DOC_f (-)	DOC (Mg C/Mg)	DOC_f	Source
Food waste	0.18	0.5	0.09		Baumeler et al., 1998
			0.167	0.571	Ramke, 2010
			0.172		BLfU, 2003
			0.229		Nelles et al., 1998
Garden	0.20	0.5	0.218	0.43	Ramke, 2010
			0.230	0.123	Ramke, 2010
Paper and cardboard	0.40	0.5	0.283		Nelles et al., 1998
			0.296		BLfU, 2003
			0.297		Nelles et al., 1998
			0.30		Ramke, 2008
			0.343		Baumeler et al., 1998
Wood and straw	0.43	0.5	0.33	0.268	Ramke, 2010
			0.38	0.014	Ramke, 2010
			0.426		Baumeler et al., 1998
Textiles	0.24	0.5	0.275		Nelles et al., 1998
			0.275		BLfU, 2003
			0.413		Baumeler et al., 1998
Disposable nappies	0.24	0.5	0.167		BLfU, 2003
			0.195		Nelles et al., 1998
Sewage sludge	0.15	0.5	0.095	0.057	Ramke, 2010
Composites	0.10	0.5	0.22		Nelles et al., 1998
			0.229		BLfU, 2003
MBP-Residues	0.023	0.5			
Municipal solid waste			0.137		Baumeler et al., 1998
			0.128		Nelles et al., 1998
			0.130		BLfU, 2003
			0.20		Barghoorn et al., 1986

25-53% of NIR values, is particularly significant. The high assumptions in NIR are of considerable importance for the determination of methane emissions, since –in combination with the landfilled waste masses and selected half-life – these would theoretically dominate methane production in German landfills. Based on an average carbon content of 300 kgC / t (DOC = 0.3), the gas formation potential for paper fractions indicate an anaerobically degradable fraction of 28% (DOC_f = 0.28, range 0.17-0.36);

- The discrepancy displayed by the gas formation potential for wood and straw fraction is even more significant, particularly with regard to wood, for which gas formation potentials of 5-14% based on the NIR gas formation potential were determined. Straw with a higher gas formation potential represented only a negligible mass fraction compared to wood;
- This basic tendency was also confirmed by comparison of the average gas formation potential of all individual organic fractions (NIR value: 231 m^3/t) from previous investigations of municipal solid and domestic waste. The gas formation potential determined in numerous investigations, generally determined under more favourable milieu conditions than in a real landfill (e.g. with regard to water balance, lack of aerobic degradation, etc.), ranged from 30-81% (average value 60%) compared to the average gas formation potential according to NIR for the year 1993.

These results are fundamental in facilitating further adaptation of the input values for use in determining methane emissions from German landfills. Similar investigations should be performed in other countries.

3.3 Evaluation of aerobic carbon degradation in landfills

In addition to anaerobic processes, the biological degradation processes that occur in landfills are also characterized by aerobic degradation as follows:

- Immediately after deposition, aerobic degradation pro-

TABLE 3: Landfill gas potential of organic fractions and landfilled waste (average gas potential of total organics in the reference year 1993) and comparison with reference results from laboratory scale investigations (all results refer to wet mass).

Waste type	German NIR (UBA, 2017)	Gas potential reference results	
	(m^3/t)	(m^3/t)	Source
Food waste	168	110	Kruse, 1994
		126	Ramke, 2010
		76-168	Spendlin, 1991
Garden waste	187	95	Tallner, 1993
		105	Ramke, 2010
		74-120	Kruse, 1994
		128	Tallner, 1993 (different materials)
		293	Ramke, 2010 (different materials)
Paper	374	123-144	Kruse, 1994 (different materials)
		95-159	Kruse, 1994
		158-182	Kruse, 1994
		168-201	Tallner, 1993
Wood and straw	402	21	Ramke, 2010
		44	Tallner, 1993
		37.5-57	Kruse, 1994
		297	Ramke, 2010 (shredded straw)
Sewage sludge	47	30	Kruse, 1994
Composite materials	93		
Ø Gas potential total organics reference year 1993	231		
Municipal solid / domestic waste		70-126	Lechner, 2004
		137	Kruse, 1994
		85-140	Ehrig et al., 1995
		120-150	Stegmann, 1982, zit. in Rettenberger & Mezger, 1992
		105-165	Jessberger 1992
		172	Tallner, 1993
		186	Pfeffer, 1974, zit. in Rettenberger & Mezger, 1992

cesses are initiated ($MCF_{begin.\ of\ landfilling}$) by entrapped oxygen in the discharged waste and oxygen supply through the open landfill surface;

- Using data present in literature, a value ranging between 0.8 (thin layer compaction and slow build-up) and 0.95 (fast build-up) can be derived for the MCF respectively $MCF_{begin.\ of\ landfilling}$ (Weber, 1990). According to the IPCC guidelines, the MCF factor is actually based on landfill site management conditions. The IPCC factor of 1 ("Anaerobic managed solid waste disposal sites") does not reflect true conditions in former German landfills;
- When gas production decreases, i.e. in line with age of deposition, increased access of atmospheric air is enabled via the landfill surface mainly by means of wind, temperature and atmospheric pressure changes. This effect is generally greater in the presence of an active gas extraction system (air intake by oversuction effect) ($MCF_{long\text{-}term}$). Moreover, this effect depends on the quality of the surface cover. Many German landfills, and the majority of old deposits, only have a soil cover, at times ameliorated through use of a mineral clay liner. Only younger landfills currently in the closure and aftercare period are equipped with an impermeable surface geomembrane capable of reducing, although not completely preventing, air access.

The following should be taken into account for the derivation of $MCF_{long\text{-}term}$:

- Under strict anaerobic conditions landfill gas would consist almost exclusively of the main components methane and carbon dioxide at a ratio of about 65 to 35 Vol.-%;
- Available oxygen introduced through air access is converted into carbon dioxide. As a result, the methane-to-carbon-dioxide ratio decreases, with nitrogen present in the landfill gas / air mixture. Figure 3 shows this correlation/relationship, illustrating the change in gas composition due to increased aerobization until the oxygen is almost completely converted. Below the red line, carbon dioxide and methane are the result of anaerobic degradation, above the red line nitrogen and carbon dioxide (with its carbon content C_{aerob}) are due to

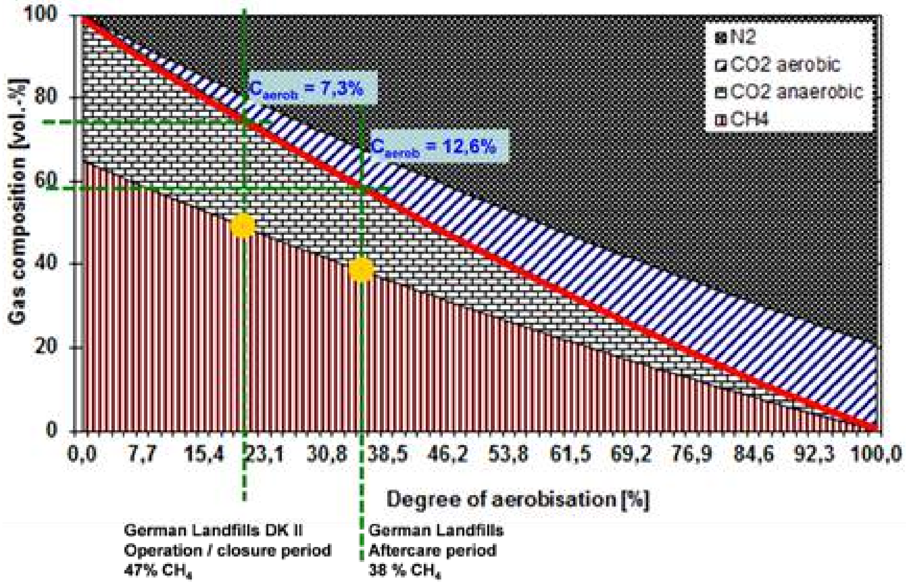

FIGURE 3: Methane content, aerobization degree and fraction of aerobic carbon conversion in German municipal waste landfills, reference year 2014 (DK II landfills correspond to MSW landfills).

air access and aerobic degradation;

- A survey carried out by the Federal Statistical Office for the year 2014 shows that the average methane content of all landfills surveyed during the operational and closure phase is 47 Vol.-%, and of landfills in the aftercare phase 38 Vol.-%. The proportion of aerobic degradation estimated for these methane concentrations in relation to total degradation of bioavailable carbon is shown in Figure 3 (7.3-12.6% of carbon conversion due to aerobically-produced carbon dioxide). For this estimation, it was assumed that oxygen consumption is dominated by aerobic decomposition, with carbon and oxygen reacting at a ratio of 1:1;
- Therefore, an actual value for $MCF_{long\text{-}term}$ $MCF_{tot.}$ of 0.93 for younger DK II landfills in the closure period and 0.87 for older landfills in the aftercare period can be deduced with a decreasing tendency (due to the increasing degree of aerobization).

4. ASSESSMENT BASED ON COLLECTED GAS VOLUMES IN LANDFILLS

4.1 Example of the gas content of an encapsulated landfill

The encapsulated German landfill “ER” displays the following conditions:

- Deposition period 1979-1991
- Area: 40 ha
- Deposition mass: 13 Mio. Mg
- Surface cover since 1995
- 95 vertical gas wells, 42 horizontal gas drainage pipes

Figure 4 shows the amount of carbon present in landfill gas extracted from this landfill compared to the amount of carbon calculated using the gas prognosis approach applied in the previous expert report (see Table 1). Using these data the degree of collection is calculated. Furthermore, Figure 4 shows the comparison of carbon degradation over time according to gas prognosis obtained according to the IPCC and German NIR approaches. In line with this evaluation it can be concluded that gas formation hypothesized using the IPCC/NIR approach, particularly from the paper and cardboard fraction, is however significantly over-estimated.

With regard to the “ER” encapsulated landfill, Figure 5 illustrates the percentage of aerobically degraded carbon estimated from methane concentrations in the extracted gas (see Figure 3). Although this landfill displays a high degree of encapsulation, the percentage of aerobic carbon conversion related to total (anaerobic and aerobic) carbon conversion increased from 2 to 14% within 10 years of application of surface capping.

4.2 Assessment based on the content of bioavailable carbon determined by solid waste sampling in landfills

Sixteen German landfills (“AI”-“OI”) with surface liners of varying gas and water permeability were drilled to obtain 449 solid waste samples. The age of deposition of solid waste samples ranged from 5 to 45 years and sampling depth was between 5 and 50 m. The samples were analysed to determine the fraction of currently bioavailable carbon and resulting methane gas formation potential L_0 with the following methods:

- Total organic carbon (TOC) according to DIN EN 13137;
- Respiration activity over 4 days according to the German Landfill Ordinance (DepV, 2009);
- Gas formation test over 21 days according to DIN 38414-8 (DepV, 2009).

Reliable estimations of current landfill gas production rates can be obtained using the results of waste sample analysis in combination with average values for the half-

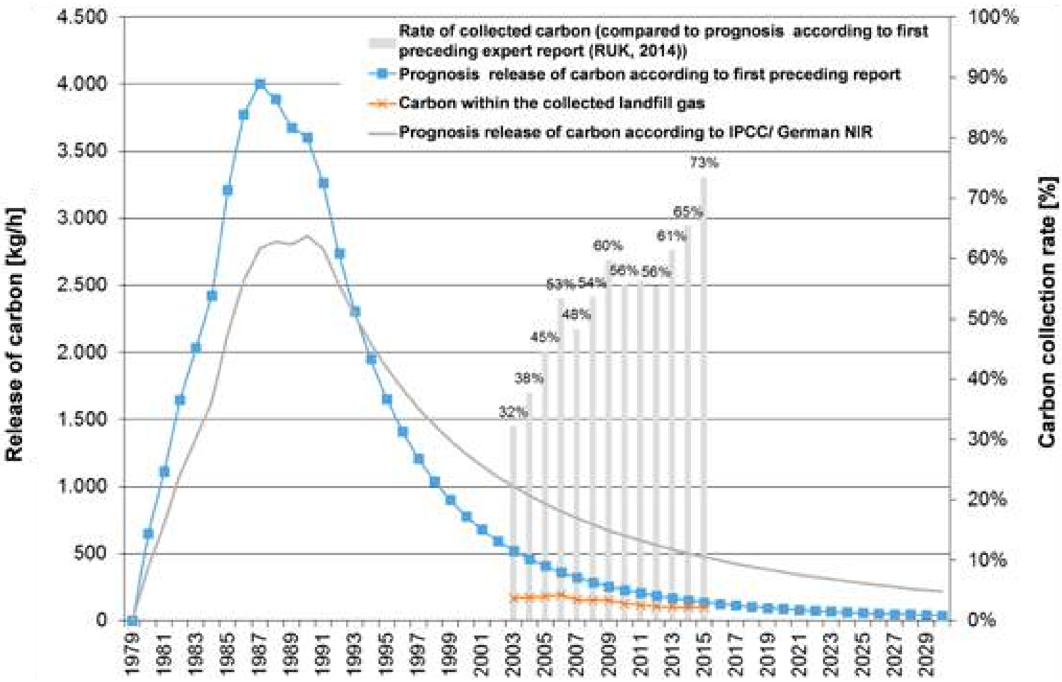

FIGURE 4: Calculated release of carbon and carbon collection rates from an "ER" encapsulated landfill (IFAS & RUK, 2017).

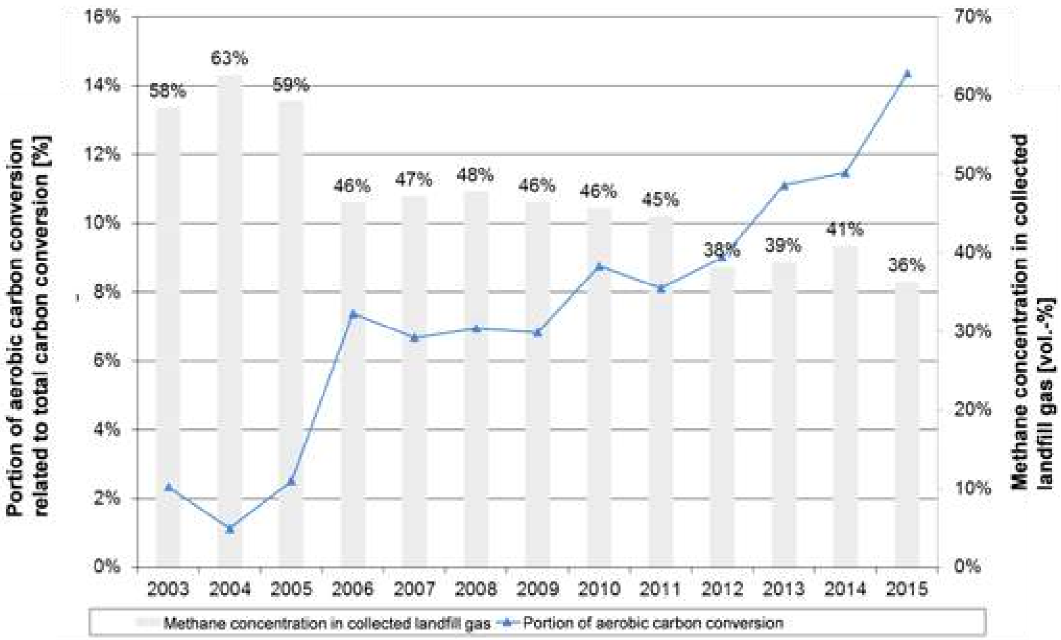

FIGURE 5: Fraction of aerobic carbon degradation in the landfill body due to introduction of atmospheric air into the "ER" encapsulated landfill (IFAS & RUK, 2017).

life. Current methane formation rate thus obtained is regarded as highly "realistic".

A comparison of methane production rates based on site-specific investigations with gas production prognosis estimated on the basis of IPCC or NIR assumption values (in particular DOC and DOC_f) confirmed that IPCC/NIR prognosis resulted in an ambiguously high estimation (approx. 2-fold higher) of methane formation rates.

At many sites, adaptation of DOC_f and half-life values (from the approach suggested in the previous expert report (Table 1) resulted in a more similar gas prognosis to that derived from solid waste investigations. As an example, this is evident in one of the 16 more closely investigated landfills:

- The landfill "LI" was filled with unpretreated municipal waste from 1971 to 2005;
- Landfill gas collection has been implemented since 1993 (Figure 6);
- By applying preliminary adjusted parameter values the

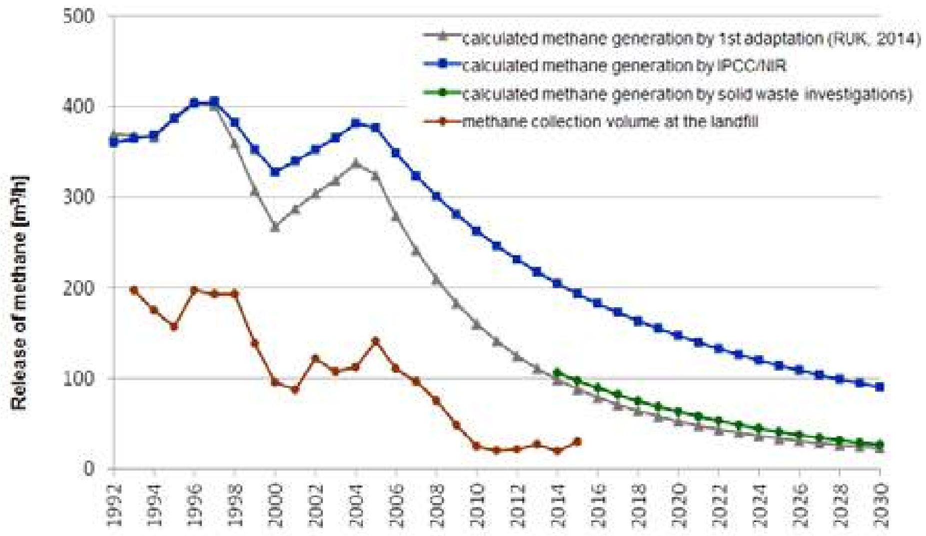

FIGURE 6: Methane production of an "LI" landfill since the year 1993; comparison with gas prog-nosis calculations using IPCC/NIR, with adapted values and results from landfill investigations.

methane production rate for the year 2014 was estimated as 99 m³ CH_4/h;

- Investigation of the solid waste samples obtained from drilling into the landfill in the year 2014 showed an average methane formation potential of 6.2 m³ CH_4/Mg DM. This resulted for the year 2014 in a methane production of 106 m³ CH_4/h at an average half-life value of approx. 8 years.

On using the above-described values for a preliminary adaptation of IPCC / NIR default values applied to predict methane production rates, and comparing the results obtained with methane production determined by solid waste sample investigations, remarkably similar curves were obtained. Moreover, average gas collection rates were in a similar order of magnitude.

4.3 Conclusions regarding the gas collection rate

A comparison of gas extraction rates yielded by almost fully gas-tight encapsulated landfills with landfills featuring a series of gas-permeable surface liners revealed an average difference in the degree of gas collection ranging from approximately 54% (encapsulated) to 44% (different gas permeabilities). This finding may indicate that landfills with a temporary – permeable – surface cover combined with a qualified gas extraction system release only marginally higher methane emissions compared to gas-tight encapsulated landfill sites.

In almost all landfills, the highest uncertainty regarding a gas prognosis is related to the lack of differentiated and reliable information on waste composition of the landfill body. This applies both to the quantity of household waste, commercial waste, sludge, construction and demolition waste, mineral waste, etc., as well as to the composition of the individual waste fractions. Results obtained in landfill investigations in Germany have demonstrated that the proportion, in particular of "paper, cardboard, cardboard packaging" and also "wood and straw" both quantitatively and in relation to methane formation potential is significantly lower than previously stated in the IPCC / German NIR standard values.

A further reduction of the calculated methane emissions, and consequent increase in the degree of gas collection, could be achieved if aerobic degradation processes were more closely monitored both at the beginning of landfilling and during landfill ageing. By including an average MCF value of 0.8 instead of 1 (as an example) as a simplified fixed factor to reflect the impact of aerobic degradation both at the start of deposition and in the long-term, the resulting increase in gas capture efficiency would be in the range of approximately 25%.

Table 4 provides an overview of the gas collection rates obtained using the different parameter sets, together with the results of solid waste sample investigations for the landfills AI - OI (different surface sealing and gas permeabilities as well as different gas extraction systems).

5. CONCLUSIONS, ADAPTATION OF GERMAN NIR DEFAULT VALUES

Conclusions of the evaluations and investigations performed:

- The review of the methodological basis for use in determining methane formation in landfills is beneficial, as the previous approaches and default values applied in the NIR (and IPCC) reports lead to methane formation rates, which do not reflect but clearly over-estimate the actual conditions of German landfills;
- Assessment of both the state of the knowledge and the literature, as well as the compilation of monitoring results and investigations of numerous waste samples from MSW landfills in Germany have provided confirmation of the above;
- A comparison of methane production rates based on site-specific investigations with gas production prognosis estimated on the basis of IPCC or NIR assumption values (in particular DOC and DOC_f) confirmed that

TABLE 4: Overview of gas collection rates obtained using different parameter sets and results of solid waste sample investigations for the landfills "AI" - "OI".

Landfill	Without adaptation DOC_f and half-life in FOD-Model of the IPCC		With first adaptation DOC_f and half-life in FOD-Model of the IPCC (RUK, 2014)		Methane formation potential L_0 from solid waste sampling, FOD-Model		Current gas collection volume
	Methane volume [m^3CH_4/h]	Collection rate [%]	Methane volume [m^3CH_4/h]	Collection rate [%]	Methane volume [m^3CH_4/h]	Collection rate [%]	Methane volume [m^3CH_4/h]
AI	186	16%	84	36%	47	64%	30
BI	185	19%	96	38%	75	48%	36
CI	131	23%	46	65%	69	43%	30
DI	145	17%	101	25%	57	44%	25
EI	131	10%	64	20%	50	26%	13
FI	202	27%	114	48%	83	66%	55
GI	200	14%	60	45%	86	31%	27
HI	669	57%	343	111% (1)	450	84%	380
II	196	36%	89	79%	112	63%	70
JI	255	16%	104	38%	94	43%	40
KI	208	14%	102	29%	95	32%	30
LI	205	15%	99	30%	106	28%	30
MI old section	1845	8%	625	24%	796	19%	148
MI new section	1703	30%	1063	48%	813	63%	511
NI	47	11%	27	19%	29	17%	5
OI	137	15%	65	31%	53	38%	20
Range		8 – 57%		19 – (111)%		19 – 84%	
Mean value with MCF = 1		21%		42%		44%	
Mean value with MCF = 0.8		26%		53%		55%	

(1) *The higher gas collection volume compared to the modified gas prognosis calculation is likely due to the uncertainty of waste composition of this landfill.*

IPCC/NIR prognosis resulted in an ambiguously high estimation (approx. 2-fold higher) of methane formation rates;

- Indeed, the estimation of current and prediction of future methane emissions for a series of closely monitored landfill sites using the modified approaches for methane formation potential and its kinetics (first preceding approach, RUK 2014) have already proven to be a good fit with the findings of biotests on solid waste samples taken from landfills;
- For a more realistic prognosis and calculation of the methane formation potential and methane formation in landfills the following parameters may be adapted for the individual organic waste fractions:
  - DOC
  - DOC_f
  - Half-life
  - MCF

The adaptations discussed for each of the relevant parameters are summarized in Table 5. In particular, a standard DOC_f value of 0.5 (50%) for all fractions is not realistic. The fraction of readily degradable waste is substantially higher than the fraction of scarcely degradable

TABLE 5: Comparison of approaches applied in the German National Inventory Report (NIR) and suggestions put forward by the authors for modified parameters to quantify methane formation.

Waste type	Values in the German NIR (UBA, 2017)			Proposed values for adaptation		
	Half-life (years)	DOC_f	DOC	Half-life (years)	DOC_f	DOC
Food waste	4	0.5	0.18	4	0.5	0.18
Garden	7	0.5	0.2	7	0.5	0.2
Paper	12	0.5	0.4	7 (rather 4)	0.5	0.4
Wood and straw	23	0.5	0.43	50	0.1	0.43
Textiles	12	0.5	0.24	10	0.4	0.24
Disposable nappies	12	0.5	0.24	10	0.4	0.24
Sewage sludge	4	0.5	0.05	4	0.5	0.15
Composite materials	12	0.5	0.1	12	0.4	0.1
MBP-Residues	12	0.5	0.023	12 (rather 4)	0.5	0.023

waste. Moreover, a simplified fixed MCF value of 0.8-0.9 instead of 1 should be considered in order to reflect the impact of aerobic degradation at the start of deposition and in the long-term.

When the adapted values are used to estimate methane emissions from German landfills, they yield a result in a range of only 50% compared to estimations obtained with the German NIR values applied to date.

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SWITCH TO SAVE? COMPARING MUNICIPAL SOLID WASTE EXPENDITURES BASED ON WASTE MANAGEMENT PROVIDER OWNERSHIP

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ABSTRACT

The debate about public vs. private provision of municipal solid waste management has been going on for several decades with no conclusive evidence in favor of either. The presence of relevant competition in the market seems to be more important than the type of the provider. In this study, we expand on this topic and use empirical evidence to show that what matters most is the willingness of the municipality to switch waste management providers. We compare the municipal solid waste expenditures of more than 60 municipalities in the Czech Republic that changed their waste management provider in 2008-2014, both before and after the change. The results show that such a change can, on average, reduce the expenditure by several percent, and change should therefore be preferred by the municipalities instead of perpetually extending contracts with the current provider. In addition, we show that it does not matter much whether the new waste management provider is a public or private company, as costs are reduced when switching either way.

1. INTRODUCTION

The issue of public vs. private provision of public services has a long history in the scientific literature. Public services usually originate in connection with two factors: the general public interest in provision of such services and the failure of the private sector in providing such services, typically due to the very high initial costs and the issue of securing sufficient revenue.

Waste management (WM) is one such service. While there is a general consensus regarding the public interest in this service, from the perspective of a private provider there is also the issue of how to persuade people to properly finance it. The common solution is that the municipality, as a public entity, is empowered with enough rights to make people pay for such a service, and then uses the collected revenues to finance it. It is then up to the municipality to delegate the service provision (Kinnaman and Fullerton, 1999), either using its own capacities or contracting out the service.

While the issue of how to raise funds for running the municipal solid waste management (MSWM) might be a simple administrative task (taxes or fees), the question of how to use these funds to secure appropriate services is much more complex.

It is important to remember the difference between the private sector and the public sector. In the private sector, the obvious goal is to make a profit, and the ability to make a profit generally results in the survival of the better service providers over those that are not able to keep up and are subsequently squeezed out by the competition. The trend towards increasing efficiency in service provision is thus secured. However, in the public sector (where waste management falls) the primary goal is the welfare of the people and not profit, although generating at least some profit is still welcome. Thus any measures that would result in improving the provided service, decreasing the related costs, and ideally the combination of both is desirable. Any evidence providing suggestions for selecting a WM provider for the municipality can therefore be useful.

One simple way to divide WM providers in municipalities is into public (contracting in) or private (contracting out) companies. The literature on the subject of public vs. private provision of WM has been examining the issue of company ownership since the 1980s when Domberger et al. (1986) examined municipalities in England but did not find any notable differences between contracting in and contracting out WM service. The only relevant factor in terms of costs was, according to the study, the existence of competitive contracting.

Since then, many studies have examined whether there

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is any significant difference between public and private provision of WM, occasionally slightly favoring one type or the other. Bel and Warner (2008) published a review in which they examined studies dealing with the effect of privatization on cost reduction in WM. Generally, they found little support for a link between privatization and cost savings, as the observed savings are not systematic. The issue identified in the review is that in the research there is rather too much emphasis on the ownership instead of on other aspects that are more important in quasi-markets such as WM with limited numbers of alternative suppliers. Cost savings are simply not systematically found when looking at the issue in terms of the WM provider organizational type. Bel and Mur (2009), Bae (2010), and Jacobsen et al. (2013) provide comparable conclusions that there is no clear evidence in favor of either one in terms of costs and the results are typically mixed (Bel et al. 2010, Simões et al. 2012).

On the other hand, the presence of competition has been identified as important (Szymanski, 1996, Gomez-Lobo and Szymanski, 2001, Bel and Warner, 2008, Jacobsen et al. 2013). Once public providers are forced to compete with private companies, they are likely to achieve comparable results (Kinnaman and Fullerton, 1999).

Bel and Warner (2008) therefore stress that instead of emphasizing the public versus private debate, primary attention should be given to the market structure and whether there is sufficient competition. In the absence of competition, savings are less likely to occur, regardless of the WM provider ownership.

Waste management in Czech municipalities is strongly affected by their size structure. It is not uncommon for a municipality to have a population of less than 1000, or even less than 500. Such small municipalities have to rely on contracting out their WM, as it does not make economic sense to have an in-house WM company. But even though the waste sector is becoming more economically attractive, many of these municipalities struggle with increasing costs, as they often have historically relied on only one provider and are reluctant to change. In many cases, they simply perpetually extend the contract with their provider, accepting regular cost increases. Due to the limited administrative capacities in the smallest municipalities, this is often the most convenient solution, although likely also the most expensive.

The efficiency of WM as a public service was examined in more detail in the Czech conditions by Ochrana et al. (2007), who focused on the role of the WM company organizational form in the overall efficiency of the service. The study analyzed the preferred form of service provision, the important criteria when selecting a WM provider, and the reasons leading to municipalities changing WM providers. The authors collected more than 900 survey replies from Czech municipalities and analyzed the answers together with the available data about related municipal expenditure. The outcome of the study is that in-house production of services appears to be the most efficient, but this is because certain related costs of service production are often not directly assigned to the production of these services by the municipalities, and therefore the reported expenditures are lower. Using only expenditures directly reported by the municipalities on these services thus yields inaccurate results, when comparing with the external provision of these services. The least efficient, on the other hand, are municipal companies that were arbitrarily selected without any competition. The overall conclusion of the study is that as long as there is competition, the form of service provider ownership actually might not matter at all.

In order to extend the current research in this area, the research question of our study concerned how switching WM providers affects municipal solid waste expenditure (MSWE) and whether the change in WM provider ownership has any role. Unlike other studies, which usually compare the differences between public and private providers in a selected time period, we use empirical evidence to examine the difference in costs before and after changing WM providers.

2. MATERIALS AND METHODS

2.1 Data

In this part, we describe the data used in this study and how we obtained them.

There were two primary data sources. Financial data were acquired from a web portal run by the Czech Ministry of Finance called MONITOR. This portal provides information about the budgets of all municipalities in the Czech Republic and presents complex aggregated data about the financial situation of individual Czech municipalities, freely available to the public. Complete detailed data for individual fiscal years can be downloaded for further analysis.

We were specifically interested in the current expenditures of municipalities on MSWM that represent the day-to-day expenditures of municipalities on MSWM provision. We do not use capital expenditures, as they include primarily occasional investment costs that happen usually only once in a few years, making it problematic to compare among the municipalities, especially those of varying sizes. On the other hand, current expenditures calculated per capita (using municipality population data available through the Czech Statistics Office) generally provide a good basis for comparing expenditures among the municipalities, as they cover approximately the same things in both smaller and larger municipalities.

However, it should be noted that the financial data provided by MONITOR are not always 100% correct. We collected municipal financial data for several consecutive years, making it possible to see developments over time and to check whether there are any issues with the data, suggested for instance by very high variances between individual years. Such issues are usually the result of a municipality reporting its financial data incorrectly. Typical examples include reporting both current and capital expenditures as current, or failing to differentiate between expenditures from certain subgroups and reporting only aggregated expenditures under the most common category for such groups.

The reasons for such mistakes are mostly municipal staff with insufficient knowledge of how to report municipal expenditures or insufficient time for detailed expenditure reporting. As the majority of municipalities in the Czech Republic are very small with populations of only a few hundred, often there is simply an insufficient administrative capacity for certain tasks.

The second data source was interviews with the local authorities from a sample of municipalities. As we are examining the effect of changing/switching WM providers, we focus only on municipalities where such a change occured. Unlike with the financial data, there is no centralized source where municipalities report how they secure their WM. We contacted over 500 municipalities in the Czech Republic, of which 70 reported a change of WM provider in the last several years. However, due to very large interannual differences caused by combining the expenditures related to building a civic amenity site with the current expenditures, we dropped four municipalities, resulting in a final sample of 66 municipalities.

Most of these municipalities use an external WM company. This makes sense, as due to their relatively small sizes, it is not economical to have their own municipal waste company. Therefore they contract a private, public, or mixed WM company. We now define "municipal", "public", and "mixed" WM ownership types as they are used throughout this study; "private" ownership is self-explanatory.

A "municipal" WM company is usually historically created by a larger municipality for which it provides MSWM; sometimes, it also provides this service for a few neighbouring municipalities. According to the interviews, such a company is usually less focused on profit and is often part of a larger municipal company generally dealing with various technical municipal services. Providing MSWM for additional municipalities serves as a way to better utilize the available infrastructure with a greater focus on profit.

In this study, a "public" WM company is one that is owned by an association of municipalities in which individual municipalities act as the shareholders based on their size and respective financial investments. Each municipality pays this company for the MSWM provision, and it also participates in the profits of the company. However, during our interviews we noted occasional disillusionment with involvement with such companies, as small municipalities have very little say compared to the few larger municipalities.

A "mixed" WM company is usually the result of the previous decision of a larger municipality to partially outsource WM provision, maintaining some participation in the decision making and profit while having an economically strong partner. In such cases, the private part of the mix is often represented by a newly created company owned by an already established player in the waste market. If relevant, this company also provides MSWM for surrounding smaller municipalities, just as with a municipal company.

Based on telephone interviews with responsible local authorities, or alternatively with local authorities with sufficient knowledge of the topic, we matched each municipality with a WM company, a time horizon when this company provided MSWM in the given municipality, and the ownership type of the company.

We then created a dataset for several consecutive years with information about municipalities and their WM companies, ownership type of the WM companies, related municipal expenditures, and any change in the position of the WM company that occurred in the examined time period.

The following three tables include some basic characteristics of the sample used in this study.

2.2 Methods

In order to be able to analyze the effect of switching WM companies, we had to adjust the data, as these were collected for a broader time horizon and thus difficult to compare directly. We adjusted the dataset in order to have data in a format reflecting municipal expenditures in the

TABLE 1: Description of the sample (with respect to the year of the provider change).

66 municipalities	Bottom value	Median	Top value	Average
Population	76	930	9 555	1 391
MSWE per capita	291 CZK	518 CZK	926 CZK	539 CZK

Source: Czech Statistical Office, Czech Ministry of Finance

TABLE 2: Year of the waste management company change, 66 municipalities.

Year	2008	2009	2010	2011	2012	2013	2014
No. of changes	2	8	5	6	14	19	12
% of the sample	3	12	8	9	21	29	18

Source: own data

TABLE 3: Waste management company ownership before/after the change, 66 municipalities.

Ownership	Private	Public	Municipal	Mixed
Before the change	50	8	6	2
% of the sample	76	12	9	3
After the change	31	29	4	2
% of the sample	47	44	6	3

Source: own data

year before changing WM provider (year -1), in the year when the WM provider was changed (year 0), in the subsequent year (year +1), etc. After this adjustment, we aligned the individual municipal data in order to have matching periods. Doing this means we do not need to consider in which absolute year the WM provider changed, as we have a relative timeline, which is more useful for our purposes. Instead of 66 changes occurring over a seven-year horizon, we now have a dataset with the WM provider change occurring in the same relative period.

With the municipal data about the WM provider change aligned to the same relative (year 0) period, we calculated the relative differences in municipal expenditure per capita from the period before the change of the WM provider in terms of year +1 and year +2.

Calculating these differences allows us to directly see how MSWE changed once the municipality switched to a different WM provider.

In addition, our sample of municipalities was divided into groups based on the change of the WM provider ownership type. We differentiate four types of WM provider ownership. After the data collection, we concluded that there are five common situations with WM provider changes in terms of ownership; these are discussed later in the study. After making this distinction, each situation can be analyzed separately and compared.

3. RESULTS AND DISCUSSION

Figure 1 shows the differences in aggregated data from municipalities in the years before and after the municipality changed its WM provider. Provided data are calculated as the MSWE per capita.

The data show that once the WM provider change occurred, the average per capita expenditure decreased in the following year on average by 6% (the median decrease was 4%). While this might not seem that significant, even such a small change can make a difference in terms of municipal finance where budgets are often very limited. If we consider such a savings for a period of several years, a municipality can save enough to make a larger investment that could further improve its WM or can alternatively tackle some other important issue in the municipality. Moreover, in municipalities that perpetually extend their contract with the WM company, it is common that MSWE increases each year by a few percent. Reductions in MSWE instead of standard annual increases thus represent even greater savings

Figure 2 shows the relative interannual changes in MSWE of individual municipalities. In this figure, we see that switching WM providers does not always lead to decreased MSWE. There may be several reasons for this. First, the new WM provider might provide a broader range of waste services, which logically results in higher costs. For instance, the collection frequency might be increased, additional waste fractions might be separately collected, etc.

Second, although being more expensive than before, the new provider might still be cheaper compared to the situation with the previous WM company. Jump increases in costs requested by the original WM providers were mentioned by several local authorities as the decisive factor in switching to a different WM provider.

Third, there might be some additional costs included in the reported MSWE by the municipality that coincidentally occurred in the same year as the WM provider change. For instance, many municipalities begun to separately collect biowaste during this period, which required purchasing composters or additional bins for biodegradable waste. Even though such purchases occur irregularly, technically they can count as current expenditure, leading to the increased reported MSWE in a given year and might result in overall increase of MSWE by several per cent.

Nevertheless, Figure 2 shows that the majority of municipalities experienced a decrease in MSWE after they switched their WM provider. Almost 30% report a decrease in MSWE by up to 10%, while an additional almost 30% report even higher MSWE reduction, with a few municipalities saving more than 40%. Few municipalities reported an increase in MSWE by over 30%, but based on our experience such an increase is very probable due to the reasons men-

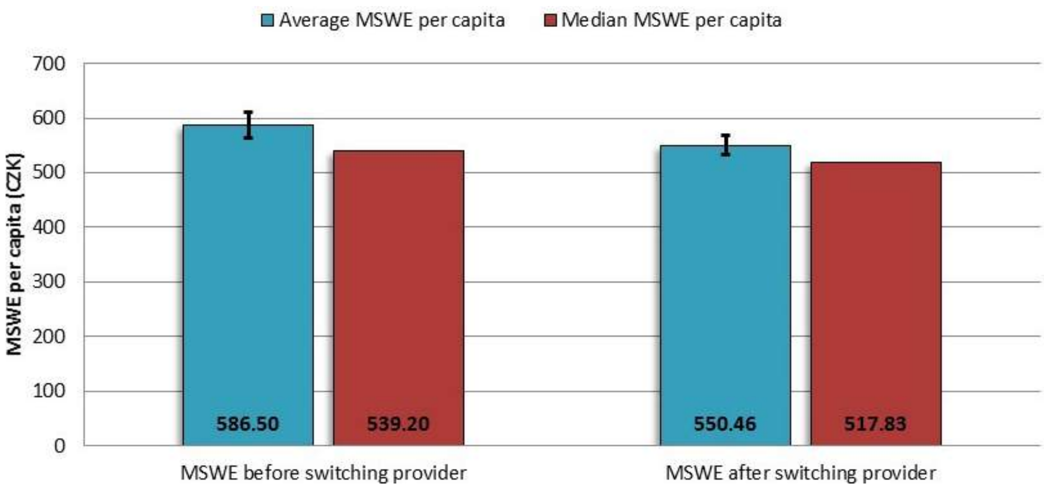

FIGURE 1: Interannual changes in MSWE after switching waste management provider (66 municipalities), standard errors for averages included - *Source: own construction*

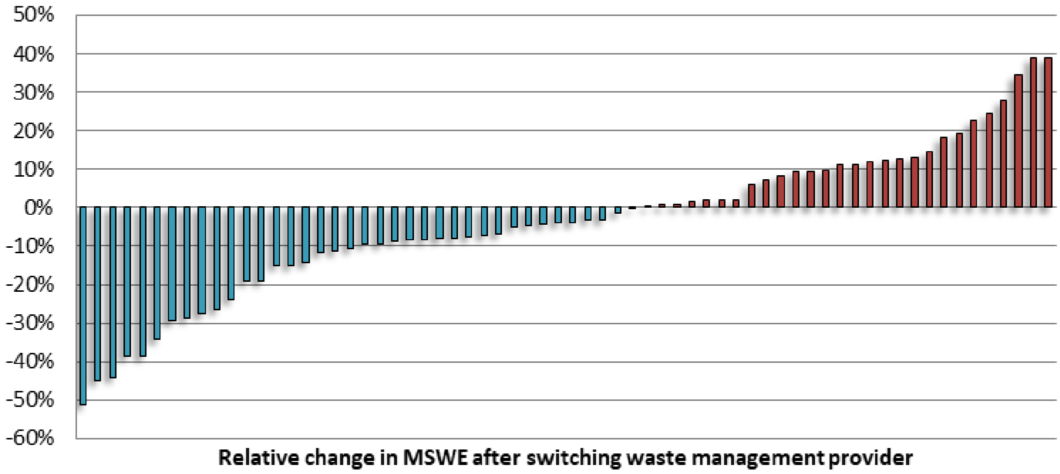

FIGURE 2: Relative changes in MSWE of individual municipalities after switching waste management providers (66 municipalities) - *Source: own construction*

tioned above. Overall, half of the municipalities from the sample did not experience a difference of more than ±10% in their MSWE.

Szymanski and Wilkins (1993) and Gomez-Lobo and Szymanski (2001) mention that while there might be great cost savings in the initial year, this advantage tends to diminish quickly in the following years, resulting again in higher costs and probably in another public tender. In part of our sample, we were able to obtain MSWE data from the second year after the change in WM provider; this is presented in Figure 3.

Figure 3 shows that after two years, the savings still exist (on average 3.5% lower MSWE than in the period before the change of the WM provider), but are beginning to diminish, which is in accordance with the mentioned literature. Competitive tendering every few years might have the potential to keep the MSWM costs down. On the other hand, each individual municipality has to decide how often it should opt for the next tendering, as such action brings additional costs to the municipality.

Jacobsen et al. (2013) suggest a biannual tendering system in order to find the provider with the best offer. One municipality in our sample utilized biannual tendering through electronic bidding applications and was able to get a much better contract than before, although this was largely due to the rather poor starting condition of WM in this particular municipality.

The Czech Republic has a very fragmented municipal structure and, in many cases, the fixed costs associated with WM provider tendering might represent a significant part of the total annual MSWE and might even exceed the potential savings. In such cases, it is actually more economical to stay with the current, albeit probably more expensive, WM provider than to look for a possibly cheaper one, and thus the WM provider change is likely to occur less frequently. The general suggestion here would still be to actively pursue public tendering, although somewhat less frequently.

A different perspective on this issue comes from the WM providers themselves. From their position, frequent changes are typically far from desirable. An ideal situation for a WM provider would probably be to have a secured

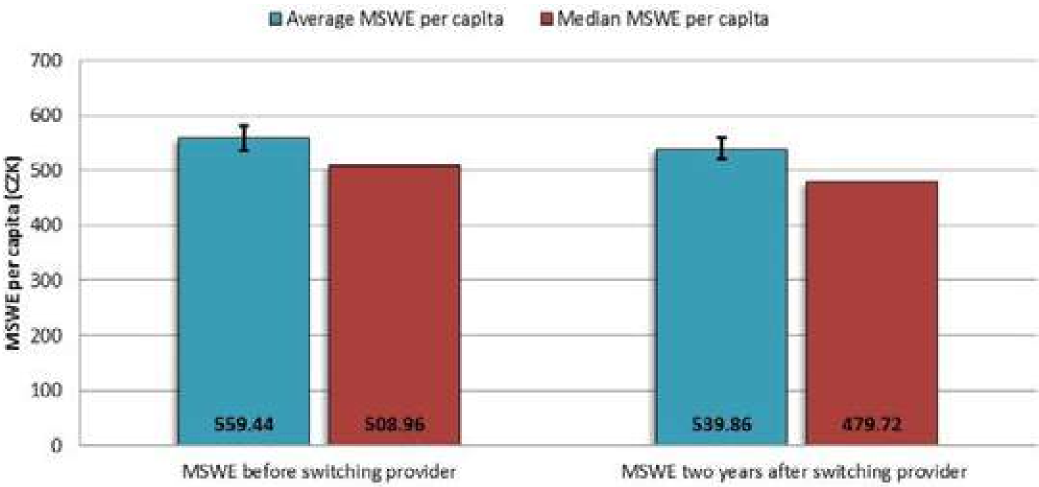

FIGURE 3: Comparison of MSWE before and two years after switching waste management provider (52 municipalities), standard errors for averages included - *Source: own construction*

contract for an infinite period with gradual increases in prices over the time. In such a situation, they could plan far into the future and not need to worry about the competition. This is the classical market situation, in which the interests of the customer (municipality) compete with the interests of the supplier (WM provider). The customer wants as much as possible while paying as little as possible, while the supplier wants the opposite. If these two sides are able to find an intersection, a deal occurs. From the perspective of the municipality as the customer, it is important to have a sufficient choice of WM providers, so that the municipality does not have to compromise that much in terms of the quality of the contracted service and the associated price. But of course, if the available offer is not good enough, the municipality always has the option of providing WM services itself, and sometimes this actually can be the best available option.

Figure 4 shows the changes in MSWE differentiated by the type of the WM provider ownership before and after the municipality changed WM providers. We can see that savings are possible in any kind of scenario, independent of the original type of WM provider ownership.

Slight savings are reported whether a municipality switches from a private WM provider to a public one (in our conditions, owned by an association of municipalities) or vice versa. Larger savings seem to be possible when switching between private waste companies and municipal waste companies, but again, the data suggest that this goes both ways. We therefore cannot draw a clear conclusion in terms of savings of whether it is better to choose a private or public waste company. The observation that there is rather little difference in waste-related costs between public and private providers is in accordance with many other authors (Domberger et al., 1986, Szymanski, 1996, Bel and Fageda, 2010), as well as with the observation that the existence of competition is much more important than the type of provider (Gomez-Lobo and Szymanski, 2001, Dijkgraaf and Gradus, 2007, Bel and Warner, 2008).

However, the highest amount of cases where MSWE increased were in individual municipalities changing from a private to a public company. This somehow contradicts Bel and Costas (2006), who suggest that intermunicipal cooperation (which is, in our case, represented by a waste company owned by an association of municipalities) might be a good alternative for small municipalities with limited potential external WM providers.

The observed MSWE increase in multiple cases when switching to a public provider might partially explain the disillusionment that some local authorities expressed in interviews after becoming a member of an association of municipalities in order to utilize the MSWM services of the related public waste company. Accordingly, it might be wise for a municipality considering a switch to a public WM company to examine whether the potential savings are truly there in comparison with the other options.

According to Massarutto (2007), even better results can be achieved when competitive tendering is used for separating more specific activities along the value chain. However, based on our experience with local authorities, such separate competions for specific activities in MSWM are very scarce. In our opinion, the problem might also be the small average municipality size in the Czech Republic: it does not make much economic sense for the waste companies to compete for only specific activities in such small municipalities, and thus the separation of MSWM into distinct activities becomes relevant only in larger municipalities. But the results in those few municipalities where separate tenders happen so far seem promising. In combination with the stronger preference for short-term contracts suggested by Simões et al. (2012) this might become a good strategy for municipalities to cut down MSWE and keep them low.

4. CONCLUSIONS

As in several previous studies, we examined the differences between public and private provision of municipal solid waste management. However, in contrast to previous studies, we did not focus on the cost difference between various types of waste management provider ownership in a single selected time period, but instead on the changes of waste management costs over time, once the municipality switched to a different waste management provider. This approach does not provide a static perspective on the

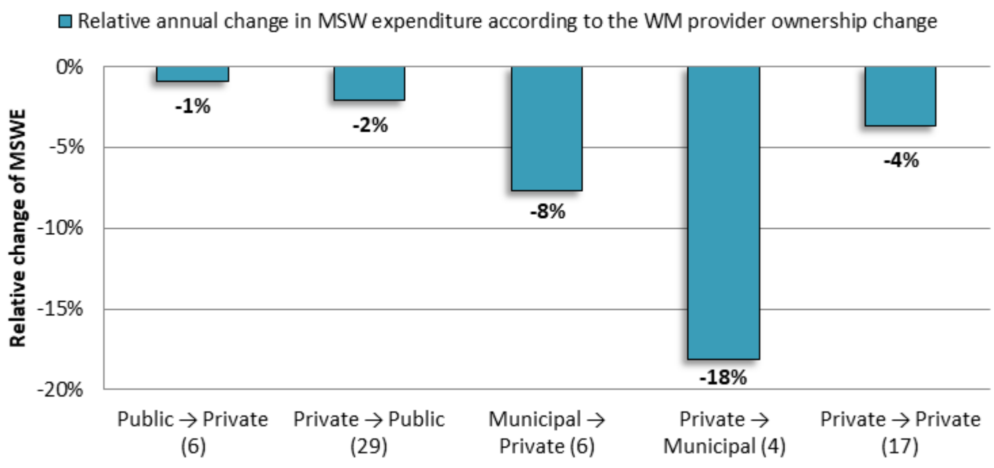

FIGURE 4: Relative changes in MSWE after switching WM provider (62 municipalities) - *Source: own construction*

matter, but in our opinion it actually provides a more important dynamic perspective using relative changes, as it overcomes the issue of various initial starting points of the municipalities before switching their providers.

Our results show that a municipality is likely to benefit from changing its waste management provider. The majority of the municipalities in our sample experienced a decrease in their waste management expenditure once they switched providers, on average by 6% in the first year after the change and in some cases by more than 20 to 30%. The comparatively lower waste expenditure level achieved under the new provider seems to hold even for the following year, although savings tend to slowly diminish, as has been suggested by other authors dealing with this issue. Based on these observations, municipalities should consider actively pursuing regular competitive tendering every couple of years depending on the actual service and the market availability. An active approach in this field seems to pay off relatively well considering the nature of municipal finances.

On the other hand, waste management costs increased in some municipalities, but our evidence indicates that this was caused typically by other factors, such as the extension of activities included in the service provided by the new company. In addition, even though in some cases the waste management costs increased with the new provider, this increase was actually likely lower than the costs would have been with the previous waste management provider, so this can still be considered as an improvement or as an actual savings.

Finally, we examined the differences in waste management expenditures when switching from public to private waste management provider and vice versa. We did not find any significant patterns. It seems that, in accordance with other authors, it actually does not matter much whether the waste provider is public or private, but whether the municipality is willing to regularly engage in competitive tendering for such services. By doing this, the municipality seems to be most likely to get the best available services at reasonable costs. The ownership of the potential service provider does not seem to play an important role, as long as these providers have competition. Sufficient competition ensures that the efficient providers will survive and be able to offer their services to the municipalities. Municipalities therefore should not be biased towards any potential service provider based on its ownership and should approach the question of what provider to choose in a pragmatic way. In this way, municipalities should be able to secure the best combination of quality, scope, and price of the provided services.

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TACKLING INTERNATIONAL AIRLINE CATERING WASTE MANAGEMENT: LIFE ZERO CABIN WASTE PROJECT. STATE OF THE ART AND FIRST STEPS

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ABSTRACT

Concerns about cabin waste date back more than two decades. However, in the past few years, several airlines and stakeholders (notably catering companies) have increased their efforts to tackle this issue. A number of factors such as low landfill disposal rates, particularly for inorganic fractions, lack of appropriate facilities and restrictive regulations had traditionally discouraged airlines and other actors to proactively look for solutions. LIFE+ Zero Cabin Waste aims to create a sustainable model to reduce, re-use and recycle (including energy recovery) waste recollected in Iberia airplane cabins in Madrid-Barajas Airport (Spain) and set the basis for its replication by other airlines and related stakeholders in the future. The objectives of this project are: properly separate the cabin waste on board, demonstrate that waste can be dealt with in a more environmentally friendly way contributing to the reduction of the high carbon footprint associated to the generation and the current inadequate management of cabin waste and set the basis for replication through standard protocols. The project, although it is in the first stage, has already been laid the bases and recommendations for future implementations and improvements that will allow reaching the proposed objectives.

1. INTRODUCTION

Waste produced in aircrafts is far from minor. According to Godson (2014), passengers worldwide produce an average of 1.43 kg of waste per trip. On the basis of the above data and the latest report of the Airports Council International (ACI), which states that there were about 7.7 billion plane passengers worldwide in 2016 (ACI, 2017), we can estimate a production of about 11 billion kg of waste produced by aircraft passengers per year.

Concerns about cabin waste date back more than two decades where characterizations of this waste stream started to be analyzed so as to highlight the hot spots and develop recycling strategies (Li et al. 2003). Despite this early concern, until now most airlines and catering companies have been recycling very little and the waste obtained is typically of low quality due to the mix of multiple waste fractions. A number of factors such as low landfill disposal rates (particularly for inorganic fractions), lack of appropriate facilities and restrictive regulations had traditionally discouraged airlines and other actors to proactively look for solutions.

However, in the last years, a change of trend can be observed. After thorough research made by the authors of this paper, it can be stated that several airlines and stakeholders (notably catering companies) have increased their efforts to tackle this issue. This is the case of Ryanair, for instance, that have promised to eliminate non-recyclable plastics from its operations by 2023. In addition to switching to biodegradable cups, wooden cutlery and paper packaging onboard, Ryanair said it would make its head offices, bases and operations plastic free (Topham, 2018). British Airways expect to decrease the amount of waste that goes to landfill and recycle 50% of waste by 2020 (British Airways, 2018). Other companies such as Alaska Airways are committed to reducing the waste from all paper, cups, bottles and cans on every domestic flight they operate (Alaska

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Airways, 2015). At this point is worth to mention that all these efforts made by aircraft operators are usually single initiatives, lacking a comprehensive and holistic approach. Nevertheless, an increasing public environmental consciousness that scrutinizes companies' behaviors as well as the progressive price increase in disposal rates are triggering more responsible solutions to this problem.

The management of catering waste is regulated both by the Waste Directive (Official Journal of the European Union, 2008) and the Packaging and Packaging Waste Directive (EC Packaging Waste Directive, 1994) since waste from meals and the packaging of those meals is produced due to the catering service and treated jointly. These two directives follow the inverted waste hierarchy pyramid).

1.1 Classification of Cabin Waste

When discussing cabin waste, it is necessary to make a preliminary clarification and distinguish between two different types of waste categories depending on its origin, namely category 1 (Cat1) and 3 (Cat3), even if, technically, both categories belong to the management of animal by-products, the so-called SANDACH waste (animal by-products not intended for human consumption) (European Parliament, 2009).

International catering waste (ICW) is not considered risky waste when the planes are traveling in EU territory only, and it is classified as Cat3. However, in flights from countries not included in EU territory, ICW is considered as animal by-product and, therefore, included in high-risk classified as Cat1. It is assumed that a potential risk of the spread of animal diseases exists, being dangerous to animal and human health if not properly disposed of. The European Parliament regulates the way in which ICW can be disposed of. Waste classified as Cat1 must be disposed of by burial in an authorized landfill according to the EU 1069/2009 Regulation (European Parliament, 2009).

1.2 Current treatment of Cabin Waste

In Madrid-Barajas Airport, such as the rest of Spanish airports, waste from flights from destinations within the EU (classified as Cat3) is formed by a mix of inorganic recoverables (light packaging plastics, cans, cartons, glass and paper) and what is assimilated to and called MSW (Municipal Solid Waste) fraction. This last is mainly composed of organic matter plus all other waste that the crew cannot separate (typically napkins, thin plastics, etc.). In the case of Iberia flights, as well as in other airline operators from Madrid-Barajas Airport, all those fractions are mixed in the same bag and accumulated in containers, which the authorized waste manager collects and brings to a sorting plant. There the inorganic recoverable materials are separated to be sent to a recycler. For the case of flights coming from outside the EU (classified as Cat1) this waste is collected in bags that are stored in containers that will be collected by the same management company, but unlike Cat3 waste, it is not sent to a sorting plant: it is directly deposited in an authorized landfill. (Figure 1).

Landfilling is a cheap way to dispose of waste, but very expensive if we take into account its environmental implications. Estimations speak of global CH_4 emissions from landfills to be 500-800 Mt CO_2-eq/y (Bogner et al., 2007). Only regarding food waste, 1.9 t CO_2-eq. (at least) are emitted per tonne of food waste, which amounts 170 Mt of CO_2-eq. (at least) emitted per year, representing ~ 3% of total EU27 GHG emissions (Bio Intelligence Service, 2010). In our project case, of the 6,000t, a third of the tons are Cat3, of which 40% (according to preliminary characterizations results) of the waste is organic matter. In addition, of the

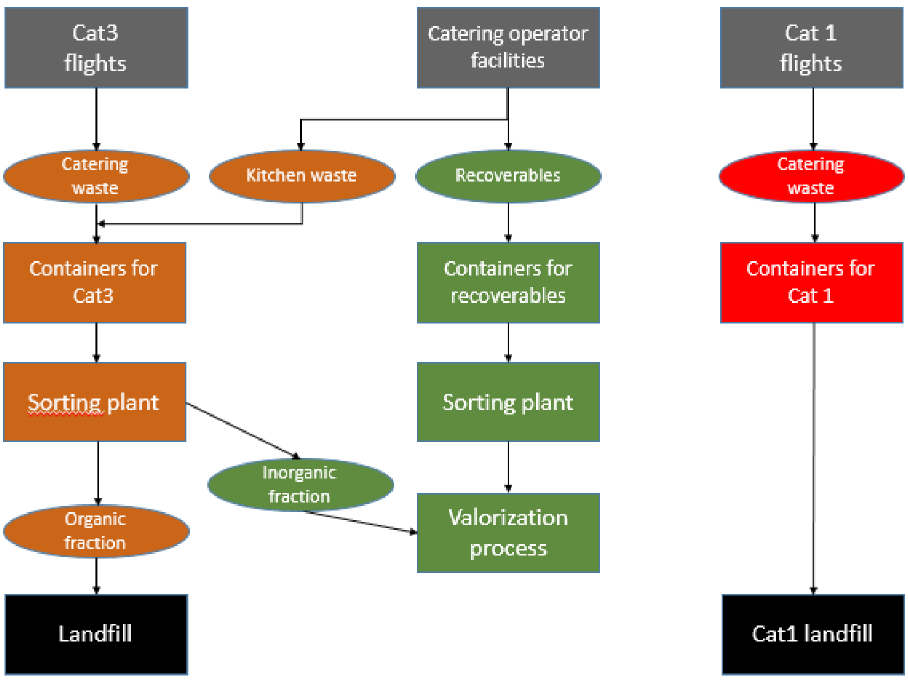

FIGURE 1: Current waste management for Cat3 and Cat1 waste.

4,000t that are generated from Cat1, 52% are organic matter, so that it ends up in landfill 2,880t annually. This translates, according to the emission factors for food waste previously shown, into 5,472 t CO_2-eq per year.

2. ZERO CABIN WASTE PROJECT

ZERO CABIN WASTE is a project founded by the Life Programme of the EU. It started in 2017 and is supposed to finalize in 2019. Table 1 shows the partners and their roles in the project.

2.1 Objectives

The project aims to create a sustainable model to reduce, re-use and recycle (including energy recovery) waste recollected in Iberia airplane cabins in Madrid-Barajas Airport (Spain) and set the basis for its replication in the future by other airlines and related stakeholders. Its final objective is to drastically reduce landfilling with at least 80%, (50% through recycling and 30% through energy recovery and compost), considering both Cat1 and Cat3 residues.

The specific objectives of the project are listed below:

- Paying more attention to the management of cabin waste. In order to reduce the amount of waste and to obtain more homogeneous waste streams that facilitate its subsequent recovery, a better classification at source is important. In this area, waste minimization must also be achieved through the implementation of good practices and eco-design measures for the menus served on board. Those measures require the involvement and the efficient coordination of all the agents involved;
- Change the legislation on the treatment of this type of waste has to follow. Currently, European legislation states that international cabin waste of animal origin must be incinerated or deposited in authorized landfills. The project aims to demonstrate that the current law is to some extent antiquated, overprotective and wasteful. By means of a sterilization treatment of Cat1, hazardous substances can be eliminated and, therefore, this type of waste can be valorized like Cat3 waste;
- Reduce the carbon footprint of the current waste management system. Landfill is the end of life option that emits more GHG (Cherubini et al. 2009). The project aims to reduce the amount of (mainly organic) waste sent to a landfill, and therefore, a reduction in GHG emissions is foreseen. The total reduction will be measured through a life cycle assessment (LCA) comparing the current management system with the proposed new system;
- Allow the replication of the new waste management system by other airlines and catering services to contribute to the reduction of the carbon footprint of its activities. This project is intended to demonstrate that with a comprehensive approach and a solid partnership between the members of the system, the waste management system can be improved.

2.2 Action plan

To achieve the objectives described above, the action plan is organized in the following stages:

- Preparatory actions. Detailed inventory of the waste flows and fractions per type of flight; analysis of potential re-use and waste minimization opportunities; consultations with key stakeholders and design of the recycling process. Current practices (processes, flows and fractions) modeled in an LCA program;
- Implementation actions. Training of crew and staff; installation of equipment adjustments; execution of the collection and separation protocol; processing of waste fractions; implementation of a pilot treatment for Cat.1 waste; and partial replication of the actions at Heathrow Airport;
- Monitoring of Technical and Environmental Progress. Technical monitoring of performance indicators (also LCA); proposed practices (processes, flows and fractions) modeled in an LCA program. At the end of the project, conclusions and recommendations will be given, including the socio-economic impact report of the project;
- Public awareness and dissemination of results. The project website and social media will be used in order to engage not only the passengers on board but also the professional stakeholders at national and EU level. Reforestation events will engage employees and clients further;
- Finally, project management will be carried out by all

TABLE 1: Partnership and project roles.

Partner	Role in the project
IBERIA	Coordinator & General project management. Leader in several preliminary and implementation actions and dissemination activities. Separation of waste onboard.
GATE GOURMET	Caterer of Iberia. First receptor of offloaded waste and responsible for first controls and further waste management. Leader of several preliminary and implementation actions. Contribute to technical monitoring and dissemination.
ECOEMBES	Responsible for sub actions concerning mainly waste characterizations, trainings and awareness-raising materials. Also in charge of conclusions and recommendations. Contribute to technical monitoring and dissemination.
BIOGAS FUEL CELL	Involved in several actions concerning waste management opportunities and design. Responsible for pilot action B5 (treatment of organic fraction Cat.1 waste). Contribute to technical monitoring and dissemination.
FERROVIAL	Mainly responsible for the management of waste in recycling plant and valorization process. Contribute to technical monitoring and dissemination.
ESCI-UPF	Involved in different actions and sub-actions as to monitor LCA related parameters. Responsible for developing a state of the art LCA for aviation industry and for compiling and monitoring project performance indicators.

partners. Project evaluation and auditing will be part of this action, as well as the after-Life communication plan.

2.3 Project innovation

Given the nature of this project, its innovations are more related to conceptual, organizational and methodological aspects, rather than to strictly technological developments. It is also worth mentioning the scale of the implementation. Companies such as Delta Airlines already recycle aluminum cans, plastic bottles, plastic trays, beverage cups, newspapers, and magazines but they only do it in a small percentage of flights operated (around 8%) and just in one international destination. This project deals with waste produced in aircrafts as a whole, looking for an integrated solution based on prevention, preparation for re-use and recycling. It also brings on board all main stakeholders involved along the whole chain and considers the impact through the life-cycle of the activities. This is a major difference in comparison to other strategies initiated by other airline companies.

It is intended to implement the actions at full scale with IBERIA's flights, both at EU and international level, having trained all members of its crew as well as Gate Gourmet's staff in Madrid and at Heathrow. Thus, creating a best practice code with a very high replication potential. To replicate, the geographical factor should be taken into account. The airlines and related companies' possibilities differ from one continent to another significantly. For example, some Asian airlines already introduce in the contracts of their crews the obligation to separate on board. We are far from this point in Europe, where cooperation of the crew remains a challenge and must be tackled tactfully and realistically. Another important difference is that, on other continents, airlines and authorities are more open to tackle the issue of Cat1 waste. This is the case of, for example, Australia or Canada, where sterilization of this kind of waste has already been successfully trialed. Consequently, the project must be understood in a European context (same legislation and culture), even if its expected outcomes could be replicated elsewhere worldwide.

The proposal of an alternative method to manage Cat1 waste which does not exist in Europe sets the highlight in the innovation of this project. At an early stage, it is foreseen to treat a small fraction of Cat1 with different methods to prove it innocuous for human and animal health, then taking the organic fraction to a bio digestion process allowing energy recovery. Afterward, the proposed management system (Figure 2) will be scaled for the treatment of Cat1 waste to industrial levels and its environmental performance will be measured trough an LCA. Implementing this integrated waste management system in which separate collection in origin takes place, with energy recovery from waste and reducing landfill disposal can guarantee high efficiency when minimizing CO2eq emissions (Calabrò, 2009), (Calabrò, Gori, & Lubello, 2015).

Although European legislation allows both incineration and landfilling as a way to manage Cat1 waste, Spanish legislation has narrowed down options to disposal in landfill. As one of the main objectives of this project is to reduce the carbon footprint of the system, bio digestion is a better option for the energy recovery of Cat1 organic matter rather than incineration as an alternative to landfilling (Eriksson et al., 2015).

Finally, in collaboration with national & EU relevant authorities, it is intended to develop an integrated best practice guideline on catering waste management that would include the new proposed valorization method.

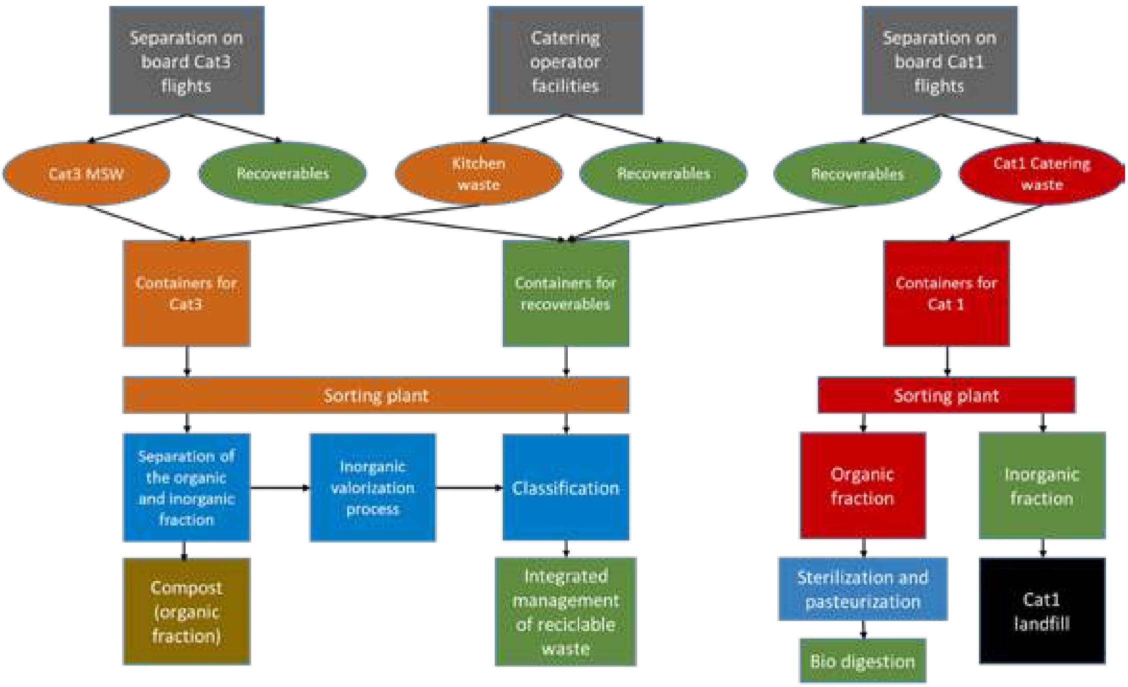

FIGURE 2: Future waste management for Cat3 and Cat1 waste.

3. DISCUSSION AND FIRST OUTCOMES OF ENVIRONMENTAL PERFORMANCE

Thorough bibliography research of LCA studies dealing with catering and aviation was performed. The use phase was found to have the greatest environmental impact, due to the kerosene burned during the flights (Horvath and Chester, 2008) (Lopes, 2010) (Howe, Kolios, & Brennan, 2013). To deal with this, the literature proposes that the impact can be reduced by making parts of the aircraft from lighter materials that would save fuel (Timmis et al., 2015). Although the manufacture of these components (carbon fiber) has a greater impact than traditional materials, (aluminum), it is largely offset by the reduction of impact during the aircraft use phase, by reducing weight (Beck et al. 2011).

However, no specific references for catering in aviation were found, although the same option of weight reduction may apply. Finally, the research was expanded to also englobe LCA studies on food and packaging in other sectors, in order to learn from eco-design alternatives other than dematerialization.

Regarding food, it was found that from the stage of agriculture until reaching the final consumer, the stage of agriculture is the one with the greatest environmental impact (Bellarby et al. 2008), followed by transportation and manufacturing (Tassielli et al. 2017). It will be crucial to take into account its origin so that, according to their associated environmental impact, increasing the design of menus with lower carbon footprint (Sim et al. 2007). The types of food that contribute most to the impact categories are those of animal origin, especially those of bovine origin (Foster et al. 2007) (Williams et al. 2006). Indeed, food of vegetable origin is the one with the least impact.

With regard to catering, comparative studies have been published between reusable and non-reusable packaging for glasses, plates and cutlery. The manufacture of reusables produces more impact than those of a single use, but it can be offset by the number of uses that the reusable ones can be given by a single container (Garrido and Alvarez del Castillo, 2007). Therefore, the number of uses together with the efficiency of the washing process, which is the stage with the greatest impact on the life cycle of the reusable containers and cutlery, will determine whether it is more beneficial to use disposable or non-disposable ones (Woods and Bakshi, 2014) (Pro.mo/Unionplast, 2009).

Due to the fact that the possible and alternative treatment of cabin waste depends largely on its composition, a characterization of the waste generated in the aircraft was done. Residues of 87 different flights were analyzed. As on some flights there is not enough waste generation to make a characterization, those flights were grouped as shown in Table 2.

As can be seen in the previous figure, flights coming from London and Medium flights that were longer than average, enough waste was generated to make characterizations out of a single airplane.

Flights were differentiated according to the length of the flight: National (flights coming from Spain), European, (those coming from EU), and International (being Short, Medium or Long depending if the flight takes more than 3, 5 or 7 hours, respectively).

Waste streams were also taken into account differentiating 5 streams as showed in Table 3.

Waste was differentiated by material and was subgrouped by the type of plastic and metal and whether it had been manipulated (the packaging, has been opened no mater if the content was consumed or not) or unmanipulated (Table 4).

The latter was important, since packaging manipulated on board is considered waste regardless its final consumption by the passenger or not. Now the composition and the amount of waste generated during every single Iberia's flight is known, as well as the generation of waste per pas-

TABLE 2: Grouping of flights for waste characterization.

Type of flight	Number of flights grouped	Number of groups	Total flights
National	5	5	25
European (Flights from London)	4 1	7 3	31
Short International	2	1	2
Medium International Longer flights	2 1	2 3	7
Long International	1	22	22
			87

TABLE 3: Different waste streams.

Stream	Description
Waste trolley	They contain waste generated during the flight, mainly coming from the sale on board
Galley	Trolleys that mainly contain baverages (water, soft drinks, wines, juices) and napkins
Business menu	Trolleys that contains the remains of the menus that have been served (trays)
Tourist menu	Trolleys that contain the remains of the tourist menus that have been served (trays)
2nd menu	Trolleys that contain the remains of the 2nd menus that have been served

TABLE 4: Waste classification.

Manipulated								Unmanipulated		
Packaging	Organic matter	Cellulose	Cutlery	Glass	Paper and cardboard	Organic Matter in packaging	Liquid in packaging	Packaging	Organic Matter	Liquid in packaging

PET	Natural HDPE	Color HDPE	PVC	Film	PP	PS	Other Plastics	Steel	Aluminium	Flexible polylaminate packaging	Wood

senger since information about the number of passengers of each airplane studied was gathered. Another article is being done with the whole study and analysis of the characterizations. Table 5 shows the waste generation per passenger depending on the flight length.

For National, European and Short International flights, most of the waste is collected in the waste trolley flow (64%) as, on these flights, no tourist class menu is served, therefore there is no tourist trolley on board. Followed by the business menu flow (31%) and the galley (5%). As for Long International flights and Medium International flights, the majority of the waste comes from the tourist flow (29%), followed by the waste flow (22%), business menu (21%), second tourist menu (15%) and galley (13%) (Table 6).

With the outcomes of all this research, an eco-design guideline for the catering services company (GG) was developed, including recommendations for changes in the configuration of the menus (reducing the amount of meat) changes in the design of some packaging items (extending the use of reusable solutions) and also other recommendations to reduce the amount of generated waste in each flight (such us asking passenger preferences when booking the flight in order to better adapt the loading of the meal on board or asking the passengers to deliver newspapers on board in order to make them available for other passengers and, therefore, reducing the amount of paper waste).

TABLE 5: Waste generation per passenger and flight.

Flight	Kg/passenger
National	0,14
European	0,25
Short International	0,23
Medium International	0,99
Long International	1,4

TABLE 6: Waste generation streams.

Type of flight	Sources	% waste
National	Galley Waste Trolley Business	8% 61% 31%
European	Galley Waste Trolley Business	5% 63% 32%
Short International	Waste Trolley Business	70% 30%
Medium International	Galley Waste Trolley Business Tourist	6% 34% 21% 39%
Long International	Galley Waste Trolley Business Tourist 2nd menu	13% 20% 21% 27% 19%

At this stage of the project, the anticipated reduction of GHG emissions has been estimated to be around 4,340t CO_2 eq. per year by using the LCA methodology. The functional unit chosen was the management of all the waste coming from the catering of Iberia aircrafts arriving in Madrid that were collected by Gate Gourmet and managed by Ferrovial during the year 2016. The burdens of the system, as Figure 3 shows, are the stages of unloading the waste from the Iberia aircrafts, transport to the GG facilities, collection of the waste by Ferrovial to take it to its selection plant, transport from Ferrovial to the different recyclers, recycling processes and the landfill. It also includes the savings associated with the production of electricity and primary secondary materials from alternative processes.

Gabi (2017) software was used for the calculations and the method of impact evaluation chosen was the one recommended by the ILCD Manual and those of the European Commission's Product Environmental Footprint Initiative, paying special attention to the environmental impact category of climate change to calculate the total carbon footprint.

The current situation has been compared with another scenario, in which Cat1 recoverables and both Cat1 and Cat3 organic matter being currently sent to landfill are managed with alternatives higher in the hierarchy: the recycling rates are 4.5 times higher and 88% of organic matter is considered to be sent to a valorization process from which biogas can be obtained as a sub-product.

4. CONCLUSIONS

Being in the early stages of the project, the preliminary outcomes are laying the groundwork for reaching the goals set. Through the state of the art analysis, we have been able to identify what the premises to be taken into account are when guiding future decisions through a life cycle perspective.

Using lighter aviation construction materials will reduce environmental impacts as a more efficient combustion will occur. Regarding catering food, menus with a greater amount of foods of vegetable origin will have a lower carbon footprint than those where there is the presence of meat, especially bovine.

LCA perspective should be taken into account when deciding what kind of material both for packaging and cutlery should be used, as the results depend on the number of uses of the reusable item, the efficiency of the washing

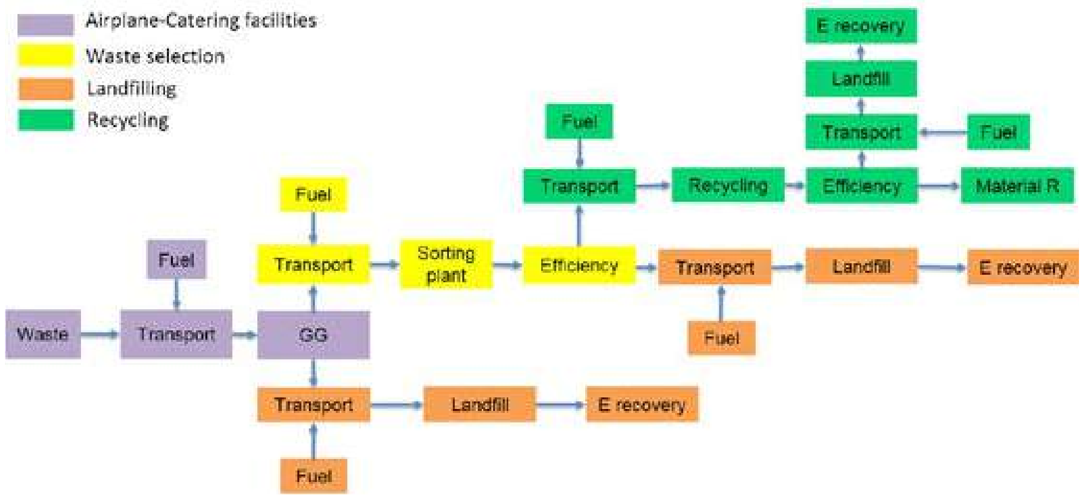

FIGURE 3: LCA stages for Cat3 and Cat1 waste management.

process and the number of washes between the uses since the washing stage is the one with the highest impact for the reusable items. In addition, single-use items fabrication has a less environmental impact and are lighter reducing emissions while flying.

In addition, the characterization study allows discover the composition of the waste and its origin, to plan an efficient and differentiated management. The outcomes of the study reveal that the distance of flight has a direct relationship between the amount of waste and the unmanipulated material generated. The majority of it, is organic matter that comes from the menus.

It is in the waste flow and in the tourist flows where most of the recoverable waste is, therefore more efforts have to be made there, for a correct separation in origin.

It is expected that with the development of the project and the implementation of measures in the current system, a substantial improvement of the entire process will be achieved. Moreover, if we take into account its more than probable replicability in other airports.

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QUANTIFICATION OF MUNICIPAL SOLID WASTE MANAGEMENT IN THE UNITED STATES – WITH COMPARATIVE ANALYSIS TO OTHER ESTIMATES

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ABSTRACT

The Environmental Research and Education Foundation (EREF) recently completed a study to quantify the amount of MSW managed in the United States. The study represents the first to use a bottom-up, facility-based methodology in the U.S. to improve accuracy and more discretely track the amount of waste managed. Over 9,000 facilities managing municipal solid waste (MSW) were identified through the course of the study (landfills, WTE incinerators, recycling facilities, composting operations, and anaerobic digesters), and information on the amount and types of waste managed were collected. Results suggest 1.4 times more MSW is managed in 2013 than previously suggested by U.S. Environmental Protection Agency (U.S. EPA) estimates. Data show that the majority of MSW was landfilled and the least amount of tonnage was composted.

1. INTRODUCTION

1.1 MSW estimates in the United States

Each year a significant amount of municipal solid waste (MSW) is generated in the United States, the collection and subsequent management of which has implications for sustainability. Worldwide the waste sector comprises approximately 18% of global anthropogenic CH_4 emissions (Bogner et al., 2007). In the U.S. waste disposal accounts for 22% of national anthropogenic CH_4 emissions (US EPA, 2010). Additionally, landfills are among the largest anthropogenic sources of CH_4 in the U.S. and are frequent targets for mitigation (Chanton et al., 2011). As such, accurately tracking both the quantity of waste diverted from and deposited in U.S. landfills is key to understanding sustainability from both materials management and global climate change perspectives.

Although both state and federal entities seek to quantify annual waste management, estimates of nationwide MSW generation and fractionation between management endpoints (i.e. landfills, incinerators, recycling facilities, and composting operations) have historically differed greatly (Tonjes and Greene, 2012). The two primary sources for nationwide MSW generation, recovery and disposal information have been the US Environmental Protection Agency's (US EPA) annual Facts and Figures report, and the biennial State of Garbage series published by Biocycle magazine through Columbia University. In 2008, the most recent year for which both sources estimated MSW generation, estimates differed by 126.9 million metric tons, or about 50% (Tonjes and Greene, 2012).

Differences between estimates are attributable to a number of factors, primarily differences in methodology and inability to resolve disparate MSW definitions. The US EPA implements a top-down (material flow) methodology in which production, import and export values are coupled with estimated product life to approximate annual waste generation. Management fractionation is approximated using data for remanufacturing (recycling), recovery (composting) and incineration, with the net assumed as landfilling. By contrast, Biocycle estimates are derived from a middle-up methodology in which state agency-provided statistics are aggregated to provide national-level data. As a result of dependency on state agency data reporting structures, these estimates are susceptible to error introduced by factors such as differences in state permitting and reporting requirements, data collection and calculation methodologies, and material types included in state definitions of MSW.

One approach to increase accuracy of waste management estimates is the use of a bottom-up facility-based methodology where tonnage and material data is aggregated across all MSW management facilities (i.e. landfills, incinerators, recycling facilities and composting opera-

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tions). By aggregating facility data, rather than state-reported statistics, tonnage data is captured from those entities not required to report to the respective states. The inclusion of material data (e.g. fraction MSW, industrial waste, construction and demolition waste) allows for the use of a consistent definition of MSW for all states. EREF has used this bottom-up facility-based methodology to estimate MSW management in 2010 and 2013 for each state and the United States as a whole (EREF, 2016).

1.2 Nomenclature

EREF: Environmental Research & Education Foundation
IPCC: Intergovernmental Panel on Climate Change
MRF: Material Recovery Facility; typically, a highly-automated facility for the processing, sortation, and baling of recyclable commodity materials
MSW: Municipal Solid Waste, i.e. waste generated in residential, commercial and institutional sectors
Non-MRF: Recycling facility not fitting the description of a Material Recovery Facility (MRF); facility for the aggregation and/or densification of recyclable commodity materials.
OECD: Organization for Economic Co-operation and Development
US EPA: United States Environmental Protection Agency
WTE: Waste-to-Energy incineration with energy recovery

2. QUANTIFYING MSW MANAGEMENT

2.1 Approach

The use of a bottom-up methodology necessitates identification of all waste management infrastructure, as no standardized count or database exists for facilities due to inconsistent notification and permitting requirements between states. Facilities were identified and reported tonnage was aggregated to estimate the amount of MSW managed at the country's landfills, waste-to-energy incinerators, composting operations, and recycling facilities.

Over 9,000 facilities managing MSW materials were identified as operational during 2013, the majority of which were associated with material recovery (i.e. recycling and composting) (Table 1). Two distinct types of recycling facilities were identified: traditional material recovery facilities (MRFs) with highly automated sorting and baling lines; and smaller material aggregators (termed "non-MRFs" in the study) which typically perform minimal sorting, may accept only limited material types (e.g. steel and aluminum cans exclusively), and little automation of the processing line.

2.2 MSW management in the U.S.

Results indicate approximately 315 million metric tons of MSW was collected in 2013, and subsequently managed at MSW facilities (Table 2). The majority (64%) was disposed of in landfills. Approximately 21% of generated MSW was recovered at recycling facilities (both highly-automated MRFs and non-MRFs). It is important to note this figure includes only commodity recyclables that are part of the US EPA definition of MSW (i.e. paper, glass, plastic, and non-scrap metals from residential, commercial, and institutional sources). An additional 6% of MSW was recovered for composting, resulting in a 27% combined rate for recycling and composting. The remaining MSW was managed at waste-to-energy facilities.

2.3 Comparison to other studies

The bottom-up tonnage estimates indicate significantly more MSW is generated, recovered, and disposed in the U.S. than previously thought, based on comparison to US EPA estimates for the same year (US EPA, 2014). Total MSW generation for 2013 was estimated by US EPA as 230.5 million metric tons of MSW, a difference of 84.3 million tons or approximately 37% (Table 3).

The largest difference between estimates exists for landfilled tonnage. This is attributable, in part, to the differences in methodology between the two estimates. Although tonnage and material data for landfills is widely available through reporting data and scale ticket measurements, the material-flow methodology from which US EPA derives its estimates does not utilize this data. Instead, landfilled tonnage is estimated as the net of estimated generation minus estimated remanufacturing, recovery, and incineration (US EPA, 2014). By contrast the EREF's facility-based methodology uses scale reports for Subtitle D landfills, providing increased granularity and accuracy. Given that Subtitle D landfills can also manage a variety of non-MSW non-hazardous wastes (i.e. construction and demolition debris (C&D), non-hazardous sludge, and industrial solid waste), one key element of this assessment was to use site-specific material data to separate MSW from non-MSW tonnage. Detailed material data was available in 14 states, representing 37% of landfilled tonnage in 2013. Data suggests one-third of material accepted at MSW landfills was non-MSW, with individual state values ranging

TABLE 1: Number of facilities identified as processing MSW during 2013.

Type of Facility	EREF	Previous Estimates
Recycling	3,913	1,652 [a]
MRFs	799	590 [b]
Composting	3,494	3,285 [c]
Landfills	1,540	1,802 [a,d]
Waste-to-Energy	81	94 [a,e]
TOTAL	9,028	6,833

[a] *Waste Business Journal (2014)*
[b] *Berenyi (2007)*
[c] *ILSR (2014)*
[d] *Includes some C&D landfills*
[e] *Includes some non-MSW incinerators, such as medical waste*

TABLE 2: Amount of MSW managed at identified facilities in 2013.

Type of Facility	MSW Managed (million metric tons)	Percent of total
Landfills	201	64%
Recycling	66.2	21%
Waste-to-Energy incineration	27.9	9%
Composting	19.3	6%
TOTAL	314.8	100%

TABLE 3: Differences between EREF and US EPA estimates for 2013.

Type of Facility	EREF Estimate (million metric tons)	US EPA Estimate (million metric tons)	Percent Difference
Landfills	201	121.8	65 %
Recycling	66.2	58.7	13 %
Waste-to-Energy	27.9	29.7	-6 %
Composting	19.3	20.3	-5 %
TOTAL	314.8	230.5	37 %

from 9-82% non-MSW (EREF, 2016b).

Acceptance of non-MSW materials occurred at all facility types, but was most common in landfills, composting operations, and non-MRF facilities (e.g. scrap metal processors accepting steel and aluminum cans from residential generators). As illustrated with Subtitle D landfills, facility-specific tonnage and material data was key to minimizing the inclusion of non-MSW materials in the EREF estimates and therefore minimizing sources of error existent in other studies (e.g. Biocycle). Results also suggest US EPA underestimates MSW managed via recycling, however to a lesser extent (Table 3). By contrast, US EPA may overestimate both MSW incineration and composting. This is likely due, in part, to the potential inclusion of non-MSW materials (e.g. industrial waste or agricultural biomass) in industry-reported statistics incorporated into the US EPA recovery figures.

A recent study corroborates the assertion that US EPA underestimates MSW landfilling, using facility data from the subset of Subtitle D landfills included in the GHG reporting database. Powell et. al. (2015) estimates 262 million metric tons was managed via landfill in 2012, whereas US EPA estimates 122 million metric tons for the same year: a difference of 115%. With independent lines of research suggesting US EPA underestimates MSW sent to landfill, it stands to reason that a bottom-up methodology currently produces the most accurate estimate of MSW-only material deposited in landfills in the U.S.

2.4 Comparison to international statistics

Recently the World Bank (2012) estimated that developed countries (denoted as OECD) generated the most MSW in the world, at about 2.2 kg/person-day. Using the EPA values, the U.S. would actually be lower than the OECD average at 2 kg/person-day. However, EREF values put U.S. per capita MSW generation at 2.7 kg/person-day, which would make the U.S. the largest global waste generator on a per capita basis, about 23% above the OECD value and nearly 2 ½ times higher than Europe (Figure 1).

In addition to MSW generation, the management of MSW also differs by country. Statistics compiled by the World Bank (2012) indicate the percentage of MSW managed via landfilling, WTE incineration, recycling, and composting by nation. Excerpted results for 8 countries, and results from EREF's bottom-up estimates for the U.S., are shown in Figure 2. MSW management in the U.S. is most similar to that of the U.K. which exhibits an identical landfilling rate (64%) and similar material recovery (i.e. recycling and composting) rate (26% compared to 27% in the U.S.). Austria reported the highest material recovery rate (71.26%). Switzerland reported the lowest landfilling rate (1%). The highest waste-to-energy incineration rate was reported in Japan (74%).

Differences in MSW management statistics between the U.S. and other countries indicate potential improvement through both a reduction in waste generation and an increase in material recovery (i.e. recycling and composting). The challenge to achieve these aims in the U.S. is

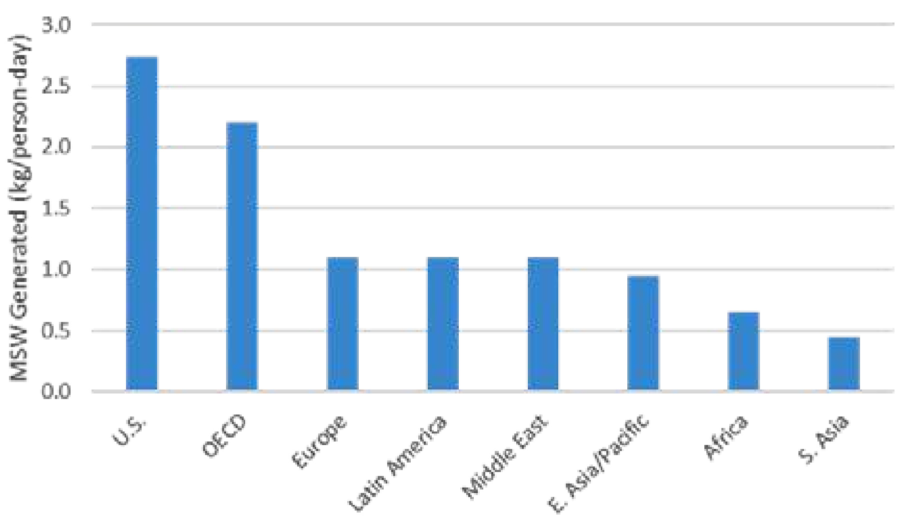

FIGURE 1: Per capita waste generation of the U.S., Europe and global regions (Note: OECD = Organization for Economic Cooperation & Development; i.e. developed countries).

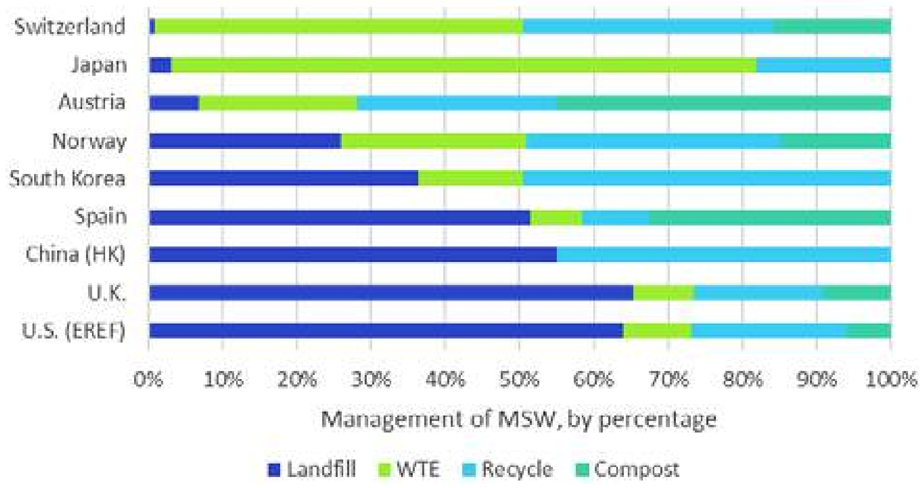

FIGURE 2: EREF estimates of U.S. MSW management compared to MSW management of selected nations (Source: World Bank, 2012).

multi-faceted, including: infrastructure, policy, and behavioral challenges. It should be emphasized that while substantial infrastructure already exists in many regions of the U.S. to divert materials from landfills, others may lack sufficient infrastructure to further increase recycling or composting rates (EREF, 2016). Policy differs across the U.S., presenting another challenge to increased recovery. For example, 53% of U.S. states ban yard waste materials from landfill while 10% mandate food waste recovery (EREF, 2015). Even in areas with sufficient infrastructure and policy drivers for recovery, challenges such as increasing recycling contamination rate exist due to participant behavior (EREF, 2016).

3. CONCLUSIONS

The use of a facility-based, bottom-up methodology is key to increasing accuracy of MSW management estimates (Powell et. al., 2015). The use of such methodology to estimate MSW managed in the U.S. suggests that 315 million metric tons of MSW were managed in 2013, or approximately 2.7 kg/capita-day. Of this, the majority of waste was landfilled, with 27% recovered via recycling and composting combined.

Results represent a 37% difference in total MSW managed compared to US EPA estimates for the same year, with the largest difference for landfilled tonnage (65% difference; Table 3). A large difference for landfilled MSW compared to US EPA has also been documented in other facility-based estimates (Powell et al, 2015). Landfills have consistently been listed as one of the largest sources of anthropogenic methane in the United States by entities such as the Intergovernmental Panel on Climate Change (IPCC) and the US EPA (US EPA, 2010). As such, accurate estimates of MSW generation and management are key to understanding the environmental impact of end-of-life material management decisions and assess the nation-wide progress toward material recovery and sustainable materials management goals. Studies suggest that current inputs from US EPA material flow models may not provide accurate data for these efforts, however, with facility-based results suggesting that managed tonnage is greater than US EPA estimates (Table 3).

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WASTE MANAGEMENT IN DEVELOPED AND DEVELOPING COUNTRIES: THE CASE STUDY OF UMBRIA (ITALY) AND THE WEST BANK (PALESTINE)

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ABSTRACT
A survey and a preliminary comparison were conducted between the waste management systems and schemes implemented in the Region of Umbria (Italy) and the West Bank (Palestine). The Region of Umbria operates in a wider political legal and economic supporting scheme, i.e., the one promoted by EU Directives. The West Bank showed all the typical economic, legal and political features of a developing country. From the economic point of view the incidence of the cost for waste collection and management with respect to the per capita GDP was 0.82% for the Region of Umbria and 1.2% for the West Bank. Although the incidence for the West Bank was higher, it was not enough to support the budget necessary for efficient waste management. A relevant aspect was the practically similar amount of organic waste generated per capita and per year in the two areas. The West Bank lacks infrastructures and adequate collection systems and there are no composting facilities. The number of mechanical sorting facilities was 0.034/10^5 inhabitants. The current recycling rate for the West Bank is about 6%. Some criticism about the sustainability of recycling and composting rates for the Region of Umbria are also highlighted. Some benefits for the West Bank, such as the introduction of home composting, are identified. This will affect both the amount and the costs of waste collected allowing the municipalities to allocate more money for separated collection of recyclables.

1. INTRODUCTION

Although having different perspectives, waste management is one of the key issues to be addressed both by developed and developing countries for achieving a sustainable implementation of the various human activities worldwide.

According to Marshall and Farahbakhsh (2013), progress in the waste management sector has been historically influenced by six key factors: public health; environment, resource scarcity, waste value, climate change and public awareness. These aspects are affected both directly, as by the emissions generated from incorrect collection and disposal of wastes (Couth and trois, 2011, 2012; Tian et al., 2013), and indirectly as a consequence of raw materials consumption and transformations (Di Maria et al. 2014).

Currently the most effective approach for waste management worldwide is based on the three R concept: Reuse, Recycle and Recovery. This was extrapolated from the wider concept of the waste management hierarchy, introduced in the European Union (EU) in 1977 by the European Commission (CEC, 1977), stating the main activities and goals to be pursued with strict hierarchic order in waste management: Prevention; reuse; recycle; recovery; disposal.

The hierarchy concept was definitively introduced in the EU legislation, becoming a fundamental component of the integrated waste management approach in the EU in 1991 by the first Directive 91/156/EEC on waste (Council Directive, 1991). This has always been confirmed throughout the years including the latest Waste Framework Directive (WFD, 2008) that introduced another important goal to be achieved within 2020 by member states: the recycling of at least of 50% of the whole waste generated. Recycling also includes the organic fraction via biological treatments that is able to generate organic fertilizer, effectively exploitable in agriculture.

Furthermore, the implementation of the waste management hierarchy was described by the European Commission (EC) as a key activity in communication n°614 (COM, 2015) concerning the EU Action Plan on circular economy. A key factor for the optimization of recycling and reuse is

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effective and efficient waste collection, consisting of efficient source segregation able to return high quality recyclables directly exploitable in industry. Municipalities are the authorities in charge of providing municipal solid waste (MSW) collection directly or by private/public companies. Currently, collection coverage in the EU15 is practically 100%.

For the year 2016 the MSW management in the EU15 was: gross generation 207,862,000 Mg; Recycling 29.5%; Composting and /or anaerobic digestion (AD) 17.4%; Incineration 29.9%; Landfilled (sanitary landfills) 23.1%. Considering that the percentage of bio-waste in the municipal solid waste (MSW) at the EU15 level is about 30%, more than 50% is currently recycled by composting or integrated processing with AD and post-composting (Di Maria et al. 2016; Smidt et al., 2011). The remaining amount could be considered quite equally shared between the amount of waste incinerated and the amount landfilled.

Waste management in most developing countries is still greatly based on uncontrolled dumping and/or littering and, domestic burning. This mismanagement leads to serious health and environmental problems (Guerrero et al., 2013; Henry et al., 2006; Sharholy et al., 2008; Al-Khatib et al., 2010). Kumar et al. (2009) showed that more than 90% of MSW in India is directly disposed of on the land in an unsatisfactory manner and collection coverage is often less than 60% (Zhang et al., 2010; Henry et al., 2006). Couth et al. (2011) reported that in Africa GHG emissions from waste management were 3 times higher than those in developed countries and similar results were also reported by Tian et al. (2013) concerning the Chinese scenario. Per capita production ranges from 0.4 kg/day to 1.0 kg/day (Zhang et al., 2010), peaking in certain areas also up to 1.7 kg/day (Manaf et al., 2009). The organic fraction content ranges from 45% up to more than 80% of the whole waste generated in developing countries, representing the main source of health and environmental concerns (Al-Khatib et al., 2010; Zhang et al., 2010; Henry et al., 2006;). In general, the waste management system consists in transporting the waste outside of cities (Marshall et al., 2013). The rapid and unplanned growth of cities has led to extreme land use problems and infrastructural challenges that have crippled the capacity of governments and local authorities to increase MSW services to meet the demands (Marshall et al., 2013). Funding and technical competency to provide an efficient waste collection service are lacking (Al-Khatib et al., 2008; Henry et al., 2006). Similarly, Guerrerio et al. (2013) confirmed that waste management failure in cities of developing countries is due to inadequate technical, environmental, financial, socio-cultural, institutional and legal aspects. A study showed that political instability, civil wars and military operations contribute to increasing the difficulties in the waste management sector. From the social point of view the extraction of recyclables from waste is largely performed by the informal sector often operating in unsafe conditions (Manaf et al., 2009). In general recycling figures were very poor, less than 10%, together with a significant lack of facilities for the treatment of the largest and most threatening waste component, the organic fraction (Kumar et al., 2009). Concerning Palestine and the West Bank area, the waste generation ranges from 0.48 kg per person per day up to 2.00 kg/per/day. Despite the high coverage (98%), more than 80% of the waste is open dumped. It is also reported that from 2% to 8% of the municipal budget is dedicated to MSW management (MSWM), indicating a low priority for this activity (Al-Khatib et al. 2007). As reported by Khatib et al. (2010) for the Nablus district the percentage of the organic fraction in the MSW was about 65%, whereas the annual fee for MSWM was about 51 USD/year.

The aim of the present work was to assess the waste management status in the Region of Umbria (Italy) and in the West Bank (Palestine), in order to quantify some social and waste management indicators able to give a more objective assessment of the problems in the two areas.

2. MATERIAL AND METHODS

2.1 Study area

The present study was carried out in the Region of Umbria (Italy) (Figure 1a) and the West Bank (Figure 1b) (Palestine). The Region of Umbria consists of about 900,000 inhabitants with a total surface area of about 8,500 km^2 characterized by the presence of large mountainous areas and temperate climatic conditions. The average per capita GDP in 2012 was about 23,00 €. The West Bank consists of about 3,200,000 inhabitants on a total surface area of about 5,640 km^2 with arid climatic conditions. The per capita GDP for 2014 was about 4,300 USD (CIA, 2017). The study was conducted following these main methods.

2.2 Data collection and waste management assessment components

An in-depth analysis was carried out of the available literature and the annual reports from national and local authorities addressing the waste management implementation status. Available reports and documents about waste management in Palestine, including relevant regulations together with direct experiences of the authors operating in that area were reviewed Cesvi NGO. For the Region of Umbria both lab scale and full scale studies were performed at some of the plants (Di Maria et al., 2016; Di Maria et al., 2015; Di Maria, 2012) as well as on the basis of waste management reports of national authorities (ISPRA, 2016). Field visits and the technical/scientific analysis of facilities and waste management service providers, recycling companies, landfills, relevant facilities and stakeholders were done.

This was also performed for the Region of Umbria (Figure 1a) through a project funded by EC grant LIF12 IT/ENV/000411 from 2014 to 2016.

The study involved many stakeholders, mainly the University of Perugia, the Region of Umbria, local governments, one municipality and two main waste management companies managing the collection system and the different facilities operating in the area considered, such as mechanical biological treatment (MBT), composting and landfill. For Palestine (Figure 1b) this was performed by Cesvi NGO, also financed by the United Nations as part of the UNRWA project (2017). During this project citizens, municipalities and waste management companies were in-

FIGURE 1: Location of the study area: (a) the Region of Umbria, and (b) the West Bank (dashed) (b) (Centanni, 2012).

volved in defining a strategic waste management plan for 2018-2023 in those areas.

These stakeholders were also directly interviewed to assess the attitude regarding waste separation, the efficiency of the collection system and the objective to be pursued.

2.3 Data analysis and study limitations

The analysis and comparison of the two areas were conducted according to the following main indicators: per capita MSW generated; per capita organic fraction generated; percentage of waste management fee/per capita GDP (%); number of mechanical treatment facilities per 10^5 inhabitants; number of composting facilities per 10^5 inhabitants; recycling percentage (%); composting percentage (%).

These indicators will give an assessment about; the current level of waste management implementation; socio-economic correlations; waste management efficiency related to material recovery and recycling; effective implementation of a waste management strategy.

In particular the incidence of the waste management fee on the per capita GDP will provide information about the relative economic sustainability of the MSWM and also about the socio- economic incidence on the efficiency of this activity. The number of treatment facilities per inhabitant is another important aspect assessing the presence of specific goals and infrastructure in the sector. Similarly composting and recycling percentages indicate the efficiency and the presence of specific goals in managing one of the largest MSW components.

The study is mainly aimed at quantifying some relevant differences and/or similarities in the two areas under investigation using the indicators defined above. This could lead to an objective evaluation of criticisms and of current performances, also highlighting some causes of low efficiencies.

3. RESULTS

3.1 The region of Umbria

3.1.1 Status of waste production and management

The amount of MSW generated in 2015 was 519 kg/per capita, 48.9% of which was collected separately at source (ISPRA, 2016). Except for the organic fraction and greens, different wastes separated at source are moved to the recycling industry directly or via specific national consortiums. National consortiums were imposed at the time of the first EU Directive 91/156/EEC on waste management for implementing the extended producer responsibility (EPR). In contrast, the organic fraction and greens from separate collection are processed in 5 biological treatment facilities operating at the regional scale and managed by waste collection and/or treatment companies for the production of an organic fertilizer in compliance with the Italian legislation (D.Lgs., 2010). Of these 5 facilities 4 are exclusively aerobic, whereas 1 consists of integrated anaerobic pre-treatment followed by a post composting phase.

According to the EU and National legislation, MSW coming from the separated collection has to be properly processed before being disposed of in landfill. These treatments are aimed to extract other recyclables and or recoverable materials from the residual wastes, including energy recovery, and to reduce their final biological reactivity. As indicated by the first EU Directive 91/156/EEC, the most suitable treatment is by incineration which is able to recovery energy and, most importantly, to reduce both the mass and the reactivity of the materials, mainly returned slags. As an alternative to incineration, even if with lower efficiency in mass and reactivity reduction, MBT was widely adopted mainly due to its lower costs.

Currently in the Region of Umbria, there are 3 MBT in operation and one mechanical sorting (MS) facility processing about 222,000 Mg/year (2015) of residual MSW. There is no incineration facility for MSW.

The amount of waste disposed of in the 3 sanitary landfills was estimated at 260,000 Mg/year in 2015.

According to national and European legislation, municipalities are charged with collecting MSW. They can operate on their own or they can entrust it to private companies after public calls. The most widespread option is by private companies participating usually not less than 50% by municipalities. The MSW fee includes the whole service from collection, to recycling and disposal. According to the recycling goal imposed by the EU (WFD, 2008) the current MSWM is strongly oriented at improving recycling.

3.1.2 Separated collection

Separated collection at source has been demonstrated to be the most efficient method for returning high quality materials suitable for high recycling efficiency. This collection methodology is practically implemented in the entire regional area.

Two main methods are currently used to achieve this aim:

- Door-to-door collection;
- Proximity collection.

Together with these two methods there is also a large use of city amenities.

A timetable appositely developed is delivered each year to users indicating the collection frequency for each waste material. The organic fraction is collected 2 to 3 times per week, whereas, the other fractions are collected every 2 to 4 weeks. Glass is collected only by the proximity method.

Currently, 43 city amenities are in operation in the regional collection, separating the following waste streams: plastics; glass; electric and electronic wastes (WEEE); plant and animal oils; paper, cardboard and multi-layer; lubricating; gardening; bulky and others.

Regarding debris from small households, only small quantities up to 150 kg per year per user are allowed. Higher amounts are classified as special waste and have to be delivered to specific recycling/disposal plants. Users who deliver wastes to the amenities are credited by economic incentives, depending both on the amount and type of waste. Furthermore, city amenities play an important role regarding the amount of waste collected separately. More than 40% of the waste collected separately arises from these facilities. With respect to 2015 the entire management of one tonne of waste cost on average 190.8 €/tonne. Some innovation was introduced in this region by the economic support of a LIFE project EMaRES (Grant n° LFE12 ENV/IT/000411). Three main innovations were introduced to demonstrate their effectiveness:

- The first was the introduction of a traceability system based on bar codes and chips for door-to-door collection. With this system it was possible to know exactly the user, the amount of waste returned and the quality. Operators had to verify the quality of waste returned and to warn citizens if they were not operating correctly. On the other hand, this system was useful for user awareness and for implementing a pay-as-you-throw (PYT) fee system;
- The second was the implementation of a dynamic collection center for increasing the amount of small electric waste and electronic equipment WEEE (mobile phones, remote controller, hair drier…), used cooking oil and batteries returned by the citizens (Figure 2). In practice this was an appositely equipped truck able to move to the citizens, according to a specific timetable, in the various places of their everyday life, such as supermarkets, shopping centers, fairs, events. This vehicle also plays an important role in the continuous awareness of the population;
- The third was the implementation of an advanced me-

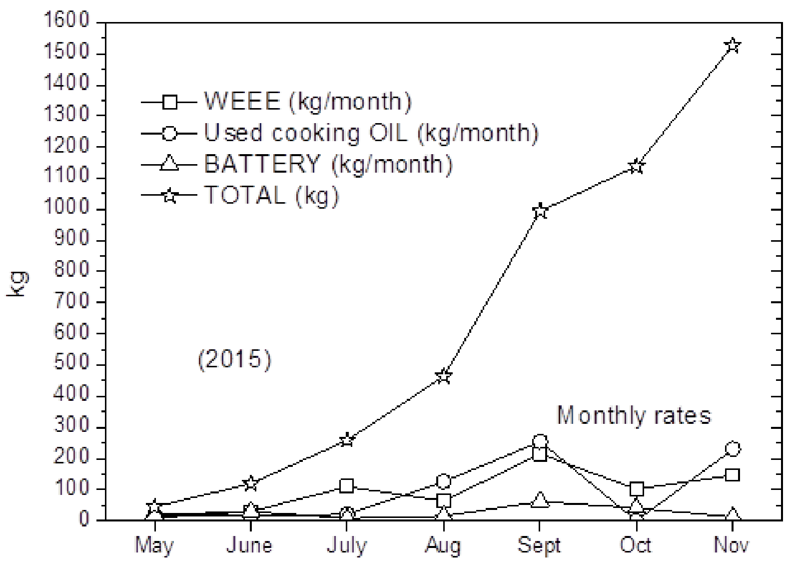

FIGURE 2: Amount of specific waste collected by the mobile collection center during its first months of operation.

chanical treatment system for the extraction of further recyclable materials from residual waste. This plant was equipped with different mechanical, physical and optical selectors and it was able to return on average from 9% to 10% of recyclables (plastics, metals and glass) from the residual waste (Di Maria et al. 2015).

3.1.3 Recycling

Waste delivered for recycling comes mainly from the separated collection. In 2015 the amount of waste material collected separately and moved to recycling was globally about 221,000 (Mg/year) (Table 1). The largest fractions were organic, paper, glass and plastic.

These materials were successively delivered to the national consortiums for recycling. The main national consortiums were CdC RAEE for WEEE and CONAI for packaging in plastics, paper, aluminium, steel, wood and glass. CONAI includes specific consortiums: CoReVe for glass packaging; CIAL for aluminium; RICREA for steel; COMIECO for paper and cardboard; RILEGNO for wood; COREPLA for plastics. In some cases, collection companies directly deliver the waste to the recycling industry.

3.1.4 Composting and anaerobic digestion

In the Region of Umbria there are five biological treatments for processing the organic fraction returned by separated collection. The achievement of end of waste criteria for this material is defined by the national legislation which imposes specific conditions on the waste source, on the process performances and on the features of the final product. For recycling, waste has to be the organic fraction separated at source (kitchen waste, green waste, restaurant). The process has to respect the following requirements: aerobic; lasting for not less than 90 days; able to guarantee 55°C for not less than 3 days (pathogen removal). Quality standards for final products are imposed by the Italian legislation for organic fertilizer (D.Lgs., 2010) (Table 2).

Currently Italian and EU legislation lacks specific criteria concerning recycling of the organic fraction by anaerobic digestion (AD). Consequently this limits the implementation of this technology in the specific sector.

At the regional level there is only one integrated AD and post-composting plant, which in 2015 processed 34,402 Mg/year of organic fraction from separated collection generating 3,810,431 Nm3 of biogas and 5,166 MWh of electrical energy. The amount of organic fertilizer generated in 2015 was 2,999 Mg. This means that the average biogas yield per single tonne was about 111 Nm^3/Mg, whereas the specific energy recovery was about 150 kWh/Mg. Sustainability of the higher investment and maintenance costs for AD was supported by economic incentives for the cost of electricity recovered accessible by new facilities until 2013. After that date, the only possible incentive achievable by AD was the generation of bio-methane. Currently a new AD facility for organic waste aimed at bio-methane generation is under construction in the Region of Umbria. Globally the amount of organic fraction processed in these 5 facilities in 2015 was 180,432 Mg with the production of 8,878 Mg of organic fertilizer.

3.1.5 Mechanical biological treatment

Currently three MBT are in operation (Figure 3) and are able to sort the MSW into two main streams, a dry stream rich mainly in plastics, paper, cardboard and textiles, and a wet stream rich mainly in the organic fraction, together with metals (both ferrous and aluminium) for extraction and recycling. For these facilities the wet stream is then successively processed in an aerobic bio-stabilization section (Di Maria, 2012).

Together with these three MBT, there is a mechanical sorting (MS) facility returning a dry, a wet and a recyclable stream (metals). For these three MBT the mass balances for 2015 were: Inlet: 165,771 Mg of MSW; Outlet: 123,877 Mg of dry fraction to landfill; Outlet: 45,034 Mg biostabilized wet fraction to landfill; Outlet: 708 Mg of ferrous metals to recycling; Outlet: 35 Mg of aluminium to recycling. For the MS the mass balance for 2015 was: Inlet: 48,629 Mg of MSW; Outlet: 27,212 Mg of dry fraction to landfill; Outlet: 19,650 Mg of wet fraction to landfill; Outlet: 85 Mg of ferrous metals to recycling. The energetic consumption for MBT was on average 30 kWh/Mg, whereas the MS facility had an average consumption of 17 kWh/Mg (Di Maria, 2012; Di Maria et al., 2015).

According to national and EU legislation, the wet fraction returned by MBT has to be adequately bio-stabilized

TABLE 1: Amount of waste materials moved to recycling in the year 2015.

Material	Amount (Mg/year)
Organic fraction and green	104,500
Paper	54,800
Glass	26,900
Plastic	17,965
Metals	5,143
Wood	8,220
WEEE	4,090
Textile	1,680
Other	3,200

TABLE 2: Mean chemical and physical features required by Italian legislation for classifying compost from bio-waste as organic fertilizer.

Parameter	Value	u.m.
Moisture Content	<50	% w/w
pH	6.0-8.5	-
TOC	>20	% on TS
TKN	-	% on TS
N organic	>80% of TKN	% on TS
C/N	<25	-
Cu	<150	ppm on TS
Zn	<500	ppm on TS
Pb	<140	ppm on TS
Germination Index	>60	%

Legend: TKN=Total Kjeldahl Nitrogen / TOC=Total Organic Carbon

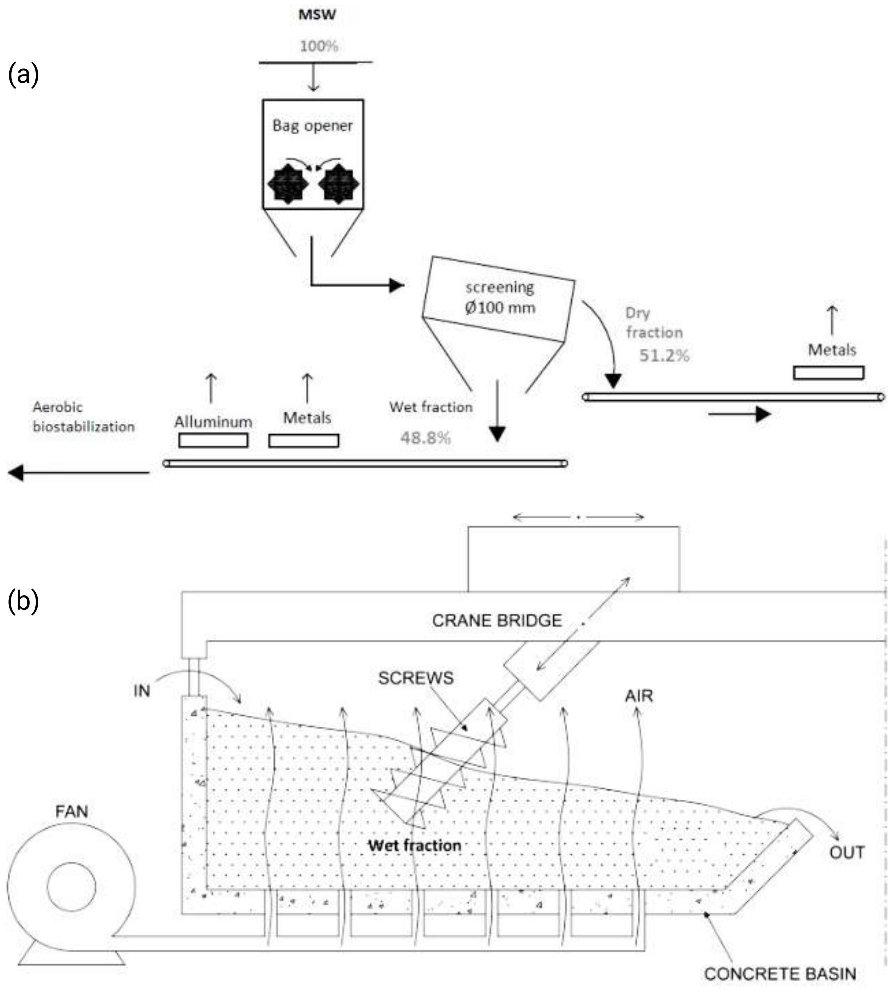

FIGURE 3: Scheme of the MBT sections: (a) mechanical sorting (MS) and (b) bio-stabilization.

for reducing the amount and the intensity of emissions during landfilling. The quality of the waste, particularly the wet fraction, is evaluated by a standardized methodology based on the dynamic respiration index (DRI) (mgO2/kgVs*h) (Di Maria and Micale, 2014). The DRI threshold imposed by national legislation for acceptance in landfill is DRI ≤ 1,000 (mgO2/kgVS*h) and even more stringent limits are imposed at the regional level. Several studies performed in similar geographical areas such as Italy and EU Member States (De Gioannis et al., 2009; Komilis et al., 1999) and in a specific geographical area (Di Maria et al., 2013) indicate a reduction in gaseous emissions of landfilled waste after bio-stabilization of up to 90% when compared to untreated waste.

3.1.6 Landfill

The amount of waste landfilled in 2015 was globally 264,504 Mg consisting of the following fractions: 12,531 Mg of MSW directly landfilled without any pre-treatment; 230,929 Mg of waste landfilled after treatment (MBT or MS); 21,062 Mg of waste different from MSW (commercial activities).

All the landfills currently operating are sanitary landfills equipped with landfill gas collection, energy recovery and a leachate collection system. Leachate treatment is performed both on site and off site (Di Maria and Sisani, 2017; Di Maria et al, 2018; Morello et al., 2016; Spagni et al., 2008).

3.2 West Bank

3.2.1 Status of waste production and management

According to ARIJ (2015), the total solid waste generated in 2015 was 0.78 Mg/day, – composed of 45-50% of household waste, 20-25% of construction and demolition waste and industrial waste, while the remaining amount was generated by commercial and institutional facilities. In 2015, 95% of households were reportedly served by a service provider, while 85% paid collection fees. Only 50% of the waste produced was landfilled, and the recycling rate was only 6%. However, these values are only estimates, and may be disputable, as indicated by Cesvi assessment activities in the sector.

A set of laws and policies regulate the solid waste management sector – including the National Strategy for Solid Waste Management 2010-2014 (MoLG, 2010), extended to 2017 and soon to be replaced. However, the legal and regulatory framework still looks incomplete and requires additional and more ready-to-implement laws and regulations.

The population of the West Bank was served by three different types of service providers. These actors were municipalities, the Joint Service Councils (JSC) for waste management and the United Nations Relief and Works Agency for Palestine Refugees in the Near East (UNRWA). Municipalities are the local authorities responsible for the service, and may decide to directly implement it or to contract a company working on its behalf. Few options are available as part of the MSW management (MSWM) service contracted out, especially to private companies, for example cardboard collection in the Tulkarem governorate. Instead, it is more frequent that municipalities of the same governorate join a publicly-owned company which replaces the municipalities for all the MSW steps required or for haulage and disposal, only. Finally, UNRWA provides refugees living in the 19 camps of the West Bank with various services, including MSWM.

Despite a certain number of recycling activities available – mainly informal – the number of MSWM facilities is limited. Three sanitary landfills – in Zharat al Finjan in the North, one in Al Minya in the South and a small one in Jericho in the East – are the only available disposal sites, apart from a large number of small and medium dumpsites, still used by small municipalities. There is a large private transfer station in Eizeriya which mainly serves the area of Israeli settlements. Waste collected in this transfer station is sent either to the Al Minya landfill – near Bethlehem - or to the Atarot sorting plant – in Israel, if it comes from Palestinian towns and villages or Israeli settlements, respectively. Small transfer stations are owned and managed by private enterprises, municipalities, and JSCs, mainly for local use. The environmental performances of the listed facilities are scarce, and all of them require immediate interventions.

There are at least, ten composting plants in the West Bank, owned by private companies or cooperatives. Some of them are grouped by the Agricultural Development Association, which provides them with some support and services. However, some of them are currently closed due to unsustainable management costs and difficulties in properly commercializing the compost produced. Few are truly sustainable, thanks to an acceptable and stable quality of the compost and a certain number of customers.

Several pre-treatment and recycling facilities are available, especially to sort, pre-treat or stock recyclables. Few examples of complete recycling are also available, and mainly for some types of plastic. Other recyclables are sent to Israel for direct recycling. These activities are mainly informal, or formalized only after some steps of the recycling process.

3.2.2 Solid waste recovery and recycling market

Solid waste recycling initiatives in West Bank depend on one hand on the entrepreneurial capacity of individuals and, on the other hand, on the intervention of international organisations. The Palestinian regulatory framework, although it implements waste prevention, recovery and recycling activities (PNA, 1999; PNA, 2010), does not propose any incentives for the establishment of such initiatives. Neither does it provide environmental performance control systems for waste storing and treatment enterprises. The legislative framework, however, permits waste recycling enterprises to be registered in the chamber of commerce of the governorate in which they are situated, although with no obligations. The registration to the chambers of commerce, finally, does not imply any specific verification from the authorities. The specific political and social conditions of the area considered also have a non-negligible influence on the waste management sector (UNCTAD, 2014; World Bank, 2008). This limits the creation of large recycling enterprises and usually this activity is performed by small, family-run, individual business with a large participation of the informal sector.

The waste most traded is metal, which is collected throughout the West Bank. Metal waste is collected by itinerant waste buyers equipped with cars and megaphones who purchase materials directly from households before it enters into the formal solid waste management system. Metal waste collected by small-size buyers in the northern area is sold to larger-size industries and transported mainly to Ramallah or Eizarya. Then from the Haifa or Akka harbours it is shipped for international export. Although in minimum quantities, metal waste from the northern West Bank is also brought to Jericho, where an important local industry transports it to Jordan, where it is recycled. Metal waste collected in the Bethlehem and Hebron governorates, instead, is transported to a large industrial pole situated in the city of Idhna, in the south-west area of the Hebron governorate. From Idhna, metal waste is traded with Israeli buyers and, similarly for the Nablus area, it is brought to the Haifa and Akka harbours. In order to increase its value, metal waste is segregated manually by type. Frequently, metal waste is separated from plastic by fire, with negative consequences on the environment and human health.

Plastic waste is collected by several categories of stakeholders in the West Bank, and it is the only waste recycled in the territory. In the West Bank, in fact, several plastic waste recycling factories are distributed mainly in the north (Nablus) and in the south (Hebron). Such factories mainly recycle HDPE, PVC and PP with archaic and unsafe procedures. Waste usually is delivered to such industries mixed by type and colour, and as flakes. Thus the quality of the final products is expected to be low. PET is rarely collected, instead, as it is exclusively traded with Israeli industries, who implement waste producer responsibility policies for PET bottles. Plastic waste is mainly segregated by households and sold to middlemen, or collected by waste scavengers on dump sites. Due to the very low revenues achievable from the trade of such waste, large enterprises and JSCs do not invest in trading it.

Cardboard waste recovery is widespread mainly in the northern and central part of the West Bank. No cardboard recycling industries are available in the territory, and this waste is exclusively traded with Israeli buyers, who also establish the price of the material. Cardboard waste is collected by informal organised scavengers and through waste recovery initiatives developed by JSCs or municipalities in collaboration with private enterprises or local organisations. Cardboard waste is usually not compacted, as the revenues obtained by the trade would not cover the purchasing and running costs of waste balers.

Only one material recovery facility is available. This facility is situated at Al Minya landfill and, at the moment, works at half of its treatment capacity (150 t/d versus 300 t/d capacity). The plant, equipped only with a trommel and a manual sorting line, mainly segregates cardboard. A second material recovery facility has been installed at the Zharat Al Finjan landfill (Jenin governorate) in 2012. The facility, managed through an agreement between private enterprises and public authorities, stopped its activities in 2014 due to political reasons. The plant, was equipped with a trommel, several manual sorting lines, shredders and balers.

Another waste recovery facility was installed by a private industry within a transfer station situated in the Nablus municipality. The plant, which was supposed to be run through a public-private partnership, never started its activities due to political and organisational reasons.

Other material recovery facilities are planned to be designed in the West Bank, and in particular within the area of the Jericho landfill and within the area of the Ramun landfill (which is still not constructed).

Organic waste treatment facilities, comprising only composting, are distributed throughout the West Bank territory, although there is a higher number in the Jenin governorate located north of the city. Existing facilities have been constructed and/or equipped through the economic support of international organisations. Two new composting plants are managed by local JSCs with the supervision of a Palestinian NGO (House of Water and Environment), and Jericho and Baytillu composting plants.

3.2.3 Role of local and international organisations in improving solid waste management

The West Bank region has enjoyed, currently as in the past, many interventions carried out by local and international organisations. Solid waste management projects, as well as studies on solid waste management systems, have been carried out with the scope of improving the environmental and sanitation performances of the territory. The majority of the interventions, however, failed due to the lack of a rigorous legislative framework, the scarce preparation of solid waste managers and lack of awareness in the population.

The NGO Nexus, in the years 2010/2011 tried to replicate Modena's separate waste collection method in a municipality of the Jenin governorate. The project had the main scope of segregating household food waste and composting it at a local cooperative, which, through the project, is equipped with machines for compost shredding and packaging. They also enjoyed several training sessions. The project, however, failed due to the lack of awareness of the households involved in the waste segregation project, who delivered very contaminated organic fractions, and due to the costs derived from material treatment. Similarly, between 2014 and 2017 Cesvi implemented a project in the north-west of the West Bank with the objective of diverting the organic market waste and the cardboard waste from final disposal and closing and remediating a municipal dump site. Through the project, with the collaboration of the University of Brescia, Cesvi successfully completed the remediation process of a dumpsite and strengthened the interest of the local JSC for the cardboard recycling programme, but failed in establishing a sustainable organic waste collection system. Commercial activities interrupted the segregation of organic waste as soon as the direct input from the NGO finished, and the local JSC did not pursue the objectives of the project.

JICA is a major organisation in the country, active in solid waste management projects, which, however, focuses its activities on the building capacity of local institutions and provision of solid waste management equipment. JICA, however, in its new operative strategy, is expecting to implement solid waste prevention, recovery and recycling programmes which also includes the population.

3.3 Discussion

Even though the per capita MSW generation was higher for the Region of Umbria, the amount of organic fraction generated was very similar for the two areas (about 180 kg/per capita/year) (Table 3). The number of specific facilities able to process this waste was practically absent in the West Bank. Only agricultural organic waste is currently composted in the entire West Bank territory through the static pile method. In some cases, especially, in Jenin, composting activities are performed only during the summer season. Nonetheless many initiatives have failed due to organisational problems and insufficient revenue for the farmers.

The number of mechanical sorting facilities per inhabitant able to manage unsorted waste was about ten times less for the West Bank compared to the Region of Umbria. Furthermore, the West Bank had a total absence of composting facilities. The lack of separated collection of the organic fraction is one of the main reasons, but also economic sustainability is another important aspect to be considered.

The MSWM fee payed by users, mainly citizens in the Region of Umbria was about 0.82% of the per capita GDP. For the West Bank area, this figure was higher, about 1.2%. As reported by Al-Khatib et al. (2010, 2007) even though the incidence on the per capita GDP was higher, the fee was not enough to cover all the MSWM costs. This leads

TABLE 3: Comparison of the main indicators for the Region of Umbria and the West-Bank-Palestine.

Indicator	Region of Umbria	West Bank
MSW per kg/capita/year	519	285
Organic fraction kg/per capita/year	180	178
MSWM fee/per capita GDP (%)	0.82	1.20 *
Mechanical treatment facilities per 10^5 inhabitants	0.44	0.034
Composting facilities per 10^5 inhabitants	0.56	0
Recycling (%)	32 [a]	6
Composting (%)	22.5 [a]	0

** Estimated on the basis of Al-Khatib et al. (2010); a=at recycling/composting facility gate.*

to the absence of separated collection and of successive proper management and treatment phases as already described above.

The lack of infrastructures, financial support and political willingness leads the West Bank to a very limited percentage of recycled waste and to practically no composting of the organic fraction.

These results pointed out that the correct management of the organic fraction of MSW is a main challenge for the MSWM in the West Bank but, also that the current absence of an adequate funding program suggests the promotion of home composting or animal feeding recycling activities. Promotion of home composting can also be an effective approach for reducing and/or avoiding collection needs and related costs. This can also be an opportunity for the municipalities to use their budgets for improving the separation at source of other recyclables such as plastics and paper. In any case the success of such activities cannot be maximized without an adequate level of awareness of citizens and of decision makers together with a strong political willing. In particular, the weakness of this last aspect can frustrate all the initiatives promoted in this sector by international agencies and NGOs.

The percentage of recycling and composting in the Region of Umbria and reported in Table 2 was related to the amount of waste delivered at the recycling and composting plants. This means that these figures are not fully representative of the effective fraction of recycling and composting. The current system implemented for monitoring this aspect is not able to give adequate information about the recycling efficiencies for each region and for specific areas. Furthermore, national data (ISRPA, 2016) indicated that for some materials such as metals, glass and paper, the recycling efficiency of waste delivered to recycling facilities was up to 90%, but for other materials such as plastics the recycling efficiency drops to less than 50%. Similar considerations can also be made for the composting activity. In general, the amount of organic fertilizer (Table 1) generated after composting treatment can also be less than 15% of the amount of organic fraction at the facility inlet. This is a consequence of the processing loss (e.g. humidity and degraded organic matter), but also a consequence of the level of impurities that needs to be removed due to the low quality of the collection phase. All these considerations lead to future discussion about the effective sustainability of given MSWM practices currently implemented in the Region of Umbria and at the EU level.

4. CONCLUSIONS

Lack of adequate infrastructures, economic budget and citizen awareness are the main causes for the low level of efficiency of waste management in developing countries, especially in the West Bank. Furthermore, the absence of a fully implemented market for the recycling industry limits the investment and the interest of the private sector. The main results pointed out that the economic viability of a more efficient waste management approach for the West Bank is difficult to be pursued due to the already quite high incidence of the fees payed when compared to the per capita GDP.

Due to the high incidence of the organic fraction on the whole amount of waste generated in the West Bank, the promotion of home composting and/or recycling via animal feeding seems to be the best recommendation. This can contribute significantly to improving waste management in this area. Both environmental/sanitary and economic benefits could be achieved by this practice. The reduction in the amount of waste to be collected can increase the budget of municipalities for improving the collection of other recyclable fractions. It is important to note that this goal is difficult to achieve without an adequate level of citizen awareness and political support.

Although there is a higher efficiency and reliable political, legal and economic supporting scheme in the Region of Umbria, some critical aspects related to its effective sustainability were detected. The high percentage of separated collection did not generally correspond to an effective high recycling rate, particularly for plastics and the organic fractiona. This opens the floor to a discussion about the level of effectiveness of current waste management practices in the Region of Umbria, in particular, and in the EU in general. The low efficiencies found for the effective recycling of some relevant waste materials in these areas suggests that other environmental and economically sound management schemes should be investigated.

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THE PRESENT AND FUTURE OF ALTERNATIVE WASTE TREATMENT APPROACHES IN THE UNITED STATES: TONNAGE AND TRENDS

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ABSTRACT

Based on a series of surveys conducted by the author and her firm over the past twenty-five years, the paper documents trends in alternative pre-disposal municipal waste treatment approaches in the United States, namely curbside recycling and energy from waste initiatives. Recycling tonnage and the corresponding percentage of the waste stream recycled has increased steadily since the 1990s, but has appeared to reach a plateau of about 28% currently. If one adds organics composting to the tonnage, the percentage rises to about 34%. While recycling of inorganic materials has encountered economic headwinds, curbside organics recycling programs appear to be increasing, leading to a rise in aerobic or anaerobic treatment programs in the future. Oppositely, conventional energy from waste initiatives have stalled in the United States. With the exception of one recently opened facility, no new plants have been built, although several have or are undergoing expansion. About 13% of the municipal waste stream is currently being processed at energy from waste plants. Within the next five years, this percentage could drop to about 10%, with numerous facility closings anticipated. In the future, it appears that the municipal waste stream in the United States will become increasingly disaggregated, resulting in a number of small to mid-sized processing plants handling these various streams. Nearly 50% of the waste stream will continue to be landfilled.

1. INTRODUCTION

The purpose of this paper is to examine trends in the pre-disposal treatment of waste in the United States, specifically as it relates to recycling, energy from waste, and the possible convergence of the two into an integrated strategy of the future. Drawing on a series of surveys conducted by the author, it will assess the direction of recycling and waste to energy in the United States. As of 2017, both national and international trends are impacting waste management in the United States. With adequate land available for landfilling waste in many regions of the country, the comparatively low price of landfilling, coupled with the low cost of energy and a volatile commodities market, there is little incentive for most localities to invest in capital intensive waste disposal alternatives. With the United States' decision to withdraw from the Paris Climate Accords, and the Trump Administration's lack of serious commitment to the mitigation of greenhouse gas emissions, there is not likely to be any major national policy initiatives to stimulate innovative waste reduction and energy conservation approaches. In the near future, states and localities will be taking the lead in implementing innovative waste management strategies.

In the United States, the federal government sets overall solid waste management policy, particularly in the regulatory arena, but it is left to states and localities to implement these regulations. There is large variation among states as to the level of commitment to alternative disposal methods. While curbside recycling has become the norm in almost all U.S. communities, most of the remaining waste in the U.S is landfilled. As shown in the latest U.S. Environmental Protection Agency (USEPA) report, 25.7% of municipal solid waste is recycled, 8.9% is composted, 12.8% is combusted with energy recovery, and the remaining 52.6% is landfilled. In 2000 the corresponding percentages were 23.0% recycled, 7.1% composted, 13.8% combusted with energy recovery and 56% landfilled. Thus, over the past 15 years, there have been some gains in the percentage of waste recycled and composted, but a decrease in the proportion of waste going to waste to energy. Corresponding-

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ly, reliance on landfilling has decreased by about 3% over the fifteen-year period. While the USEPA has promulgated the waste hierarchy of re-use, reduce, recycle, energy recovery, landfilling, there are no national directives compelling states or localities to implement the hierarchy in any particular way.

The remainder of the paper will focus on waste treatment prior to ultimate disposal. It will delve further into the numbers with respect to post consumer recycling and waste to energy. The general finding is that in the United States, recycling rates have reached a plateau due to various challenges confronting the industry. Reliance on energy from waste facilities has been declining in the last five years and will continue to do so over the next five-year period. Existing plants are facing the multiple challenges of age, low energy prices, stable landfill prices and lack of government policies to support the industry through subsidized energy pricing or other programs. Growth in alternative disposal methods will mainly occur in the treatment of food waste and other organics, since a number of localities are implementing source separated organics (food and plant waste) collection and other food waste reduction programs.

A mention should be made of additional energy from waste initiatives occurring on solid waste landfills throughout the United States, by which landfill gas is collected, cleaned and converted to electricity or used directly as a medium BTU fuel directly in boilers or as a high BTU pipeline quality gas. According to the United States Environmental Protection Agency, as of January 2018 there are 632 operational landfill gas recovery projects in the United States, with 40 more either under construction or planned. The majority of these projects generate electricity, 74%, with 19% creating a medium BTU gas for direct use and 6% producing a high BTU pipeline quality gas. A few projects are also generating a synfuel for use in vehicles. In total, these landfills are producing about 2200 MW of electricity. Some federal tax credits were available to assist these projects, but they have expired as of December 2016. Because the paper focuses on waste treatment or diversion prior to landfill, landfill gas to energy, while a highly successful means of generating energy from waste will not be a topic of this discussion going forward.

2. METHODOLOGY

The data in the paper is obtained from a series of surveys undertaken by the author and staff through her firm Governmental Advisory Associates Inc. Beginning in the 1980s, Governmental Advisory Associates Inc. conducts periodic surveys of waste to energy and recycling facilities in the United States. A detailed questionnaire has been developed and is periodically administered by telephone to plant operators, public officials, and private firms which own the facilities. The surveys cover technical aspects of the plant, types of equipment, types and quantities of materials processed, as well as the financial and contractual arrangements regarding capital and operating costs, waste input and product sales. In addition, information is garnered from state and local government reports, including financial audits of facilities, government budgets and annual operating reports.

Annual operating reports from plants, state and local reports, municipal or district council minutes, white papers, budgets, consultant reports have been stored for each plant. Each detailed questionnaire with notes are also stored for observation or review at Governmental Advisory Associates, Inc. Westport Connecticut.

3. RECYCLING

3.1 Changing market forces impacting U.S. recycling facilities

Curbside recycling has become the norm in nearly all metropolitan centers in the United States. Even in rural or semi-rural areas, most citizens have access to recycling drop-off containers or a drop-off center. In the residential sector, single stream curbside collection is the predominant form of collection. Residents place their post-consumer fiber and recyclable metal, glass and plastic in a single receptacle without further sorting. The materials are transported to centralized materials recovery facility (MRF) for processing and distribution to markets. In addition, many localities have extended recycling collection programs to the multi-family, commercial and institutional sectors. While the U.S. has seen the expansion of recycling, it is also experiencing challenges to this system. Single stream recycling has broadened the array of materials accepted in the curbside bin and increased the quantity recycled, but it has also placed technological and financial strains on sorting facilities. Residual rates have increased at the same time that markets are demanding a high level of sorting accuracy and product quality. Attaining quality requirements necessitates investments in capital equipment and labor. However, the end markets for much of the recycled product are volatile and often not robust enough to support processing costs. Thus, plants and local users must find methods to share the economic risks of a recycling program, creating budget stress on local government decision-makers and trimming profit margins of participating private firms.

The changes in the international and national environment over the last decade have had substantial and dramatic impacts on the U.S. recycling industry. The years of the Great Recession (2008-2010) battered the world economy, resulting in depressed commodity prices and lower than average waste and recycling volumes. Other economic forces have also worked to disrupt the recycling industry. The oil market has a direct impact on plastic production cost. When oil prices are high, recycled plastic is an attractive substitute for virgin plastics. As prices fall, virgin plastic surpasses recycled plastic as an input. Oil prices in the United States, while plunging in 2009, began rising steadily after 2010 through 2014. By 2013 the price of oil had recovered from the recession, trading in the range of $96 per barrel only to begin falling again in 2014. By 2016 the price of oil had plunged to $48 due to a slowdown in Asian economic growth and demand, a strengthening U.S. dollar, and the increased production of shale oil in North America. As of 2017, prices have remained weak. Reflecting these changes, the average revenue for recyclable containers fell from a high of about $160 per ton in 2012 to $66 per ton in 2015. Weak revenues from plastics acts as a drag on prices for other recyclable containers, pulling down overall recycling revenues, forcing MRF operators and local governments to re-evaluate their recycling programs.

In addition, the Asian markets for recyclables are becoming more discerning and careful about products they are importing. In February 2013, China implemented Operation Green Fence followed by National Sword in 2016-2017 to ensure that only quality plastics and paper were shipped from U.S. MRFs to be used by Chinese companies. Customs checks have been placed on imported materials, with the major focus on plastics and electronics. Chinese inspectors have been sent to U.S. container ports and large processing facilities to monitor shipments. To meet standards, U. S. sorting facilities have invested in upgraded machinery and quality control measures. Product purity has increased, but sorting costs have also increased. Plastics and paper exports have been affected. There may be additional bans of other materials such as scrap metal, in order to build up the domestic Chinese materials recycling industry, should China fully implement the bans it is exploring.

Furthermore, shifts in consumer habits as well as the evolution of packaging is reconfiguring the recycling stream. The amount of newsprint, once a mainstay of recycling programs has declined sharply, as people move to internet-based news. Many MRFs are no longer baling newsprint and are shipping only mixed paper bales. Oppositely, there is an increasing amount of old corrugated cardboard in the stream as consumers abandon brick-and-mortar stores, relying on internet sites for their purchases. Light weighting of packaging has decreased amounts of tin and aluminum and increased the reliance on plastics of various types. Plastics are more difficult to sort and depending on the variety of plastic grades in the stream, require additional labor or capital or both.

U.S. recycling facilities are becoming increasingly automated, with the widespread adoption of optical sorters, ballistic separators and, in a few instances, robotic sorters. Nevertheless, certain materials, create issues with sensitive machinery. Glass if not properly handled can cause problems on the sort line, as can plastic bags and multi-resin plastic containers. Recycled glass requires a regional or local market. Its relatively low market value and heavy weight make it economically infeasible to ship long distances. The result is that various curbside recycling programs are eliminating glass. Similarly, some programs are prohibiting plastic bags and other types of hard to recycle plastics from the recycling bin. Moisture and other contaminants can impact the fiber sort, leading to increased residuals. The average residual rate for single stream facilities is in the range of 17 to 25% of total incoming materials.

In response to these challenges, MRF operators are being forced to re-negotiate contracts with their customers or re-write new contracts in order to share market risks. When commodity prices were high, MRF operators were able to pay a premium for incoming recyclables and tolerated a broader range of materials with variations in quality. In the current economic environment, operators are being forced to charge a tipping fee to cover their costs, sharing revenues with customers only if market prices for recyclables go above a certain threshold. Faced with climbing residue rates, some MRFs and localities are deciding to cut back on materials accepted in the curbside program, add additional quality control personnel, and educate residents as to the precise materials which belong in the recycling bin.

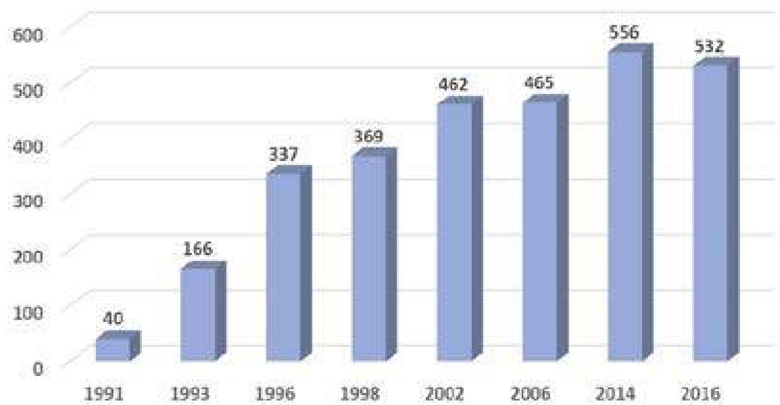
In part as a result of these world economic trends as well as developments in the national solid waste sector, the U. S. recycling industry has been experiencing the same consolidation sweeping many industries, from banking and telecommunications to health care. Some MRFs have closed due to poor economics, market saturation, antitrust considerations, or the elimination of service. Others have stopped processing and have been re-configured as transportation centers, where materials are baled and shipped to larger, regional processing plants. Despite these economic hurdles, the industry continues to look to the future. There has been ongoing innovation in sorting technology with improvements in the speed and accuracy of sorting and automated feedback systems to spot and in some instances self-correct problems on the processing lines. Robotic technology has been introduced into MRFs, further automating sorting functions. There is a drive to continue the extension of recycling into the construction, commercial and food waste sector. Source separated food and yard waste collection has been implemented in many localities on the West Coast and is being piloted in various communities across the country. In some instances, a convergence of recycling and waste to energy is taking place as localities are looking to use the organic fraction of the waste stream as feedstock for gasification or other energy producing plants.

3.2 Status of recycling in the United States

The move to widespread municipal recycling in the United States coincided with the growth of environmental awareness that began in the 1960s. Citizen activism and concern over polluted rivers, air, land soiled by unregulated landfills “dumps”, and the overuse of dangerous pesticides documented by Rachel Carson’s seminal work The Silent Spring published in 1962, culminated in the creation of the U.S. Environmental Protection Agency (USEPA) by President Nixon in 1970. It was formed, in part, to implement the National Environmental Policy Act (NEPA), passed in 1969 to establish national environmental goals, conduct research on the extent of various types of pollution and means to curb them, and issue grants to states and localities to curb pollution. From 1970 to 1974 a number of national policies and regulations were put in place to arrest environmental damage and preserve and conserve environmental resources. Through federally mandated solid waste management plans, states began to encourage recycling and energy from waste as a means to reduce waste and conserve resources. Furthermore, the USEPA began a decades long initiative to close sub-standard municipal waste “dumps”. Through the 1980s, municipal recycling was focused on five major materials: newsprint, corrugated cardboard, tin cans, aluminum beverage containers, glass food and beverage containers. While there were some curbside collection programs, most recycling consisted of public areas where such materials could be dropped off. Non-governmental organizations often conducted newspaper or can collection drives to augment their charitable activities. However, by the 1990s, as the federal and state governments increased their focus on resource conservation and waste diversion from landfills, curbside collection of recyclables became popular and spread throughout the country. Plastics became a prominent part of the recycling bin. Processing facilities began to be built to sort the materials being collected from households and business. Figure 1 shows the number of such materials processing facilities through 2016. The dip in 2016 is due to closures as well as consolidation across the industry.

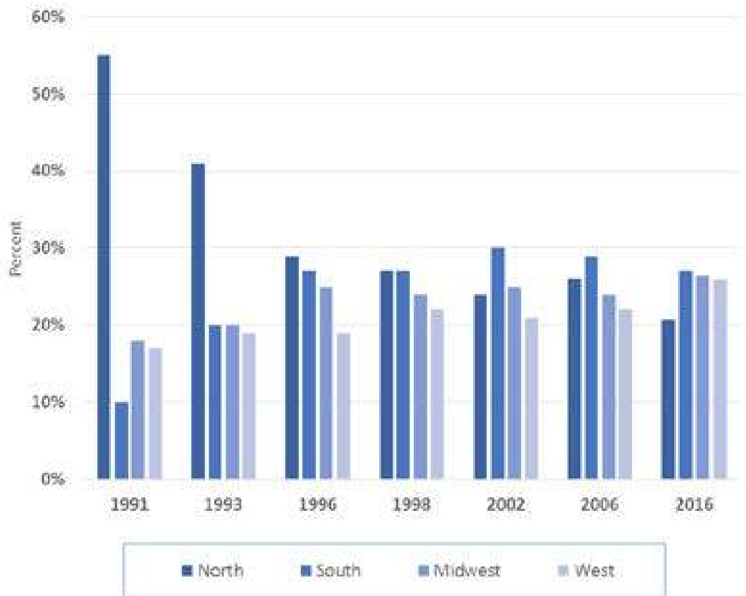
The northeast region of the United States, which encompasses the New England states of Vermont, New Hampshire, Maine, Massachusetts, Rhode Island and Connecticut, New York, Pennsylvania, New Jersey, Delaware was one of the first areas to adopt curbside recycling and construct processing facilities. However, as shown in Figure 2, by the late 1990s, processing facilities and the curbside programs they serviced were distributed relatively evenly over all regions in the United States.

The early curbside programs required residents and businesses to pre-sort their recyclables into various containers. Usually there were separate bins for newspaper, cardboard, tin and aluminum cans, glass, and later plastic. Over the years due to economic pressures and technological innovation, the level of pre-sorting of recyclables decreased and the range of materials accepted for recycling increased. Currently, in most localities, citizens do not have to place each type of material into separate bins. Rather they have adopted single stream collection and processing. Residents place all recyclables, fiber, metal, plastic and glass in a single container. The result has been an increase in recyclable tonnage both per facility and in total across



Source: Governmental Advisory Associates Inc. Westport CT

FIGURE 1: Number of materials recovery facilities in the United States.



Source: Governmental Advisory Associates Inc. Westport CT

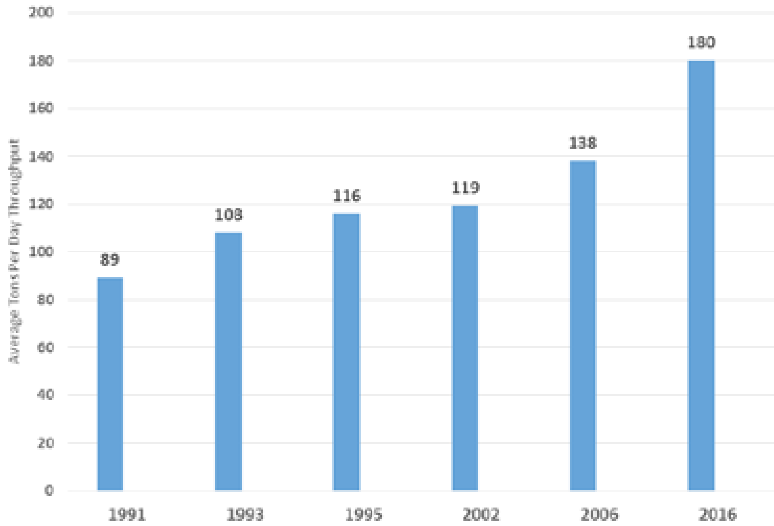
FIGURE 2: Distribution of materials processing facilities by region over time.

all facilities. Figures 3 and 4 indicate this growth in tons processed per facility as well as total municipal solid waste recycled tonnage processed annually. In 2017, 70% of the multi-material processing facilities in the United State rely on single stream recyclables for their input stream. This compares to 27% of such facilities a decade ago and 15% in 2000.

The implementation of single stream collection has forced recycling facilities to invest in upgraded sorting technology to handle the mixed stream. Smaller facilities have been forced out of business as processing plants have become regionalized. The average capital costs to construct a recycling facility have more than doubled from $6,000,000 in 2006 to $15 million in 2016. Sophisticated screening technology, intricate digital controls, optical sorters and in a few instances robotic sorters have contributed to the cost. Nevertheless, as mentioned above, material reject percentages have also increased from an average of about 6% for a facility, where fiber and containers were collected separately to an average of 17%-32% for a single stream sorting plant. Much of the residual percentage is composed of unmarketable glass and mixed plastics.

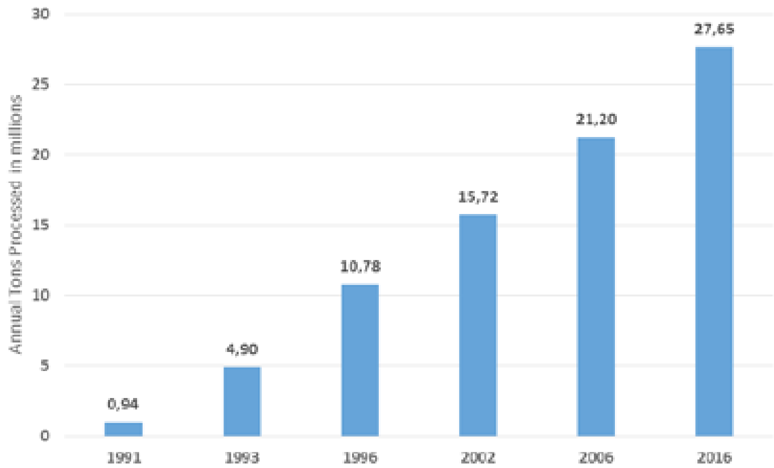
3.3 Future recycling trends and tonnage

According the USEPA, recycling rates in the United States have held relatively steady over the last five years. While the election of President Trump in 2016 has injected an element of uncertainty over the direction of national environmental policy and created some potential state and local budget concerns, several developments indicate that the recycling percentage may increase. Certain states continue to forge ahead with innovative and forceful waste management approaches. California had initially mandated that 50% of waste must be diverted from landfill, through source reduction, recycling and composting by 2000. In 2012 the state passed AB341 mandating com-



Source: Governmental Advisory Associates Inc. Westport CT

FIGURE 3: Average throughput per processing facility-tons per day.



Source: Governmental Advisory Associates Inc. Westport CT

FIGURE 4: Total tons processed annually at materials processing facilities (millions of tons).

mercial recycling and moving the waste diversion goal to 75% solid waste diversion by 2020. Oregon passed a new recycling law in 2015, updating recycling goals for its localities to 52% by 2020 and 55% by 2025. More specifically, it set recovery goals for food and plastic at 25% by 2020. In 2012, Vermont enacted Act 148, Universal Recycling and Composting Law, which bans designated recyclables from landfills as of 2015. By 2020 all food scraps generated by residents will be banned from landfills. In 2014 Minnesota expanded its recycling requirements to cover all commercial establishments producing a certain threshold of trash in the seven-county metro area. The recycling goal for the area was increased from 60 to 75%. As of October 2014, Massachusetts has mandated that large food waste generators must separate food waste to be sent to a composting, animal feed, or waste conversion plant. It is imposing a statewide goal of 30% trash reduction by 2020. The nation's largest city, New York is implementing separate curbside collection of organic waste and is mandating separate food waste collection from large food generators. As of July 2016, all large-scale food generators must have a separate organics collection. By 2018, all New Yorkers will have separate organics curbside collection or access to a convenient drop off site.

If source separated organics collection continues to be adopted by states and municipalities, U.S. recycling rates would jump substantially. The food waste stream makes up 14.9% or 38.4 million tons of the 258 million tons of municipal solid waste generated in 2014. Currently only 1.96 million tons or 5.1% of total food waste generated is collected for composting or other treatment. If that percentage were increased to 25% in the next five years, the overall recycling rate would increase to about 41%, landfilling would fall to 49% of total waste generated, with about 5 million additional food waste tons being diverted from landfill.

A third possible development which may impact recycling in the future is the implementation of mixed waste processing plants. These facilities process a fully mixed solid waste stream, including organics and inorganics. Under this model, citizens no longer do any sorting. All disposal items are thrown into one bin and organics and other recyclables are separated at the plant. There are currently 47 of such facilities, most of which are located in California. Five additional mixed waste plants are being planned in California, handling mainly commercial waste and a few are being planned in the Middle- Atlantic region. In Minnesota, energy-from-waste facilities are planning or have added front end materials sorting capacity, to separate high value materials prior to combustion. With the advance of sorting technologies, including the ability to sort organic from inorganic waste, such plants become technologically feasible. Some of the west coast mixed waste plants are being planned with an adjacent anaerobic digestion facility. The challenges facing these types of plants are mainly economic and institutional. The initial capital costs are high, and revenues from material and energy sales may not cover the operational costs. In fact, a promising plant constructed in Alabama was forced to close after only a few months of operation, due not to technical failure, but economic issues related to lower than planned commodity prices. The $30 million plant, developed by a private firm, was recently purchased by the City of Montgomery for $625,000. In addition, even if the economics work, there is opposition to this type of plant from environmental and recycling groups. There is a fear that under this model, citizens no longer will need to pay attention to what they discard, diminishing the concept of waste reduction. After receiving a one million dollar grant from a non-profit organization to examine the feasibility of such a plant, the city of Houston Texas had to abandon the idea. Citizen opposition was such that it did not proceed. Nevertheless, if this type of plant were to be built in parts of the United States with low levels of recycling participation, it might boost recycling tonnage and landfill diversion.

Finally, most relevant to future trends in recycling is the re-imagining of waste stream management that is currently occurring. The USEPA through its Sustainable Materials Management (SMM) Program is moving away from a focus on disposal of unwanted materials to the appropriate handling and marketing of the various material streams that compose the waste stream . Its three strategic priorities are 1) focusing on sustainable building through use of environmentally sensitive materials; 2) developing sustainable food management initiatives through supporting alternatives to the landfill disposal of waste food and encouraging methods to reduce food waste; 3) Continuing to support sustainable packaging through improved product design, life cycle analysis. As localities move to different types of collection systems, such as source separated organics collection, they are re-thinking their materials processing infrastructure. Some communities are moving to a two-bin collection system comprised of organics in one bin and inorganics in the other. The organics are sent to an anaerobic digester or composting facility for the production of energy or compost material. The inorganics go to a processing facility, where valuable materials are separated. Residuals may go to landfill or to an energy from waste plant. Other communities are adopting a three-bin system, where organics and soiled paper are placed in one bin, non-contaminated inorganic recyclable materials in a second bin, with the remaining discards placed in the third bin. Should such systems take hold across the country, one could anticipate and major increase in recycling tonnage, energy from waste facilities, as well as a major reduction in waste going to landfill. These types of systems create a natural synergy between recycling and energy from waste.

4. ENERGY FROM WASTE

4.1 Market forces impacting energy from waste in the United States

The shifts occurring in the re-thinking of waste management approaches are having a distinct impact on energy from waste initiatives in the United States. With the potential disaggregation of the waste stream into component categories, organic, inorganic recyclables, residuals, there is new focus on gasification technologies using the organic or residual stream. As the waste stream becomes more segmented, any new facility may have to be scaled down from those existing energy from waste facilities that are combusting a less segmented stream. As of 2014, energy from waste facilities are processing about 12.8% of municipal waste generated in the United States. This percentage reached a high of 14% in 2000 and has hovered in the 12% range since that time. An average energy from waste plant, handling municipal solid waste combusts about 1100 tons per day, producing about 28 megawatts of electricity. The growth of the energy from waste initiative in the United States grew out of the turbulent 1970s, driven in part by the Middle East oil embargo and the birth of the environmental movement. In the midst of soaring oil prices, the federal government began to encourage alternative energy projects, including energy from waste plants. Various financial and tax policies were enacted to stimulate the development of such facilities. Under the 1978 Public Utility Regulatory Policies Act, which sought to promote energy conservation and use of renewable energy, power utilities were required to purchase electricity from qualifying facilities (generating under 80MW of power) that used waste, biomass, or other renewable fuels. Rates paid to QFs were to be equal to the "avoided cost" to the utility, defined as the incremental energy and capacity cost the utility would have incurred but for the purchase from the qualifying facility. With the high cost of oil during that period of time and fuel shortages projected into the future, waste to energy facilities were able to enter into long term, 20 to 25-year power purchase agreements with utilities at advantageous rates.

Furthermore, during this same period the country was turning its focus to cleaning up the environment and preventing further environmental degradation through air, water, and land pollution. As cited previously, the United States through the newly formed Environmental Protection Agency and its predecessor departments mandated the closure of sub-standard landfills throughout the country. Numerous facilities shut down, driving up landfill costs and forcing state and local officials to look at alternatives. The USEPA assisted in these efforts, providing technical assistance and grants to localities looking to procure waste to energy plants or implement other types of resource and energy conservation programs. Given the favorable regulatory and policy environment through the mid-1980s, states and localities implemented the construction of energy from waste plants. By 1990, 127 of these plants had begun operations with another 63 in the planning stages.

In the 1990s, the U.S. EPA turned its attention from encouraging the development of energy from waste plants to regulating the potential harmful air pollutant effects of such plants. Of particular concern were the carcinogenic effects of dioxins and furans emitted during the combustion process, the toxicity of incinerator ash, and the monitoring and testing of these impacts. By 1995, the U.S EPA had promulgated stringent new air emission standards, requiring energy from waste facilities to install maximum available control technology (MACT) to control for particulate emissions, dioxins, furans, nitrous oxide, sulfur dioxide, heavy metals and other harmful pollutants. These standards are to be revisited every five years. Emissions standards for certain substances continue to be adjusted downward as new control technologies have been developed. These regulations forced many plants to make costly upgrades to their air pollution control and management systems.

Additional policy changes impacting energy from waste initiatives were occurring during this period. A national tax reform package enacted in 1986, eliminated favorable future tax incentives for investment in energy from waste plants. Also, by the 1990s, the energy supply picture had begun to change. The U.S utility industry was substantially de-regulated. New sources of oil were found and utilities turned to alternative fossil fuels such as coal and natural gas. Counter to earlier predictions, energy prices began to fall. Individual state utility commissions charged with setting the avoided cost rates at which energy from waste facilities could sell their electricity moved to a competitive bidding method. Due to decreasing energy prices, as energy from waste facilities renewed their power sales contracts, their electricity revenues fell dramatically. In many cases, energy from waste plants began to sell power on the open market, without the benefit of a long-term, stable, above market power sales agreement.

Just as the price of energy failed to continue its predicted rise, a similar development occurred with solid waste disposal prices. Beginning in the 1970s, and continuing through the 2000s, the number of municipal solid waste landfills dropped from approximately 10,000 to 1900. The modern landfill of today is strictly regulated by federal, state, local governments for leachate control, liner construction and methane gas control. With the decline in landfill numbers, it was expected that the shrinking disposal capacity, would cause landfill disposal prices would rise. Prices did more than double from 1980 through 1995 to $50.00/ton; however, in the mid-1990s landfill prices began a slow decline, leveling off to about in the $48-$50/ton range. (2014 dollars). Landfills that met federal standards were able to expand and new large landfills opened. When they were first constructed in the 1980s and 1990s, energy from waste facilities were anticipating stable, subsidized electricity prices and rising waste disposal fees. However, with a largely de-regulated energy and waste disposal market, these energy from waste plants have been forced to keep their disposal fees competitive, placing additional downward pressure on their revenues.

Adding to downward pressure on energy from waste revenues, the Supreme Court of the United States in its decision of C&A Carbone v. Town of Clarkstown, struck down flow control, the power of a locality to direct all waste generated within its confines to a specific facility. The court held that flow control violated the freedom of interstate commerce, since the plaintiffs were forced to use a waste disposal facility within the town that was more costly than alternative facilities out of state. Energy from waste facilities relied on flow control to ensure that they had an adequate waste flow at set disposal prices. This decision was modified with a later decision that permitted flow control if the disposal facility was publicly owned and operated. However the overall impact of these court rulings was that many energy-from-waste projects were forced to decrease their disposal tip fees, as long term waste delivery contracts between municipalities and plants expired.

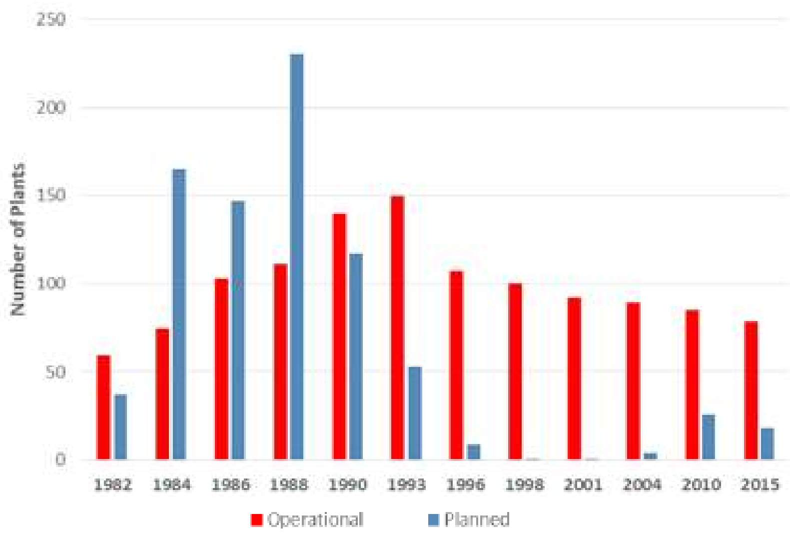
With a single exception, all 78 energy from waste plants currently operating in the United States have been built in the 1980s or early 1990s. Many have been substantially upgraded and can continue to operate into the future, but others are reaching the end of their operational life. Their economic future is further muddied by various long term contractual arrangements that are expiring, both for the sale of their energy product as well as for the incoming waste. Facilities are being forced to compete in the waste disposal market, with inherent limits as to how much they can charge for tip fees. With prevailing landfill rates in the range of $50.00 per ton, it is difficult for plants to charge rates above those prevailing in their area. The challenging economic picture is exacerbated by continuing low energy and recycled metal prices. Since very few states are offering electric rate subsidies based on the use of waste as a renewable fuel, plants are confronting declining or flat energy revenues. Furthermore, certain states such as California, New York, New Jersey have placed a moratorium on the construction of new waste to energy plants using combustion or have limited these plants' access to renewable energy credits. While waste generation rates have held steady, the segmentation of waste streams to food waste and recyclables is diverting materials from existing energy from waste plants. Many have excess capacity, which adds to their uncertain economic future.

4.2 Current status of energy from waste plants in the United States

Reliance on conventional municipal waste combustion to energy is declining in the United States due to the economic and political challenges cited above. While in a few places plants have expanded, only a single facility in Palm Beach County, Florida has been newly built since 1995. Figure 5 shows the number of facilities in operation and planned by year since 1982. The growth and maturation of the industry can be clearly seen. Through 1988, the number of planned facilities outstripped the number of operating plants, while the number of operating plants also grew steadily. The years 1990 to 1993 were a turning point. The number of operating plants peaked and the number of planned facilities dropped drastically. In 1993 there were 150 operating energy from waste plants in the United States. By 2015, that number had been nearly halved to 78. Also by 1996, planning for new facilities had essentially stopped. However, as of 2010, there has been some change in the direction. In conjunction with source separated organics collections, communities have begun to examine the feasibility of anaerobic digestion. A few of these plants are being constructed. In addition, various types of waste gasification or other conversion plants for certain waste streams are being developed. These are largely small pilot projects, depending on a segregated organic or residual plastics waste stream. There are currently no planned conventional waste combustion plants, relying on an unsegregated municipal waste stream.

While numbers of plants have declined, total tonnage processed by energy from waste projects has held steady over the last decade. Many of the first wave of closures in 1993 were in specific reaction to the newly promulgated air pollution control regulations. Smaller or older facilities did not have the financial strength to invest in the necessary air pollution control systems to meet the new standards. Tonnage processed grew through 1995, when energy from waste processed about 32 million tons of waste or about 14.5% of the municipal waste stream. Since 2006 total tons processed has hovered around 30 million tons annually. As of 2015, this represents about 11.6% of total tons of waste generated.

Energy-from-waste plants are located mainly in the northeastern and southern regions of the United States. Northeastern states with their dense population centers and high landfill prices were early adopters of energy from waste technologies. In the south, Florida embraced the concept of energy from waste, looking to divert waste from landfills. Figure 6 shows the distribution of plants by region over time. What is most striking is that by 2016, nearly two thirds of existing plants are located in the northeast



Source: Governmental Advisory Associates Inc. Westport CT

FIGURE 5: Number of operational and planned energy from waste plants by year.

or the south. Due to the existence of less costly landfills with large capacities, the West and Midwest regions of the country have largely moved away from conventional energy-from-waste as a disposal alternative. Figures 7 and 8 reflect the challenging revenue environment confronted by energy from waste plants. Average tip fees charged by energy from waste plants, while increasing steadily to about $92.00 per ton in 1994 ($2016) began to decline after that point reaching their current average of about $61.00 per ton. Similarly, electricity revenues have also declined from a high of 10.31 cents per kilowatt-hour in 1989 ($2016) to about 6.60 cents per kilowatt-hour in 2016. Without any subsidies on electricity pricing under renewable portfolio standards or other renewable energy incentives, or any policies or regulations that might significantly drive up the cost of landfilling, energy from waste plants face an uncertain economic future in many parts of the United States. In addition, the high initial capital investment of $300,000 per design ton and average operating costs of $99.00/ton, inclusive of debt service, make economic feasibility problematic for any new plant that might be developed in most regions of the country. To the extent that there is downward pressure due to declining waste flows or declining prices, the facility has to compensate by raising tip fees. This is challenging in a competitive environment.

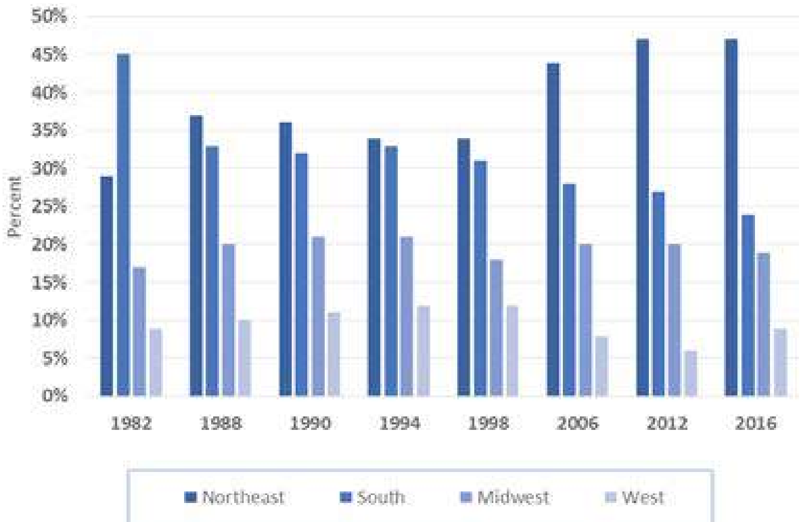
4.3 Future Energy-from-Waste trends and tonnage

The projections for conventional waste to energy plants in the near future are not favorable. No new plants are planned due to declining waste quantities, high capital costs, citizen opposition, and siting issues. Of the 78 operating plants, 16 may close within the next five years, representing about 4.3 million tons of annual throughput. Oppositely, three plants in Lee County, Florida, Pasco County, Florida, and Kent County Michigan are planning expansions, and other plants are anticipating increased throughput. The net loss of energy from waste capacity within the next five years is expected to be about 3.1 million tons. Total waste processed annually from conventional waste to energy plants will total about 27 million tons, dropping to 10% of the municipal solid waste generated in the United States, rather than the 12.8% it is today.

While conventional energy-from-waste through combustion is declining in importance as a waste management alternative, gasification and anaerobic digestion plants appear to be the wave of the future. Gasification technology is viewed as a means to capture energy from waste without the toxic impacts of air emissions control and ash disposal that characterize waste combustion. Gasification facilities can be modular, operating at lower tonnage levels, to be scaled up to meet increased demand. Furthermore, gasification in theory achieves greater thermal efficiencies than combustion, resulting in higher energy production per input ton of waste than conventional waste to energy plants.

The drawback of waste gasification or anaerobic digestion is that it requires a high level of pre-sorting of waste to ensure that the resultant waste stream is of sufficient quality to be treated. Moving from bench scale to commercial operation of such plants has proved challenging. One of the first waste to bio-fuel plants to operate at commercial scale recently opened in Alberta Canada. The facility is designed to handle 100,000 metric tons annually of post recycled, pre-sorted waste. It has been producing methanol from the non-organic fraction of the waste that is sorted and sized to create a refuse derived fuel. Methanol production has been at lower levels than anticipated and there have been delays in moving to the production of ethanol, due to problems with the pre-sorting of the waste. Nevertheless, similar projects are being planned in Montreal Canada and Rotterdam in Holland. Other countries such as France and Japan have been operating gasification plants for several years.

Similarly, in conjunction with source separated organic collection programs that are being adopted in various states, local governments are looking to anaerobic diges-



Source: Governmental Advisory Associates Inc. Westport CT

FIGURE 6: Distribution of energy from waste facilities by region over time.

Source: Governmental Advisory Associates Inc. Westport CT

FIGURE 7: Average per ton tip fees charged by energy from waste facilities by year in 2016 dollars.

Source: Governmental Advisory Associates Inc. Westport CT

FIGURE 8: Average electricity rates obtained by energy from waste facilities in cents per kilowatt-hour by year in 2016 dollars.

tion to treat the organic waste stream. Most of these plants also require extensive pre-sorting. With the production of biogas and digestate, anaerobic digestion creates value from waste, without harmful emissions and a large amount of residual by-product, which requires landfilling. While such facilities are common in parts of Europe, there are only 15 plants in the U.S. solely dedicated to commercial and residential food. Improper waste sorting, difficulties in securing long term waste supply contracts and low energy revenues have made these plants difficult to finance and operate.

5. CONCLUSIONS

The United States is experiencing a paradigm shift in waste management. It is moving away from waste management as a disposal problem to waste management as a materials flow issue. The waste stream is being disaggregated into its component parts, i.e. organic, inorganic (recyclable, non-recyclable), residual, with various forms of treatment proposed for each stream. The overall goal is to decrease greenhouse gas emissions to the extent possible, to reduce waste and to reuse and recycle at maximum levels. Land disposal is no longer viewed as a waste disposal option, but the final resting place of low value residuals from various treatment and sorting processes. Under this paradigm, the bifurcation between recycling and energy from waste is blurring or disappears completely. Different levels of sorting will be required for each stream, whether such sorting occurs at the point of generation (residence or business) or at a centralized sorting and processing facility. Based on the sorting technology that is being developed, it may be more efficient to end curbside collection of separated streams. Separation and processing could occur at a fully automated plant, after which materials could be sent to an anaerobic digester, compost facility, gasification or other energy from waste plant, or directly to end markets.

Various factors cloud this future picture. There is no national policy or systems of incentives to encourage state and localities to aggressively move to higher rates of re-use or recycling or waste conversion to energy. Policies and regulation vary by state and in some instances by locality. Electricity generated from combustion of solid waste does not qualify for renewable energy credits in many states. In other states, it qualifies for a substantially reduced subsidy. Waste gasification projects are given more beneficial treatment in most states, but levels of subsidy also vary by state. Certain states, such as California, Minnesota, Vermont, Massachusetts have implemented aggressive policies to meet landfill diversion and recycling goals, whereas other states have implemented less stringent policies.

In many areas of the United States, landfills remain the cheapest and most available disposal option. At disposal rates of \$25-\$35 per ton in areas of the Midwest or West, it is difficult for a local government to make the case to invest scarce public funds in alternative disposal options. In fact, due to the financial uncertainty that local governments face, some have dropped curbside recycling programs entirely and others have scaled back their program to cover only those materials with stable markets. It may be that sorting and waste treatment technology is currently outstripping economic feasibility in the United States. Optical sorters, sophisticated screens, computerized feedback loops, robotic sorters achieve efficiencies, but require high levels of throughput and maintenance. The result may be a high- quality end product with low value or quantity. Aggregating the various materials stream for processing at a centralized facility may achieve the necessary throughput to support a highly automated plant, but may result in a contaminated feedstock that degrades market price.

When one looks at waste management as a sustainable materials management strategy, there is a level of instability built into the approach. Waste continues to be generated at a given level, but materials markets are highly volatile. Revenues from the sale of materials or energy are not easily predicted, which makes budgeting within a local government difficult. While private companies often operate in the environment of commodity price swings, through future markets and other mechanisms, such behavior is not typical for a local government. Even if the entire waste management operation is privatized, once a private entity begins to lose money, there is no guarantee that the company will remain in business. Local government managers must take a conservative approach, since it ultimately their responsibility that waste be collected and disposed in a safe manner. In the near future it is states and localities that will serve as laboratories for future waste management strategies.

The data presented provides opportunities for continued research, which were beyond the scope of this paper. Are oil prices solid predictors of average recycling container revenues and if so, what kind of hedging strategies can localities and firms develop to protect themselves from market volatility. Similarly, do commodity prices drive recycling levels or is the implementation of recycling and other waste management alternatives driven by other factors beyond the materials market.

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WASTE MANAGEMENT IN POST-SOVIET COUNTRIES: HOW FAR FROM THE EU?

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ABSTRACT
After the collapse of the Soviet system, every new independent state selected its own way of development, own goals and speed of transformation. Dramatic changes were linked not only to the political and economic sphere, but also to environmental governance as a whole and waste management in particular. 25 years later the authors looked at 6 post-soviet countries and analysed the situation in the sector of municipal solid waste management by comparing this with EU member states (some of them have a socialistic past). We used BiPRO approach (BiPRO, 2012) and looked for answers related to the question: how far developed is the current state of waste management in post-soviet countries compared to EU members? Which factors define the potential efficiency of waste management system and its full conformity with the situation in "old" EU member states? The overall scores of 6 post-soviet countries range from 2 (Georgia) to 11 (Belarus). The common reasons for these low scores in all mentioned countries are weak waste management policies, and landfilling as a main way of waste disposal, the lack of economic instruments for stimulating reducing of waste generation and recycling, as well as underdeveloped infrastructure for waste treatment facilities. Specific problems for post-soviet countries are, for example, the high share of landfilled biodegradable waste, incomplete coverage of waste collection systems, the lack of forecasting of waste quantities and planning of waste management, preserved obsolete soviet approach to tariff policy, statistical accounting and administrative procedures in the sector of waste management. The improvement of waste management systems should aim at the legislative ban on the disposal of municipal solid waste at landfills, the re-establishment of a separate waste collection system (disestablished after USSR collapse), the establishment of economic and financial mechanisms supporting the waste processing sector and stimulating the population to reduce waste generation.

1. INTRODUCTION

The issue of municipal solid waste management is an urgent problem of urban management and environmental governance in the countries with different level of social and economic development. Constant growth of consumption goes along with an increase of waste generation all over the world. The strategic goals of waste management are becoming recycling, minimization and waste avoidance. The main challenge of the environmental governance is municipal solid waste management (MSWM) linked to the quality of waste collection, removing and recycling, as well as the efficiency of the institutions for waste management.

The geographical focus of the paper is on post-soviet countries. After the collapse of the Soviet system, every new independent state selected its own way of development. Dramatic changes were linked not only to the political and economic sphere, but also to the environmental governance as a whole and waste management in particular. The speed of transformation was quite different in different countries: some of them transformed fast and dramatically (Russia and Ukraine), some of them saved a lot of societ performances of waste management system (Belarus), others had middle speed of transformation (Kazakhstan and Moldova), and Georgia has changed the goal of transformation drastically. In present post-soviet countries have different GDP, incomes and economic growth (table 1). The speed of the transformation, as well as level of

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TABLE 1: General information about analysed countries*

	Belarus	Russia	Kazakhstan	Ukraine	Moldova	Georgia
Total area	207,600 km^2	17,098,242 km^2	2,724,900 km^2	603,550 km^2	33,851 km^2	69,700 km^2
Population	9,549,747 (2017)	142,257,519 (2017)	18,556,698 (2017)	44,033,874 (2017)	3,474,121 (2017)	4,926,330 (2017)
Urbanization	77.4% (2017)	74.2% (2017)	53.2 % (2017)	70.1% (2017)	45.2% (2017)	54% (2017)
GDP (purchasing power parity)	$175.9 billion (2017)	$4 trillion (2017)	$474.3 billion (2017)	$366.4 billion (2017)	$20.07 billion (2017)	$39.32 billion (2017)
GDP - real growth rate	0.7% (2017)	1.8% (2017)	3.3% (2017)	2% (2017)	4% (2017)	4% (2017)
GDP - per capita (PPP)	$18,600 (2017)	$27,900 (2017)	$26,100 (2017)	$8,700 (2017)	$5,700 (2017)	$10,600 (2017)
Population below poverty line	5.7% (2016)	13.3% (2015)	2.7% (2015)	24.1% (2010)	20.8% (2013)	9.2% (2010)

** Data from web-site Index Mundi https://www.indexmundi.com/*

social and economic development, was a reason to choose the following 6 countries for analysis: Belarus, Russia, Kazakhstan, Ukraine, Moldova and Georgia. Some current data about mentioned countries is represented in table 1. Mentioned countries have different square and population, and very different GDP. At the same time, all of them have middle level of GDP per capita and similar real growth rate (excluding Belarus). Three of them (Belarus, Russia and Ukraine) have high level of urbanization (more than 70 %), and Kazakhstan, Moldova and Georgia have a middle level of the urbanization (45 – 54 %). Moreover, all of them chosen different goals of the development: Moldova and Ukraine try to integrate fully with EU, Belarus, Russia and Kazakhstan are developing a strong economic and political partnership (The Eurasian Economic Community), Georgia provides own independent policy. So, analysed countries are characterazied by diverse social, economic and political conditions at present times, but have common soviet past, that why the assessment of the MSWM systems could be interesting for the identification of driving factors and effective tools of the waste policy implementation.

The waste generation in total and waste generation per capita in analysed countries are presented in table 2. The main characteristics of the MSWM system in the mentioned countries are (1) landfilling as a main method of waste management; (2) tariff policy based on the "normative of waste generation" for the waste collection and removing per capita; (3) significant over-use of the equipment; (4) under-development of recycling capacities; (5) littering of urban areas; (6) development of the informal and illegal sector for collection and treatment of recyclables. In spite of common issues in the waste management sector, every analysed state has own specifics and features of the MSWM system.

The main goal of the research was to analyse the current state and level of development of the MSWM system in 6 post-soviet countries, identify strong and weak points of national waste policy, and to compare results with EU countries. Comparisons with EU countries could be useful for identifying the efficiency of national MSWM systems, analyzing more sufficient instruments and tools of MSW management, driving factors of waste policy implementation.We assume that analysing and comparing post-soviet countries with each other and EU members could allow identifying implementation gaps and improve national waste policy and MSWM system performances.

2. METHODS AND MATERIALS

The research is based on the BiPRO approach developed under the EU project "Support to Member States in improving waste management based on assessment of Member States' performance", project number 070307/2011/606502/SER/C2. The final report on screening of waste management performance of EU member states was published in 2012 (BiPRO, 2012).

The list of the criteria was developed based on the LD 99/31/EC and WFD 2008/98/EC. The set of criteria is reflecting the main elements and legal requirements stemming from the Directives in the field of waste management. Criteria were divided on 5 groups: (1) compliance with the waste management hierarchy reflecting the real situation; (2) existence and application of legal and economic instruments to support waste management according to the waste hierarchy; (3) existence and quality of an adequate network of treatment facilities and future planning for municipal waste management; (4) fulfilment of the targets for diversion of biodegradable municipal waste from landfills and (5) number of infringement procedures and court cases concerning non-compliance with the EU waste legislation. For each from 16 criteria two, one or zero points could be achieved according with the table in ANNEX 1. Overall

TABLE 2: Waste generation in analysed countries in 2014

	Belarus	Russia	Kazakhstan	Ukraine	Moldova	Georgia
Waste generation, mln t	4	56,68	3,5	9,2	0,7	no data
Waste generation kg per capita	421,7	385,6	200	215,7	199,3	no data

score was received as a sum of all criteria score. Individual criteria points were defined empirically based on the data observation in BiPRO (2012). In current paper the meaning of the points were saved for better understanding the situation in MSW management in post-soviet countries in compare with EU. The initial data for the assessment was collected from available statistical data, analytical reports, and reviews for the period 2010-2014, as well as from the analysis of national regulative and normative documents. The list of used sources for the assessment is represented in ANNEX 2. The fifth group of criteria was not assessed (explanations in ANNEX 1), and the overall scores of the EU countries from (BiPRO, 2012) were re-calculated without the mentioned criteria group. Overall score was received as a sum of all criteria score.

3. MUNICIPAL SOLID WASTE SYSTEM IN ANALYSED POST-SOVIET COUNTRIES

Main performances of the MSWM system in the analysed post-soviet countries are represented in table 3. The assessment according to BiPRO approach and the interpretation of the physical performances of MSW system are represented in table 4. The scoring, sources and way of the calculation could be found in the ANNEX 1.

In all mentioned countries the waste generation is increasing on the background of the growth of consumption (NSC RB, 2017; Sycheva & Asadcheva, 2013; MEP Kz, 2015; SSS U, 2016; NBS RM, 2016). In Georgia data on waste generation and treatment are not collected systematically. The constatnt growth of waste generation is a common problem of all analysed countries and reflects a global trend of overconsumption and waste generation. The problem of outstripping growth of waste generation over consumption is typical for EU countries also, including leaders in the treatment of municipal solid waste. Only in such countries as Austria, the Netherlands, Denmark and Luxembourg the growth of MSW is the only indicator that is equal to zero amid significant progress in all other areas of improving the waste management system (BiPRO, 2012).

The waste quantity per capita in analysed countries differs from about 200 kg in Ukraine, Moldova and Kazakhstan to about 400 kg in Belarus and Russia (table 2). There is no data on waste per capita in Georgia. We can't say, that mentioned figures on waste generation per capita reflect the real situation adequately. The common issue for analysed 6 countries is the lack of accurate estimations of the total waste generation and waste per capita due to specifics of statistic recording. Statistic recording takes into account only the amount of collected and removed waste by special enterprises; there is no 100-% coverage of waste collection system in all overviewed countries (especially in the rural areas); there is a lack of official data and assessment of waste flows in the informal and illegal sector. Moreover, in some cases data from local level are not transmitted correctly to the national level and may contain significant mismatching (see, for example SSS U, 2016 and MRDCH U, 2015).

Almost all MSW is landfilled in post-soviet countries: up to 100 % in Georgia and Moldova, 94 % in Kazakhstan and Ukraine, about 90 % in Russia and about 80 % in Belarus (table 3). The level of recycling in Ukraine, Russia and Kazakhstan is less than 8 %, and in Belarus is about 20 % (table 3). In the Republic of Moldova, the data on the volume of recycled waste is not under statistical monitoring. The data on the material recycling in Georgia is not available in open sources. There are a few incineration plants in Belarus, Ukraine and Russia built for energy production, but their capacity is not enough to play a significant role in the MSW treatment: according to statistic data the level of energy recovery is about 1-3 % (table 3). Kazakhstan is only planning to construct incineration plants. The widespread use of landfilling links, first of all, to very low fee for waste disposal, especially in comparison with recycling or energy recovery. The payment for removing MSW is less than 35 €/t in all analysed countries (table 3). The low tariffs are a legacy of old soviet approach to the payment for removing and treatment of solid waste. The approach is based on the "normative of waste generation per capita" and established tariffs for communal services. The growth of the service costs is based, as a rule, on the artificial increasing mentioned "normative per capita" because the tariffs on com-

TABLE 3: Performances of the waste management system of analysed countries*

	1.1	1.2	1.3	1.4	1.5	1.6	2.1	2.2	2.3	3.1	3.2	3.3	3.4	3.5	4.1	4.2
Criteria Countries	Decoupling indicator	WPP in place	% Re-cyc-ling	% re-cove-ry	% dis-pos-al	% recycling	Ban/ Restrictions	€/t	PAYT	% coverage	WMP	WMP	WMP	% compliane	% target	% biodegrad.
Belarus	coupling	Yes	19	1	80	20	Restrictions	9	No	85	Yes	No data	Yes	76	No	No data
Russia	decoupling	Yes	7	3	90	3	Restric-ti-ons	Less 35	No	No data	Yes	Yes	Yes	8	No	No data
Kzakhstan	coupling	No	6	0	94	4	Restric-ti-ons	Less 35	No	Less 50	Yes	Yes	Yes	6	No	No data
Ukraine	decoupling	Yes	<3	<3	94	3	Restric-ti-ons	2	No	77	un-der-ca-pacity	No	No data	less 75	No	No data
Moldova	decoupling	Yes	No data	no data	up to 100	No data	No	12	No	Less 60	un-der-ca-pacity	Yes	Yes	0	No	No data
Georgia	NA	Yes	No data	0	up to 100	No data	No	No data	No	Less 70	un-der-ca-pacity	No data	No data	less 75	No data	No data

** Conducted by authors as a result of the analysis documents, statistical data and analytical report (see ANNEX 2).*

TABLE 4: The results of the assessment of the MSWM system in post-soviet countries.

Indicator	Belarus	Russia	Kazakhstan	Ukraine	Moldova	Georgia
1 Compliance with the waste management hierarchy reflecting the real situation						
Criterion 1.1: Level of decoupling	1	0	1	0	0	N/A
Criterion 1.2: Existence of own waste prevention programme	2	2	0	2	2	2
Criterion 1.3: Amount of municipal waste recycled	1	0	0	0	0	0
Criterion 1.4: Amount of municipal waste recovered (energy recovery)	1	1	0	1	0	0
Criterion 1.5: Amount of municipal waste disposed	0	0	0	0	0	0
Criterion 1.6: Development of municipal waste recycling	2	1	1	1	0	0
2 Existence and application of legal and economic instruments to support waste management according to the waste hierarchy						
Criterion 2.1: Existence of nationwide ban/restrictions for the disposal of municipal waste into landfills	1	1	1	0	0	0
Criterion 2.2: Total typical charge for the disposal of municipal waste in a landfill	0	0	0	0	0	0
Criterion 2.3: Existence of pay-as-you-throw (PAYT) systems for municipal waste	0	0	0	0	0	0
3 Existence and quality of an adequate network of treatment facilities and future planning for municipal waste management						
Criterion 3.1: Collection coverage for municipal waste	0	0	0	0	0	0
Criterion 3.2: Available treatment capacity for municipal waste in line with the EU waste legislation	1	0	0	0	0	0
Criterion 3.3: Forecast of municipal waste generation and treatment capacity in the WMP	0	1	1	0	1	0
Criterion 3.4: Existence and quality of projection of municipal waste generation and treatment in the WMP	1	1	1	0	1	0
Criterion 3.5: Compliance of existing landfills for non-hazardous waste with the Landfill Directive	1	0	0	0	0	0
4 Fulfillment of the targets for diversion of biodegradable municipal waste from landfills						
Criterion 4.1: Fulfillment of the targets of the Landfill Directive related to biodegradable municipal waste going to landfills	0	0	0	0	0	0
Criterion 4.2: Rate of biodegradable municipal waste going to landfills	0	0	0	0	0	0
Overall score	**11**	**7**	**5**	**4**	**4**	**2**

munal services are socially sensitive component (especially in the situation of low incomes and significant share of poors in the country) and their increasing is regulated by the national governments. Such conditions do not allow developing recycling or energy recovery effectevly, and moreover, the implementation of the PAYT systems is not profitable for service providers under existing tariff policy. It is no surprise that PAYT systems are not implemented in the analysed countries, and there is no ban on landfilling.

Many landfills do not meet modern environmental requirements or do not have all necessary documents and permissions. For example, in Russia only 8% of MSW landfills meet environmental requirements (IFC's the World Bank Group, 2010); 90 % of existing landfills are operated without a license (Ecoportal, 2015); in Kazakhstan there are 4284 landfills and dumps: and only 459 from this number meet environmental requirements and sanitary standards and are provided with all necessary documentation (MEP Kz (2015). In the field of landfilling next typical discrepancies are (on the example of Kazakhstan, MEP Kz (2015): 1) the lack of synthetic or clay liners at the majority of the waste disposal sites; 2) widespread disposal of MSW together with industrial, medical and others types of toxic and hazardous waste; 3) unsystematical compaction and interleaving of the stored waste with isolated layer (clay) or the lack of it; 4) the lack of system for collection of leachate and landfill gases (including methane); 5) excessive usage of many landfills and dumps which exceed their capacity; 6) lack of monitoring; 7) discrepancy of requirement of sanitary rules and sanitary protection zone. In Ukraine, municipal solid waste landfills are a source of contamination of the surrounding rural areas: as a result of their operation may deteriorate the sanitary state of soils, the quality of groundwater and air (Makarenko, Budak, 2017).

Current regulations for design, construction and operation of landfills as well as their enforcement significantly differ from the EU Landfill Directive. The national requirements are not comparable with EU regulations, that why the final score for this criterion is very low in all analyzed countries.

In all analyzed countries the capacity for MSW treatment and recycling is underdeveloped and the list of recycling technologies is short. For example, according to (Cleandex, 2010), there were 39 waste sorting plants in operation (beginning of 2010) in Russia. Their average capacity is about 180 000 tons per year, which is comparable with the amount of waste generated in a small town (IFC's the World Bank Group, 2010). Recycling plants in Russia, Kazakhstan, Ukraine, Moldova, Georgia are private, in Belarus they belong to state. Recycling plants in mentioned countries meet similar problems (on the example of Belarus, Lysuho & Eroshina; 2011): (1) high cost of recycling products with relatively low quality; (2) poor quality of the waste for recycling due to the lack of effective waste sorting; (3) the prevalence of manual labor with involving marginal groups, (4) the competition with illegal recycling sector. In spite on noted problems, the recycling sector is fast developing in all analyzed countries. Its growth is particularly impressive in Belarus, where for the last five years the capacity of recycling plants has increased by almost 20 %. In Ukraine there is a huge recycling potential, waste treatment is provided both in formal and informal way. There are lots of companies dealing with waste recycling in Ukraine but with no official monitoring, accounting and control. Therefore, it could be observed the lack of statistical data in open sources. That was the reason of low scoring for Ukraine.

Biodegradable waste is not a point for MSW management in the analyzed countries. The generation, landfilling or treatment of the biodegradable waste is not controlled. Moreover, there is not definition of such kind of the waste in the national legislations (see documents in ANNEX 2). There is a lack of reliable statistical data on the biodegradable waste in the countries, that is why this criteria has score "0" in the overall scoring. Almost all biodegradable waste is landfilled in all analyzed countries. The share of the biodegradable waste varies from the place of their generation: its share is much larger in the multi-story apartments; and such kind of waste is practically not met in the waste from private households where biodegradable waste is traditionally used for composting or incineration (NSC RB, 2017; Sycheva & Asadcheva, 2013; MEP Kz, 2015; SSS U, 2016; NBS RM, 2016).

It should be noted that the system of the collection of "food waste" was established in the USSR. The "food waste" was collected at the multi-story apartments and then transported to the livestock breeding complexes for animal fattening. After the USSR collapse this system was destroyed due to reasons of hygienic and sanitary safety as well as due to changes in animal fattening technologies. The revival of such system for "food waste", of course in the modernized form adapted to modern conditions, could be greatly improved the MSWM system and decreased the share of the landfilling biodegradable waste.

Economic instruments for MSWM regulation are underdeveloped in all overviewed countries. For example, in Russia it was recognized the special value of public-private partnership for the implementation of major infrastructure projects and programs. However, until now there was no even one integrated project united all components of MSW management (collection and removal, disposal, recycling, landfilling) at the level of urban agglomeration and / or the subject of the Federation (IFC's the World Bank Group, 2010). In Belarus under the President's Decree № 313 "On Some Issues of Consumer Waste Disposal", the procedure for implementation of EPR is established.

National programs, normative and regulative documents on MSW management are approved in Belarus (MHU RB 78, 2014), Ukraine (WMP U, 2004), Russia (MNRE RF 298, 2013), Moldova (NWMS RM, 2013). The National program of modernization of the MSWM system in Kazakhstan (MP Kz, 2014) was canceled in the September, 2016. It should be mentioned that approved national strategies on MSW management is one of the advantages of Belarus, Ukraine, Russia, Moldova and Georgia, since more than half of the EU members (17 States) do not have national documents on MSW management and use EU directives. From the other hand, as was pointed in report (BiPRO, 2012), approved national policy and legislative documents on MSW management do not guarantee an efficiency of MSWM system due to governance gaps and implementation deficits. All of these could be pointed in analysed countries: in spite of approved national strategies on MSW management, the situation with MSW was not radically changed (NSC RB, 2017; Sycheva & Asadcheva, 2013; MEP Kz, 2015; SSS U, 2016; NBS RM, 2016).

The weak component of the MSWM system in all countries is the forecasting and planning in the waste sector. As was already noted, the capacity of the recycling plants is underdeveloped. At the same time there is no clear strategy for developing of the recycling capacity due to the lack of the reliable assessment of the waste generation of different types as well as the forecasts of economically feasible recycling and extraction of the secondary raw materials (MHU RB 78, 2014; WMP U, 2004; MNRE RF 298, 2013; NWMS RM, 2013). Approved national strategies, programs and plans include, of course, elements of the forecasting and planning, but they are not detailed (ibid). In analyzed countries there are no established integrated plans of MSWM at the local level. As a result, it could be stated that the MSWM system in analyzed post-soviet countries is not effective.

4. COMPARISONS WITH EU COUNTRIES

The overall score of MSWM system in analyzed post-soviet countries is presented in Fig. 1 (analysed countries are showed by red bars). The results are corresponding with EU countries of the third group with the lowest score – Latvia, Cyprus, Romania, Lithuania, Malta, Bulgaria and Greece.

The analysis of the weakness of the MSWM systems in the EU countries of the third group highlighted the similar problems as in the analysed post-soviet states. The common features of the MSWM systems are (1) weak policy, especially with respect to the ban of the landfilling and reg-

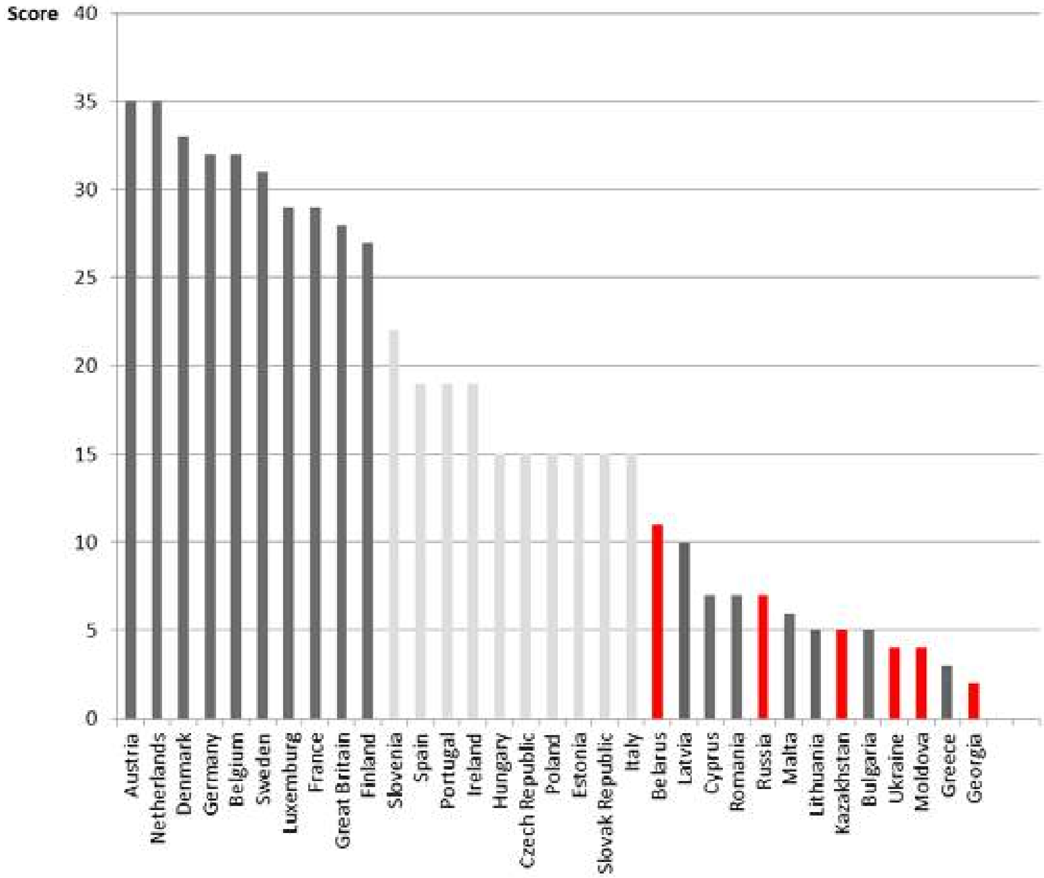

FIGURE 1: Comparative assessment of the municipal solid waste management system in European countries (drawn by authors based own research (red bars) and BiPRO, 2012).

ulation of the biodegradable waste treatment; (2) the lack of the economic instruments for stimulating the reducing the waste generation and recycling; (3) not 100-% coverage by the formal system of the waste collection and removing; (4) governance gaps and implementation deficits of local waste management plans and programs.

Despite attempts to transfer to waste-to-energy, landfilling is still a problem in Greece (81%) and Latvia (79%), Lithuania and Spain (reaching 55% each), where landfilling is regarded the cheapest option in terms of investment (Malinauskaite et al., 2017) as well as in post-soviet countries. Authors (Malinauskaite et al., 2017) suggest, that if the government introduces a high tax and landfilling fee, it may just be that it is more economically viable to reuse waste in order to produce energy than depositing it in landfills as the example of Estonia proves . It seems, that the increase of the landfilling fee could be one of the solution for the increasing efficiency of waste policy in analysed post-soviet countries. For example, evaluation of the implementation of the landfill tax has shown a correlation between tax rate dynamics and the reduction in amounts of waste disposed in Latvia and Estonia (Klavenieks, Blumberga, 2017). All countries from the first group with the most effective waste policy in EU (dark grey bars with the highest scores in Figure 1) have landfill fee more than 80 €/t (BiPRO, 2012), it looks as one more proof of the efficiency this economic instrument.

The landfill tax is not the only way to reduce waste disposal. As was mentioned in EEA (2007), the most important policy tools used to reduce waste disposal in landfills are landfill ban, separate collection systems of MSW, and deposit refund schemes as well as landfill tax. The second waste policy option for analysed countries is the landfill ban. If we look at the results of BiPRO assessment (BiPRO, 2012), we could find, that the most impressive results of the solid waste policy implementation were achieved in the countries with ban on MSW landfilling (Austria, Netherlands, Germany, Denmark, Belgium, Sweden, Luxemburg) in contrast with results of Latvia, Cyprus, Romania and Greece where there is no the ban on MSW landfilling. It should be mentioned, (based on the example of the Netherlands) not only the tax is essential, but also the availability of technological alternatives (Klavenieks, Blumberga, 2017). If the first group with the highest scores demonstrates “sufficient treatment capacity” (BiPRO, 2012), then the third group of EU countries (as well as post-soviet states) are “highly depending on landfilling, other treatment options are rarely in place” (ibid). Based on the experience of EU countries, we could conclude that the development of the sufficient treatment capacity is a key point for successful implementation of MSW policy.

The main governance gaps and implementation deficits of waste policy in post-soviet and EU countries are political issues (Likhacheva, Skryhan, Shkaruba, 2017; Malinauskaite et al., 2017). While waste management and prevention policies are defined in all countries, a further focus to consider waste as a source is lacking (Malinauskaite et al., 2017). The further improvment of waste policy should be linked to overcoming implementation deficits of the waste policy and articulating the goals of waste management system (for example, choosing the waste-to-energy or recycling strategy) and set up nessesary legal, economic and financial tools and instruments.

After post-soviet period some effective tools and instruments of MSW management got lost (for example, treatment of biodegradable waste). Further improving waste policy in analysed countries should focus on the re-establishment of some elements of the soviet waste management system.

Significant disadvantages of the assessed the MSWM system in the post-soviet states are the lack of reliable data on the amount and composition of the waste. The overall score for the post-soviet countries could have higher values, if the relevant statistic data would be available in acomparable form. The changes in the statistic accounting and reporting could be considered as a measure to increase the efficiency of the MSWM system. During post-soviet period the legislation was changed as well as statistic forms and data. These changes were not always successful. For example, in Russia the term "MSW" was included in the definition of the "consumption waste". The result is the lack of statistic data or extremely generalized and insufficient information about MSW. It is even more difficult to find and compile information about recyclables because the statistic data is not separated recyclables from consumption waste and recyclables from production waste (SP RF, 2014). In Ukraine there are two different official sources of information about collected, treated and disposed waste amount: State Statistics Service and Ministry of Regional Development, Construction and Housing and Communal Services. State Statistics Service registers household and similar waste (household and similar wastes - wastes produced in the process of people activity in the inhabited and uninhabited buildings (solid, bulky, repair, liquid, except waste associated with the production activities of enterprises) and that are not used in the place of their accumulation) while Ministry of Regional Development, Construction and Housing and Communal Services accounts municipal solid waste generated in households and entities. Additionally, some data on waste management which can be different from above mentioned are published in regional reports of the Ministry of Ecology and Natural Resources of Ukraine (SSS U, 2016; MRDCH U, 2015). The difficulties in data interpretation can influence on the decision-making process, forecasting of future tendencies etc.

5. CONCLUSIONS

The MSWM systems in post-soviet countries have low efficiency. Their efficiency level is comparable with EU countries of the third group – Latvia, Cyprus, Romania, Lithuania, Malta, Bulgaria and Greece. Essential shortcomings of the MSWM systems in analysed countries are: (1) insufficient legislation and regulation: the lack of the ban for landfilling, the lack of the regulation of the biodegradable waste, weak system of the forecasting and planning, outdated tariff policy and statistic accounting; (2) undeveloped capacity for recycling and treatment; (3) the lack of the effective economic instruments for the stimulating the recycling and reducing the waste generation.

During post-soviet period in analyzed countries the national strategies or other regulative documents on MSW management were developed and approved, but in general the MSWM system retains the list of soviet features (the service fees, the organization of the waste collection, removing, treatment and technic regulation). A number of effective soviet tools and practices have been lost (the collection system for recyclables, the collection of food waste, awareness raising activities, etc.). The establishment of the institutional instruments in the new social, economic and political conditions has not yet been completed, in consequence the governance gaps and implementation deficits can be observed.

BiPRO approach is based on the EU legislation and its aims, and obviously does not coincide with the objectives and legislation of the post-soviet countries. BiPRO aproach is usefull for brief screening and compare of MSWM systems in different countries, but it requires a list of quantitative data. Established forms statistical reporting in analysed post-soviet countries as well as open access to data do not allow to estiminate correctly the BiPRO criteria. So we can not be sure that the worse situation in the field of MSWM in Georgia, and in Belarus it is much better than that in other analysed cuntries. The further step for the research will be the development of a methodological approach based on waste policy goals and statistical reporting of post-soviet countries for adequate analysis of MSWM system.

AKNOWLEDGEMENTS

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ANNEX 1: Methodology of the assessment

Methodology was developed under implementation of the project "Support to Member States in improving waste management based on assessment of Member States' performance". The project aims at contributing to the improvement of the waste management practices in Member States in accordance with the principles of EU waste WFD (2008). Results of the assessment were represented in BiPRO (2012).

The set of criteria is reflecting the main elements and legal requirements stemming from the Directives in the field of waste management. All criteria were divided into 5 groups. The group 5 "Number of court cases or infringements concerning non-compliance with the EU waste legislation" was excluded from current assessment because analysed post-soviet countries are not a part of EU, that why EU legislation is not obligatory for countries and it is impossible to identify number of infringement procedures and court cases concerning non-compliance with the EU waste legislation. For each from 16 criteria two, one or zero points could be achieved according with the table below. Overall score was received as a sum of all criteria score.

ANNEX 2: Sources for the calculation and the assessment of the criterion

Belarus

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TABLE: Indicators and its way of the calculation

Indicator	Scoring	Way of calculation / source of data
1. Compliance with the waste management hierarchy reflecting the real situation		
Criterion 1.1 Level of decoupling of municipal waste generation from household final consumption expenditure	Reducing of Waste generation – 2, increasing of consumption is slower, than waste generation – 1, waste generation is equal to increasing of consumption– 0 All 27 MS will be ordered descending (highest decoupling rate first) 9 MS with highest rate: 2 /9 MS with medium rate: 1 /9 MS with lowest rate: 0	Calculation according to methodology and decoupling indicator EC (2011). Evolution of (bio-)waste generation/prevention and (bio-) waste prevention indicators, Annex F, chapters 7.4 and 7.14. In order to take into account decreasing driving forces the formula has been adapted as follows: = the decoupling indicator for a time interval of five years from y-5 to y = the slope of the linear regression of the waste generation (environmental pressure) over the last five years | EP expressed as an index with y-5 = 100 = the slope of the linear regression of the private consumption expenditure (driving force) over the last five years | DF expressed as an index with y-5 = 100 D>0: decoupling | D ~0: coupling | D<0: reverse decoupling Source: Source: national statistical yearbooks and reports (ANNEX 2)
Criterion 1.2: Existence of own waste prevention programme (WPP) or equivalent existence in WMP or other (environmental) programmes	Does a waste prevention programme exist? Does an equivalent exist in WMP or other (environmental) programmes? YES: 2 / NO: 0	Source: analysis of national normative and regulative documents (ANNEX 2)
Criterion 1.3: Amount of municipal waste recycled (material recycling and other forms of recycling including composting)	How much municipal waste is recycled in a particular year (in %)? >39 % :2, 19-39 %: 1, <19 % : 0 All 27 MS will be ordered descending (highest % of municipal waste recycling first) 9 MS with highest rate (above 39 %): 2 /9 MS with medium rate (between 19 % and 39%): 1 / 9 MS with lowest rate (below 19 %): 0. Weighting is applied for the criterion; for overall scoring the received score is doubled.	Source: national statistical yearbooks and reports (ANNEX 2)
Criterion 1.4: Amount of municipal waste recovered (energy recovery)	How much municipal waste is recovered (energy recovery) in a particular year (in %)? >17 % :2, 1-16 %: 1, <0 % : 0 All 27 MS will be ordered descending (highest % of municipal waste recovery first) 9 MS with highest rate (above 17 %): 2 /9 MS with medium rate (between 1 % and 16 %): 1 / 9 MS with lowest rate (below 1 %): 0	Source: national statistical yearbooks and reports (ANNEX 2)
Criterion 1.5: Amount of municipal waste disposed (deposit onto or into land and incinerated without energy recovery)	How much municipal waste was disposed of (deposit onto or into land and incinerated without energy recovery in a particular year in %)? < 49,5 % :2, 49,5-75 %: 1, >75 % : 0 All 27 MS will be ordered ascending (lowest % of MSW disposal first) 9 MS with lowest rate (below 49.5 %): 2 / 9 MS with medium rate (between 49.5 % and 75 %): 1 / 9 MS with highest rate (below 75 %): 0.	Source: national statistical yearbooks and reports (ANNEX 2)
Criterion 1.6: Development of municipal waste recycling (material recycling and other forms of recycling including composting)	What was the development of recycling of municipal waste during the last three years (in %)? Recycling rate increased min. 5 % or total rate is min. 40 % over the last three years: 2 Recycling rate increased over the last three years, but increasing rate is below 5 %: 1 Rate of recycling is decreasing or zero in last three years: 0	Source: national statistical yearbooks and reports (ANNEX 2)
2. Existence and application of economic instruments to support waste management according to the waste hierarchy		
Criterion 2.1: Existence of nationwide ban/restrictions for the disposal of municipal waste into landfills	Is a ban / are restrictions for the disposal of municipal waste applied? YES: 2 / Restrictions: 1 / NO: 0	Source: analysis of national normative and regulative documents (ANNEX 2)
Criterion 2.2: Total typical charge for the disposal of municipal waste in a landfill	How much is charged for landfilling municipal waste (€/t)? < 35: 0, 36-100: 1, > 100: 2 9 MS with highest rate (more 100 €/t): 2 /9 MS with medium rate (between 36-100 €/t): 1 /9 MS with lowest rate (less 35 €/t): 0	Source: analysis of national normative and regulative documents (ANNEX 2)
Criterion 2.3: Existence of pay-as-you-throw (PAYT) systems for municipal waste	Is a PAYT system for municipal waste in place? Yes, covering the whole territory: 2 / Yes, not covering all municipalities: 1 / No: 0 In case no information is available in the consulted reference document, a score of 0 applies.	Source: national statistical yearbooks and reports (ANNEX 2)

Indicator	Scoring	Way of calculation / source of data
3. Existence and quality of an adequate network of treatment facilities and future planning for municipal waste		
Criterion 3.1: Collection coverage for municipal waste	Is information about capacity available? / Does an under capacity exist? Under capacity: No: 2 / Partly 1 / Yes: 0 In case no information is available in the reference documents, a score of 0 applies.	Source: national statistical yearbooks and reports (ANNEX 2)
Criterion 3.2: Available treatment capacity for municipal waste in line with the EU waste legislation (including disposal and incineration)	Is information about capacity available? / Does an under capacity exist? Under capacity: No: 2 / Partly 1 / Yes: 0 In case no information is available in the reference documents, a score of 0 applies.	Source: national statistical yearbooks and reports (ANNEX 2)
Criterion 3.3: Forecast of municipal waste generation and treatment capacity in the WMP	Is under capacity to be expected according to information contained in the WMP? No: 2 / Partly 1 / Yes: 0 In case no information is available in the WMP, a score of 0 applies.	Source: national statistical yearbooks and reports (ANNEX 2)
Criterion 3.4: Existence and quality of projection of municipal waste generation and treatment in the WMP	Is information on the future development of municipal waste generation and treatment in the territory included in the WMP? Yes, in high quality: 2 / Yes: 1 / No: 0	Source: analysis of national normative and regulative documents (ANNEX 2)
Criterion 3.5: Compliance of existing landfills for non-hazardous waste with the Landfill Directive	Which percentage of landfills for non-hazardous waste is compliant with the requirements of the Landfill Directive (in %)? 100 %: 2 / at least 75 %: 1 / below 75 %: 0	Source: national statistical yearbooks and reports (ANNEX 2)
4. Fulfilment of the targets for diversion of biodegradable waste from landfills		
Criterion 4.1: Fulfillment of the targets of the Landfill Directive related to biodegradable municipal waste going to landfills	Is the first target on reducing biodegradable municipal waste disposed of in landfill reduced to at least 75 % fulfilled? Yes: 2 / No: 0	Source: national statistical yearbooks and reports (ANNEX 2)
Criterion 4.2: Rate of biodegradable municipal waste going to landfills	Rate of biodegradable municipal waste going to landfills: less 40 % - 2, 40-75 % - 1, more 75 % or the lack of data - 0	Source: national statistical yearbooks and reports (ANNEX 2)

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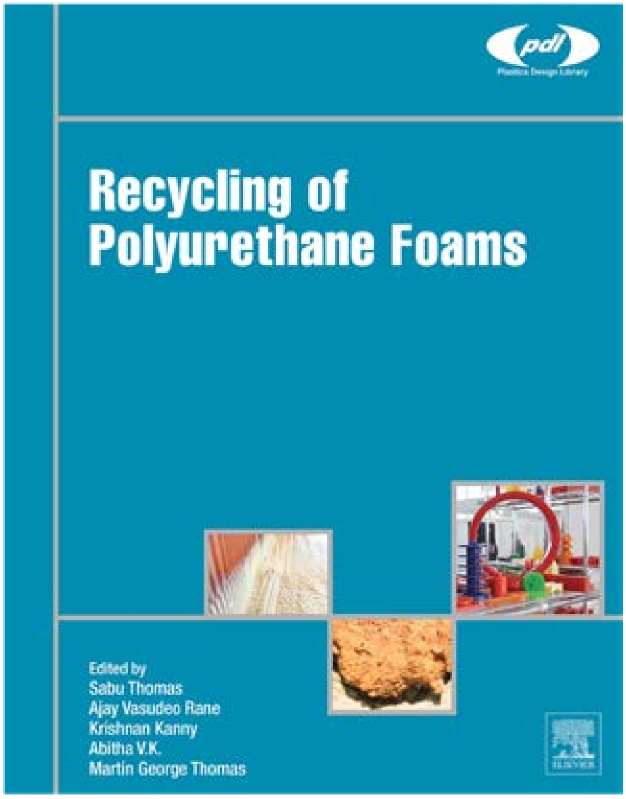

RECYCLING OF POLYURETHANE FOAMS

Edited by Sabu Thomas, Ajay Vasudeo Rane, Krishnan Kanny, V.K. Abitha and Martin George Thomas

Since its invention by Otto Bayer and his group in 1937, and following production with the purpose of coating aircrafts during World War II, Polyurethane (PU) foams have been used extensively in furniture, insulation panels, medical appliances, automotive interiors and in a number of consumer products for daily use. The impressive development of polymer and plastic industry results in billions of tons of polyurethane produced annually, which generates the need for the concurrent development of recycling strategies for waste products.

Recycling of Polyurethane foams, recently published by Elsevier within the PDL (Plastics Design Library) book series, encompasses 133 pages of essential information for students, researchers and practitioners interested in the fundamental processes and technologies in the PU recycling industry. The main idea behind the book, which consists of 10 chapters by different authors, is to provide comprehensive information on all aspects of the cycle of PU products, from manufacturing to end use, recycling, thermal treatment and landfill disposal.

The book starts with a broad introduction to polymers, including its classification and its history. The descriptions of different types of plastics and their application follows, with an overview of traditional recycling technologies and the challenges and opportunities for improvement.

The focus then moves onto PU foams, with a thorough description of the chemistry behind the production and of the specific properties at the base of the global success of PU foams. Recycling concepts are introduced and details are given in the following chapters about mechanical and chemical recycling methods. In particular, mechanical treatment is described, from the reduction of PU waste scrap into particles (regrinding) to the addition of binders for rebonding, adhesive pressing and compression or injection molding. Three full chapters are devoted to the comprehensive description of chemical treatment methods which allow depolymerization to occur and monomer production for further use in production processes. The chemistry and reaction schemes behind glycolysis, hydrolysis, ammonolysis and aminolysis of PU foams are thoroughly presented and discussed. Combined methods are introduced and their potential to reduce drawbacks is described, underlining the need for further innovation and the limits of current approaches.

State of the art technologies are reported, along with recently patented processes involving different chemicals and based on new concepts limiting the production of undesired compounds during recycling, as results from the analysis of most recent literature.

A thorough comparative assessment of Life Cycle Analysis studies of PU foam wastes is carried out, providing insights into the improvement of the environmental performance of PU foams thanks to the replacement of traditional blowing agents with new ones, with negligible global warming potential.

The last chapter focuses on advances in construction applications of PU foam wastes, including the use of triturated PU waste for the production of coating materials, modified bitumen and PU-based adhesives, providing not only for a reduction of production costs but also improving properties such as thermal conductivity, durability and long term behavior in comparison with traditional products.

Overall, this book offers a collection of excellent contributions covering all aspects of the life cycle of PU foams, structured in a convincing way with numerous links between the chapters. The result is an essential manual which leads both the experienced reader and the newcomer through an exciting path, unveiling the science and technology of state of the art PU production and recycling processes, shedding

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new light on the limits of current approaches, advances in research and future opportunities for closing the material cycle.

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A PHOTO, A FACT, AN EMOTION

"Cows and rivers are sacred in Hindu culture. Here in a river in Kathmandu, Nepal, a cow is washed in a holy river that runs north towards the mountains where the gods live. Traditionally, left over food was thrown in the river as a tribute to the gods but with the rise of plastic, a ritual that was once beneficial for nature is no suffocating the very places that Nepali people hold sacred."

"SACRED WASTE"

Kathmandu, Nepal

Nicholas Dunning, New Zealand

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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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